

# **The Effects of Noise on Biodiversity (NO0235)**

**Final Report for Defra**

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# Contents

1	Executive Summary .....	2
2	Aims and objectives.....	4
3	Introduction.....	5
3.1	Background.....	5
3.2	What is anthropogenic noise? .....	5
3.3	UK noise and biodiversity policy.....	6
3.4	Applicability of standard human noise assessment to wildlife .....	7
3.5	UK Biodiversity Action Plan Priority Species.....	8
4	Methodology.....	8
4.1	Literature searches .....	8
4.2	Studies considering impact of anthropogenic noise on UK priority species .....	9
4.3	Extrapolations.....	9
4.3.1	Audiograms and noise spectra.....	9
4.3.2	Non-anthropogenic noise sources and UK species.....	10
4.3.3	Phylogenetic and ecological extrapolation.....	11
5	Impacts of anthropogenic noise on UK PS and SPI.....	11
5.1	Overview.....	11
5.2	Representation of noise sources .....	12
5.3	Taxonomic representations.....	13
5.4	Representation of impacts of noise measured.....	13
5.5	Strength of evidence for impacts of noise.....	14
5.6	Individual fitness.....	15
5.7	Summaries for taxonomic groups.....	16
5.7.1	Birds .....	16
5.7.2	Terrestrial mammals.....	17
5.7.3	Reptiles .....	18
5.7.4	Amphibians .....	18
5.7.5	Fish.....	19
5.7.6	Terrestrial invertebrates .....	19
6	Moving forward: considerations for future work.....	20
6.1	Taxonomic biases .....	20
6.2	Study design .....	20
6.3	Noise quantification .....	21
6.4	Individual level studies .....	21
6.5	Community and ecosystem level studies .....	21
7	References.....	22

# 1 Executive Summary

Anthropogenic (man-made) noise increased across the globe in the 20<sup>th</sup> Century and is now recognised as a major environmental change in the 21<sup>st</sup> Century. As part of the EC Environmental Noise Directive, the UK has strived to reduce the impact of anthropogenic noise on humans. Policy has also extended to include impacts of noise on the marine environment. Defra commissioned this review to collate the literature on the impacts of anthropogenic noise on non-marine UK species, with a particular focus on UK Priority Species (UK PS) and Species of Principal Importance (SPI) for England, Northern Ireland, Scotland and Wales. The project had four key objectives:

**Objective 1: Identify and review relevant literature.** Collate information on the main sources of anthropogenic noise, the known impacts of such noise on UK species in general and UK PS and SPI in particular, the hearing abilities of UK PS and SPI, and the known impacts of noise (not specifically anthropogenic) on species that are similar phylogenetically and ecologically to UK PS and SPI.

**Objective 2: Make an informed assessment of the strength of evidence.** Consider the literature obtained from Objective 1 and independently assess the strength of conclusions that can be drawn about the potential impacts of anthropogenic noise.

**Objective 3: Use indirect information to assess the likely impact of anthropogenic noise on priority species.** Enhance the limited evidence base available from studies directly assessing the impact of anthropogenic noise on UK PS and SPI by: combining information about the hearing capabilities of UK PS and SPI with knowledge about major sources of ecologically relevant anthropogenic noise; drawing conclusions from studies examining the impact of non-anthropogenic noise; and extrapolating from research investigating the impact of anthropogenic noise on species that are phylogenetically and ecologically similar to UK PS and SPI.

**Objective 4: Identify patterns and gaps in the findings.** Tabulate key information, identify patterns and discuss knowledge gaps relating to the potential impact of anthropogenic noise.

Comprehensive searches were made of the peer-reviewed literature and supplemented by web searches of publication lists from 25 UK wildlife organisations. Peer-reviewed papers and unpublished reports directly addressing the potential impacts of anthropogenic noise on UK PS and SPI were assessed for their strength of evidence and assigned one of three categories. Since impacts on individual fitness are of most relevance and importance for population viability, studies were also assigned to one of three categories depending on what conclusions could potentially be drawn in this regard. Available audiograms of UK PS and SPI were compared with noise spectra for each ecologically relevant noise source, with overlaps in auditory frequency responses of the species and dominant frequency of the noise source indicating potential vulnerability. Potential impacts of anthropogenic noise on UK PS and SPI were extrapolated from known responses of these species to non-anthropogenic noise sources and from the effect of anthropogenic noise on phylogenetically and ecologically similar species from the UK and around the world.

In total, 86 publications directly addressing the impact of anthropogenic noise on non-marine species were identified, of which 16 focused on UK PS and SPI. Overall patterns were generally similar whether considering studies on species throughout the world, on all UK species or on UK PS and SPI only: the literature is currently dominated by studies on road traffic noise, on birds and on behavioural impacts. Inclusion of extrapolatory evidence from the combination of audiograms and noise spectra, from non-anthropogenic noise data and from studies on phylogenetically and ecologically similar species allowed consideration of the potential impact of noise for a total of 79 UK PS and SPI.

The major finding is that a strong evidence base does not exist regarding the potential impact of anthropogenic noise on non-marine UK PS and SPI. Definite conclusions could be made only about the reed bunting (*Emberiza schoeniclus*), which exhibits shifts in song frequency in response to road traffic noise. It is also likely that foraging in brown long-eared bats (*Plecotus auritus*), singing in European robins (*Erithacus rubecula*), house sparrows (*Passer domesticus*), starlings (*Sturnus vulgaris*) and bullfinches (*Pyrrhula pyrrhula*), and the behaviour of common toads (*Bufo bufo*) are affected by road traffic noise to some degree. Common issues preventing strong conclusions for other species include a lack of sufficient controls to rule out potential confounding factors (e.g. changes in the behaviour of animals near roads may be the consequence of differences in lighting, disturbance or habitat differences, rather than noise) and the use of acoustic measurements that are more relevant to humans than the auditory capabilities of the study species. In addition, hardly any studies directly considered how anthropogenic noise might impact individual fitness; while several more studies provided good proxies for fitness, definite conclusions in this regard would also be premature.

To make a fair assessment of how much anthropogenic noise affects non-marine wildlife in general, and UK PS and SPI in particular, will therefore require further empirical work. Such work should ideally address the current taxonomic bias towards studies on birds, include carefully designed experimental studies (while bearing in mind that such research on species of conservation priority raises some ethical issues), quantify the noise sources of relevance in a way that relates to the hearing capabilities of the study organism, look beyond short-term studies to consider chronic and repeated exposure, focus on response indicators that can inform models of population viability, and investigate impacts at community and ecosystem levels as well as how individuals are affected.

## 2 Aims and objectives

The overall aim of this project was to assess the current state of knowledge relating to the likely impact of anthropogenic noise on UK non-marine species, and especially UK Priority Species (UK PS) and Species of Principal Importance (SPI) for England, Northern Ireland, Scotland and Wales. Specific objectives were as follows:

**Objective 1: Identify and review relevant literature.** Collate information on the main sources of anthropogenic noise in the UK, the known impacts of anthropogenic noise on UK species in general and UK PS and SPI in particular, the hearing abilities of UK PS and SPI, and the known impacts of noise (not specifically anthropogenic) on species that are similar phylogenetically and ecologically to UK PS and SPI.

This objective was fully met. Information was obtained on the main sources of anthropogenic noise (section 3.2), including representative noise spectra where possible (Appendix A). A comprehensive compilation of the peer-reviewed literature investigating potential impacts of anthropogenic noise on UK species was made, with additional studies also sourced from the grey literature (36 studies in total, 16 of which focused on UK PS and SPI; Appendix B). Audiograms (providing information on the hearing ability of an organism) were obtained for 15 UK PS and SPI (Appendix C). A comprehensive search was made of the peer-reviewed literature relating to impacts of anthropogenic noise on species worldwide, resulting in an additional 50 studies (Appendix B); research examining the impact of non-anthropogenic noise on UK PS and SPI (8 studies; Appendix D) and on terrestrial invertebrates worldwide (selected because of the paucity of any direct work on this taxonomic group; 2 studies, Appendix D) was also collated.

**Objective 2: Make an informed assessment of the strength of evidence.** Consider the literature obtained from Objective 1 and independently assess the strength of conclusions that can be drawn about the potential impacts of anthropogenic noise.

This objective was fully met. All studies directly examining the potential impact of anthropogenic noise (i.e. the 86 studies in Appendix B) were assessed and categorised on a three-point scale according to the strength of evidence provided.

**Objective 3: Use indirect information to assess the likely impact of anthropogenic noise on priority species.** Enhance the limited evidence base available from studies directly assessing the impact of anthropogenic noise on UK PS and SPI by: combining information about the hearing capabilities of UK PS and SPI with knowledge about major sources of ecologically relevant anthropogenic noise; drawing conclusions from studies examining the impact of non-anthropogenic noise; and extrapolating from research investigating the impact of anthropogenic noise on species that are phylogenetically and ecologically similar to UK PS and SPI.

This objective was fully met. All available audiograms for UK PS and SPI were combined with noise spectra from relevant major anthropogenic noise sources, and an assessment made of the likelihood that each species is vulnerable to each noise source. Conclusions about the likely impact of anthropogenic noise on UK PS and SPI were made from studies examining how non-anthropogenic noise affects such species and terrestrial invertebrates worldwide, and from research investigating the impacts of anthropogenic noise on species throughout the world.

**Objective 4: Identify patterns and gaps in the findings.** Tabulate key information and use these tables to identify patterns (e.g. whether specific taxa appear more vulnerable to anthropogenic

noise) and gaps in our knowledge (e.g. whether particular groups are under-represented in the literature on anthropogenic noise).

This objective was fully met. Detailed summaries of our findings are provided in tabulated and graphical form (see section 5 and Appendices E and F). The main patterns relating to noise sources studied, taxonomic biases, types of impact considered and strength of conclusions possible are discussed. Moreover, an assessment of what can be concluded about potential fitness implications is provided for each study. Suggestions are made in relation to these elucidated patterns about research that would be potentially valuable in the future.

## 3 Introduction

### 3.1 Background

Anthropogenic (man-made) noise increased across the globe in the 20<sup>th</sup> Century and is now recognised as a major environmental change in the 21<sup>st</sup> Century. In terrestrial environments, the prevalence of transportation networks, resource extraction and urban development is much greater today than in the past (see Barber et al. 2010; Laiolo 2010). Likewise, freshwater environments have suffered from increases in recreational boat activity, while fish species that inhabit estuarine areas or spend some time at sea may also be subjected to rising noise levels from coastal windfarms and construction work (see Popper & Hastings 2009; Slabbekoorn et al. 2010). All these sources contribute to greater noise levels in potentially important wildlife habitats. Since the implementation of the EC Environmental Noise Directive (END) in 2002 (<http://ec.europa.eu/environment/noise/directive.htm>), there has been an increasing awareness of anthropogenic noise, with policies being developed to protect and improve human health. Policy has also extended to include impacts of noise on the marine environment (<http://www.defra.gov.uk/publications/2011/09/30/pb13654-marine-policy-statement/>). Defra has commissioned this review to collate the literature on the impacts of anthropogenic noise on non-marine UK species, with a particular focus on species of conservation priority.

### 3.2 What is anthropogenic noise?

Anthropogenic noise is defined as unwanted sound generated by humans. As the term ‘unwanted’ is dependent upon the perception of each individual, this definition is highly subjective. However, the following are widely recognised as the main sources of anthropogenic noise in the non-marine environment (although non-marine species may also be affected by many more anthropogenic noise sources, such as construction and demolition, military activity, quarries, waste disposal, wind turbines and pile-driving):

- Road traffic – Perhaps the most prevalent and chronic source of environmental noise, road traffic noise is predominantly generated by a combination of vehicle engine and tyre-tarmac interaction noise. As part of the UK’s implementation of the END, noise has been mapped for major UK roads with over 6 million vehicle passes per year. This provides strategic information and estimated A-weighted sound levels for day, evening and night at locations across England<sup>1</sup>, Northern Ireland<sup>2</sup>, Scotland<sup>3</sup> and Wales<sup>4</sup>. Although traffic load, composition and road surface can differ dramatically between sites, the spectral content of the generated noise is generally considered to fall in the frequency range below 3 kHz, with minimal energy at higher frequencies (see Appendix A).

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<sup>1</sup> <http://services.defra.gov.uk/wps/portal/noise>

<sup>2</sup> <http://www.noiseni.co.uk/>

<sup>3</sup> <http://www.scottishnoisemapping.org/>

<sup>4</sup> <http://new.wales.gov.uk/topics/environmentcountryside/epg/noiseandnuisance/environmentalnoise/?lang=en>

- Air traffic – Aircraft generate noise during over-flight, take-off and landing operations. Noise levels for aircraft are strictly regulated by the EU and major airports in England<sup>1</sup>, Northern Ireland<sup>2</sup> and Scotland<sup>3</sup> have been mapped to show average daily, evening and nightly A-weighted sound levels; there are no major airports in Wales. It is difficult to obtain details on the spectral content of this noise source as standard protocols for human impacts require A-weighted sound levels only, but the general consensus is that peak, or maximum, noise lies below 5 kHz (see Barber et al. 2010 for overview; see Appendix A).
- Rail traffic – As with air traffic and road traffic noise, rail traffic noise across parts of England<sup>1</sup>, Northern Ireland<sup>2</sup>, Scotland<sup>3</sup> and Wales<sup>4</sup> has been mapped using A-weighted sound levels for day, evening and night. Beyond busy city stations, rail traffic noise can be relatively transient depending upon the number of trains servicing the route (see Appendix A).
- Industrial – Industrial noise includes sounds generated by industrial plants and machinery. The effect of industrial noise on members of the public is controlled through the Integrated Pollution Prevention and Control regime (now driven by the Industrial Emissions Directive), and through the statutory nuisance legislation provisions of the Environmental Protection Act 1990.
- Neighbourhood – Includes music, public houses and clubs and other incidental noises generated by humans, the characteristics of which are variable.
- Boat – Engine noise and noise generated by onboard devices. Boat noise in freshwater habitats is relatively poorly quantified (although see Amoser et al. 2004), but energy appears concentrated at frequencies below 6 kHz. Due to the sound transmission properties of water, noise in this environment can travel great distances (see Appendix A).

### 3.3 UK noise and biodiversity policy

#### *England*

The Noise Policy Statement for England (<http://archive.defra.gov.uk/environment/quality/noise/policy/documents/noise-policy.pdf>) applies to all forms of noise including environmental noise, neighbour noise and neighbourhood noise. It sets out the long term vision of Government noise policy, which is to “promote good health and a good quality of life through the effective management of noise within the context of Government policy on sustainable development”. This long term vision is supported by the following aims, through the effective management and control of environmental, neighbour and neighbourhood noise within the context of Government policy on sustainable development:

- avoid significant adverse impacts on health and quality of life;
- mitigate and minimise adverse impacts on health and quality of life; and
- where possible, contribute to the improvement of health and quality of life.

The NPSE provides the necessary framework to enable decisions to be made regarding what is an acceptable noise burden to place on society. The NPSE has been confirmed in the Natural Environment White Paper 2020 (NEWP) as the means by which noise is to be considered in England and its approach can be seen in new and emerging policy affecting a wide range of areas from planning to health. The NEWP states a mission is to halt overall biodiversity loss, support healthy well-functioning ecosystems and establish coherent ecological networks, with more and better places for nature for the benefit of wildlife and people.

The biodiversity strategy for England, Biodiversity 2020, states the desired outcomes for biodiversity and priority actions that will be taken to achieve them. Consideration of noise impacts on species may be relevant to the following key outcomes:

- By 2020 measures will be in place so that biodiversity is maintained, degradation halted and restoration is underway – more resilient and coherent and resilient ecological networks (includes Nature Improvement Areas, and capacity to withstand pressures such as climate change), healthy well-functioning ecosystems, delivering multiple benefits.

- By 2020 an overall improvement in status of our wildlife will have prevented further human-induced extinctions of known threatened species (linked to integrated landscape scale approaches as well as more targeted efforts to identify and protect threatened species-including genetic diversity, farmed and crop wild relatives).

Biodiversity 2020 lists priority actions under four main themes, one of which is Theme 3: reduce environmental pressures- integrate consideration of biodiversity within the sectors which have the greatest potential for direct influence, and reduce direct pressures. There is insufficient evidence to indicate that noise is causing significant adverse impacts on biodiversity, however the above measures and the policy set out in the NPSE provide a useful framework if more evidence becomes available. The White Paper also gives a commitment to work with its transport agencies and key delivery partners to contribute to the creation of coherent and resilient ecological networks, supported, where appropriate, by organisation-specific Biodiversity Action Plans, where consideration of noise impacts might be important.

