

Exploring behavioural responses of shorebirds to impulsive noise

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Abstract

Despite recent research interest in the consequences of chronic noise for animal populations, the effects of impulsive noise are less well understood. Determining wildlife responses to impulsive sound is pertinent in coastal areas where development of port and power generation industries may result in disturbance from impulsive sounds such as percussive piling, especially around estuaries which may support internationally important numbers of bird species. Discussions between regulators, planning authorities and regional development agencies over precautionary levels of impulsive sound emitting activities are common, yet we're unaware of any study which provides guidance on acceptable noise limits that would keep disturbance to shorebirds to a minimum. This experimental study intentionally disturbed birds at a high tide roost site, an agricultural field adjacent to the sea wall on the south bank of the Humber estuary in northern England, using an impulsive sound, and their behavioural responses were recorded. The researcher sounded an air-horn at ever decreasing distances towards a mixed species flock of shorebirds. While visual disturbance from the experimenter was taken into consideration in the methods, we cannot statistically separate their effects in this study. An ordinal logistic regression model demonstrated a statistically significant positive relationship between the decibel (dB(A)) level experienced by the birds and the behavioural response observed. In principle, this model can be used to predict the probability of a particular behavioural response at a given dB(A) level. Estimates of sound levels which included calculation of geometric and atmospheric attenuation were superior over more simple measures in predicting behavioural responses. This study has implications for assessing possible disturbance caused by impulsive noise, and adds

to growing evidence that the effects of noise may be an important, albeit not straightforward, consideration in management decisions made for shorebirds and other avian species.

Key words: coastal development, disturbance, human-wildlife conflict, Humber Estuary, impulsive noise, waders.

There has been increasing concern raised over the consequences of human-induced noise to animal populations and the integrity of protected areas (Brumm 2004; Brumm & Slabbekorn 2005; Habib *et al.* 2007; Barber *et al.* 2010). Determining the severity of the impacts of disturbance from human activity on wildlife is essential for appropriate management decisions (Gill 2007), yet quantifying these has proven challenging. Risk assessment for potentially disturbing human activity (*e.g.* construction) is based on the precautionary principle, and the absence of scientific knowledge does not necessarily result in postponement of mitigation (Myers 1993). There is generally an assumption that noise disturbance is harmful to birds and that there is a safe threshold noise level at which noise does not have a significant effect on birds using a site; moreover, this may be extrapolated as being constant across all species and sites. Yet given differences in background noise levels and the potential for birds to habituate to some types of disturbance, the response to auditory disturbance (and thus the maximum noise level at which a behavioural response is seen to be absent) is likely to vary quite substantially between species, across sites and seasons, and for different types and levels of sound.

There has been recent research into the effects of noise exposure on wildlife. Direct

effects of chronic noise exposure include hearing loss, increased production of stress hormones and hypertension (Barber *et al.* 2010). Indirect effects include the masking of acoustic signals such as calls and sounds which may otherwise lead to alert behaviour (Slabbekorn & Ripmeester 2008), as well as the reduced perception of unintentional sounds which may attract predators (Barber *et al.* 2010). It has also been suggested that exposure to chronic noise may result in increased vigilance in order to compensate for decreased auditory awareness (Quinn *et al.* 2006). All of these effects of chronic noise have clear deleterious consequences on resident or breeding animals at an individual level, and reductions in population density with proximity to noise sources are also well documented (Reijnen *et al.* 1995; Forman & Alexander 1998), as well as possible effects on demography (Habib *et al.* 2007). Minimising the effect of noise from human activity on wildlife using protected areas therefore is important for the management of these sites. Yet most conservation issues relating to noise in coastal areas concern impulsive noise disturbing wintering or migrating populations during the short-lived construction phase of a development, where the consequences are less well known.

Impulsive noise may be defined as a short duration noise extending over a range of

frequencies. The effects of impulsive noise on wildlife behaviour are pertinent to coastal areas experiencing intensive development due to the requirement of ground stabilising construction methods such as percussive piling. This process produces periodic impulsive noise of high intensity which has led to concerns over changes in distribution and abundance of coastal birds due to disturbance (Cutts *et al.* 2008). At present there is little evidence for how impulsive noise relates to associated behavioural responses, such as dispersal, movement or site abandonment. Regulation of impulsive noise at or adjacent to protected areas therefore is based upon previous, largely non-noise based, disturbance research, and needs to be verified.

The effects of impulsive noise may also differ between species due to differing energy requirements and food sources. For instance, Common Redshank *Tringa totanus* are regarded as being more vulnerable to the effects of disturbance and extreme weather than other waders, due to their requiring more feeding time to offset the energetic costs of foraging (Mitchell *et al.* 2000). Prevailing geographical and environmental conditions also affect behavioural responses and thus the effects of disturbance; birds are more likely to respond if there is suitable habitat present nearby to which to retreat (Gill *et al.* 2001). Likewise, in the case of impulsive noise, propagation of a potentially disturbing sound is highly dependent upon meteorological conditions. So, for example, there may be occasions when the effects of impulsive sound are more likely to result in site abandonment because they are emitted upwind of a roost

site and the nearest available alternative roost is downwind. The wide range of environmental and physiological factors affecting species-specific and site-specific costs in responding to disturbance increases the likelihood that a fixed cap on sound emissions, to minimise disturbance to shorebirds and other species, may fail to meet its objectives.

