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RESEARCH ARTICLE

Anthropogenic noise exposure in protected natural areas: estimating the scale of ecological consequences

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Abstract The extensive literature documenting the ecological effects of roads has repeatedly implicated noise as one of the causal factors. Recent studies of wildlife responses to noise have decisively identified changes in animal behaviors and spatial distributions that are caused by noise. Collectively, this research suggests that spatial extent and intensity of potential noise impacts to wildlife can be studied by mapping noise sources and modeling the propagation of noise across landscapes. Here we present models of energy extraction, aircraft overflight and roadway noise as examples of spatially extensive sources and to present tools available for landscape scale investigations. We focus these efforts in US National Parks (Mesa Verde, Grand Teton and Glacier) to highlight that ecological noise pollution is not a threat restricted to developed areas and that many protected natural areas experience significant noise loads. As a heuristic tool for understanding past and future noise pollution we forecast community noise utilizing a spatially-explicit land-use change model that depicts the intensity of human development at sub-county resolution. For road noise, we transform effect distances from two studies into sound levels to begin a discussion of noise thresholds for wildlife. The spatial scale of noise exposure is far larger than any protected area, and no site in the continental US is free form noise. The design of observational and experimental studies of noise effects should be informed by knowledge of regional noise exposure patterns.

Keywords Anthropogenic noise · Wildlife · Acoustical fragmentation · Acoustic modeling · Soundscape

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Introduction

Marine scientists have been alarmed about the potential ecological effects of increasing background sound levels from anthropogenic noise in the world's oceans for decades (Payne and Webb 1971; Nowacek et al. 2007; Weilgart 2007; Clark et al. 2009). This special issue highlights the emerging concern in the terrestrial ecological literature that noise pollution can degrade ecosystem integrity (Patricelli and Blickley 2006; Warren et al. 2006; Slabbekoorn and Ripmeester 2008; Barber et al. 2010; Pijanowski et al. 2011). For example, over a dozen studies have assessed the impacts of traffic volume on breeding bird density. A qualitative review of these studies indicates a clear negative relationship between traffic on roads and birds breeding near roads (Reijnen and Foppen 2006). Although many factors could mediate the effects of roads on ecological communities, two research groups (Reijnen and Foppen 1995; Reijnen et al. 1996, 1997; Forman et al. 2002) have presented evidence that a major source of long distance impacts to bird densities (effect distances from 120 to 1,200 m) is the noise produced by vehicles.

Although the road ecology literature is extensive, differences in interpretation remain. Roedenbeck et al. (2007) claim that the evidence is weak for a population-level effect of roads on terrestrial organisms. Fahrig and Rytwinski (2009) counter this conclusion by emphasizing that it is based on evidence more than a decade old. In their broader review of 79 modern studies, covering 131 species, Fahrig and Rytwinski find that the number of negative effects of roads on animal abundance outnumber the positive effects by a factor of 5. A meta-analytical study, based on 49 data sets spanning 234 mammal and bird species, found that bird populations decline within 1 km of roads and other infrastructure and mammals decline within 5 km (Benítez-López et al. 2010).

Because road ecology work has not controlled for other confounding factors (e.g., road mortality, visual disturbance, chemical pollution, habitat fragmentation, increased predation and invasive species along edges) we do not know the independent contribution of noise to these effects. Yet, over the last 8 years, a set of studies has continuously made the case that noise significantly alters terrestrial animal behaviors and distributions (Barber et al. 2010). The first of these studies, showed that great tits (*Parus major*) increase the minimum frequency of their song in urban noise, a source dominated by roadway noise (Slabbekoorn and Peet 2003). Subsequently, several birds, frogs and one squirrel have been shown to alter vocalization characteristics (including frequency shifts) in anthropogenic noise. These adjustments likely reduce the impacts of masking on signal transmission (see reviews: Patricelli and Blickley 2006; Warren et al. 2006; Slabbekoorn and Ripmeester 2008; Barber et al. 2010). Interestingly, birds that vocalize at low frequencies, in the band of road noise, show lower occupancy near roads than birds with higher frequency calls (Goodwin and Shriver 2010; also see Reindt 2003). Although there have been a few recent suggestions (Bermúdez-Cuamatzin et al. 2010; Gross et al. 2010; Nemeth and Brumm 2010; Ripmeester et al. 2010), we do not understand the reproductive consequences of signal adjustments in anthropogenic noise.