Biodiversity 2020 includes actions under Theme 4, improving our knowledge base, including working collaboratively across Defra and the relevant agencies to direct research investment within Government to areas of highest priority to deliver the outcomes and priorities set out in this strategy. This research project will help towards improving this knowledge base.

#### *Scotland, Wales and Northern Ireland*

The devolved administrations also have policies in place to address the adverse effects of environmental noise on human health and wellbeing. Environment, transport, planning and health officials in devolved government, agencies and local authorities work with one another to develop and implement action plans under the Environmental Noise Directive and ensure that noise is appropriately embedded in other areas of policy where appropriate.

The 2012 Green Paper “Sustaining a Living Wales” (<http://wales.gov.uk/consultations/environmentandcountryside/sustainingwales/?lang=en>) proposes an ecosystem approach to environmental regulation and management in Wales. This will mean considering and regulating the environment and its health as a whole rather than dealing with individual aspects separately. It will mean weighing up and setting priorities for the many competing demands on natural resources to provide different services to society and taking steps that will help to maximise the environmental, economic and social opportunities available. Welsh Government has collaborated with Defra in the conception and oversight of the current project in recognition that decision-makers require greater clarity on the effects of noise on the natural environment.

Scotland is currently reviewing the Scottish Biodiversity Strategy to make sure it can deliver UN and EU biodiversity targets for 2020. This will promote an ecosystem approach to biodiversity conservation that demands an holistic approach to managing the environment and the range of pressures on biodiversity. A full consultation will take place over summer 2012.

A review of the Northern Ireland Biodiversity Strategy is being taken forward to reflect both international and EU obligations and targets for halting biodiversity loss by 2020. In tandem with Defra, other devolved administrations and the ROI, Northern Ireland is looking to integrate government action to collectively safeguard and restore biodiversity. These actions will be recorded in a revised Biodiversity Strategy which will be the subject to full consultation later in 2012.

### 3.4 Applicability of standard human noise assessment to wildlife

Although standardised protocols for environmental noise assessment are crucial in evaluating impacts and enforcing environmental protection policy for humans, their applicability to wildlife is



limited; it is not possible simply to infer the impacts of anthropogenic noise on wildlife from the human literature. This is because the hearing ranges and sensitivities of non-human animals can be very different from those of humans (Heffner & Heffner 2007). For example, bats use ultrasonic frequencies (Popper & Fay 1995) and most bird species have relatively insensitive hearing compared to humans (Dooling et al. 2000), although some can hear sound at levels too quiet for humans to detect (e.g. barn owl *Tyto alba*, Knudsen 1981; Dyson et al. 1998). Studies on humans have understandably used methodology that tailors the quantification of anthropogenic noise to our hearing capabilities: for example, the use of microphones limited to the human hearing range (20 Hz – 20 kHz) and the implementation of frequency filters effectively mimicking human auditory sensitivity (A-weighting). As such, noise measurements may cover only part of the relevant acoustic range for other species. Moreover, species differences in behaviour, physiology and ecology, in addition to hearing capabilities and perception, mean that extrapolations from human studies can provide only a limited understanding of the potential impact of anthropogenic noise on wildlife. Studies aimed at collecting data on how anthropogenic noise impacts wildlife are therefore important and there has been an increase in such work in recent years (see Popper & Hastings 2009; Barber et al. 2010; Laiolo 2010; Slabbekoorn et al. 2010 for reviews).

### 3.5 UK Biodiversity Action Plan Priority Species

The UK Biodiversity Action Plan (BAP) was developed as the Government's response to the Convention on Biological Diversity (CBD), which the UK signed up to in 1992. The aim of the BAP was to identify priority species, conserve and protect existing biological diversity, and to enhance it wherever possible. A list of UK priority species (UK PS) and habitats was developed, which was revised in 2007 (<http://jncc.defra.gov.uk/page-5705>), and the outcomes of suggested actions to protect UK PS species are reviewed every 3-5 years. Since devolution in Northern Ireland, Scotland and Wales, the UK BAP has been implemented at a devolved administration level, allowing regional differences in conservation priority to be addressed. To identify the UK's regional differences in conservation priority, England<sup>5</sup>, Northern Ireland<sup>6</sup>, Scotland<sup>7</sup> and Wales<sup>8</sup> independently list Species of Principal Importance (SPI), highlighting to public bodies the highest conservation precedence in each country. Although environmental noise is recognized as a threat to biodiversity in a global context (see Barber et al. 2010), its effects in the UK remain to be evaluated.

## 4 Methodology

### 4.1 Literature searches

To identify papers of relevance in the peer-reviewed scientific literature, combinations of the following search terms were initially inputted into Web of Knowledge and Google Scholar: 'noise', 'acoust\*', 'audio\*', 'hearing', 'bird\*', 'mammal\*', 'amphibian\*', 'reptile\*', 'fish\*', 'invertebrate\*' (an '\*' acts as a wild card allowing broader searching). Papers were included in the review if they directly addressed anthropogenic noise impacts on any species, contained audiograms for UK PS and SPI, or assessed the impact of non-anthropogenic noise sources on UK species or any terrestrial invertebrates (as there were a paucity of other relevant studies on this taxonomic group). Only publications referring to noise in the title, abstract or keywords were included in the study (e.g. studies focusing on the impact of roads, such as Huijser & Bergers (2000) and Berthinussen & Altringham (2011), were not included). References within these initially identified publications, and papers citing them, were also considered to provide a comprehensive inclusion of the anthropogenic

<sup>5</sup> <http://www.naturalengland.org.uk/ourwork/conservation/biodiversity/protectandmanage/habsandspeciesimportance.aspx>

<sup>6</sup> [http://www.doeni.gov.uk/niea/biodiversity/sap\\_uk.htm](http://www.doeni.gov.uk/niea/biodiversity/sap_uk.htm)

<sup>7</sup> <http://www.snh.gov.uk/protecting-scotlands-nature/biodiversity-scotland/>

<sup>8</sup> <http://www.biodiversitywales.org.uk/species-35.aspx>

noise literature globally, and on UK PS and SPI in particular. Web searches and direct access to publication lists from 25 UK organisations, including the Royal Society for the Protection of Birds, the British Trust for Ornithology, the Mammal Society and the Environment Agency, provided information of relevance to UK PS and SPI from the grey literature. Studies in the grey literature not directly assessing anthropogenic noise impacts on UK PS and SPI were not sourced as this would have entailed an unfeasible (in terms of time) global search of relevant organisations.

Noise spectra were obtained from literature identified in the searches described above. Where those studies did not provide sufficient spectral information for particular noise sources of interest, additional literature was collated as above by using combinations of the following search terms: 'noise', 'anthropogenic', 'acoust\*', 'spectr\*', 'frequency', 'sound'. If multiple studies provided spectra for the same type of noise source, up to eight were included, with preference given to studies using flat-response microphones, a broad frequency bandwidth and repeatable methodology. The typical frequency range containing the highest amplitudes (dominant frequency) was determined for each noise source by comparing the spectra sourced.

## 4.2 Studies considering impact of anthropogenic noise on UK priority species

Peer-reviewed papers and unpublished reports directly addressing the potential impacts of anthropogenic noise on UK PS and SPI were assessed for their strength of evidence and assigned one of three categories. Category 1: publications with strong evidence for impacts, or confirmed lack of impacts, of anthropogenic noise on the study subjects. These studies were well designed with suitable controls and large sample sizes. Category 2: studies in which impacts of anthropogenic noise are indicated rather than confirmed, usually due to correlative results. These studies are generally well designed, but do not provide a definitive test that noise is the cause of changes observed. Category 3: work providing little or no evidence to support anthropogenic noise either having or not having an impact. These studies included many confounding factors, small sample sizes and/or weak analytical methodology.

Ultimately, it is impacts on individual fitness that are of most relevance and importance for population viability; fitness describes the ability of an organism to survive and reproduce, with the fittest individuals surviving to produce many offspring. Hence, studies were also assigned to one of three categories depending on what conclusions could potentially be drawn about individual fitness. 'Direct' evidence was provided by studies showing that individual fitness has been impacted by anthropogenic noise. This may be through changes in, for example, clutch size, nest success or offspring survival. Studies classified as using a 'proxy' show changes that have the potential to impact fitness. For example, if the presence of anthropogenic noise decreases foraging efficiency this may lower the survival and reproductive success of the individual. However, animals might compensate for this effect on foraging, by foraging at a quieter time of day or moving to a habitat away from the noise source, and thus individual fitness might not ultimately be affected. Other examples of proxies for fitness include alterations in acoustic communication and physiological changes. Studies categorised under 'none' show no evidence that anthropogenic noise impacts the fitness of the subject animals. Outcomes classified as such would be differences in distribution and theoretical studies showing no empirical support for their conclusions.

## 4.3 Extrapolations

### 4.3.1 *Audiograms and noise spectra*

An audiogram is a measure of the hearing range and sensitivity of an organism. On a diagram, the area above a threshold curve depicts what can be heard by the animal and the curve itself represents

the minimum audible level at any tested frequency (see Figure 1 for examples). The ‘best’ frequency for an animal is that at which the lowest sound level produces a response. Acoustic thresholds can be determined from either behavioural/physiological responses (e.g. through changes in heart rate) or from neural responses (e.g. by measuring electrical responses in peripheral auditory nerves or the central nervous system, such as in brain regions where auditory processing is conducted). Each type of audiogram is likely to produce slightly different threshold levels, but methods are generally comparable (Kenyon et al. 1998; Szymanski et al. 1999; Yuen et al. 2005). All measurement methods were included in this report to allow a more comprehensive representation of UK PS and SPI, but behavioural thresholds may be more ecologically relevant to the present review. It must also be noted that standard protocols for obtaining audiograms involve the use of anechoic (echoless) recording chambers, which are likely to produce thresholds with different sensitivity to those produced under field conditions (see Schmidt & Römer 2011).

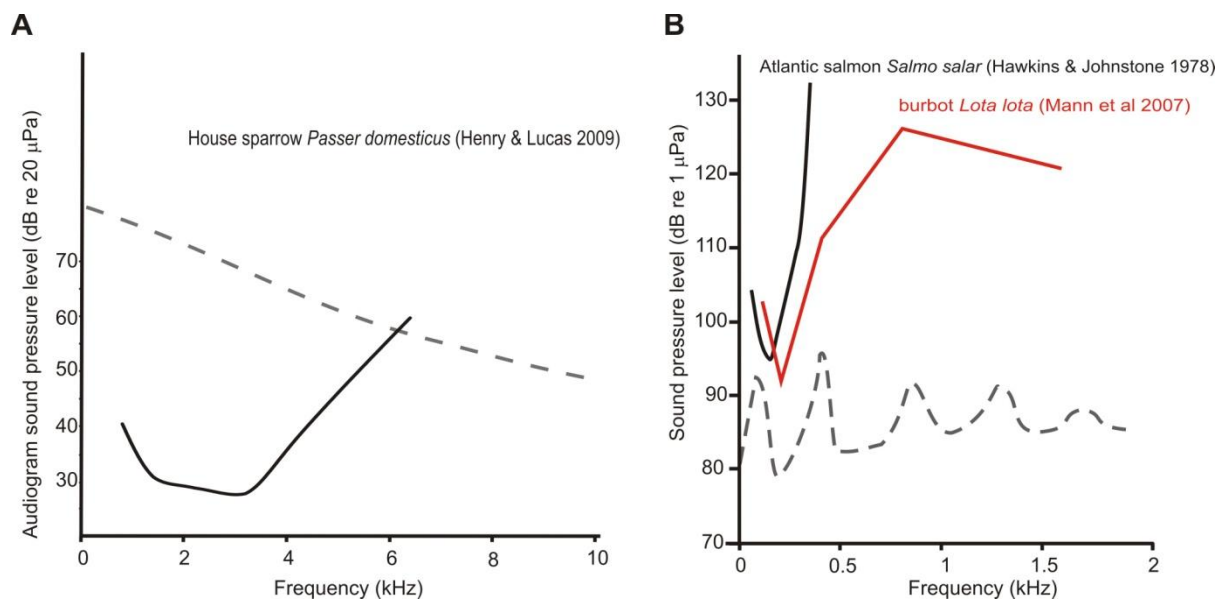


Figure 1. Examples of audiograms and anthropogenic noise spectra. A) Audiogram for the house sparrow (*Passer domesticus*) (solid line; from Henry & Lucas 2009) plotted against road traffic noise spectrum (dashed line; from Schaub et al. 2008). B) Audiograms for the Atlantic salmon (*Salmo salar*) (solid black line; from Hawkins & Johnstone 1978) and burbot (*Lota lota*) (solid red line; from Mann et al. 2007) plotted against boat noise spectrum (dashed line; from Amoser et al. 2004). Values on the y-axis in both (A) and (B) represent audiogram sound pressure levels as given in the relevant papers and do not relate to the noise spectra, which are plotted to indicate frequency distribution and not to give absolute amplitude values.

To assess the vulnerability of UK PS and SPI to anthropogenic noise, frequency sensitivities from all available audiograms of relevant species (see Appendix C) were compared with the dominant frequency of noise spectra for each ecologically relevant noise source (see Figure 1). Ecologically relevant sources were noises found in the environment inhabited by a particular species (i.e. terrestrial noise sources for birds, reptiles, invertebrates and mammals; aquatic noise sources for fish; and both terrestrial and aquatic sources for amphibians). Overlaps in auditory frequency response and dominant frequency of the noise source indicate that the animal can potentially hear the noise and that it may therefore have an impact.

#### 4.3.2 Non-anthropogenic noise sources and UK species

The impact and response of UK PS and SPI to any environmental noise, whether it is anthropogenic or ‘natural’, was considered (see Kight & Swaddle 2011). Studies examining the impact of non-anthropogenic noise on UK PS and SPI were sourced from the peer-reviewed scientific literature (see Appendix D). Additionally, studies on non-anthropogenic noise in species outside the UK were

sourced for terrestrial invertebrates due to the paucity of studies addressing anthropogenic noise impacts on UK invertebrate species.

Non-anthropogenic noise sources that were considered include white noise (broadband noise of equal loudness across frequency) and sine tones, as well as chorus noise from other animals and other naturally occurring background noises. Extrapolation was based on whether the UK PS or SPI responded to the sound (i.e. noise generates a response in the animal) and the frequency content, where known, of the noise source in comparison to frequency content of the main anthropogenic noise sources (section 3.2).

#### 4.3.3 *Phylogenetic and ecological extrapolation*

Peer-reviewed anthropogenic noise literature on UK non-priority species and other species globally were assessed for their strength of evidence and implications for fitness (see section 4.2). These studies on non-UK priority species were used to infer potential impacts of anthropogenic noise on phylogenetically related or ecologically similar UK PS and SPI. Phylogenetic similarity was restricted to the level of taxonomic order, with the exception of some teleost fishes (alis shad *Alosa alosa*, twite shad *Alosa fallax*, vendace *Coregonus albula*, pollan *Coregonus autumnalis* and arctic charr *Salvelinus alpinus*) and terrestrial invertebrates (Diptera) which were extrapolated at the level of taxonomic class due to a paucity of direct studies, audiograms or studies using alternative noise sources on these species or groups. Species considered ecologically similar shared foraging strategy and habitat type, as well as occurring within the same taxonomic class.

## 5 Impacts of anthropogenic noise on UK PS and SPI

### 5.1 Overview

Over the past 30 years, and particularly the last decade, there has been a considerable increase in the number of studies in the peer-reviewed literature addressing the impact of anthropogenic noise on non-marine wildlife across the globe, with 31% (excluding publications from 2012 to date) of studies sourced published in 2011 alone (Figure 2). However, that marked increase has not been mirrored by studies specifically examining UK PS and SPI (Figure 2). Research directly addressing the potential impact of anthropogenic noise was found for 24 UK PS and SPI (from 16 publications, including three in the grey literature), but definite conclusions can only be made about the reed bunting (*Emberiza schoeniclus*) (see section 5.7.1); it is likely that anthropogenic noise also impacts singing in European robins (*Erithacus rubecula*), a Scottish SPI (Appendix E). Inclusion of extrapolatory evidence from the combination of audiograms and noise spectra, from non-anthropogenic noise data and from studies on phylogenetically and ecologically similar species allows suggestions about the potential impact of noise for an additional 55 species, bringing the total to 79 (Appendix F). However, clear-cut conclusions remain rare: in addition to reed buntings and European robins, some evidence for an impact of noise is indicated for the brown long-eared bat (*Plecotus auritus*), house sparrow, starling (*Sturnus vulgaris*), bullfinch (*Pyrrhula pyrrhula*) and common toad (*Bufo bufo*) (Appendix F).