This project aims to develop a probabilistic approach for assessing the consequences of impulsive noise thresholds for roosting shorebirds. It explores the behaviour of multi-species shorebird flocks (comprised of Common Gull *Larus canus*, Curlew *Numenius arquata*, Golden Plover *Pluvialis apricaria* and Lapwing *Vanellus vanellus*) in response to varying sound levels, and determines the probability that a given level of impulsive noise will lead to one of an increasing range of behaviours, from no observed effect to the birds flying from the study area. It is hypothesised that increased impulsive noise will lead to more energetically expensive alarm or fleeing behaviour. This approach should provide government conservation agencies with better information for regulating impulsive noise thresholds for different shorebird species or sites (*i.e.* for minimising industrial and recreational disturbance levels), should be acceptable to stakeholders and also contribute to the growing literature on this important issue for wildlife management.

Materials and methods

Study site and species information

Great Britain is internationally important for wintering waders and wildfowl due to its

extensive estuarine habitats and mild climate, and the Humber Estuary has been estimated to account for around 170,000 of these birds, including internationally important numbers of several species (English Nature 2004; Musgrove *et al.* 2007). The estuary is designated as a Special Protection Area (SPA) under the Birds Directive (EC Directive on the conservation of wild birds; 79/409/EEC), as a Special Area of Conservation (SAC) under the Habitats Directive (Council Directive 92/43/EEC of 21 May 1992) and qualifies as a Wetland of International Importance under the Ramsar Convention. The estuary is also a busy commercial waterway and the south bank of the estuary includes some of the most industrialised parts of England, comprised of chemical works, oil refineries and power stations (Gameson 1982; English Nature 2004). Development of regional industry is important for employment and the local economy, but its proximity to the Humber Estuary SPA gives rise to concern over the potential consequences of chronic and impulsive noise from industry for birds on SPA.

Given the protection accorded to a number of coastal sites around Britain, permission may be required for any plan or project to be conducted on or near them (Natural England, pers. comm.), including experimental research. The study described here, which included noise disturbance of wintering shorebirds, was granted permission but required that our activity would have a negligible effect on the long-term protected status of adjacent sectors of the Humber Estuary SPA. Due to the difficulty in gaining both regulator (Natural

England) and owner permission for multiple sites, experimental disturbance was carried out from January–March 2010 on a single site close to Stallingborough, northeast Lincolnshire (53°35'N, 0°08'W). The status of the site (which is currently agricultural but earmarked for development) facilitated permission being obtained for the study near Stallingborough. Permission was also granted on the basis that the study was being made at the end of the winter, when flock sizes were smaller, and our agreeing to conduct disturbance events only when the weather was mild (*i.e.* in the absence of strong winds, and when temperatures were above zero; see below).

The study site consisted of a large farmland field, close to a number of industrial power plants, used by several species of shorebirds as a high tide roost. Over the study period the area was used by small flocks (range = 4–34 individuals, median = 14) of Common Gull, Curlew, Golden Plover and Lapwing. While these numbers were as expected for the time of year, this site and the surrounding area attracts large numbers of waterbirds during migration and winter; for example, a peak November count of > 4,000 Golden Plover was recorded in 1999 (Austin *et al.* 2008).

Sound monitoring and behavioural observations

Two Casella CEL-593 sound level analysers were used for monitoring both ambient and impulsive noise levels, using a 1/3 octave band filter to measure the energy average L_{Aeq} (defined as the equivalent continuous sound which would contain the same sound

energy as the time-varying sound over the same time period). Use of 1/3 octave band measurement provides better resolution, in terms of the frequency components of a given sound, than broad band or octave band; on the CEL-593 this constitutes a range of 33 bands ranging from 12–20,000 hz as defined by the International Standards Organisation (ISO) (ISO 266:1997), each with an associated decibel level. The CEL-593 also includes an event mode whereby any sound detected over a particular threshold level will have its length and maximum level recorded separately from the background measurement in progress. A threshold level of 71dB(A) was found to be appropriate for detecting most intentional disturbances whilst preventing false detection of variation in background noise levels or other incidental noise such as gusts of wind. The high threshold was required due to the close proximity of operational power production plants and associated traffic. A-weighting of decibels was used when calculating overall broadband sound levels. This weighting scheme, which is based on the inverse of a curve of sound intensities perceived as being equally loud to humans, has also been found to approximate to hearing in birds (Dooling & Popper 2007).

Fieldwork was carried out on days forecasted for fair weather and winds of < 5 m/s in order to ensure accuracy of sound monitoring. On four visits where wind speed was > 5 m/s fieldwork was abandoned because, despite wind shields being used, extraneous noise is created as higher speed winds move over the microphone. Six days of sampling were

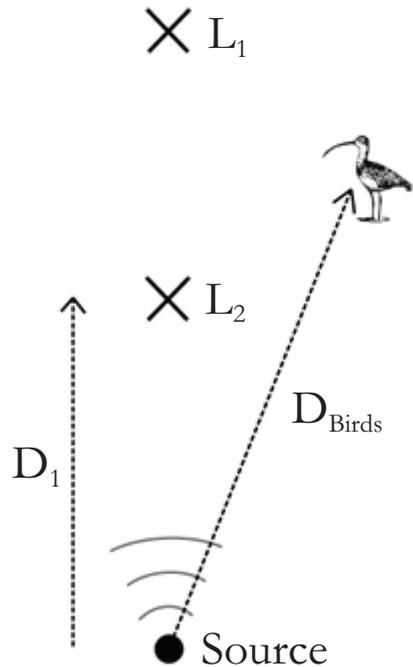


Figure 1. Placement of sound recording instruments ($X L_1$ and $X L_2$) relative to the source of the sound. By using sound levels recorded at a known distance (e.g. D_1) to calculate the power of the source it is possible to determine approximately the decibel (dB(A)) level experienced at other distances (D_{Birds}).

completed in 2010, on 30 January, 20, 25 and 27 February, and 7 and 12 March 2010. The instruments were set up at least 2 h prior to high tide while birds were feeding on the estuary, in order to avoid unintentional disturbance. They were placed in the centre of the field at 165 m from either end, at a height of 1 m, in order to provide maximum coverage for possible locations of roosting birds (Fig. 1). Observers then left the field and waited for high tide. The instruments were left running to provide an average

ambient noise level (L_{Aeq}) estimate for the measurement period (approximately 1–2 h), and because it was not possible to manipulate the equipment once the birds had arrived without causing unintentional disturbance. Shortly after high tide the number of birds present on the site was counted and their position relative to the two instruments (measured from the first location by compass) was recorded, for use in estimating the decibel level they experienced. The birds show great fidelity to their roosting location within the field and the maximum angle between the observer, the downrange meters and the birds was 35° (see Appendix A).