There is convincing evidence from four research groups that have isolated noise as a driving ecological force behind changes in bird habitat use. Two groups have taken advantage of 'natural experiments' in oil and gas fields, where noisy compressor stations and very similar, yet much quieter, well pad installations exist in matched habitats. Findings from these systems have shown that noisy compressor stations result in a one-third reduction in songbird density (Bayne et al. 2008), reduced pairing success and altered age structuring in ovenbirds (*Seiurus aurocapilla*; Habib et al. 2007) and substantial changes in avian species richness and predator–prey interactions (Francis et al. 2009, 2011), compared to controls. Blickley et al. (2011) used a controlled noise playback study at sage grouse (*Centrocercus urophasianus*) leks to demonstrate that experimentally-applied noise from natural-gas drilling and roads resulted in 38 and 75% declines, respectively, in peak male attendance compared to controls. Finally, traffic noise has been shown to directly impact reproductive success in great tits (*P. major*; Halfwerk et al. 2011).

These studies make it clear that, while other factors are also at play, noise is likely a major driver behind the ecological impacts of roads (although see Summers et al. 2011). We highlight road effects in this "Introduction" section as they are among the most widely documented of any anthropogenic disturbance associated with high levels of noise. However, roads are far from the only source of noise. Airways, railways, energy development, motorized recreation and human community noise all contribute to modern background sound levels. To explore the spatial extent of noise exposure and to highlight tools available for investigating the ecological impacts of noise, this paper presents a series of models involving different sources and geographic contexts. We use three noise propagation software platforms to model energy development, aircraft and roadway noise in or near protected natural areas. Our efforts are focused in protected lands to highlight that anthropogenic noise pollution is not an ecosystem threat restricted to developed areas and that, in fact, many natural areas experience significant noise loads, particularly along transportation corridors (also see Barber et al. 2010; Lynch et al. 2011). We also present an initial effort at modeling noise pollution over even broader spatial and temporal scales, specifically a continental scale community-noise model that integrates an empirically-derived relationship between human population density and day–night sound level (DNL; Table 1; EPA 1974a) with a predictive land-change model (Theobald 2005; Bierwagen et al. 2010). We emphasize that soundscape research must begin to operate at landscape scales to unequivocally demonstrate the ecological ramifications of anthropogenic noise pollution.

Methods

Sound propagates geometrically and spreading loss alone produces level declines of about 6 dB per doubling of distance for a point source (such as a natural gas compressor), and 3 dB per doubling of distance for a line source (such as a road). Atmospheric absorption (a function of temperature, humidity and elevation), temperature and wind gradients, vegetation, terrain profiles and ground characteristics are additional influences on the simplest case of geometrical spreading (Rossing 2007). The models we present here differ in their incorporation of these factors.

Energy development noise

We used SPreAD-GIS (The Wilderness Society, San Francisco, CA) to model potential noise propagation from oil and natural gas well compressors near Mesa Verde National Park in southwestern Colorado, USA (37°14′N, 108°28′W). SPreAD-GIS is an open-source application written in Python and implemented as a toolbox in ArcGIS software (ESRI, Redlands, CA)