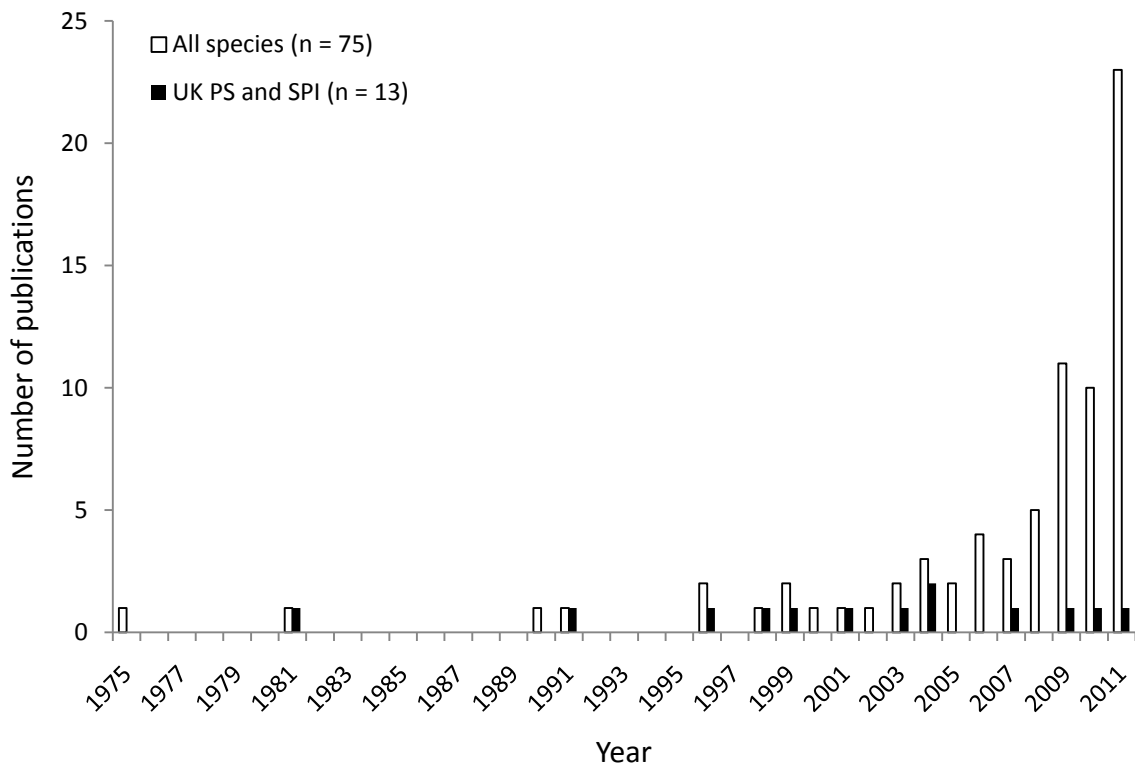


Figure 2. Number of publications in the peer-reviewed literature related to anthropogenic noise effects on non-marine wildlife.

## 5.2 Representation of noise sources

The main source of anthropogenic noise studied globally (Appendix B) is road traffic (60% of studies), followed by aircraft noise (15%) and industrial noise (13%) (Figure 3A). This pattern, with road traffic as the predominant anthropogenic noise source, was mirrored in the UK (Figure 3B) and for UK PS and SPI (Figure 3C).

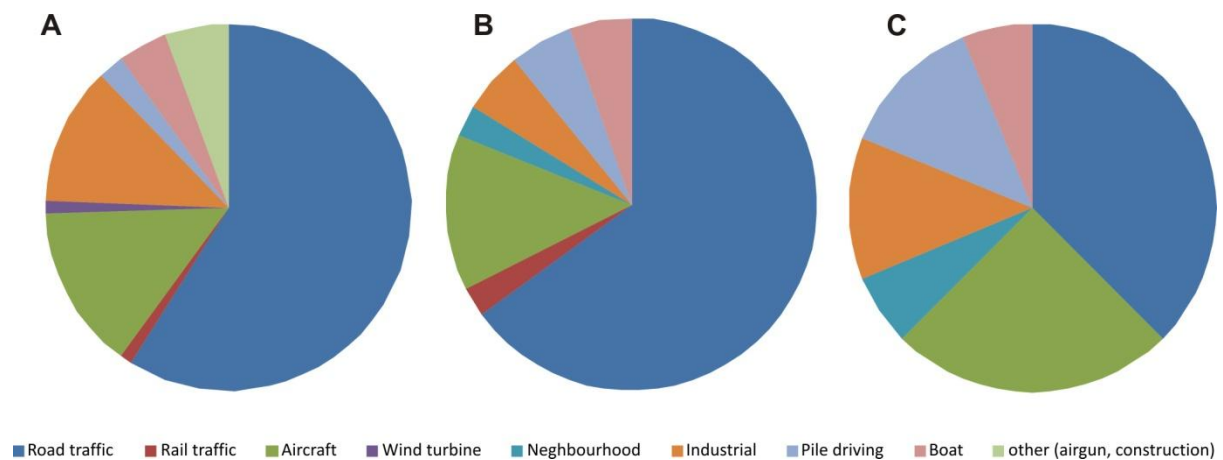


Figure 3. Representation of anthropogenic noise sources studied. A) Studies on species throughout the world (n = 90). B) Studies on UK species (n = 37). C) Studies on UK PS and SPI only (n = 16). Some studies used more than one noise source and were therefore represented multiple times across categories in this figure.

### 5.3 Taxonomic representations

Studies directly assessing the potential impact of anthropogenic noise show a strong taxonomic bias, with similar patterns apparent globally (Figure 4A) and in the literature on all UK species (Figure 4B) and only UK PS and SPI (Figure 4C). The majority of the work has been on birds, with a few studies on amphibians, freshwater fish and mammals, especially bats (See Appendices B, E & F); there is a severe paucity of research on reptiles and terrestrial invertebrates.

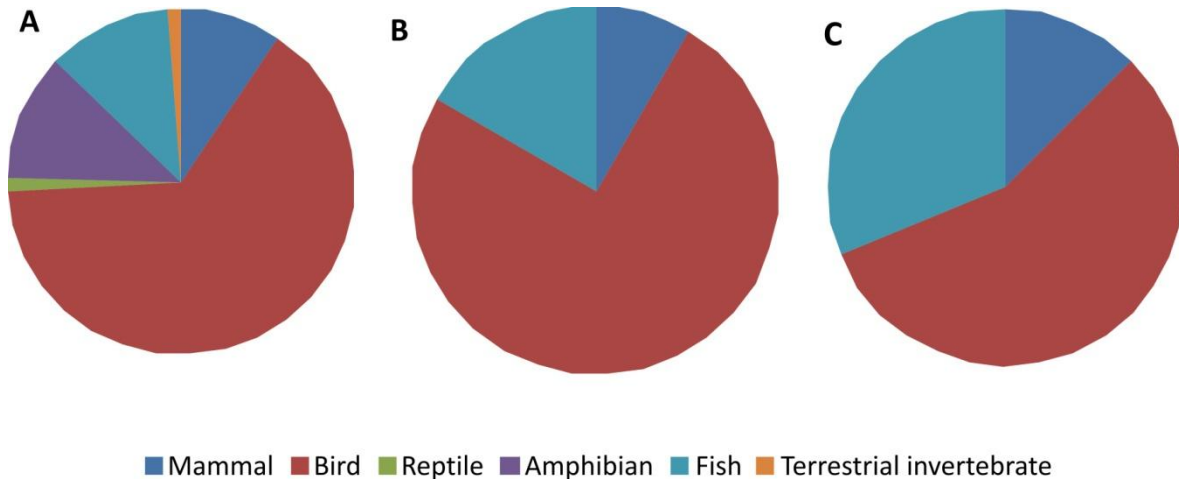


Figure 4. Representation of taxonomic groups in the literature directly assessing impacts of anthropogenic noise. A) Studies on species throughout the world (n = 85). B) Studies on UK species (n = 37). C) Studies on UK PS and SPI only (n = 16). Two studies focused on more than one taxonomic group and were thus included more than once in these figures.

### 5.4 Representation of impacts of noise measured

Studies on the impacts of anthropogenic noise both globally and in the UK are dominated by behavioural measures (Figure 5). The main behavioural impacts observed were changes in acoustic communication parameters, foraging behaviour and vigilance, and movement patterns (Appendix B). Twenty-five percent of studies on UK PS and SPI are theoretical (Figure 5C), often based on extrapolation and inferring impacts from overlaps between hearing ranges and the frequency and energy content of anthropogenic noise sources. Community-level studies also feature relatively strongly, especially when considering UK PS and SPI (Figure 5). Physiological impacts (e.g. metabolic rate, endocrine responses, auditory physiology) are not commonly measured and constitute only 6% of impacts in anthropogenic noise studies globally (Figure 5A).

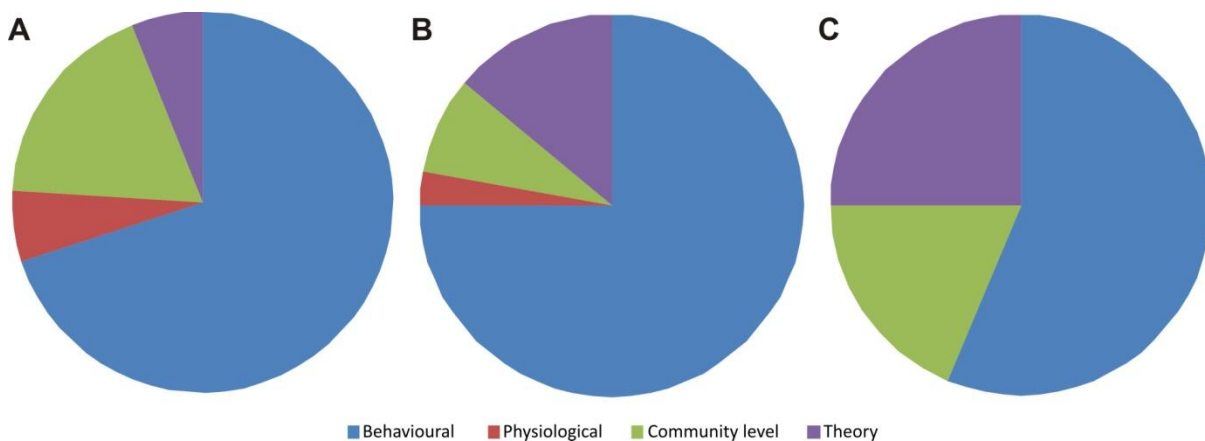


Figure 5 Representation of impacts measured in the anthropogenic noise literature directly (Appendix B). A) Studies on species throughout the world (n = 83). B) Studies on UK species (n = 37). C) Studies on UK PS and SPI only (n = 16). Some studies focused on more than one taxonomic group and were thus included more than once in these figures.

## 5.5 Strength of evidence for impacts of noise

There are few anthropogenic noise studies, both globally and in the UK, that allow strong conclusions about the impact of anthropogenic noise on non-marine animals (Figure 6). Studies from around the world (Figure 6A) and the UK as a whole (Figure 6B) share similar proportions of publications in each evidence category. Studies on UK PS and SPI, however, provide an overwhelming lack of strong evidence for or against noise impacts (Figure 6C).

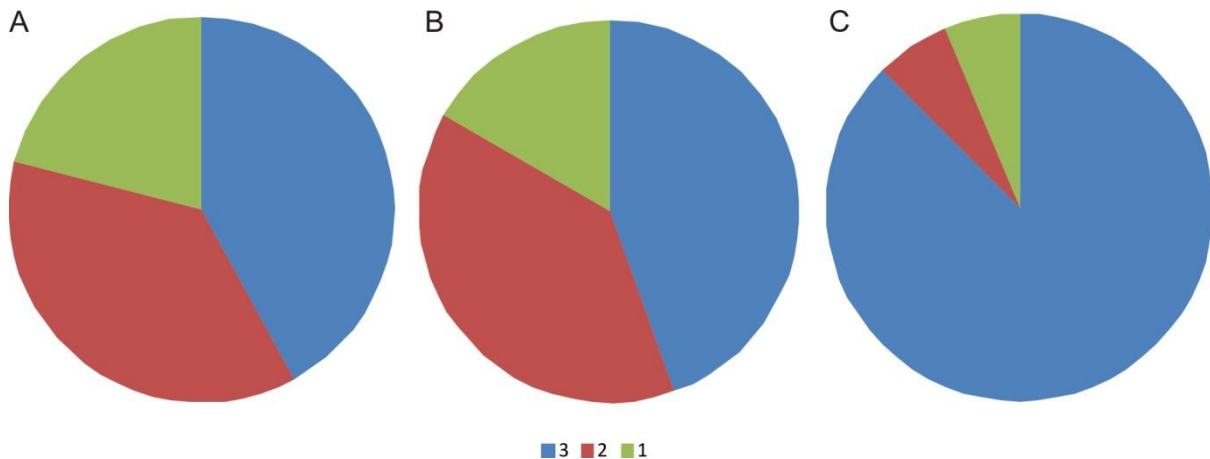


Figure 6. Strength of evidence provided by the literature directly assessing the impact of anthropogenic noise. Colours represent categories described in section 4.2 (i.e. 1 = strong evidence). A) Studies on species throughout the world (n = 81). B) Studies on UK species (n = 36). C) Studies on UK PS and SPI only (n = 16). Studies in press that could not be accessed in full were not included (n = 2).

Some key issues that prevent strong conclusions recur often throughout the literature on anthropogenic noise impacts. The most common of these is a failure to provide suitable controls for confounding factors. Confounding factors are variables that could contribute towards variation in the study subjects. For example, roads are noisy, but they also have high levels of disturbance, pollution and light, and provide an edge habitat. Studies often compare the responses of animals near a noisy road with those in a control area, either a quieter road or a site at a greater distance from the road, but such a situation does not allow any differences to be conclusively attributed to noise. For noise to be implicated as causal, all other variables must be controlled for; that is to have a control road with all other characteristics (light, pollution, traffic load, etc.) at a similar level, but differences in noise levels being the only factor that differs between sites. This may often be impossible to achieve in practice, but it hampers strong conclusions.

Careful controls can be designed in laboratory experiments (e.g. Bee & Swanson 2007; Schaub et al. 2008; Siemers & Schaub 2011), although the ecological validity of such work can sometimes be questioned. Recent studies have highlighted that it is also possible to provide strong evidence for the impact of noise utilising natural experiments: Francis et al. (2009; 2011a, b, c) and Bayne et al. (2008) have shown that anthropogenic noise impacts birds at both the species and community level. As an example of a study with strong evidence, Francis et al. (2011a) measure differences in the acoustic behaviour of two different species of tyrant flycatcher (Tyrannidae) at gas wells either with or without noisy compressors. As the wells are comparable in both structure and surrounding habitat, and thus differ only in noise production, this system provides an excellent experiment under field conditions. Their results indicate different occupancy rates between species, with the grey flycatcher (*Empidonax wrightii*), a species with limited vocal plasticity, found less frequently at noisy compressor sites; the ash-throated flycatcher (*Myiarchus cinerascens*), in contrast, shows no occupancy differences between noise treatment and control, and an increase in song frequency under the noise treatment. Although such natural experimental situations may be rare, it provides a working paradigm and examples of good experimental practice for future research programmes.

Aside from the importance of suitable controls, it is also vital to quantify the noise source in the most appropriate way for the study species. As noted in section 3.3, there are well-established protocols for assessing environmental noise levels where humans are the study subject. This has led to abundant availability of sound-level meters, microphones and loudspeakers with weighting filters designed to mimic human hearing sensitivities, and consequently many studies investigating the impact of anthropogenic noise on non-human animals present sound levels and frequency spectra that are not representative of the auditory capabilities of their study animal. As the hearing range of most species remains unknown, and weighting filters for individual species are not common (but see Nedwell et al. 2003 for an example in Atlantic salmon), studies should record the noise source of interest with a broad bandwidth and a flat-response microphone to capture all frequencies that may be detected by the study species, and should also present the resulting noise spectra.