Starting at the end of the field furthest from the majority of birds present (typically around 250 m) an air horn was used as a sound source to cause intentional disturbance (FPS air horn, 114dB(A) measured at 2 m from source; www.screwfix.com). The air horn was sounded for a standardised 3 s per replicate. Responses by the birds were classed as follows:

- 0 – No behavioural changes observed
- 1 – Behavioural change (*e.g.* vigilance or alarm calls) but not flight
- 2 – Flew but soon returned to the site
- 3 – Flew and abandoned the site.

The number of birds showing each type of behavioural response within the flock was also recorded. If there was no observable behavioural change in any of the birds (or else after a non-flight response had ended) the distance to the instruments was decreased by 20 m, and the horn was sounded again. If the sound resulted in a flight response then we would wait until the

birds returned, or could be confirmed as having abandoned the site before proceeding. Birds that flew out of site were considered to have abandoned the study area. Five minutes were allowed to pass between replicates, starting from when birds were settled to reduce the effects of visual disturbance (*i.e.* the birds seeing the researcher), which is also likely to affect the behaviour of the birds (Rees *et al.* 2005; Bregnballe *et al.* 2009). Once all birds had abandoned the site the instruments were turned off and removed. Sound data were downloaded using the software dB5 (Casella USA), and exported to an Excel spreadsheet containing the time, date, distance to each instrument, species, flock size and the birds' response to each intentional disturbance event. The distance from the source to each flock was determined, based on their position relative to the two instruments.

Downrange noise and atmospheric attenuation

In order to obtain an approximate measure of the decibel level of the intentional disturbance experienced by a given flock of birds, an estimate of the attenuation of the sound over distance due to geometrical divergence (spherical spreading in the free field from a point sound source, based on the distance from the source to the receiver, A_{div}) and atmospheric absorption (attenuation coefficient dependent upon temperature, pressure and humidity, A_{atm}) was calculated. A power level (L_W) for the source was calculated based on the readings at each instrument, which was then used to determine the approximate decibel level received by the birds at a given distance

(L_{Birds}). Full details of the calculations and our justification for using a spherical spreading model of attenuation in the present study are given in Appendix A.

Statistical analysis

Ordinal logistic regression analysis was carried out in R 2.11.1 using the package Design 2.3-0 (R Development Core Team 2010). Behavioural response was ranked by severity (0–3) to be used as an ordinal dependant variable. A maximal model (MAM) including all possible factors and covariates (Response $\sim L_{\text{Birds}} + \text{Species} + \text{Flock size} + \text{Temperature} + \text{Pressure} + \text{Humidity}$) was reduced sequentially by dropping non-significant terms. No interactions were included due to low sample sizes. The MAM could then be used to make predictions of the probability of a particular behavioural response at a given level of L_{Birds} . The proportional odds assumption, which states that the coefficient between pairs of outcome groups remains constant (*i.e.* the coefficient between response 0 and 1 is equal to that of 1 and 2), was tested using a graphical approach suggested by Harrell (2001) and was found to hold. A chi-squared goodness of fit was performed to test the fit of the model to the original data.

As birds disturbed at the start of the study were potentially the same birds disturbed on subsequent days, and the same birds were also likely to be consecutively disturbed within days, there was a high probability of pseudoreplication in the dataset. The ordinal logistic package used to predict probabilities of multiple behaviour types over the scaled impulsive noise levels

was not able to account either for repeated measures or for random effects to control for this. Generalized linear mixed models (lmer from R package “lme4” 0.999375-35) and generalized estimating equations (geeglm from R package “geepack” 1.0–17) were used to examine whether the relationship between L_{Birds} , species and behavioural response would remain when including random effects to account for pseudoreplication (see Appendix B). While the effect of L_{Birds} was reduced on taking into account repeated measures made across days, it remained positive and statistically significant. The ordinal logistic regression results are therefore provided on the basis that this pseudoreplication does not appear to detract significantly from the qualitative conclusions of this study, while allowing presentation of the results in a probabilistic format (see later). Methods to combine the ordinal regression with repeated measures and random effects are under development (see Appendix B).