Table 1 Glossary

Term	Definition
1/3rd octave spectrum	Sound level measurements obtained from a contiguous sequence of 1/3rd octave spectral bands. 1/3rd octave bands approximate the auditory filter widths of humans
A-weighting; dBA or dB(A)	A method of summing sound energy across the frequency spectrum of sounds audible to humans. A-weighting deemphasizes very low and high frequencies by approximating the inverse of a curve representing sound intensities that are perceived as equally loud (the 40 phon contour)
dB (decibel)	A logarithmic measure of sound level
DNL (L_{dn})	DNL is the equivalent A-weighted sound level (LEQ) over a 24 h period with a 10 dB penalty for noise between 2200 and 0700 h
Frequency (Hz and kHz)	For a sinusoidal signal, the number of pressure peaks in 1 s (Hz). A continuous measure of frequency is the speed of sound divided by wavelength
L_d	Daytime sound level is the equivalent A-weighted sound level over daylight hours, usually a 15 h period between 0700 and 2200 h
LEQ (L_{eq})	The level of a constant sound over a specific time period (e.g., 24 h LEQ) that has the same energy as the actual (unsteady) sound over the same interval
L_{max} (Max dBA)	The maximum A-weighted sound level, with a specified integration interval, measured or modeled during an event
L_x (exceedance percentiles)	The dB level exceeded × percent of the time for a given measurement period
SEL (L_{AE})	Sound exposure level is the integration of all the acoustic energy (dBA) in an event referenced to 1 s

Table 2 Input and output data for energy extraction, aircraft, and road noise models

	Energy	Aircraft	Road
Software	SPreAD-GIS	NMSim	CadnaA
Sound metric	1/3rd octave bands; 15 s LEQ	dBA 1 s LEQ	dBA; DNL, L^d, others possible
Input data			
Terrain	DEM	DEM	DEM
Ground	NLCD	Built-in	Built-in
Vegetation	NLCD	No	No, but possible
Temperature (°F)	80.5	40	50
Relative humidity (%)	22.5	65	70
Wind speed (mph)	7.4	No	No, but possible
Wind direction (°)	242.2	No	No, but possible
Source sound level	Measured	Built-in	Built-in
Ambient sound	Measured	Built-in	No, but possible
Other			Traffic mix/volume
Output data			
Raster resolution (m)	30	500	50
Raster extent (km^2)	1,960	4,576	7.3
Receiver height (m)	No	1.5	1.5
Processing time (h)	5.5	<2	9

If input or output variables are adjustable within each software package, the value or type of data used in our models is listed or indicated as 'measured'. Otherwise, 'no' or 'built-in' denotes the inability of the software package to use or alter that input or output variable. See "Methods" section for other details

DEM Digital Elevation Model, *NLCD* National Land Cover Database, for sound metric abbreviations see Table 1

Note the newest version of NMSim allows Vegetation, Wind Speed and Wind direction inputs

that incorporates data on land cover, topography, and weather conditions to model spatial patterns of noise propagation around one or multiple sound sources (Reed et al. 2011).

We used previously compiled data on oil and natural gas well locations within 10 km of the Mesa Verde National Park boundary (Leu et al. 2008). Based on mean compressor densities of 0.8 compressors/km^2 observed in nearby areas of the San Juan Basin (C. Francis, personal communication), we assumed that 42.8% of these wells had an operating compressor. We randomly assigned compressors to 64 of the 149 well locations and estimated their cumulative potential noise propagation for one-third octave frequency bands (0.125–2 kHz) throughout the 1,960 km^2 study area. Ambient sound conditions were estimated from continuous acoustic monitoring data for a nearby park with similar habitat conditions (Natural Bridges National Monument; S. Ambrose, personal communication), and models simulated mean daytime weather conditions from 1 to 7 June 2010 (National Climatic Data Center, U.S. Department of Commerce, Asheville, NC). Sound source levels were calculated from field recordings of operating natural gas well compressors (C. Francis, personal communication), which had an average power of 125 dB between 125 and 2,000 Hz, over 15 s at a distance of 30 m. We summarized cumulative propagation results for all frequency bands processed by SPreAD-GIS (0.125–2 kHz) as unweighted (dB) and A-weighted sound levels (dBA), 15 s LEQ. For a summary of input and output data see Table 2.