A study by Schaub et al. (2008) on bat foraging sets a good standard for quantification of road traffic noise in a way relevant to the study species; bat auditory sensitivity is known to differ substantially from human hearing (Popper & Fay 1995). Schaub et al. (2008) measured traffic noise between 0 and 50 kHz with a flat-response microphone. Moreover, they quantified the number of vehicles, vehicle type and distance from the noise source to describe fully the sound used in their experiments. As noise sources can be variable (e.g. number and composition of vehicles, distance measured), quantification of the noise source not only strengthens the study by enabling the frequency content of the signal to be assessed, but allows future research to consider the vulnerability of different animals to different sources of anthropogenic noise.

As research into the impacts of anthropogenic noise continues to grow, studies are generally improving with common experimental and analytical flaws being ironed out, and this is crucial if strong conclusions are to be drawn.

## 5.6 Individual fitness

The vast majority of studies investigating anthropogenic noise have, to date, considered relatively short-term effects. Ultimately what is needed, however, are the likely impacts on survival and reproductive success; that is, assessment of fitness at the individual level. Some short-term effects (e.g. increased predation risk) can be translated relatively easily into these ultimate consequences presenting direct fitness implications; others (e.g. foraging behaviour) need more careful consideration because animals may be able to compensate in quieter periods, and thus there may be no direct link between short-term effects and long-term consequences (see Bejder et al. 2006a, b). However, some of these behaviours are probably reliable proxies of fitness. Most of the anthropogenic noise literature globally and in the UK reports on fitness proxies or behaviours with no clear implications for fitness (Figure 7A, B), and there are no studies on UK PS or SPI that address direct fitness impacts (Figure 7C).

One recent study has attempted to assess impacts of noise on individual fitness in great tits (*Parus major*). Halfwerk et al. (2011) found that male songs of lower frequency, preferred by females at the peak of their fertility in quiet conditions, are poor at eliciting female emergence from the nest at dawn when there is increased urban noise; emergence behaviour correlates with numbers of extra-pair copulations (i.e. mating success of the singing male). Songs of higher frequency, known to be produced under such noisy conditions, more successfully elicited female emergence. The implication is that males with higher pitched songs have greater fitness than males with lower pitch songs in noisy conditions. This case study makes an excellent start at addressing fitness implications and future research should extend their example to address more long-term fitness consequences that could inform population level models.



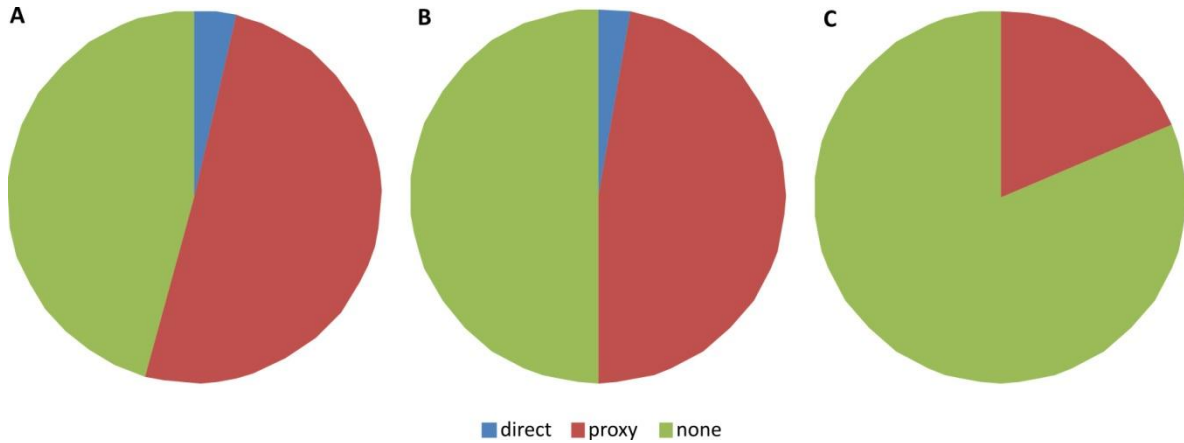


Figure 7. Fitness implications in the literature directly assessing the impact of anthropogenic noise. Colours represent categories described in section 4.2. A) Studies on species throughout the world (n = 81). B) Studies on UK species (n = 36). C) Studies on UK PS and SPI only (n = 16). Studies in press that could not be accessed in full were not included (n = 2).

The majority of publications combine a lack of fitness impacts with little conclusive evidence supporting impacts of noise (Figure 8). The few studies that do directly address fitness consequences or fitness proxies also tend to provide stronger evidence for impacts of anthropogenic noise (Figure 8). This trend for fitness implications being provided by well designed experiments is seen at the global level (Figure 8A), but is not so clearly discernable in UK PS and SPI due to the paucity of studies providing strong evidence or fitness consequences (Figure 8B).

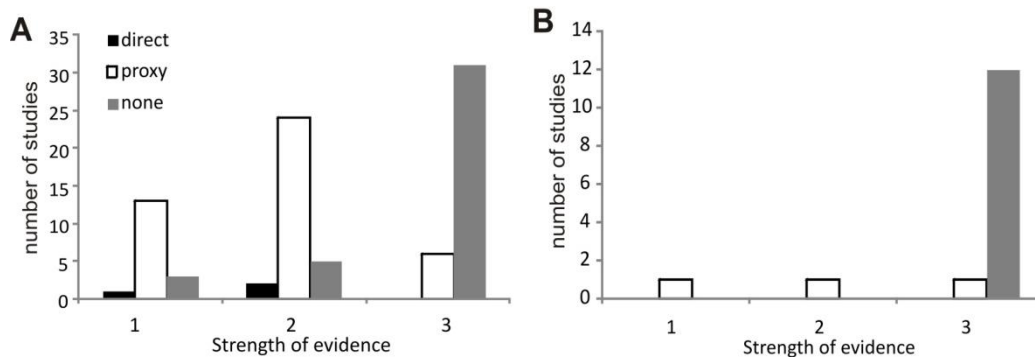


Figure 8. Strength of evidence and fitness implications in the literature directly assessing the impact of anthropogenic noise. A) Studies on species throughout the world (n = 81) divided according to strength of evidence (i.e. 1 = study with strong evidence; see section 4.2) and the three fitness categories. B) As in (A) but for UK PS and SPI only (n = 16). Studies in press that could not be accessed in full were not included (n = 2).

## 5.7 Summaries for taxonomic groups

### 5.7.1 Birds

This taxonomic group has received the most research attention with respect to the impacts of anthropogenic noise. Sixty-five percent of all studies obtained in the literature search were on birds (Figure 4), and this bias was reflected in the UK, with nine of the 16 studies (56%) directly examining anthropogenic impacts on UK PS and SPI considering birds (Figure 4).

Most direct anthropogenic noise studies on birds have addressed the impact of road traffic noise, with song frequency shifts under noisy conditions a common finding (see Patricelli & Blickley 2006). These frequency shifts are seen in the reed bunting, a UK PS, both under natural conditions and through experiment manipulation (Gross et al. 2010). Song frequency shifts are a well established phenomenon and have been the subject of study in other UK bird species, such as great tits, blackbirds (*Turdus merula*) and chiffchaffs (*Phylloscopus collybita*) (see Brumm 2006 for a review). A change in song frequency, as a sexually selected trait, can only serve as a proxy for changes in fitness, but a recent study has indicated that reproductive success in great tits is affected by shifts in song frequency (Halfwerk et al. 2011). Whether this affects long-term population viability, however, is still unknown. There is evidence that the European robin, a Scottish SPI, is more likely to sing at night in noisy urban areas (Fuller et al. 2007), again providing a proxy for fitness implications, but as the study is correlative, anthropogenic noise cannot be identified conclusively as the cause.

Other studies on UK PS and SPI have focused on abundance and breeding bird density at noisy roadsides compared to control sites (Reijnen 1996; Rheindt 2003; Peris & Pescador 2004). While these studies implicate noise as a factor in their reported results, it is difficult to draw firm conclusions because there is no control for confounding factors such as overall disturbance levels, lighting and air pollution. Moreover, fitness implications are difficult to extract from such data.

The impact of aircraft noise has also been studied in several birds on the UK PS and SPI lists (Burger 1981; Ellis & Ellis 1991; Trimper et al. 1998; Ward et al. 1999). As with assessing the impact of road traffic noise, it is difficult to eliminate confounding factors without experimental manipulation of the noise source. These studies, being field-based and generally observational, therefore do not provide great weight when drawing conclusions about the impact of aircraft noise on UK wildlife.

The hearing range of birds is largely restricted to a bandwidth of up to 10 kHz with sensitivities at best frequencies comparable to those of mammals, but outside this range sensitivity is considerably lower (Dooling et al. 2000). The audiograms of UK PS and SPI (house sparrow, bullfinch, starling, barn owl) show best frequencies ranging from 2-6 kHz (Schwartzkopf 1949; Dooling et al. 1986; Dyson et al. 1998; Henry & Lucas 2009), overlapping at least in part with the dominant frequencies of road traffic noise and aircraft noise.

### 5.7.2 Terrestrial mammals

There is some direct literature concerning anthropogenic noise impacts on UK PS and SPI mammals (13% of studies; Figure 4), but, as is also reflected globally, this taxonomic group is under-represented (9% of studies; Figure 4). Direct studies on the badger (*Meles meles*), a water vole (*Arvicola* sp.) (Iglesias et al. 2011) and Daubenton's bat (*Myotis daubentonii*) (Shirley 2001), UK PS and SPI, do not empirically quantify sound levels and, like many other studies, are unable to remove confounding factors in the explanation of their results. These studies therefore cannot provide strong evidence for any direct impacts or lack of impact of anthropogenic noise on UK PS and SPI.

Assessments of the impact of road traffic noise on a species of gleaning bat (the greater mouse-eared bat *Myotis myotis*) represent some of the best work on the influence of anthropogenic noise in mammals (Schaub et al. 2008; Siemers and Schaub 2011). Rather than using echolocation for the detection and localisation of prey (echolocation is still used for orientation), this species listens for prey-generated sounds and gleans food items from the ground or other substrate. These bats avoid foraging when exposed to playback of road traffic noise, but when noise is unavoidable they show reduced foraging efficiency. Greater mouse-eared bats use the same foraging strategy as the brown long-eared bat (Swift & Racey 2002; Siemers & Swift 2006), which is on the UK PS list. It can be inferred therefore that foraging efficiency in this species is likely to be influenced by the presence of road traffic noise.

In contrast to gleaning bats, echolocating bats appear to be at relatively low risk of direct impacts of anthropogenic noise (Tressler & Smotherman 2009). Audiograms indicate that the best frequencies of these bats are high above the dominant frequencies of the main sources of anthropogenic noise (road traffic, aircraft). Likewise, the polecat (*Mustela putorius*) audiogram shows a best frequency (8-11 kHz) above the dominant frequency content of road traffic and aircraft noise, and therefore expected impacts to this species are relatively low. A study on the California ground squirrel (*Spermophilus beecheyi*) indicated that vigilance behaviour may be modified by the presence of noisy wind turbines (Rabin et al. 2006). The California ground squirrel is in the same family as the red squirrel (*Sciurus vulgaris*), therefore vigilance behaviour in this UK PS may be influenced by the presence of wind turbines here. However, the prevalence of on-shore wind turbines in the UK is currently low, although numbers are increasing, and as the American study contained many habitat differences between treatment and control sites (Rabin et al. 2006), it would be difficult to imply that noise in isolation would cause these changes in the red squirrel; extrapolation therefore provides only weak evidence for any impact of noise in this species.

### 5.7.3 Reptiles

No publications directly assess the impact of anthropogenic noise on UK PS or SPI reptiles (Figure 4). Only one publication directly assesses the impact of anthropogenic noise on reptiles, but due to shortcomings found commonly in the anthropogenic noise literature (see section 5.5) and a lack of identification down to species or genus level, few implications can be drawn from this work.

There is also no strong extrapolatory evidence for reptiles. The best frequencies in lizard hearing are between 1 and 3 kHz (Saunders et al., 2000) and their sensitivity does not extend far above 8 kHz (Manley 2011). Information on the auditory sensitivity of the sand lizard (*Lacerta agillis*), a UK PS, suggests that this species is no exception, with no behavioural responses observed above 8 kHz (Berger 1924). The low frequency sensitivity of reptiles may leave them vulnerable to impacts from anthropogenic noise such as road traffic which exhibits dominant low frequencies.

### 5.7.4 Amphibians

There are no publications that directly assess the impact of anthropogenic noise on UK PS or SPI amphibians (Figure 4; Appendix E). All implications for UK PS and SPI therefore must come from extrapolation (Appendix F). Globally, however, amphibians are represented in 12% of anthropogenic noise studies (Figure 4).

Anurans, the taxonomic order containing both frogs and toads, constitute 12% of publications on anthropogenic noise in non-marine wildlife worldwide (Figure 4). Responses to anthropogenic noise are not consistent across species: some, such as *Rana taipehensis* (Sun & Narins 2005), appear to have plastic calling behaviours that allow them to modify their responses and thus minimise the likelihood of masking, while others, such as the European tree frog (*Hyla arborea*) (Lengagne 2008), do not modify the structure of their calls to compensate for noisy conditions. It is therefore difficult to predict the impact of noise on the UK PS and SPI.

Audiograms have been produced for the common toad (Walkowiak et al. 1981), and show best frequencies below 2 kHz and thus within the dominant frequency range of all noise sources reported here. It is therefore likely that noise can be heard by this species; indeed, in response to white noise, the common toad increases locomotion and escape behaviours (Llusia et al. 2010).

### 5.7.5 Fish

In comparison to the wider literature, direct studies on freshwater UK PS and SPI are relatively well represented (12% of global studies; 31% of UK PS and SPI studies; Figure 4). However, the evidence for anthropogenic noise impacts is relatively weak with extrapolatory methods and low sample sizes preventing definite conclusions.

Audiograms exist for three UK PS fish (European eel (*Anguilla anguilla*), burbot, Atlantic salmon; Appendix C), and these suggest that boat noise is not only within the hearing range of all three, but that the dominant frequency for boat noise is near the best frequencies of these species. Evidence from non-anthropogenic noise studies suggests that noise can serve to deter some species from its proximity (Appendix F).

For fish it is important to consider not just the pressure component of a sound, the component familiar to humans, but also particle motion. Pressure and particle velocity are both integral components of any sound wave with pressure waves dominating the sound field further from the source (far field) and particle velocity dominating close to the source location (near field). Particle motion can be detected as velocity, displacement or acceleration. Fish are known to detect particle acceleration using sensory cells within the lateral line system on their skin surface. Only a subset of fish have evolved specialised pressure sensitive ears, but around two-thirds of freshwater species are thought to be hearing specialists with the potentially quieter freshwater environments perhaps favouring the evolution of pressure reception (Nelson 1994; Amoser & Ladich 2005). It is therefore important to measure both components of the sound wave and determine the impacts of anthropogenic noise on each detection system.

### 5.7.6 Terrestrial invertebrates

There is an extreme lack of knowledge when it comes to the impact of anthropogenic noise on terrestrial invertebrates. There are no direct studies on species within the UK (Figure 4; Appendix E) and only one paper which directly investigates impacts of anthropogenic noise within this group. Shieh et al. (2011) assessed the impact of road traffic noise on cicada choruses in Japan, finding that peak frequency of calls was increased in urban environments. Other work addressing impacts of noise on invertebrates focuses on environmental background sources, such as the communication signals of other species, usually in the biotically noisy rainforests of the tropics (Schmidt et al. 2011; Schmidt & Römer 2011).

The hearing sensitivity and capability of the vast majority of invertebrates remain unknown (Hoy et al. 1998). This group can, however, be crudely divided into those that can detect sound pressure (Hoy & Robert 1996) and those which are sensitive to particle velocity (Tautz 1979). Crickets and grasshoppers (order Orthoptera) are the most conspicuous insects in the UK that use acoustic cues for communication. This group are known to receive the pressure component of a sound wave and use calls to attract mates, with their hearing system characterised relatively well compared to other taxonomic orders within this class (Bailey 1993). Indeed, audiograms have been produced for two orthopteran UK PS, the field cricket (*Gryllus campestris*; Nocke 1972) and the wart-biter cricket (*Decticus verrucivorus*; Kalmring 1978), indicating best frequencies of 4-20 kHz. As their best frequency is relatively high compared to the dominant frequency of road traffic and aircraft noise, they may not be at high risk from anthropogenic noise, but work suggests that their ears are tuned to a broad range of frequencies (Schmidt et al. 2011) and signal detection under noisy conditions may be made more difficult.