Linear discriminant analysis (R package “MASS” 7.3–7) was conducted as a *post-priori* assessment of the possibility of increasing the availability or complexity of information useful for predicting bird behaviour. This would be of value for determining the most suitable methods for obtaining accurate estimates of local behavioural responses to impulsive noise disturbance. Behavioural response was the categorical dependent variable while species and various measures of L were used as factors. The rate of grouping accuracy (%) was calculated by comparing predicted and observed responses for each case. By using different measures of L in separate analyses

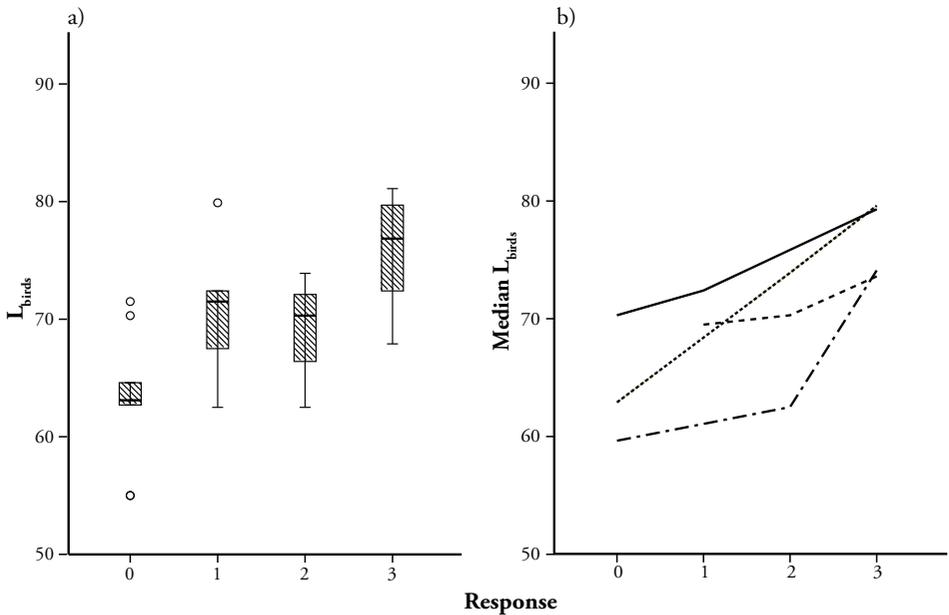


Figure 2. a) Boxplot showing the relationship between behavioural response and dB level experienced by intentionally disturbed birds. The line in the centre of the box represents a median value, and the top and bottom represent 25th and 75th percentiles. Whiskers represent minimum and maximum values excluding points that fell outside 95% confidence intervals. b) The relationship between behavioural response and median dB level experienced by four different species of bird which were intentionally disturbed. Full line = Curlew, dotted line = Golden Plover, dashed line = Common Gull, Dot-dash = Lapwing.

it was possible to analyse the reliability of using the recorded sound level at a single instrument (L_1 and L_2) or arithmetic averages between two microphones (L_{1+2}) in prediction of behavioural response.

Results

Relationship between L_{Birds} and behavioural response

The relationship between L_{Birds} and behavioural response is shown in Fig. 2a. L_{Birds} was highly variable for no observable behavioural response (median = 63.1 dB(A), range = 54.9–71.5 dB(A) with a high

proportion of extreme outliers. Flight with all birds abandoning the site (median = 76.85 dB(A), range = 67.9–81.1 dB(A)) appears to occur at notably higher noise levels than any other response, though its lowest value does fall below the median of the other two behavioural responses. If separated by species it is clear that flight with abandonment is indeed seen at higher levels, though there is inter-specific variation in the thresholds observed (Fig. 2b). The noise levels that elicit non-flight behavioural (median = 71.5 dB(A), range = 62.4–79.1 dB(A)) and flight with return (median = 70.3 dB(A), range = 62.4–73.9 dB(A))

responses appear to show similar values on grouping species (Fig. 2a), but not when the species are considered separately (Fig. 2b). In addition to the apparent difference in thresholds between species, there is also a notable difference in the slope of the association; for example, the slope for the Common Gull is notably less than that observed for Golden Plover. In terms of its behavioural response, Lapwing appears to be the species most sensitive to intentional disturbance, while Curlew is the most tolerant (Fig. 2b). There was no detectable effect of sensory loading (*e.g.* a decrease in thresholds due to repeated treatments), but see Appendix B.

Ordinal logistic regression

There was a significant relationship between dB(A) level experienced (L_{Birds}) and behavioural response (Wald $Z = 3.58$, Coefficient = 0.277, $P < 0.001$). All other factors were found to be non-significant. Testing for an interaction between L_{Birds} and species, which preliminary data exploration suggested may be useful in describing variation in response (Fig. 2b), was not possible in the present study due to low sample sizes. In a model containing L_{Birds} and species as factors, the latter was found to be non-significant (Wald $Z = 1.60$, Coefficient = 0.776, $P = 0.11$, n.s.). Goodness of fit tests indicate that the MAM does not differ significantly from the observed values for behavioural response ($\chi^2_{66} = 68.82$, $P > 0.38$, n.s.). The ordinal logistic regression model was also used to predict probabilities of behavioural responses for given levels of L_{Birds} (Fig. 3). Intentional disturbance at very low dB(A) levels is

highly unlikely to elicit a behavioural response, while at above 65.5 dB(A) a behavioural response of some kind becomes more likely to occur than no response. At above 72.2 dB(A) flight with abandonment of the site becomes the most likely outcome of the disturbance. If non-response and non-flight response were taken to be relatively harmless, and flight responses potentially costly (in terms of energy expenditure), then for those species studied at the site a costly outcome becomes more likely at = 69.9 dB(A).

Linear discriminant analysis

The accuracy of the prediction of observed values differed by the measure of dB(A) used, as shown in Table 1. L_{Birds} which is the measure including attenuation calculations, was the most accurate at predicting the correct response (64.5%). Readings at single instruments without calculations (L_1 and L_2) were less accurate than this (38.7% and 54.8%, respectively), as was using a mean L value of the two instruments (51.6%). The predictions given by using L_{Birds} were within a single measure of the response (*e.g.* predicting a type 0 or type 1 response for cases where no behavioural response was observed) for the vast majority of cases (93.6%), and were almost as successful at predicting harmless against harmful responses (as defined, Fig. 3) 87.1% of the time.