Aircraft noise

We used NMSim (Noise Model Simulation; Wyle Laboratories, Inc., Arlington, VA) to simulate an aircraft overflight of Grand Teton National Park, Wyoming, USA. NMSim was developed to complement the Federal Aviation Administration's Integrated Noise Model (INM) and the Department of Defense's NoiseMap and model 3-dimensional in-flight spectral noise energy, account for terrain and barrier effects, and combine the effects of multiple noise sources. We digitized a departing and an arriving path into the Jackson Hole airport based on the Boeing 757 flight track information provided by the NPS and built-in INM sound source information in NMSim. Weather values input into the model represent the average of summer values. Receiver point locations included Gros Ventre Campground (43°37′N, 110°40′W), Jenny Lake Campground (43°45′N, 110°44′W), Signal Mountain Campground (43°50′N, 110°37′W) and Lizard Creek Campground (44°0′N, 110°41′W; see Supplementary Fig. 1). Ambient noise level was a continuous value throughout the model extent, the NMSim default value of 17.6 dBA. The departing flight was followed 3 min later by an arriving flight. This allowed for a continuous model that did not include sound source overlap. We calculated noise exposure metrics for each of the receiver sites identified in the model inputs and used the full grid output (0.5 km resolution) to create a video of the overall sound source propagation across the study area in dBA, 1 s LEQ. For a summary of input and output data see Table 2.

Roadway noise

We used CadnaA (Computer Aided Noise Abatement; DataKustik, Greifenberg, Germany) to simulate daytime sound levels (L_d) from vehicle traffic along the Going to the Sun Road in Glacier National Park, Montana, USA (48°67′N, 113°83′W). CadnaA is a commercially available noise modeling software package for calculating and visualizing propagation of industrial, road, rail, and aircraft noise.

To build a road-noise model in CadnaA we input the following information: traffic counts, vehicle mix, topography, and road geometry. The National Park Service collected traffic data along the Going to the Sun Road in Glacier National Park using a TRAX Apollyon automatic traffic counter and classifier from August 5, 2009 to August 10, 2009 (5 days). From these data, we calculated the average traffic count, average vehicle mix, and average traffic speed for input into the model. Weather input data was kept at CadnaA default values. We converted a Digital Elevation Model (DEM) to a contour shapefile using ArcGIS 9.3. We assigned elevation values from the contours as height values for the digital terrain model and we assigned the road a relative elevation of 1 m above ground. These steps insured that CadnaA did not mistakenly let the road dip underground with rapid elevation changes. We used CadnaA's implementation of the Federal Highway Administration's Traffic Noise Model (TNM), using 3D distance for calculation. The TNM module within CadnaA does not take foliage into account, so it was not possible to incorporate vegetation-mediated sound attenuation into the model. To calculate the L_d output grid, we assigned daytime hours based on the daylight hours during the period of traffic data collection. Daytime ranged from 0500 to 1959. The average number of vehicles per hour was 217.2. We defined the resolution of the output grid as 50 m on a side, with receiver heights at 1.5 m. For the remaining calculation configuration values, including weather input data, we used the CadnaA defaults (see Table 2).

Community noise model and forecast

In 1974 the Environmental Protection Agency (EPA) developed a formula that depicts DNL due to general human activity. This formula is based on 100 measurement sites in 14 cities located throughout the US. Within each city, sound measurements were taken across the largest possible gradient of population densities using 1970 census tracts. Sites were also selected to capture neighborhoods near and far from busy roads but, freeways and airports were avoided to get a more general picture of urban residential areas (for more details see, EPA 1974a). DNL is the equivalent A-weighted sound level over a 24 h period with a 10 dB penalty for noise during between 2200 and 0700 h. Although DNL does not provide the clearest information regarding impacts to wildlife (see "Acoustic metrics and developing thresholds" section), this formula represents the only attempt to empirical quantify the relationship between population density and sound level in the US.