Particle velocity receivers have been best characterised in two-winged flies (Diptera), where it is known that their hair-like, flagellar ears are limited to the low frequencies (<1 kHz) that dominate anthropogenic noise (Tautz 1979; Göpfert & Robert 2001, 2002, 2007). As the particle velocity

component of a sound wave only dominates in close proximity to a sound source it may be inferred that this type of hearing system is less likely to be impacted by background and anthropogenic noise than a pressure receiver. However, mosquitoes (*Toxorhynchites brevipalpis*; Göpfert & Robert 2001) and fruit flies (*Drosophila melanogaster*; Göpfert & Robert 2002), and possibly more species (Riabinina et al. 2011), are known to amplify quiet stimuli which effectively increases their sensitivity to distant sounds of low frequency (<1 kHz). The impact of anthropogenic noise on this taxonomic group therefore remains unknown and difficult to infer.

## 6 Moving forward: considerations for future work

A growing number of studies are being published on anthropogenic noise effects on wildlife globally. Methodologies and study design are generally improving and broader questions are being asked. However, a strong evidence base is still far from apparent, especially with respect to particular groups of organisms, such as UK PS and SPI. Below are outlined five key areas for consideration moving forwards.

### 6.1 Taxonomic biases

The strong taxonomic bias towards anthropogenic noise research on birds (see Figure 4) needs to be redressed if conservation priorities are to be successfully managed across taxa. Reptiles and invertebrates are particularly underrepresented in the literature, with little known about the impact of noise on these groups even globally. Terrestrial invertebrates complete a range of ecosystem services and are important prey for numerous species. Moreover, due to their relatively short life-spans and small size, invertebrates provide potentially ideal models for longer-term studies examining directly the fitness impacts of anthropogenic noise at both the individual and the population level, rather than needing to infer such consequences from behavioural proxies.

### 6.2 Study design

Much of the anthropogenic noise research conducted to date does not allow strong conclusions (as discussed in section 5.5). The lack of appropriate controls is perhaps the most critical issue in this regard. Experimental manipulations are usually required to control for confounding effects and acoustic playbacks provide a potential way of determining whether noise causes a response. Field studies provide the most ecologically relevant data on the impacts of noise to wildlife, but experimental manipulations are often harder in such conditions. Combinations of carefully designed laboratory and field studies may provide complementary evidence in this regard; natural 'experiments' where treatment areas differ only in their noise output are also ideal, but likely to be rare. As well as assessing responses to noise, it is important to examine the hearing sensitivity of the study species. Without this information, it is not possible to conclude whether a lack of response is because an animal simply cannot hear the relevant noise or because it potentially tolerates the noise at that level. Representative audiograms can be obtained by measuring the hearing thresholds of a relatively small number of individuals, which is particularly important for vulnerable species where access to large populations may not be an option.

Conducting experiments on species of conservation priority raises some ethical issues. Costs and benefits of the information obtained from empirical study must be evaluated in conservation terms, for each vulnerable species independently. Moreover, with ethical considerations in mind, rigorous experimental design must be implemented and the most crucial questions addressed to produce the maximum research outputs from a minimum impact on priority species. Difficulties in studying certain threatened species may be circumvented by utilising regional differences in abundance, and using established captive populations. For example, the badger is a SPI in Scotland, and yet in the

south-west of England it is more common. Perhaps future research could avoid placing burden on already struggling SPI populations by utilising abundances in other regions both in the UK and wider afield, with impacts to critically threatened species extrapolated from studies on ecologically and phylogenetically similar species.

### 6.3 Noise quantification

To allow the strongest conclusions, studies need to quantify the level, duration and frequency content of the anthropogenic noise source examined. As the hearing capabilities and frequency ranges of most species remain unknown, these should be measured using a flat-response microphone with a broad bandwidth, allowing determination of important parameters outside the range of human hearing. Although A-weighted sound levels allow the comparison with noise data collected with human impact assessment as its goal, this type of filter has limited relevance for most other species as hearing ranges and sensitivities can vary considerably between taxa. Building a database of un-weighted, broad bandwidth noise spectra would be useful both in enhancing the strength of evidence for the study itself and for enabling extrapolation to studies on other wildlife in the future. If comparison to the human anthropogenic noise literature is required, the use of flat-response microphones enables appropriate filter application during analysis.

### 6.4 Individual level studies

The majority of studies on non-human animals to date have focused on the impact of acute noise, but as most anthropogenic noise sources emit sound over extended time periods, the impact of repeated or chronic noise exposure is of greater biological relevance. Experimental assessment is vital to examine possible changes in response arising from, for example, habituation, tolerance and sensitisation. Interactions between noise and other stressors also need to be understood. It is currently unknown whether the impacts of anthropogenic noise are heightened, lessened or remain the same when animals are also exposed to such situations as high disturbance or light and chemical pollution. Moreover, fitness implications across all taxa need addressing if long-term impacts of noise on populations are to be predicted; assessments about the viability and survival of a species as a whole are needed to inform conservation priorities. Studies directly investigating population viability can be logistically difficult, requiring continual assessments over long time periods. It is possible, however, to use data on fitness consequences at the level of the individual to parameterise theoretical models making predictions about outcomes at a population level. Such agent-based modelling has previously been applied to environmental resource management, ecological and conservation issues (McLane et al. 2011). If modelling such as this can be introduced to anthropogenic noise research, individual-based fitness studies would be able to indicate conservation priorities without the immediate requirement for long-term data that is not likely to become available in the near future.

### 6.5 Community and ecosystem level studies

Although assessing impacts of noise at the species level is important, investigations of interactions among species are also vital. At the dyadic level (i.e. interactions between two species), noise may affect the two species differently. For example, if a predator species is impacted in a more detrimental manner than its prey (Siemers & Schaub 2011), the reproductive success of the latter may be enhanced in noisy environments. Francis et al. 2009 have found, for instance, that the nest success of certain bird species increased at noisy treatment sites compared to a quiet control, due to a decrease in the abundance of predators. Noise might also be expected to affect community structure. To date, there have been relatively few attempts to consider how anthropogenic noise affects biodiversity *per se* (but see Bayne et al. 2008; Francis et al. 2009; Herrera-Montez & Aide

2011) and findings are mixed and potentially taxon-specific: for example, Herrera-Montez & Aide (2011) found that although avian biodiversity declined in noisy areas, anuran biodiversity was not significantly affected. Finally, very recent work has provided the first evidence that anthropogenic noise could affect ecosystem services: Francis et al. (2012) have shown that noise can influence pollination and seed dispersal. Interactions at the community and ecosystem level are clearly more complex than when considering single species, but assessing the potential impact of anthropogenic noise at all levels is crucial for a full understanding of this global issue.

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## Appendix A: Literature sources and illustrative noise spectra for major types of non-marine anthropogenic noise

### Road traffic noise

#### Illustrative noise spectra from:

Schaub, A. Ostwald, J. & Siemers, B. M. (2009). Foraging bats avoid noise. *Journal of Experimental Biology* **116**, 3174-3180.

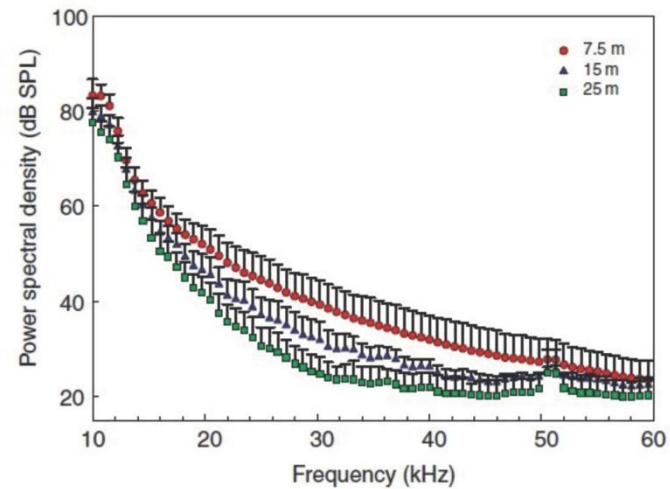


Fig. 2. Mean power spectral densities (PSDs) for 50 cars and 50 trucks recorded at a highway in 7.5 m, 15 m and 25 m distance from the middle of the right lane. Microphone height was 1.5 m. Error bars display the standard deviation.

In Fig. 2, the values on the x-axis (Frequency) were incorrectly labelled as scaling from 10 to 60 kHz.

The correct axis scale is 0 to 50 kHz.

**Other sources of noise spectra:**

Hanna D., Blouin-Demers G., Wilson D. R. & Mennill D. J. (2011). Anthropogenic noise affects song structure in red-winged blackbirds (*Agelaius phoeniceus*). *Journal of Experimental Biology* **214**, 3549-3556.

Hu, Y. & Cardoso, G. C. (2010). Which birds adjust the frequency of vocalizations in urban noise? *Behavioural Ecology* **20**, 1268-1273.

Nemeth E. & Brumm H. (2009). Blackbirds sing higher-pitched songs in cities: adaptation to habitat acoustics or side-effect of urbanization? *American Behaviour* **78**, 637-641.

Nemeth E. & Brumm H. (2010). Birds and anthropogenic noise: are urban songs adaptive? *American Naturalist* **176**, 465-475.

Parris K. M., Velik-Lord M. & North J. M. A. (2009). Frogs call at a higher pitch in traffic noise. *Ecology and Society* **15**, article 25.

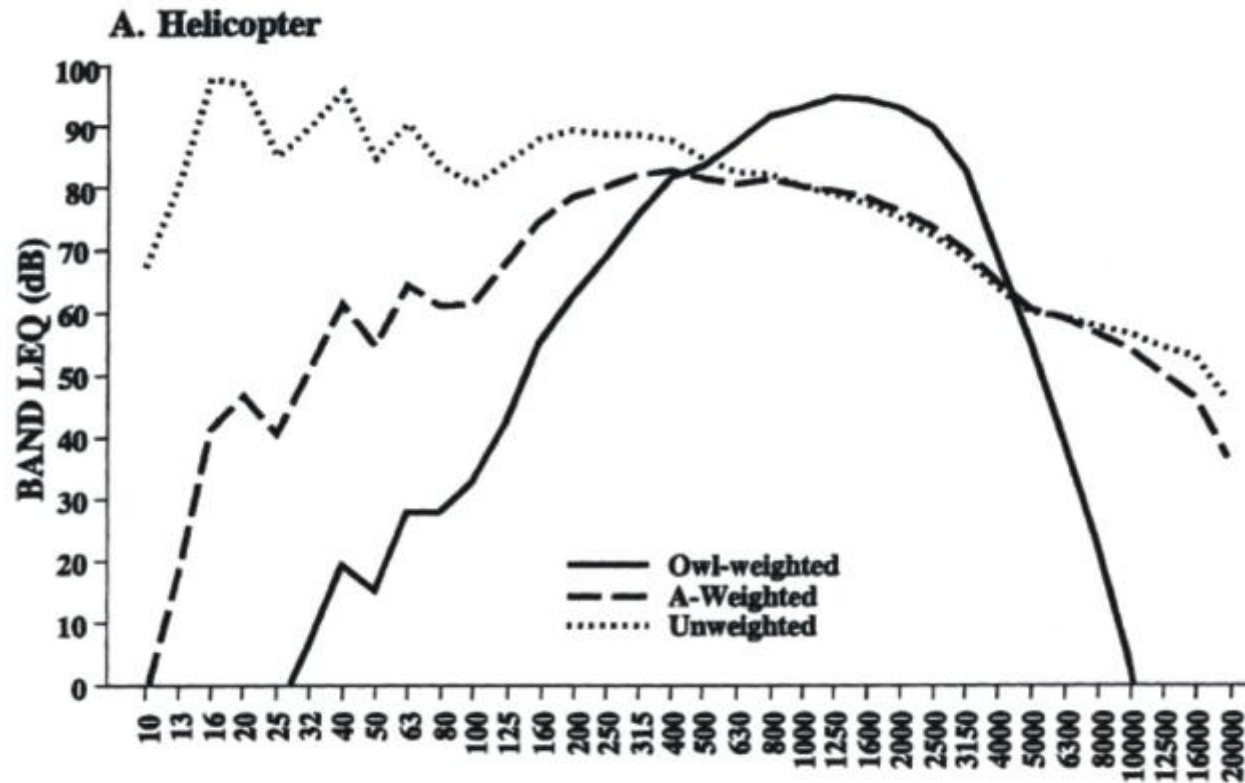
Pohl, N. U., Slabbekoorn, H., Klump, G. M. & Langemann U. (2009). Effects of signal features and environmental noise on signal detection in the great tit, *Parus major*. *Animal Behaviour* **78**, 1293-1300.

Shieh, B., Liang, S. Chen, C., Loa, H. & Liao, C. (2011). Acoustic adaptations to anthropogenic noise in the cicada *Cryptotympana takasagona* Kato (Hemiptera: Cicadidae). *Acta Ethologica*

### Air traffic noise

#### Illustrative noise spectra from:

Delaney, D. K., Grubb, T. G., Beier, P., Pater, L. L. & Reiser, M. H. (1999). Effects of helicopter noise on Mexican spotted owls. *Journal of Wildlife Management* **63**, 60-76.



#### Other sources of noise spectra:

Brown, A. (1990). Measuring the effect of aircraft noise on sea birds. *Environment International* **16**, 587-592.

Pons, J. & Santiago, J. S. (1999). Normalized noise spectra of aircraft take-off and landing operations. *Deutsche Gesellschaft für Akustik (DEGA)*.

Sun, J. W.C. & Narins, P. M. (2005). Anthropogenic sounds differentially affect amphibian call rate. *Biological Conservation* **121**, 419-427.

**Rail traffic noise**

**Illustrative noise spectra from:**

Halfwerk, W., Holleman, L. J. M., Lessells, C. M. & Slabbekoorn, H. (2011). Negative impact of traffic noise on avian reproductive success. *Journal of Applied Ecology* **48**, 210-219.

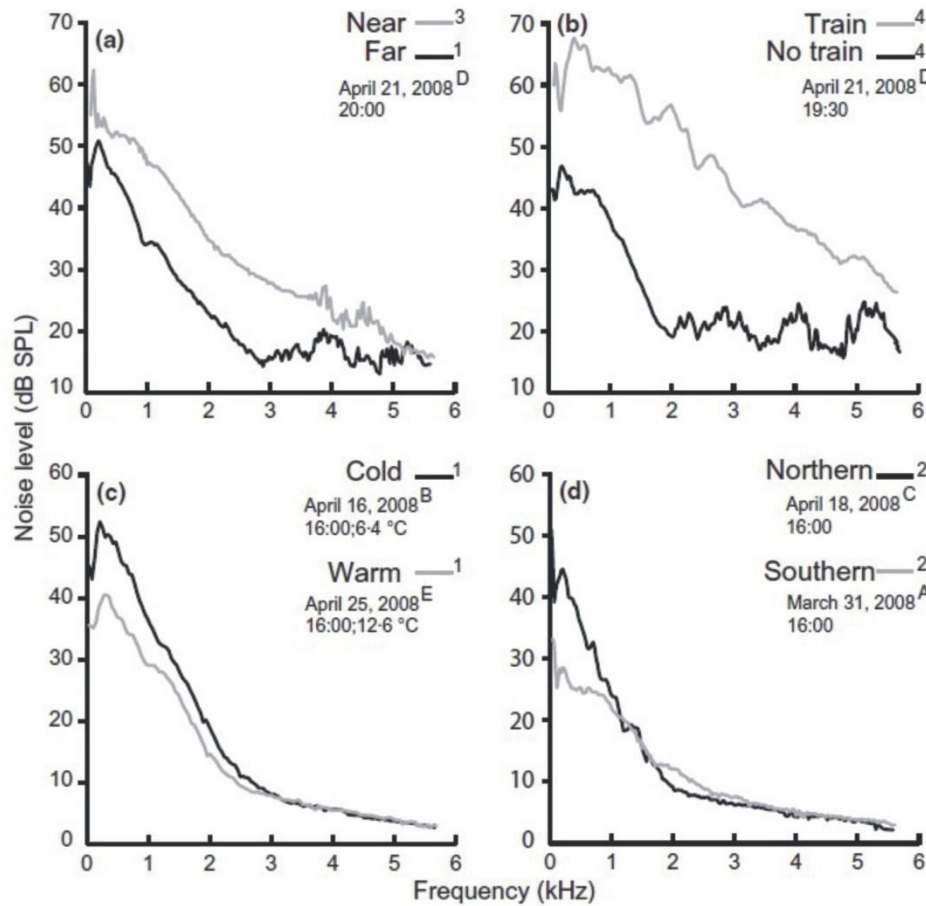


Fig. 2. Variation in sound profiles across different environmental conditions. a) power-spectrographic example comparing sound profiles near to (±100 m), and far from (±700 m), the motorway. At larger distances, the high-frequency components of traffic noise are more attenuated and even disappear above ±3 kHz. b) recordings made near the railway (±100 m from the track and ±1 km from the motorway) shortly before and during the passage of a train. c) comparison of sound profiles on days with different temperatures, but similar wind conditions illustrates large effect of weather conditions on noise levels. d) comparison of sound profiles on days with opposite wind directions, but similar temperature and wind speed. Small numbers refer to locations illustrated in Fig. 1. Capital letters refer to recording days illustrated in Fig. 3.