Discussion

Noise and behaviour

All bird species subject to intentional disturbance showed a range of behavioural responses, with the probability of a more

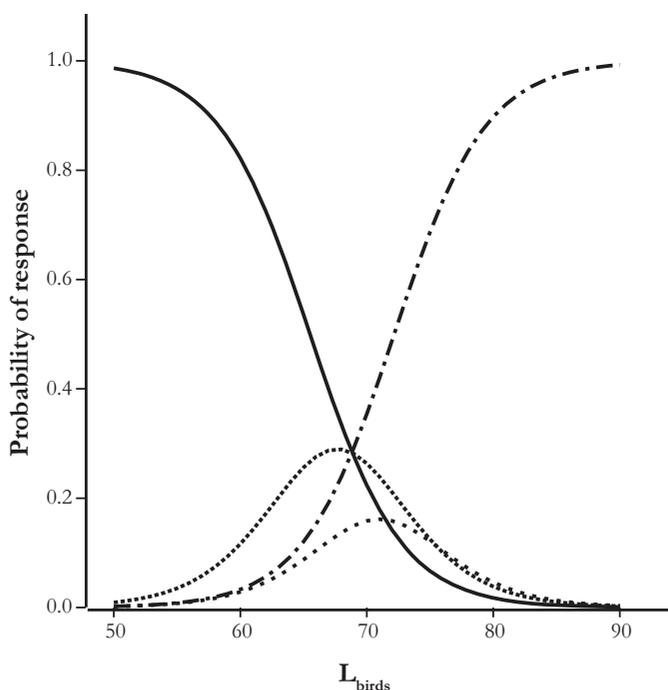


Figure 3. Probability of behavioural response based on dB(A) level experienced based on predictions from an ordinal logistic regression model. Solid line = no behavioural response observed (0), dotted line = behavioural change, but did not fly (1), dashed line = flew but returned to area (2), dot-dashed line = flew and abandoned the area (3).

Table 1. Accuracy of predictions of behavioural response to intentional disturbance for four different measures of decibel level using a linear discriminant analysis.

Measure of decibel level	% Prediction accuracy		
	Exact	Within 1 response level	Non-flight <i>versus</i> flight response
L_1	38.7%	61.3%	45.2%
L_2	54.8%	87.1%	67.7%
L_{Mean}	51.6%	90.3%	71.0%
L_{Birds}	64.5%	93.6%	87.1%

energetically costly behavioural response occurring increasing with noise levels experienced. The severity of the response observed also increased with noise levels, in a species-specific manner. If we assume that behavioural responses that include flight are energetically costly, could potentially affect the survival or productivity of an individual bird and thus contribute to negative trends in bird numbers locally, then impulsive noise limits should be restricted to $< 69.9\text{dB}$ at this site (but see discussion of caveats below). There was no significant difference in the behavioural responses of the four shorebird species considered in this study in response to impulsive noise, but this may be attributable to the low sample size.

The behavioural response of birds to human activity is thought to depend upon associated predation risk and energetic cost (Yasue 2005), with humans perceived as predators (Frid & Dill 2002), yet birds are known to respond to impulsive noise without visual stimuli. As remote noise-making equipment was not used in this project, it is likely that the increase in severity of behavioural responses with noise levels was to some extent confounded by visual stimuli. D_{Birds} (the distance from the sound source to each flock) was non-significant in the ordinal logistic regression model, but when used as a predictor in the linear discriminant analysis it was similar to a single instrument at predicting behavioural responses. At face value this suggests that behavioural responses largely depended upon the visual stimulus of the researcher approaching the birds. However, this relationship is more likely to be due to D_{Birds} showing a relatively strong negative

correlation with L_{Birds} (the noise level experienced by the birds). This is unsurprising as the calculated power level (L_{W}) of the source was approximately the same for each repeat, and attenuation increases with distance. In a practical situation where impulsive noise events may vary considerably in both power and frequency components D_{Birds} is likely to be a poor predictor of behavioural response. Nevertheless, there is likely to be a threshold distance where L_{Birds} is low enough as not to be perceived against background noise, but determining this accurately in practice would require knowledge of the likely sound sources, the frequency components they produce, and atmospheric conditions. Behavioural responses resulting from walking towards the flock, prior to the use of the air horn were rare. Where birds responded we paused until they had returned and settled or until they had left the area before continuing with the experimental noise with remaining birds. Nevertheless, it is likely that a combination of visual and auditory disturbance stimuli resulted in the responses observed, particularly in this study area where chronic ambient noise exposure (average $L_{\text{eq}} = 68\text{dB(A)}$) may have resulted in increased vigilance (Quinn *et al.* 2006). This combined effect may have resulted in increase in the severity of responses observed; use of remote sound-making equipment and also testing the effects of approaching the birds without using noise (*i.e.* quantifying the visual factor) will help to clarify this issue.

Behavioural responses to intentional noise disturbance appear to differ in a

species-specific manner, though an increased sample size will be required to determine whether this effect is biologically relevant. Blumstein *et al.* (2005) suggested that body size be used as a predictor of responses to disturbance, with body mass positively correlating with the distance at which birds fly from a predator. Contrary to this hypothesis, the largest species in this study, Curlew, appeared to be the species most tolerant to noise disturbance, while Lapwing showed the lowest apparent thresholds. This may be due to differences in hearing, or to differences in perceived predation risk posed by impulsive noise, as opposed to approaching pedestrians. Alternatively, there may be a difference in the amount of feeding opportunities required to compensate for the flight response relative to the time of the tide, the distance to the nearest alternative roost in this linear landscape and the costs associated with flight for species of different size (Mitchell *et al.* 2000). Common Gulls had relatively low noise thresholds, with the least steep slope between non-flight and flight response. When abandoning the site at high tide, Common Gulls were able to retreat to the adjacent estuary as they are able to swim on the water, while waders would have to fly to a different inland roost site as their feeding habitat is not accessible at high tide. This may represent a difference in energetic costs affecting inter-specific responses to disturbance, though it will be necessary to test this assumption at different sites with different associated costs.