We used an updated version of Theobald's (2005; Bierwagen et al. 2010) Spatially-Explicit Regional Growth Model (SERGoM) to depict housing densities at a 100 m resolution for the conterminous United States (U.S.). In addition to current (2010) housing densities, SERGoM also depicts past and future housing densities through its linkage to U.S. Census data. Additional details about SERGoM are described elsewhere (Theobald 2005). Housing densities in SERGoM are classified into 12 categories that depict the number of housing units per unit area (0 units, >160 ac/unit, 80–160 ac/unit, 40–80 ac/unit, 20–40 ac/unit, 10–20 ac/unit, 5–10 ac/unit, 1.7–0.5 ac/unit, 0.5–1.7 ac/unit, 2–5 units/ac, 5–10 units/ac, and >10 units/ac). We assumed no housing units occurred in undevelopable public lands. While SERGoM depicts housing units, calculating a community-noise level required us to convert housing units to human population densities. To do this conversion, we first calculated the median of these 12 housing-unit ranges and then multiplied the median housing unit value by 2.83, which is the mean number of people living in a single house based on the 2000 U.S. Census. We then input these population densities into the EPA (1974a) formula:

$$\mathrm{DNL} = 22 + 10\ \log(\rho)$$

where ρ is the population density value we obtained from our transformation of SERGoM housing densities. Finally, we transformed the continuous values output from this model into integers and blocked DNL values into deciles for display. All spatial analyses were conducted with ArcGIS 9.3.1 software (ESRI 2009).

Transforming road effect distances to sound levels

We borrow an approach first used by Reijnen and Foppen (1995) and transform the threshold distances from roadways at which animal density or relative abundance were observed to decline into 24 h LEQ values using CadnaA software for two comprehensive studies (frogs, Eigenbrod et al. 2009; birds, Forman et al. 2002). In ArcGIS 9.3 we created a flat, straight road 10 km in length. Receiver points were referenced from the center-point of the road. We imported these shapefiles into CadnaA and assigned receiver height. We assigned a distance of 2.5 km as the search radius for the source and for the receivers, and as the maximum distance from source to receiver. We designated 0700–2159 as daytime and 2200–0659 as nighttime. We used the TNM method for road noise calculations. To calculate a 24-h LEQ, we used the DNL metric with no penalty for nighttime. To calculate the standard DNL we used the 10 dB nighttime penalty for the 2200–0659 h. For traffic scenarios with $\geq$10,000 cars per day we assigned the road type as national 3.02, and for scenarios with less than 10,000 cars per day we used the local 3.02 road designation. National 3.02 roads assume 15% truck traffic during the day and 25% at night. Local 3.02 roads assume 10% truck traffic during the day and 5% at night. Car traffic accounted for the remainder of the traffic mix. For all scenarios we applied a vehicle speed of 100 km/h. For each receiver site, CadnaA computed the average daily traffic noise in dBA.

Results and discussion

The models presented here show that noise exposure can be forecast across a broad range of spatial scales. This capability contrasts with the most decisive studies of noise impacts to date, which have focused on small scale behavioral and ecological responses. While more work at these levels of detail is crucial to clarify how noise affects wildlife, new studies must be developed to address ecological responses at landscape scales to unambiguously demonstrate that effective noise management must take the regional perspective of noise sources into account. Furthermore, our modeling efforts demonstrate significant increases in background sound levels for protected natural areas as described below for natural gas compressor, aircraft overflight and roadway noise sources. This underscores the importance of quantifying the benefits of noise reduction for biodiversity preservation and ecosystem health to maximize protected area effectiveness.