## Boat noise (freshwater)

### Illustrative noise spectra from:

Amoser, S., Wysocki, L. E. & Ladich F. (2004). Noise emission during the first powerboat race in an Alpine lake and potential impact on fish communities. *Journal of the Acoustical Society of America* **116**, 3789-3797.

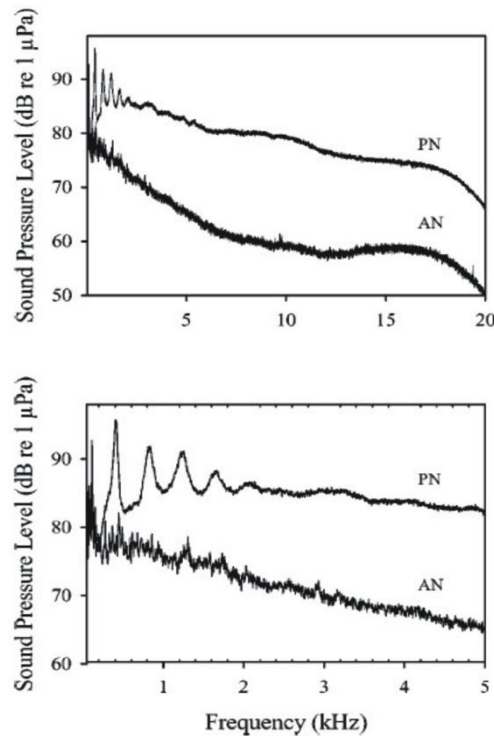


FIG. 4. Noise spectra recorded during the different conditions in Lake Traunsee during the powerboat race in 2003. Shown are the mean spectra computed from 107 s AN and 350 s PN. The lower graph gives a detail of the noise spectra within the hearing range of fishes (50–5000 Hz).

### Other sources of noise spectra:

Purser, J. & Radford, A. N. (2011). Acoustic noise induces attention shifts and reduces foraging performance in three-spined sticklebacks (*Gasterosteus aculeatus*). *PLoS ONE* **6**, e17478.

Wysocki, L., Dittami, J. & Ladich F. (2006). Ship noise and cortisol secretion in European freshwater fishes. *Biological Conservation* **128**, 501-508.



**Appendix B: List of all publications assessing anthropogenic noise impacts in non-marine species**

Species category	Reference:
UK PS/SPI	Amoser, S., Wysocki, L. E. & Ladich, F. (2004). Noise emission during the first powerboat race in an Alpine lake and potential impact on fish communities. <i>Journal of the Acoustical Society of America</i> <b>116</b> , 3789-3797.
Non-UK	Arevalo, E. J. & Newhard, K. (2011). Traffic noise affects forest bird species in a protected tropical forest. <i>Revista De Biología Tropical</i> <b>59</b> , 969-980.
Non-UK	Bayne, E., Habib, S. & Boutin, S. (2008). Impacts of chronic anthropogenic noise from energy-sector activity on abundance of songbirds in the boreal forest. <i>Conservation Biology</i> <b>22</b> , 1186-93.
Non-UK	Bee, M. A. & Swanson, E. M. (2007). Auditory masking of anuran advertisement calls by road traffic noise. <i>Animal Behaviour</i> <b>74</b> , 1765-1776.
Non-UK	Bermúdez-Cuamatzin, E., Ríos-Chelén, A. A., Gil, D. & Garcia, C. M. (2011). Experimental evidence for real-time song frequency shift in response to urban noise in a passerine bird. <i>Biology Letters</i> <b>7</b> , 36-38.
Non-UK	Bermúdez-Cuamatzin, E., Ríos-Chelén, A.A., Gil, D. & Garcia, C. M. (2009). Strategies of song adaptation to urban noise in the house finch: syllable pitch plasticity or differential syllable use? <i>Behaviour</i> <b>146</b> , 1269-1286.
Non-UK	Brown, A. (1990). Measuring the effect of aircraft noise on sea birds. <i>Environment International</i> <b>16</b> , 587-592.
UK	Brumm, H. (2004). The impact of environmental noise on song amplitude in a territorial bird. <i>Journal of Animal Ecology</i> <b>73</b> , 434-440.
UK PS/SPI	Burger, J. (1981). Behavioural responses of herring gulls <i>Larus argentatus</i> to aircraft noise. <i>Environmental Pollution</i> <b>24</b> , 177-184.
Non-UK	Cardoso, G. C. & Atwell, J. W. (2011). On the relation between loudness and the increased song frequency of urban birds. <i>Animal Behaviour</i> <b>82</b> , 836-831.
UK	Chesser, R. K., Caldwell, R. S. & Harvey, M. J. (1975). Effects of noise on feral populations of <i>Mus musculus</i> . <i>Physiological Zoology</i> <b>48</b> , 323-325.
Non-UK	Cunnington, G. M. & Fahrig, L. (2010). Plasticity in the vocalizations of anurans in response to traffic noise. <i>Acta Oecologica</i> <b>6</b> , 436-470.
Non-UK	Delaney, D.K., Grubb, T.G., Beier, P., Pater, L.L. & Reiser, M. H. (1999). Effects of helicopter noise on Mexican spotted owls. <i>Journal of Wildlife Management</i> <b>63</b> , 60-76.
UK	Diaz, M., Parra, A. & Gallardo, C. (2011). Serins respond to anthropogenic noise by increasing vocal activity. <i>Behavioural Ecology</i> <b>22</b> , 332-336.
Non-UK	Dowling, J. L., Luther, D. A. & Marra, P. P. (2012). Comparative effects of urban development and anthropogenic noise on bird songs. <i>Behavioural Ecology</i> <b>23</b> , 201-209.
UK PS/SPI	Ellis, D. H. & Ellis, H.E. (1991). Raptor responses to low-level aircraft and sonic booms. <i>Environmental Pollution</i> <b>74</b> , 53-83.

Appendices for Project NO0235 'The Effects of Noise on Biodiversity'

Non-UK	Francis, C. D., Ortega, C. P. & Cruz, A. (2009). Noise pollution changes avian communities and species interactions. <i>Current Biology</i> <b>19</b> , 1415-1419.
Non-UK	Francis, C. D., Ortega, C. P. & Cruz, A. (2011). Different behavioural responses to anthropogenic noise by two closely related passerine birds. <i>Biology Letters</i> <b>7</b> , 850-852.
Non-UK	Francis, C. D., Ortega, C. P. & Cruz, A. (2011). Vocal frequency change reflects different responses to anthropogenic noise in two subspecies tyrant flycatchers. <i>Proceedings of the Royal Society B</i> . <b>278</b> , 2025-2031.
Non-UK	Francis, C. D., Paritsis, J., Ortega, C. P. & Cruz, A. (2011). Landscape patterns of avian habitat use and nest success are affected by chronic gas well compressor noise. <i>Landscape Ecology</i> <b>26</b> , 1269-1280.
Non-UK	Francis, C., Kleist, N. J. & Ortega, C. (2012). Noise pollution alters ecological services: enhanced pollination and disrupted seed dispersal. <i>Proceedings of the Royal Society (in press)</i> .
UK PS/SPI	Fuller, R., Warren, P. & Gaston, K. (2007). Daytime noise predicts nocturnal singing in urban robins. <i>Biology Letters</i> <b>3</b> , 368-370.
Non-UK	Goodwin, S. E. & Shriver, W. G. (2011). Effects of traffic noise on occupancy patterns of forest birds. <i>Conservation Biology</i> <b>25</b> , 406-411.
Non-UK	Grafe, T. U., Döbler, S. & Linsenmair, K. E. (2002). Frogs flee from the sound of fire. <i>Proceedings of the Royal Society B</i> . <b>269</b> , 999-1003.
Non-UK	Graham, A. L. & Cooke, S. (2008). The effects of noise disturbance from various recreational boating activities common to inland waters on the cardiac physiology of a freshwater fish, the largemouth bass ( <i>Micropterus salmoides</i> ). <i>Aquatic Conservation - Marine and Freshwater Ecosystems</i> <b>18</b> , 1315-1324.
UK PS/SPI	Gross, K., Pasinelli, G. & Kunc, H. P. (2010). Behavioral plasticity allows short-term adjustment to a novel environment. <i>American Naturalist</i> <b>176</b> , 456-464.
Non-UK	Habib, S., Bayne, E. & Boutin, S. (2007). Chronic industrial noise affects pairing success and age structure of ovenbirds <i>Seiurus aurocapilla</i> . <i>The Auk</i> <b>44</b> , 176-184.
UK	Halfwerk, W. & Slabbekoorn, H. (2009). A behavioural mechanism explaining noise-dependent frequency use in urban birdsong. <i>Animal Behaviour</i> <b>78</b> , 1301-1307.
UK	Halfwerk, W., Bot, S., Buikx, J., van der Velde, M., Komdeur, J., ten Cate, C. & Slabbekoorn, H. (2011). Low-frequency songs lose their potency in noisy urban conditions. <i>Proceedings of the National Academy of Sciences</i> <b>108</b> , 14549-14554.
UK	Halfwerk, W., Holleman, C., Lessells, C. & Slabbekoorn, H. (2011). Negative impact of traffic noise on avian reproductive success. <i>Journal of Applied Ecology</i> <b>48</b> , 210-219.
UK	Hall, C. (1989). Noise measurement at Pitlochry power station. Scottish Fisheries working paper no.12/89.
UK	Hamao, S., Watanabe, M. & Mori, Y. (2010). Urban noise and male density affect songs in the great tit <i>Parus major</i> . <i>Ethology Ecology and Evolution</i> <b>23</b> , 111-119.
Non-UK	Hanna, D., Blouin-Demers, G., Wilson, D. R. & Mennill, D. J. (2011). Anthropogenic noise affects song structure in red-winged blackbirds ( <i>Agelaius phoeniceus</i> ). <i>Journal of Experimental Biology</i> <b>214</b> , 3549-3556.
Non-UK	Herrera-Montes, M. I. & Aide, T. M. (2011). Impacts of traffic noise on anuran and bird communities. <i>Urban Ecosystems</i> <b>14</b> , 415-427.

Appendices for Project NO0235 'The Effects of Noise on Biodiversity'

Non-UK	Holt, D. E. & Johnston, C. E. (2012). The effect of noise on acoustic communication in <i>Cyprinella venusta</i> . <i>Bioacoustics (in press)</i> .
Non-UK	Hoskin, C. J. & Goosem, M. W. (2010). Road impacts on abundance, call traits, and body size of rainforest frogs in northeast Australia. <i>Ecology and Society</i> <b>15</b> , article 15.
Non-UK	Hu, Y. & Cardoso, G. C. (2009). Are bird species that vocalize at higher frequencies preadapted to inhabit noisy urban areas? <i>Behavioural Ecology</i> <b>20</b> , 1268-1273.
Non-UK	Hu, Y. & Cardoso, G. C. (2010). Which birds adjust the frequency of vocalizations in urban noise? <i>Animal Behaviour</i> <b>79</b> , 863-867.
UK PS/SPI	Iglesias, C., Mata, C. & Malo, J. E. (2011). The influence of traffic noise on vertebrate road crossings through underpasses. <i>AMBIO</i> <b>41</b> , 193-201.
Non-UK	Kaiser, K. & Hammers, J. L. (2009). The effect of anthropogenic noise on male advertisement call rate in the neotropical treefrog, <i>Dendropsophus triangulum</i> . <i>Behaviour</i> <b>146</b> , 1053-1069.
Non-UK	Kaiser, K., Scofield, D. G., Alloush, M., Jones, R. M., Marczak, S., Martineau, K., Oliva, M. A & Narins, P. M. (2011). When sounds collide: the effect of anthropogenic noise on a breeding assemblage of frogs in Belize, Central America. <i>Behaviour</i> <b>148</b> , 215-232.
Non-UK	Lackey, M. A., Morrison, M. L., Loman, Z. G., Fisher, N., Farrell, S. L., Collier, B. A. & Wilkins, R. N. (2011). Effects of road construction noise on the endangered golden-cheeked warbler. <i>Wildlife Society Bulletin</i> <b>35</b> , 15-19.
Non-UK	Lengagne, T. (2008). Traffic noise affects communication behaviour in a breeding anuran, <i>Hyla arborea</i> . <i>Biological Conservation</i> <b>141</b> , 2023-2031.
Non-UK	Lowry, H., Lill, A. & Wong, B. B. M. (2011). Tolerance of auditory disturbance by an avian urban adapter, the noisy miner. <i>Ethology</i> <b>117</b> , 490-497.
UK PS/SPI	Mann, D., Cott, P. & Horne, B. (2009). Under-ice noise generated from diamond exploration in a Canadian sub-arctic lake and potential impacts on fishes. <i>Journal of the Acoustical Society of America</i> <b>126</b> , 2215-2222.
UK	Mendes, S., Colino-Rabanal, V. J. & Peris, S. J. (2011). Bird song variations along an urban gradient: The case of the European blackbird ( <i>Turdus merula</i> ). <i>Landscape and Urban Planning</i> <b>99</b> , 51-57.
UK	Mockford, E. J. & Marshall, R. C. (2009). Effects of urban noise on song and response behaviour in great tits. <i>Proceedings of the Royal Society B</i> . <b>276</b> , 2979-2985.
UK	Mockford, E. J., Marshall, R. C. & Dabelsteen, T. (2011). Degradation of rural and urban great tit song: Testing transmission efficiency. <i>PLoS ONE</i> <b>6</b> , e28242.
UK PS/SPI	Nedwell, J. & Edwards, B. (2002). Measurements of underwater noise in the Arun River during piling at County Wharf, Littlehampton. <i>Report number 513 R 0108</i>
UK PS/SPI	Nedwell, J., Turnpenny, A., Langworthy, J. & Edwards, B. (2003). Measurements of underwater noise during piling at the Red Funnel Terminal, Southampton, and observations of its effect on caged fish. <i>Subacoustech Report 558 R 0207</i> .
UK	Nemeth, E. & Brumm, H. (2009). Blackbirds sing higher-pitched songs in cities: adaptation to habitat acoustics or side-effect of urbanization? <i>Animal Behaviour</i> <b>78</b> , 637-641.
UK	Nemeth, E. & Brumm, H. (2010). Birds and anthropogenic noise: Are urban songs adaptive? <i>American Naturalist</i> <b>176</b> , 465-475.