The study was made at a time of winter when the flock sizes in the study area were small and relatively invariable in order to

gain the permissions needed to undertake work. This may have affected the estimates of noise thresholds; although flock size was not found to be significant in predicting behavioural responses in this analysis, other studies indicate that it influences foraging efficiency and flight distances in response to human disturbance in waterbirds (Mori *et al.* 2001; Yasue 2005).

Practical considerations and applications

Despite the caveats regarding confounding variables, some recommendations and guidelines can be provided to developers, government conservation agencies and non-governmental conservation organisations. The study set out to explore whether simple behavioural observations can provide a probabilistic framework of behavioural responses of shorebirds to potentially disturbing elements of the human environment. We have, for a given site, shown that the likelihood of eliciting a particular alarm or flee behaviour in roosting shorebirds can be determined by estimating noise thresholds that take into account distance between the noise emission and the roost and atmospheric conditions, *i.e.* what the birds actually hear down-range of a sound source. Our discussions with those involved in the development and conservation of estuarine environments suggest that this site-based approach would be preferable to adopting a precautionary principle across sites. We therefore recommend that research-led estimates of the risks of noise disturbance to shorebirds be put into practice more widely.

It is important that the effects of impulsive noise on wildlife be addressed, particularly in areas undergoing extensive development. Deleterious effects of chronic noise exposure have been suggested to begin at levels as low as 55–60 dB(A) (Dooling & Popper 2007), though data on physiological effects are lacking. The effects of chronic or impulsive noise on the population density of wintering birds is also poorly understood, yet should be addressed if development of modern energy sectors around the UK coastline is to be harmonised with the UK's commitment to protecting internationally important wetland sites.

It is unclear whether it is the perceived change in impulsive noise in relation to ambient noise, or simply the level of the impulsive sound itself which causes the behavioural responses. This is important in how limits are defined and comparison studies with quieter sites will be necessary to determine which of these best represents birds' recognition of impulsive sound. Nevertheless the relationship between predicted probability of a behavioural response and the estimated sound levels resulting in a behavioural response (L_{Birds}) provides a reasonable best practice method for regulators and regional authorities to define noise limits based on a likelihood of risk, for a given site based on its ambient noise level. Of course, even if another site has a similar level of ambient noise (L_{eq}), the host of environmental and physiological factors affecting behavioural responses will no doubt change the relationship, but L_{eq} is relatively easy to measure in practice.

The linear discriminant analysis indicates that L_{Birds} , which takes account of noise

attenuation, is superior to simple measures of dB(A) level by single instruments at predicting behavioural responses (Table 1). We are not the first study to draw attention to the need to quantify the “dose” or received noise at the position of the animal for studies of anthropogenic disturbance to be meaningful (Nowacek *et al.* 2001; Barber *et al.* 2010). While it would not be possible to determine the precise dB(A) experienced by flocks relative to instruments during monitoring activity, this may improve accuracy if likely distances, sounds and atmospheric conditions were considered prior to monitoring.

Conclusions

This study represents a first step towards addressing the ecological, environmental and regulatory issues affecting how noise limits are defined. The role of habituation, differences between species responses to different sounds, seasonal differences, energetic effects, carry-over effects and the relationship between ambient and impulsive noise levels should be assessed in a systematic fashion in order to provide a full understanding of the effects of human induced noise on wintering shorebirds. However, where the aim is to provide meaningful limits that minimise disturbance effects of coastal development on wildlife and to reduce precautionary limits where conflicts between industrial, regulatory and conservation stakeholders often arise, experimental approaches such as this one are useful in controlling for some of the vast amount of confounding factors influencing disturbance, its effects and its consequences.