Noise from oil and gas removal, like most anthropogenic noise, has greater acoustic energy at low frequencies (Francis et al. 2009; Barber et al. 2010), and low frequency sounds attenuate more gradually by distance than do sounds at higher frequencies. Thus, noise from energy development near protected area borders is likely to propagate into those lands. In a model of potential natural gas well compressor noise near Mesa Verde National Park we estimated that noise propagated from 64 compressors elevated sound levels by a mean of 34.8 dBA 15 s LEQ above ambient conditions (Fig. 1). Along the eastern border of the park, adjacent to the greatest density of compressor locations, sound levels increased by a mean of 56.8 dBA 15 s LEQ above ambient conditions. Although our model is only hypothetical, because locations and operating status of oil and gas well compressors is proprietary

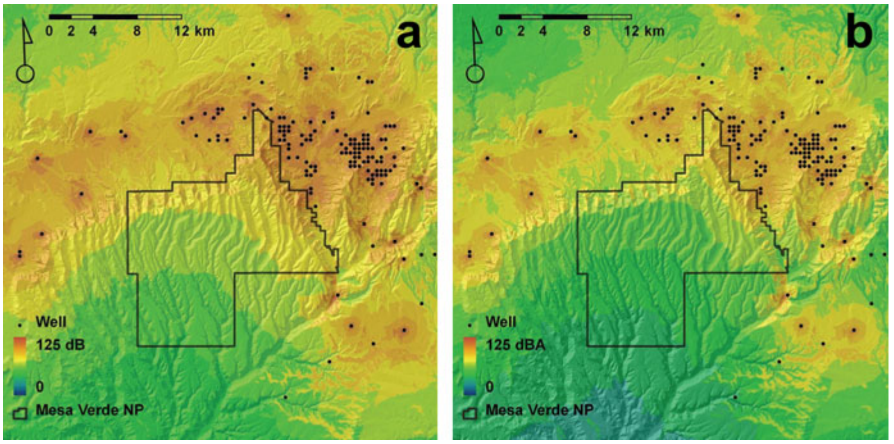

Fig. 1 Extent of potential noise propagation from oil and natural gas well compressors (*black dots*) in the San Juan Basin near Mesa Verde National Park in southwestern Colorado, USA. Results are summarized for the frequency spectrum from 0.125 to 2 kHz as **a** unweighted sound levels (dB) and **b** weighted sound levels (dBA) 15 s LEQ

information, it is based on empirical noise sources and ambient sound measurements. Thus it illustrates the possible extent of area affected by noise from energy extraction. Wind energy extraction also has the potential to significantly elevate sound levels and propagate low-frequency noise over natural areas and across protected area boundaries. Conditions with high winds aloft and little wind at the surface or wind turbines on ridgelines above areas that are topographically shadowed from natural wind noise, have the potential to experience significantly elevated background sound levels from operating turbines (K. Kaliski personal communication, also see Kaliski and Duncan 2008). Concomitant wildlife studies and wind farm noise mapping have not been performed.

High-flying jet noise can be heard, nearly everywhere in the continental US. Other sources of airway noise, including general aviation and air tours over protected natural areas, can also dominate the soundscape of some lands (Fig. 2b). Our models of two Boeing 757s landing and taking off at Jackson Hole Airport, WY, the only airport located in a National Park (Fig. 3 and Supplementary Animation 1), show that maximum levels received at some locations exceed 60 dBA 1 s LEQ. Most studies that have assessed the effects of low-flying aircraft on wildlife have performed behavioral or physiological studies and, if sound levels were quantified, reported acute exposures or event-based metrics (e.g., SEL or L_{max}; for definitions see Table 1). No study, to our knowledge, has quantified chronic exposure to elevated background sound levels from aircraft (for a review of acute studies see Efroymson and Sutter 2001).

Perhaps the most pervasive source of anthropogenic noise comes from roadway traffic. Based on bird road-effect distances, Forman (2000) estimated that 20% of the US is ecologically affected by the road network. Forman's estimate used road effect zones of approximately 300 m for rural roads, with a traffic volume of 10,000 cars per day and 810 m for urban roads, with a traffic volume of 50,000 cars per day. A recent meta-analysis extends this effect distance to 1 km for birds and 5 km for mammals (Benítez-López et al. 2010). Couple these distances with the rapid pace of increasing traffic in the US and the current area impacted by roads undoubtedly exceeds 20%.