Appendices for Project NO0235 'The Effects of Noise on Biodiversity'

Non-UK	Parris, K. M. & Schneider, A. (2009). Impacts of traffic noise and traffic volume on birds of roadside habitats. <i>Ecology and Society</i> <b>14</b> , article 29.
Non-UK	Parris, K. M., Velik-Lord, M. & North, J. M. A. (2009). Frogs call at a higher pitch in traffic noise. <i>Ecology and Society</i> <b>15</b> , article 25.
Non-UK	Penna, M. Gormaz, J. P. & Narins, P. M. (2009). When signal meets noise: immunity of the frog ear to interference. <i>Naturwissenschaften</i> <b>96</b> , 835-843.
UK PS/SPI	Peris, S. J. & Pescador, M. (2004). Effects of traffic noise on passerine populations in Mediterranean wooded pastures. <i>Applied Acoustics</i> <b>65</b> , 357-366.
UK	Pohl N., Slabbekoorn H., Klump G. & Langemann U. (2009). Effects of signal features and environmental noise on signal detection in the great tit, <i>Parus major</i> . <i>Animal Behaviour</i> <b>78</b> , 1293-1300.
UK	Pohl, N., Leadbeater, E., Slabbekoorn, H., Klump, G. & Langemann, U. (2012). Great tits in urban noise benefit from high frequencies in song detection and discrimination. <i>Animal Behaviour</i> (in press).
Non-UK	Popper, A. N., Smith, M. E., Hanna, B. W., Macgillivray, A. O., Austin, M. E. & Mann, D. A. (2005). Effects of exposure to seismic airgun use on hearing of three fish species. <i>Journal of the Acoustical Society of America</i> <b>117</b> , 3958-3971.
Non-UK	Potvin, D. A., Parris, K. M. & Mulder, R. A. (2011). Geographically pervasive effects of urban noise on frequency and syllable rate of songs and calls in silvereyes ( <i>Zosterops lateralis</i> ). <i>Proceedings of the Royal Society B</i> <b>278</b> , 2464-2469.
Non-UK	Rabin, L., Cross, R. & Owings, D. (2006). The effects of wind turbines on antipredator behavior in California ground squirrels ( <i>Spermophilus beecheyi</i> ). <i>Biological Conservation</i> <b>131</b> , 410-420.
UK PS/SPI	Reijnen, R., Foppen, R. & Meeuwssen, H. (1996). The effects of traffic on the density of breeding birds in Dutch agricultural grasslands. <i>Biological Conservation</i> <b>75</b> , 255-260.
UK PS/SPI	Rheindt, F. E. (2003). The impact of roads on birds: Does song frequency play a role in determining susceptibility to noise pollution? <i>Journal of Ornithology</i> <b>144</b> , 295-306.
UK	Ripmeester, E., Kok, J., van Rijssel, J. C. & Slabbekoorn, H. (2010). Habitat-related birdsong divergence: a multi-level study on the influence of territory density and ambient noise in European blackbirds. <i>Behavioural Ecology and Sociobiology</i> <b>64</b> , 409-418.
UK	Ripmeester, E., Mulder, M. & Slabbekoorn, H. (2010). Habitat-dependent acoustic divergence affects playback response in urban and forest populations of the European blackbird. <i>Behavioural Ecology</i> <b>21</b> , 876-883.
Non-UK	Schaub, A., Ostwald, J. & Siemers, B. M. (2008). Foraging bats avoid noise. <i>Journal of Experimental Biology</i> <b>211</b> , 3174-3180.
Non-UK	Seger, K. D., Rodewald, A. D. & Soha, J. A. (2011). Urban noise predicts song frequency in Northern cardinals and American robins. <i>Bioacoustics</i> <b>20</b> , 267.
Non-UK	Shieh, B., Liang, S., Chen, C., Loa, H. & Liao, C. (2011). Acoustic adaptations to anthropogenic noise in the cicada <i>Cryptotympana takasagona</i> Kato (Hemiptera: Cicadidae). <i>Acta Ethologica</i> June DOI 10.1007/s10211-011-0105-x.
UK PS/SPI	Shirley, M. D. F., Armitage, V. L., Barden, T. L., Gough, M., Lurz, P. W. W., Oatway, D. E., South, A. B. & Rushton, S. P. (2001). Assessing the impact of a music festival on the emergence behaviour of a breeding colony of Daubenton's bats ( <i>Myotis daubentonii</i> ). <i>Journal of Zoology (London)</i> <b>254</b> , 367-373.

Appendices for Project NO0235 'The Effects of Noise on Biodiversity'

Non-UK	Siemers, B. & Schaub, A. (2011). Hunting at the highway: traffic noise reduces foraging efficiency in acoustic predators. <i>Proceedings of the Royal Society B</i> <b>278</b> , 1646-1652.
UK	Slabbekoorn, H. & den Boer-Visser, A. (2006). Cities change the songs of birds. <i>Current Biology</i> <b>16</b> , 2326-2331.
UK	Slabbekoorn, H. & Peet, M. (2003). Birds sing at a higher pitch in urban noise. <i>Nature</i> <b>424</b> , 267.
Non-UK	Song, J., Mann, D., Cott, P., Hanna, B. & Popper, A. N. (2008). The inner ears of Northern Canadian freshwater fishes following exposure to seismic air gun sounds. <i>Journal of the Acoustical Society of America</i> <b>124</b> , 1360-1366.
Non-UK	Stone, E. (2000). Separating the noise from the noise: A finding in support of the "Niche Hypothesis," that birds are influenced by human-induced noise in natural habitats. <i>Anthrozoos</i> <b>13</b> , 225-231.
Non-UK	Summers, P. D., Cunnington, G. M. & Fahrig, L. (2011). Are the negative effects of roads on breeding birds caused by traffic noise? <i>Journal of Applied Ecology</i> <b>48</b> , 1527-1534.
Non-UK	Sun, J. W.C. & Narins, P. M. (2005). Anthropogenic sounds differentially affect amphibian call rate. <i>Biological Conservation</i> <b>121</b> , 419-427.
Non-UK	Swaddle, J. & Page, L. (2007). High levels of environmental noise erode pair preferences in zebra finches: implications for noise pollution. <i>Animal Behaviour</i> <b>74</b> , 363-368.
UK PS/SPI	Trimper, P.G., Standen, N.M., Lye, L.M., Lemons, D., Chubbs, T. E. & Humphries, G. W. (1998). <i>Journal of Applied Ecology</i> <b>35</b> , 122-130.
UK	Verzijden, M. N., Ripmeester, E., Ohms, V. R., Snelderwaard, P. & Slabbekoorn, H. (2010). Immediate spectral flexibility in singing chiffchaffs during experimental exposure to highway noise. <i>Journal of Experimental Biology</i> <b>213</b> , 2575-2581.
UK PS/SPI	Ward, D. H., Stehn, R. A., Erickson, W. P. & Dereksen, D. V. (1999). Response of fall-staging brant and Canada geese to aircraft overflights in southwestern Alaska. <i>Journal of Wildlife Management</i> <b>63</b> , 373-381.
Non-UK	Weisenberger, M. E., Krausman, P. R., Wallace, M. C., DeYound, D. W. & Maughan, O.E. (1996). Effects of simulated jet aircraft noise on heart rate and behavior of desert ungulates. <i>Journal of Wildlife Management</i> <b>60</b> , 52-61.
Non-UK	Wood, W. E. & Yezerinac, S. M. (2006). Song sparrow ( <i>Melospiza melodia</i> ) song varies with urban noise. <i>The Auk</i> <b>123</b> , 650-659.
UK	Wysocki, L., Dittami, J. & Ladich, F. (2006). Ship noise and cortisol secretion in European freshwater fishes. <i>Biological Conservation</i> <b>128</b> , 501-508.
Non-UK	Zollinger, S.A., Goller, F. & Brumm, H. (2011). Metabolic and respiratory costs of increasing song amplitude in zebra finches. <i>PLoS ONE</i> <b>6</b> , e23198.
Non-UK	Zollinger, S.A., Goller, F. & Brumm, H. (2012). The energetics of singing in noise - metabolic and respiratory costs of increasing song amplitude. <i>Bioacoustics</i> (in press).
Non-UK	Zurcher, A. A., Sparks, D. W. & Bennett, V. J. (2010). Why the bat did not cross the road? <i>Acta Chiropterologica</i> <b>12</b> , 337-340.

**Appendix C: List of publications providing audiograms for UK priority species and species of principal importance**

Speices	Audiogram type	Appendix C - UK PS and SPI audiograms
Sand lizard ( <i>Lacerta agillis</i> )	Behavioural	Berger, K. (1924). Experimentelle studien Uber schallperzeption bei Reptilien. <i>Zeitschrift fur Vergleichende Physiologie</i> <b>1</b> , 517-540
Pipistrelle ( <i>Pipistrellus sp.</i> )	Neurophysiology	Brown, A. (1973). An investigation of the cochlear microphonic response of two species of echolocating bats: <i>Rousettus aegyptiacus</i> (Geoffroy) and <i>Pipistrellus pipistrellus</i> (Schreber). <i>Journal of Comparative Physiology</i> <b>83</b> , 407-413.
Brown long-eared bat ( <i>Plecotus auritus</i> )	Neurophysiology	Coles, R. B., Guppy, A., Anderson, M. E. & Schlegel, P. (1989). Frequency sensitivity and directional hearing in the gleaning bat, <i>Plecotus auritus</i> (Linnaeus 1758). <i>Journal of Comparative Physisology</i> <b>165</b> , 269-280
Starling ( <i>Sturnus vulgaris</i> )	Behavioural	Dooling, R. J., Okanoya, K., Downing, J. & Hulse, S. (1986). Hearing in the starling ( <i>Sturnus vulgaris</i> ): Absolute thresholds and critical ratios. <i>Bulletin of the Psychonomic Society</i> <b>24</b> , 462-464.
Barn owl ( <i>Tyto Alba</i> )	Behavioural	Dyson, M., Klump, G. & Gauger, B. (1998). Absolute hearing thresholds and critical masking ratios in the European barn owl: a comparison with other owls. <i>Journal of Comparative Physiology A</i> . <b>182</b> , 695-702.
Atlantic salmon ( <i>Salmo salar</i> )	Neurophysiology	Hawkins, A. & Johnstone, A. (1978). The hearing of Atlantic salmon <i>Salmo salar</i> . <i>Journal of Fish Biology</i> <b>13</b> , 655-673.
House sparrow ( <i>Passer domesticus</i> )	Neurophysiology	Henry, K. & Lucas, J. (2009). Vocally correlated seasonal auditory variation in the house sparrow ( <i>Passer domesticus</i> ). <i>Journal of Experimental Biology</i> <b>212</b> , 3817-3822.
European eel ( <i>Anguilla anguilla</i> )	Behavioural	Jerko, H., Enger, P. S. & Sand, O. (1989). Hearing in the eel ( <i>Anguilla anguilla</i> ). <i>Journal of Comparative Physiology A</i> . <b>165</b> , 455-459.
Wart-biter cricket ( <i>Decticus verrucivorus</i> )	Neurophysiology	Kalrmring, K., Lewis, B. & Eichendorf, A. (1978). The physiological characteristics of the primary sensory neurons of the complex tibial organ of <i>Decticus verrucivorus</i> L. (Orthoptera, Tettigonioidae). <i>Journal of Comparative Physiology</i> <b>127</b> , 109-121.
Polecat	Behavioural	Kavanagh, G. & Kelly, J. (1988). Hearing in the ferret ( <i>Mustela putorius</i> ): effects of primary auditory cortical

Appendices for Project NO0235 'The Effects of Noise on Biodiversity'

( <i>Mustela putorius</i> )		lesions on thresholds for pure tone detection. <i>Journal of Neurophysiology</i> <b>60</b> , 879-88.
Greater horse-shoe bat ( <i>Rhinolophus ferrumequinum</i> )	Behavioural	Long, G. & Schnitzler, H. (1975). Behavioural audiograms from the bat , <i>Rhinolophus ferrumequinum</i> . <i>Journal of Comparative Physiology</i> <b>100</b> , 211-219.
Burbot ( <i>Lota lota</i> )	Neurophysiology	Mann, D., Cott, P., Hanna, B. & Popper, A. N. (2007). Hearing in eight species of northern Canadian freshwater fishes. <i>Journal of Fish Biology</i> <b>70</b> , 109-120.
Field cricket ( <i>Gryllus campestris</i> )	Neurophysiology	Nocke, H. (1972). Physiological aspects of sound communication in crickets ( <i>Gryllus campestris</i> L.). <i>Journal of Comparative Physiology</i> <b>80</b> , 141-162.
Bullfinch ( <i>Pyrrhula pyrrhula</i> )	Behavioural	Schwartzkopf, J. (1949). Uber sitz und leistung vonn gehor und vibrationssinn bei vogeln. <i>Zeitschrift fur Vergleichende Physiologie</i> <b>31</b> , 527-603.
Common toad ( <i>Bufo bufo</i> )	Neurophysiology	Walkowiak, W., Capranica, R. R. & Schneider, H. (1981). A comparative study of auditory sensitivity in the genus <i>Bufo</i> . <i>Behavioural Processes</i> <b>6</b> , 223-237.

**Appendix D: List of sourced publications relating to impact of non-anthropogenic noise**

<b>Species</b>	<b>UK PS or SPI? (Y/N)</b>	<b>Reference</b>
Atlantic salmon ( <i>Salmo salar</i> )	Y	Knudsen, F. R., Enger, P. S. & Sand, O. (1992). Awareness reactions and avoidance responses to sound in juvenile Atlantic salmon, <i>Salmo salar</i> . <i>Journal of fish Biology</i> <b>40</b> , 523-534.
Atlantic salmon ( <i>Salmo salar</i> )	Y	Knudsen, F.R., Enger, P. S. & Sand, O. (1994). Avoidance responses to low frequency sound in downstream migrating Atlantic salmon smolt, <i>Salmo salar</i> . <i>Journal of Fish Biology</i> <b>45</b> , 227-233.
Common toad ( <i>Bufo bufo</i> )	Y	Llusia, D., Márquez, R. & Beltrán, J. F. (2010). Non-selective and time-dependent behavioural responses of common toads ( <i>Bufo bufo</i> ) to predator acoustic cues. <i>Ethology</i> <b>116</b> , 1146-1154.
River lamprey ( <i>Lampetra fluviatilis</i> ), Smelt ( <i>Osmerus eperlanus</i> ), European eel ( <i>Anguilla anguilla</i> )	Y	Maes, J., Turnpenny, A. W. H., Lambert, D. R., Nedwell, J. R., Ollevier, F. & Parmentier, A. (2004). Field evaluation of a sound system to reduce estuarine fish intake rates at a power plant cooling water inlet. <i>Journal of Fish Biology</i> <b>64</b> , 938-946.
European eel ( <i>Anguilla anguilla</i> )	Y	Sand, O., Enger, P. S., Karlsen, H. E., Knudsen, F. & Kvernstuen, T. (2000). Avoidance responses to infrasound in downstream migrating European silver eels, <i>Anguilla anguilla</i> . <i>Environmental Biology of Fishes</i> <b>57</b> , 327-336.
Fruit fly ( <i>Drosophila montana</i> )	N	Samarra, F. I.P., Klappert, K., Brumm, H. & Miller, P. J.O. (2009). Background noise constrains communication: acoustic masking of courtship song in the fruit fly <i>Drosophila montana</i> . <i>Behaviour</i> <b>146</b> , 1635-1648.
<i>Paroecanthus podagrosus</i> , <i>Diatrypa</i> sp.	N	Schmidt, A. & Römer, H. (2011). Solutions to the cocktail party problem in insects: Selective filters, spatial release from masking and gain control in tropical crickets. <i>PLoS ONE</i> <b>6</b> , e28593.
Field cricket ( <i>Gryllus campestris</i> )	Y	Schmidt, A., Klaus, R. & Römer, H. (2011). High background noise shapes selective auditory filters in a tropical cricket. <i>Journal of Experimental Biology</i> <b>214</b> , 1754-1762.



Appendices for Project NO0235 'The Effects of Noise on Biodiversity'

Greater scaup ( <i>Aythya marila</i> )	Y	Whisson, D. A. & Takekawa, J., Y. (2000). Testing the effectiveness of an aquatic hazing device on waterbirds in the San Fransisco bay estuary of California. <i>Waterbirds</i> <b>23</b> , 56-63.
Curlew ( <i>Numenius arquata</i> ), Lapwing ( <i>Vanellus vanellus</i> )	Y	Wright, M. D., Goodman, P. & Cameron, T. (2010). Exploring behavioural responses of shorebirds to impulsive noise. <i>Wildfowl</i> <b>60</b> , 150-167.

## Appendix E: Direct evidence for impacts of anthropogenic noise on UK priority species and species of principal importance

**Appendix E1.** Direct evidence for impacts of anthropogenic noise on UK priority species and species of principal importance sorted by species. Noise levels are the range measured in the paper/report. Evidence for possible impacts of anthropogenic noise is split into three categories. Category 1 includes studies that show clear impacts or lack of impacts of anthropogenic noise. Category 2 indicates impacts may be likely but cannot be confirmed (e.g. there are confounding factors that may explain results). Category 3 is used where studies do not produce sufficient evidence to implicate anthropogenic noise as the cause of results; conclusions on possible impacts of noise are therefore weak.