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References

- Austin, G.E., Collier, M.P., Calbrade, N.A., Hall, C. & Musgrove, A.J. 2008. Waterbirds in the UK 2006/2007: The Wetland Bird Survey. BTO/WWT/RSPB/JNCC, Thetford, UK.
- Barber, J.R., Crooks, K.R. & Fristrup, K.M., 2010. The costs of chronic noise exposure for terrestrial organisms. *Trends in Ecology and Evolution* 25: 180–189.
- Beale, C. & Monaghan, P. 2004. Behavioural responses to human disturbance: a matter of choice? *Animal Behaviour* 68: 1065–1069.
- Blumstein, D.T., Fernández-Juricic, E., Zollner, P.A. & Garity, S.C. 2005. Inter-specific variation in avian responses to human disturbance. *Journal of Applied Ecology* 42: 943–953.
- Bregnballe, T., Aaen, K. & Foxx, A.D. 2009. Escape distances from human pedestrians by staging waterbirds in a Danish wetland. *Wildfowl* Special Issue 2: 115–130.
- Brumm, H. 2004. The impact of environmental noise on song amplitude in a territorial bird. *Journal of Animal Ecology* 73: 434–440.
- Brumm, H. & Slabbekoorn, H. 2005. Acoustic communication in noise. *Advances in the Study of Behaviour* 35: 151–209.
- Cutts, N., Phelps, A. & Burdon, D. 2008. Construction and Waterfowl: Defining Sensitivity, Response, Impacts and Guidance. Report to Humber Industrial Nature Conservation Association, Waterside, Lincs, UK.
- Dooling, R.J. & Popper, A.N. 2007. The effects of highway noise on birds. Report to California Department of Transportation, Contract No. 43A0139. Environmental Acoustics LLC, Rockville, Maryland, USA.
- Elgar, M.A. 1988. Predator vigilance and group size in mammals and birds: a critical review of the empirical evidence. *Biological Reviews* 64:13–33.
- English Nature, 2004. EC Directive 79/409 on the Conservation of Wild Birds: Special Protection Area (SPA) – Humber Estuary. English Nature, Wakefield, UK.
- English Nature, 2004. Reasons for recommendation as a possible Special Area of Conservation: Humber Estuary. English Nature, Wakefield, UK.
- Forman, R.T.T. & Alexander, L.E. 1998. Roads and their major ecological effects. *Annual Review of Ecology, Evolution and Systematics* 29: 207–231.
- Frid, A. & Dill, L.M. 2002. Human-caused disturbance stimuli as a form of predation risk. *Conservation Ecology* 6: 11.
- Fujii, T. 2007. Spatial patterns of benthic macrofauna in relation to environmental variables in an intertidal habitat in the Humber estuary, UK: developing a tool for estuarine shoreline management. *Estuarine, Coastal and Shelf Science* 75: 101–119.
- Gameson, A.L.H., 1982. Description of estuary and surveys. In A.L.H. Gameson (ed.), *The quality of the Humber Estuary, 1961–1981*, pp. 1–4. Yorkshire Water Authority, Leeds, UK.
- Gill, J.A., Norris, K. & Sutherland, W.J. 2001. Why behavioural responses may not reflect the population consequences of human

- disturbance. *Biological Conservation* 97: 265–268.
- Gill, J. 2007. Approaches to measuring the effects of human disturbance on birds. *Ibis* 149 (Suppl. 1): 9–14.
- Habib, L., Bayne, E.M. & Boutin, S. 2007. Chronic industrial noise affects pairing success and age structure of ovenbirds *Seiurus aurocapilla*. *Journal of Applied Ecology* 44: 176–184.
- Harrell, F.E. 2001. *Regression Modeling Strategies: With Applications to Linear Models, Logistic Regression and Survival Analysis*. Springer, New York, USA.
- International Standards Organisation. 1993. *Acoustics – Attenuation of sound during propagation outdoors – Part 1: Calculation of the absorption of sound by the atmosphere*. ISO 9613-1: 1993. http://www.iso.org/iso/catalogue_detail.htm?csnumber=17426.
- International Standards Organisation. 1997. *Acoustics – Preferred frequencies for measurements*. ISO 266: 1997. http://www.iso.org/iso/catalogue_detail.htm?csnumber=1350.
- Mitchell, C.R., Patterson, D., Boyer, P., Cunningham, P., McDonald, R., Meek, E., Okill, J.D. & Symonds, F. 2000. The summer status and distribution of Greylag Geese in north and west Scotland. *Scottish Birds* 21: 69–77.
- Mori, Y., Sodhi, N.S., Kawanishi, S. & Yamagishi, S. 2001. The effect of human disturbance and flock composition on the flight distances of waterfowl species. *Journal of Ethology* 19: 115–119.
- Musgrove, A.J., Collier, M.P., Banks, A.N., Calbrade, N.A., Hearn, R.D. & Austin, G.E. 2007. *Waterbirds in the UK 2005/06: The Wetland Bird Survey*. BTO/WWT/RSPB/JNCC, Thetford, UK.
- Myers, N. 1993. Biodiversity and the Precautionary Principle. *Ambio* 22: 74–79.
- Nowacek, S.M., Wells, R.S. & Solow, A.R. 2001. Short-term effects of boat traffic on bottlenose dolphins, *Tursiops truncatus*, in Sarasota Bay, Florida. *Marine Mammal Science* 17: 673–688.
- Quinn, J.L., Whittingham, M.J., Butler, S.J. & Cresswell, W. 2006. Noise, predation risk compensation and vigilance in the chaffinch *Fringilla coelebs*. *Journal of Avian Biology* 37: 601–608.
- R Development Core Team. 2010. *R: A language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL <http://www.R-project.org>.
- Rees, E.C., Bruce, J.H. & White, G.T. 2005. Factors affecting behavioural responses of whoopers swans (*Cygnus c. cygnus*) to various human activities. *Biological Conservation* 121: 369–382.
- Reijnen, R., Foppen, R., Ter Braak, C. & Thissen, J. 1995. The effects of car traffic on breeding bird populations in woodland. III. Reduction of density in relation to proximity of main roads. *Journal of Applied Ecology* 32: 187–202.
- Slabbekoorn, H. & Ripmeester, E.A. 2008. Birdsong and anthropogenic noise: implications and applications for conservation. *Molecular Ecology* 17: 72–83.
- Yasue, M. 2005. The effects of human presence, flock size and prey density on waterbird foraging rates. *Journal of Ethology* 23: 199–204.

Appendix A: Calculation of impulsive noise disturbance experienced by birds, taking account of atmospheric attenuation and directionality.

This Appendix describes the methods used to calculate the L_{Birds} level from the instrument recordings.