Taxa	Scientific name (* indicates Scottish Species of Principal Importance)	Common name	Noise source	Noise level	Evidence		
					Definite impacts	Confirmed lack of impact	Possible impacts
Mammal	<i>Arvicola sp.</i>	Water vole	Road traffic	Not measured	N	N	3
Mammal	<i>Meles meles*</i>	Badger	Road traffic	Not measured	N	N	3
Mammal	<i>Myotis daubentonii*</i>	Daubenton's bat	Neighbourhood	Not measured	N	N	3
Bird	<i>Alauda arvensis</i>	Sky Lark	Road traffic	38-59 dB(A)	N	N	3
Bird	<i>Branta bernicla</i>	Brent goose	Aircraft	highest amps >80 dB(A)	N	N	3
Bird	<i>Carduelis cannabina</i>	Linnet	Road traffic	39-69 dB (weight not specified)	N	N	3
Bird	<i>Coccothraustes coccothraustes</i>	Hawfinch	Road traffic	Not measured	N	N	3
Bird	<i>Emberiza calandra</i>	Corn bunting	Road traffic	39-69 dB (weight not specified)	N	N	3
Bird	<i>Emberiza schoeniclus</i>	Reed bunting	Road traffic	45.7-65.6 dB(A)	Y	N	1
Bird	<i>Erithacus rubecula*</i>	Robin	Road traffic	49-57 dB(A)	N	N	2
Bird	<i>Falco peregrinus*</i>	Peregrine falcon	Aircraft	82-114 dB(A)	N	N	3
Bird	<i>Limosa limosa</i>	Black-tailed godwit	Road traffic	39-59 dB(A)	N	N	3
Bird	<i>Larus argentatus</i>	Herring gull	Aircraft	72-116dB(A)	N	N	3
Bird	<i>Lullula arborea</i>	Woodlark	Road traffic	39-69 dB (weight not specified)	N	N	3
Bird	<i>Motacilla flava</i>	Yellow wagtail	Road traffic	39-59 dB(A)	N	N	3
Bird	<i>Pandion haliaetus*</i>	Osprey	Aircraft	52-94 dB(A)	N	N	3
Bird	<i>Passer domesticus</i>	House sparrow	Road traffic	39-69 dB (weight not specified)	N	N	3
Bird	<i>Sturnus vulgaris</i>	Starling	Road traffic	39-69 dB (weight not specified)	N	N	3
Bird	<i>Troglodytes troglodytes</i>	Wren	Road traffic	Not measured	N	N	3

Appendices for Project NO0235 'The Effects of Noise on Biodiversity'

Bird	<i>Turdus philomelos</i>	Song thrush	Road traffic	Not measured	N	N	3
Fish	<i>Coregonus lavaretus</i>	Whitefish	Boat	up to 128 dB re 1 µPa	N	N	3
Fish	<i>Lota lota</i>	Burbot	Industrial	max. over 149 dB (peak re 1 µPa)	N	N	3
Fish	<i>Salmo trutta</i>	Brown trout	Pile driving	134 dB re 1 mPa @ 400m	N	N	3
Fish	<i>Salmo salar</i>	Atlantic salmon	Pile driving	source level up to 192 dB re 1 µPa	N	N	3

**Appendix E2.** Direct evidence for impacts of anthropogenic noise on UK priority species and species of principal importance sorted by noise source. Noise levels are the range measured in the paper/report. Evidence for possible impacts of anthropogenic noise is split into three categories. Category 1 includes studies that show clear impacts or lack of impacts of anthropogenic noise. Category 2 indicates impacts may be likely but cannot be confirmed (e.g. there are confounding factors that may explain results). Category 3 is used where studies do not produce sufficient evidence to implicate anthropogenic noise as the cause of results; conclusions on possible impacts of noise are therefore weak.

Noise source	Noise level	Taxa	Scientific name (* indicates Scottish Species of Principal Importance)	Common name	Evidence		
					Definite impacts	Confirmed lack of impact	Possible impacts
Road traffic	Not measured	Mammal	<i>Arvicola sp.</i>	Water vole	N	N	3
Road traffic	Not measured	Mammal	<i>Meles meles*</i>	Badger	N	N	3
Road traffic	38-59 dB(A)	Bird	<i>Alauda arvensis</i>	Sky Lark	N	N	3
Road traffic	39-69 dB (weight not specified)	Bird	<i>Carduelis cannabina</i>	Linnet	N	N	3
Road traffic	Not measured	Bird	<i>Coccothraustes coccothraustes</i>	Hawfinch	N	N	3
Road traffic	39-69 dB (weight not specified)	Bird	<i>Emberiza calandra</i>	Corn bunting	N	N	3
Road traffic	45.7-65.6 dB(A)	Bird	<i>Emberiza schoeniclus</i>	Reed bunting	Y	N	1
Road traffic	49-57 dB(A)	Bird	<i>Erithacus rubecula*</i>	Robin	N	N	2
Road traffic	39-59 dB(A)	Bird	<i>Limosa limosa</i>	Black-tailed godwit	N	N	3
Road traffic	39-69 dB (weight not specified)	Bird	<i>Lullula arborea</i>	Woodlark	N	N	3
Road traffic	39-59 dB(A)	Bird	<i>Motacilla flava</i>	Yellow wagtail	N	N	3
Road traffic	39-69 dB (weight not specified)	Bird	<i>Passer domesticus</i>	House sparrow	N	N	3

Appendices for Project NO0235 'The Effects of Noise on Biodiversity'

Road traffic	39-69 dB (weight not specified)	Bird	<i>Sturnus vulgaris</i>	Starling	N	N	<b>3</b>
Road traffic	Not measured	Bird	<i>Troglodytes troglodytes</i>	Wren	N	N	<b>3</b>
Road traffic	Not measured	Bird	<i>Turdus philomelos</i>	Song thrush	N	N	<b>3</b>
Aircraft	highest amps >80 dB(A)	Bird	<i>Branta bernicla</i>	Brent goose	N	N	<b>3</b>
Aircraft	72-116dB(A)	Bird	<i>Larus argentatus</i>	Herring gull	N	N	<b>3</b>
Aircraft	82-114 dB(A)	Bird	<i>Falco peregrinus*</i>	Peregrine falcon	N	N	<b>3</b>
Aircraft	52-94 dB(A)	Bird	<i>Pandion haliaetus*</i>	Osprey	N	N	<b>3</b>
Industrial	max. over 149 dB (peak re 1 µPa)	Fish	<i>Lota lota</i>	Burbot	N	N	<b>3</b>
Neighbourhood	Not measured	Mammal	<i>Myotis daubentonii*</i>	Daubenton's bat	N	N	<b>3</b>
Boat	up to 128 dB re 1 µPa	Fish	<i>Coregonus lavaretus</i>	Whitefish	N	N	<b>3</b>
Pile driving	134 dB re 1 mPa @ 400m	Fish	<i>Salmo trutta</i>	Brown trout	N	N	<b>3</b>
Pile driving	source level up to 192 dB re 1 µPa	Fish	<i>Salmo salar</i>	Atlantic salmon	N	N	<b>3</b>

## Appendix F: All evidence for impacts of anthropogenic noise on UK priority species and species of principal importance

Includes evidence from direct studies and extrapolations from audiogram, non-anthropogenic noise and phylogenetic/ecologically related species. Evidence from different sources is indicated (Y) in the columns under the heading 'Evidence'. Strength of evidence is split into categories 1, 2 & 3. Category 1 indicates studies that show a clear response or lack of response to noise from direct studies only; extrapolations are not included in this category. Category 2 indicates studies that show some evidence for impacts or lack of impacts of noise on the species. Several extrapolations indicating the same conclusion would also fall into this category. Category 3 indicates studies that provide only weak evidence that anthropogenic noise does or does not impact the species. Weak evidence may be due to confounding factors or other flaws in experimental design in direct studies or due to only an extrapolated impact in isolation. Grey shading indicates noise restricted to habitat outside the species' range.

Taxa	Scientific name (* indicates Scottish species of principal importance)	Common name	Evidence				Strength of evidence for possible impacts							
			Direct	Audio-gram	Phylo-genetic /eco-logical	Non-anthro noise	Road traffic	Air traffic	Wind turbine	Rail traffic	Neigh-bour-hood	Indust-rial	Boat	Pile driving
Mammal	<i>Arvicola sp.</i>	Water vole	Y				3	N/A	N/A	N/A	N/A	N/A		
Mammal	<i>Barbastella barbastellus</i>	Barbastelle bat			Y		3	N/A	N/A	N/A	N/A	N/A		
Mammal	<i>Capreolus capreolus</i>	Roe deer			Y		N/A	3	N/A	N/A	N/A	N/A		
Mammal	<i>Cervus elaphus</i>	Red deer			Y		N/A	3	N/A	N/A	N/A	N/A		
Mammal	<i>Meles meles*</i>	Badger	Y				3	N/A	N/A	N/A	N/A	N/A		
Mammal	<i>Mustela putorius</i>	Polecat		Y			3	3	3	3	N/A	N/A		
Mammal	<i>Myotis bechsteinii</i>	Bechstein's bat			Y		3	N/A	N/A	N/A	N/A	N/A		
Mammal	<i>Myotis daubentonii*</i>	Daubenton's bat	Y				N/A	N/A	N/A	N/A	3	N/A		
Mammal	<i>Nyctalus noctula</i>	Noctule			Y		3	N/A	N/A	N/A	N/A	N/A		
Mammal	<i>Pipistrellus sp.</i>	Pipistrelle		Y	Y		3	3	3	3	N/A	N/A		
Mammal	<i>Plecotus auritus</i>	Brown long-eared bat		Y	Y		2	3	3	3	N/A	N/A		

Appendices for Project NO0235 'The Effects of Noise on Biodiversity'

Mammal	<i>Rhinolophus ferrumequinum</i>	Greater horse-shoe bat		Y	Y		3	3	3	3	N/A	N/A		
Mammal	<i>Rhinolophus hipposideros</i>	Lesser-horseshoe bat			Y		3	N/A	N/A	N/A	N/A	N/A		
Mammal	<i>Sciurus vulgaris</i>	Red squirrel			Y		N/A	N/A	3	N/A	N/A	N/A		
Bird	<i>Acrocephalus paludicola</i>	Aquatic warbler			Y		3	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Acrocephalus palustris</i>	Marsh warbler			Y		3	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Alauda arvensis</i>	Skylark	Y		Y		3	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Anthus trivialis</i>	Tree pipit			Y		3	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Aythya marila</i>	Greater scaup				Y	3	3	N/A	N/A	N/A	N/A		
Bird	<i>Branta bernicla</i>	Brent goose	Y				N/A	3	N/A	N/A	N/A	N/A		
Bird	<i>Carduelis cabaret</i>	Lesser redpoll			Y		3	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Carduelis cannabina</i>	Linnet	Y		Y		3	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Carduelis flavirostris</i> subsp. <i>bensonorum/pipilans</i>	Twite			Y		3	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Coccothraustes coccothraustes</i>	Hawfinch	Y		Y		3	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Emberiza calandra</i> subsp. <i>calandra/clanceyi</i>	Corn bunting	Y		Y		3	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Emberiza cirius</i>	Cirl bunting			Y		3	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Emberiza citrinella</i>	Yellowhammer			Y		3	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Emberiza schoeniclus</i>	Reed bunting	Y		Y		1	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Erithacus rubecula*</i>	Robin	Y		Y		2	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Falco peregrinus*</i>	Peregrine falcon	Y				N/A	3	N/A	N/A	N/A	N/A		

Appendices for Project NO0235 'The Effects of Noise on Biodiversity'

Bird	<i>Lanius collurio</i>	Red-backed shrike			Y		<b>3</b>	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Larus argentatus</i>	Herring gull	Y				N/A	<b>3</b>	N/A	N/A	N/A	N/A		
Bird	<i>Locustella luscinioides</i>	Savi's warbler			Y		<b>3</b>	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Locustella naevia</i>	Grasshopper warbler			Y		<b>3</b>	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Loxia scotica</i>	Scottish crossbill			Y		<b>3</b>	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Limosa limosa</i>	Black-tailed godwit	Y				<b>3</b>	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Lullula arborea</i>	Woodlark	Y		Y		<b>3</b>	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Motacilla flava</i>	Yellow wagtail	Y		Y		<b>3</b>	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Muscicapa striata</i>	Spotted flycatcher			Y		<b>3</b>	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Numenius arquata</i>	Curlew				Y	<b>3</b>	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Pandion haliaetus*</i>	Osprey	Y				N/A	<b>3</b>	N/A	N/A	N/A	N/A		
Bird	<i>Passer domesticus</i>	House sparrow	Y	Y	Y		<b>2</b>	<b>3</b>	<b>3</b>	<b>3</b>	N/A	N/A		
Bird	<i>Passer montanus</i>	Eurasian tree sparrow			Y		<b>3</b>	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Phylloscopus sibilatrix</i>	Wood warbler			Y		<b>3</b>	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Poecile montanus subsp. kleinschimdti</i>	Willow tit			Y		<b>3</b>	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Poecile palustris subsp. palustris/dresseri</i>	Marsh tit			Y		<b>3</b>	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Prunella modularis subsp. occidentalis</i>	Hedge accentor (Dunnock)			Y		<b>3</b>	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Pyrrhula pyrrhula</i>	Bullfinch		Y	Y		<b>2</b>	<b>3</b>	<b>3</b>	<b>3</b>	N/A	N/A		

Appendices for Project NO0235 'The Effects of Noise on Biodiversity'

Bird	<i>Sterna dougallii</i>	Roseate tern			Y		N/A	3	N/A	N/A	N/A	N/A		
Bird	<i>Sturnus vulgaris</i>	Starling	Y	Y	Y		2	3	3	3	N/A	N/A		
Bird	<i>Troglodytes troglodytes</i>	Wren	Y		Y		3	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Turdus philomelos</i>	Song thrush	Y		Y		3	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Turdus torquatus</i>	Ring ouzel			Y		3	N/A	N/A	N/A	N/A	N/A		
Bird	<i>Tyto Alba*</i>	Barn owl		Y	Y		3	3	N/A	3	N/A	N/A		
Bird	<i>Vanellus vanellus</i>	Lapwing				Y	3	3	N/A	N/A	N/A	N/A		
Reptile	<i>Lacerta agillis</i>	Sand lizard		Y	Y		3	3	3	N/A	N/A	N/A		
Reptile	<i>Zootoca vivipara</i>	Common lizard			Y		3	N/A	N/A	N/A	N/A	N/A		
Amphib-ian	<i>Epidalea calamita</i>	Natterjack toad			Y		3	3	N/A	N/A	N/A	N/A	N/A	N/A
Amphib-ian	<i>Bufo bufo</i>	Common toad		Y	Y	Y	2	2	3	3	N/A	N/A	3	3
Amphib-ian	<i>Pelophylax lessonae</i>	Pool frog			Y		3	3	N/A	N/A	N/A	N/A	N/A	N/A
Fish	<i>Anguilla anguilla</i>	European eel	Y	Y	Y	Y						N/A	3	N/A
Fish	<i>Alosa alosa</i>	Alis shad			Y							N/A	3	N/A
Fish	<i>Alosa fallax</i>	Twite shad			Y							N/A	3	N/A
Fish	<i>Cobitis taenia</i>	Spined loach			Y							N/A	3	N/A
Fish	<i>Coregonus albula</i>	Vendace			Y							N/A	3	N/A
Fish	<i>Coregonus autumnalis</i>	Pollan			Y							N/A	3	N/A
Fish	<i>Coregonus lavaretus</i>	Powan/whitefish	Y		Y							N/A	3	N/A
Fish	<i>Lampetra fluviatilis</i>	River lamprey				Y						N/A	3	N/A
Fish	<i>Lota lota</i>	Burbot	Y	Y	Y							3	3	N/A
Fish	<i>Osmerus eperlanus</i>	Smelt			Y	Y						N/A	3	N/A
Fish	<i>Salmo trutta</i>	Brown trout	Y		Y	Y						N/A	3	3



Appendices for Project NO0235 'The Effects of Noise on Biodiversity'

Fish	<i>Salmo salar</i>	Atlantic salmon	Y	Y	Y	Y							<b>3</b>	<b>3</b>	<b>3</b>
Fish	<i>Salvelinus alpinus</i>	Arctic charr			Y								N/A	<b>3</b>	N/A
Invert-ebrate	<i>Gryllotalpa gryllotalpa</i>	Mole cricket			Y		<b>3</b>	<b>3</b>	<b>3</b>	N/A	<b>3</b>	N/A			
Invert-ebrate	<i>Gryllus campestris</i>	Field cricket		Y	Y	Y	<b>3</b>	<b>3</b>	<b>3</b>	<b>3</b>	<b>3</b>	N/A			
Invert-ebrate	<i>Decticus verrucivorus</i>	Wart-biter cricket		Y	Y		<b>3</b>	<b>3</b>	<b>3</b>	<b>3</b>	<b>3</b>	N/A			
Invert-ebrate	<i>Cicadetta montana</i>	New forest cicada			Y		<b>3</b>	N/A	N/A	N/A	N/A	N/A			
Invert-ebrate	<i>Stethophyma grossum</i>	Large marsh grasshopper			Y		<b>3</b>	<b>3</b>	<b>3</b>	N/A	<b>3</b>	N/A			
Invert-ebrate		Dipteran flies			Y		<b>3</b>	<b>3</b>	<b>3</b>	N/A	<b>3</b>	N/A			