Attenuation calculation

In order to determine the decibel level of the intentional disturbance experienced by a given flock, an estimate of the attenuation of the sound over distance due to geometrical divergence (A_{div}) and atmospheric absorption (A_{atm}) had to be calculated. An approximate sound power level (L_{W}) for the source was calculated based on the readings at each instrument, which could then be used to determine the decibel level at a given distance (L_{Birds}) by adding the two attenuation coefficients:

$$L_{\text{Birds}} = L_{\text{W}} + A_{\text{div}} + A_{\text{atm}}$$

The geometrical divergence accounts for spherical spreading in the free-field from a point sound source, based solely on the distance (d , in metres) from the source to the receiver:

$$A_{\text{div}} = \left[-20 \log_{10} \left(\frac{d}{d_0} \right) + 11 \right]$$

It is recognised that the assumption of spherical spreading (*i.e.* assuming that sound energy radiates from the source in an equivalent way in all directions) when the air horn is a directional sound source, and thus the omission of directionality in the divergence calculation for L_{Birds} , is a limitation of the methods used in this study which will be addressed in future studies. Meanwhile, the significance of omitting directionality on this occasion was considered to be reduced by both the sound level meters and the birds being positioned within a (maximum) 35° cone in front of the sound source.

The attenuation due to atmospheric absorption is given by:

$$A_{\text{atm}} = \frac{\alpha d}{1000}$$

where α is the atmospheric attenuation coefficient in decibels per km and d is distance in metres as above. This was calculated using an online calculator provided by the National Physical Laboratory (<http://resource.npl.co.uk/acoustics/techguides/absorption/>), based on equations described in ISO 9613-1. Measurements of temperature, pressure and % humidity were obtained for each day from a private weather station at Grimsby.

As atmospheric attenuation varies in relation to the frequency of the sound in question, L_{W} values had to be calculated for the source for each 1/3 octave band recorded in order to give a meaningful power level:

$$L_{\text{W}1} = 10 \log_{10} \left[\sum_j^{33} 10^{0.1[L_1 - A_{\text{div}} - A_{\text{atm}}]} \right]$$

where j is an index indicating thirty-three 1/3 octave band frequencies and L_1 is the decibel level for a single band observed at the first instrument. It was necessary to log transform and then back-transform when summing decibels as they are logarithmic units. This calculation was also carried out for the second instrument to give a second estimate of the power level (*i.e.* $L_{\text{W}2}$).

Values for $L_{\text{W}1}$ and $L_{\text{W}2}$ were then compared; in ideal conditions these would be identical as they are both estimates for the same source, but variables not accounted for in the calculations (*e.g.* reflections from the ground and the effects of wind) lead to variation. Where these differed by > 5dB the L_{W} for the closest instrument to the source was used, as this was likely to be less affected by other factors. If the two L_{W} estimates were within this range the mean value was used. These estimates could then be used to calculate the decibel level experienced by the birds, as shown in the first equation.

Appendix B. Accounting for Pseudoreplication.

As sampling occurred at the same site on multiple days, and birds were exposed to multiple noise replicates within those days, it is possible that the variance in behavioural responses observed were due to either, or both, of these factors. This pseudoreplication is not accounted for in the minimum adequate model described in the results section (for reasons described below); however, the analyses below suggest that the positive relationship between L_{Birds} and behavioural response remains significant even when accounting for these factors.

The results of generalized linear mixed models (GLMM) using the package “lme4” in R are shown in Tables B1 and B2. In the first, noise replicate was specified as a random term; in the second, day was specified as a random term, with noise replicate as a nested term within day. As with the results of the main analysis, species was found to be non-significant (Table B3). The results of a generalized estimating equation model (GEE) using the package “gee” are shown in Table B4. Noise replicates was included as a random term.

Table B1. Relationship between L_{Birds} and behavioural response from generalized linear mixed model including noise replicates as a random term.

	Estimate (s.e.)	Z value	P
Intercept	-22.02 (8.63)	-2.55	0.011
L_{Birds}	0.34 (0.13)	2.62	0.009

Table B2. Relationship between L_{Birds} and behavioural response from generalized linear mixed model including day and noise replicates as random terms.

	Estimate (s.e.)	Z value	P
Intercept	-33.83 (13.00)	-2.60	0.009
L_{Birds}	0.51 (0.19)	2.66	0.008

Table B3. Maximal model of the GLMM shown in Table B2.

	Estimate (s.e.)	Z value	P
Intercept	-63.40 (24.93)	-2.54	0.011
L_{Birds}	0.93 (0.35)	2.63	0.009
Species	0.93 (1.01)	0.92	0.35 (n.s.)

Table B4. Relationship between L_{Birds} and behavioural response from generalized estimating equation model.

	Estimate (s.e.)	Wald	P
Intercept	-6.71 (1.47)	20.98	<0.001
L_{Birds}	0.13 (0.02)	49.81	<0.001
Day 2	-0.13 (0.55)	0.06	0.81 (n.s.)
Day 3	-0.67 (0.63)	1.13	0.29 (n.s.)
Day 4	0.38 (0.56)	0.45	0.50 (n.s.)
Day 5	0.26 (0.54)	0.23	0.63 (n.s.)
Day 6	-0.07(0.54)	0.02	0.90 (n.s.)

While use of statistical approaches such as GLMM and GEE allow us to take account of the apparent pseudoreplication in the data, they do not simultaneously provide predicted probabilities for each potential behavioural responses for a given value, as outlined in the main text for multiple ordinal logistic regression. This feature was central to the idea of a risk-based analysis relevant to impact assessments, and while the model outputs should be interpreted with care, we feel it was justified to take this approach to statistical analysis. For future studies with similar requirements, i.e. repeated measures in a proportional odds logistic regression to allow for random effects and serial auto-correlation (see R package “repolr”), may represent an appropriate method if it is developed to allow predictions across all ordinal responses simultaneously (Parsons *et al.* 2009).

Reference

Parsons, N.R., Costa, M.L., Achten, J. & Stallard, N. 2009. Repeated measures proportional odds logistic regression analysis of ordinal score data in the statistical software package R. *Computational Statistics and Data Analysis* 53: 632–641.