Road impacts extend to protected natural areas, where traffic exposure can be severe (Fig. 2c). However, many roads in protected areas experience low to moderate traffic volumes. Our model of 1 day's traffic along the Going to the Sun road in Glacier National Park is based on just over 3,700 vehicles (Fig. 4 and Supplementary Animation 2). Even at this modest traffic volume, example locations receive significant sound levels: 41.8 dBA L_d at 500 m from the road and 37.5 dBA L_d at 1,000 m from the road. A comprehensive study of road traffic and grassland birds in Massachusetts found no effect on bird presence for this traffic volume, although regular breeding was reduced 400 m from the road (Forman et al. 2002). At traffic levels where we might not expect significant habitat degradation, fragmentation remains a concern. In a telemetry study of bat movements along a very busy roadway in Germany (84,000 vehicles/day), only 3 of 34 gleaning bats crossed the road (Kerth and Melber 2009). Gleaning

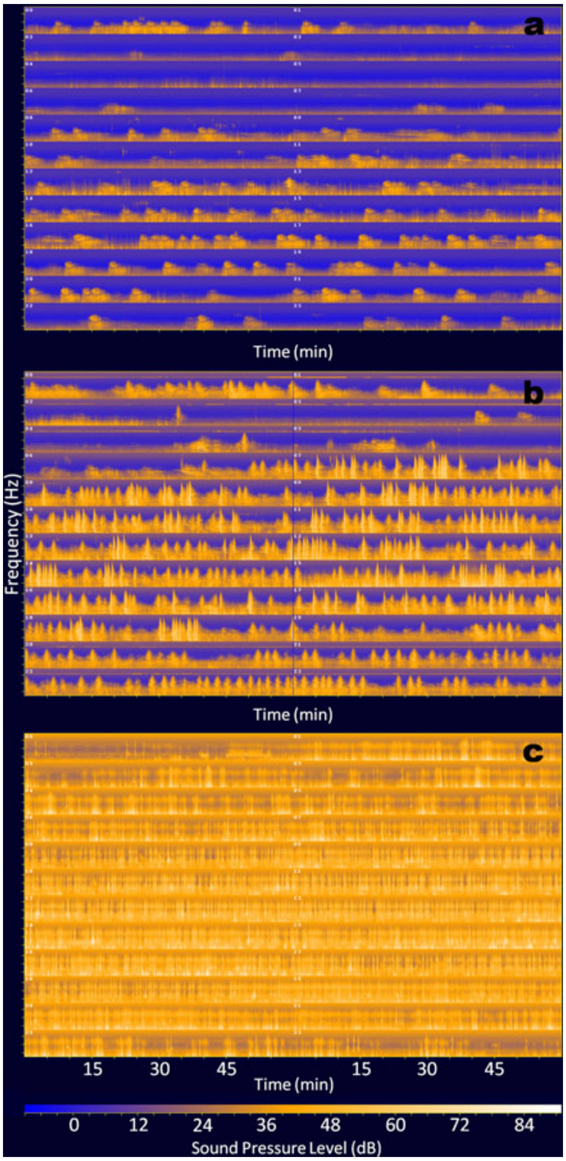

Fig. 2 24-h Spectrograms displaying high flying jet traffic in Grand Canyon National Park against an otherwise quiet background (**a**), helicopters in Lake Mead National Recreation Area (**b**), and road noise in Rocky Mountain National Park against a background already elevated by stream (rushing water) noise (**c**). Notice that despite this background the road noise significantly elevates the level. Each panel displays 1 s, 1/3 octave spectrum sound pressure levels, with 2 h represented horizontally in each of 12 rows. Frequency is shown on the y axis as a logarithmic scale extending from 12.5 Hz to 20 kHz, with the vertical midpoint of each row corresponding to 500 Hz. The z axis (*color*) describes sound pressure levels in dB (unweighted). (Color figure online)

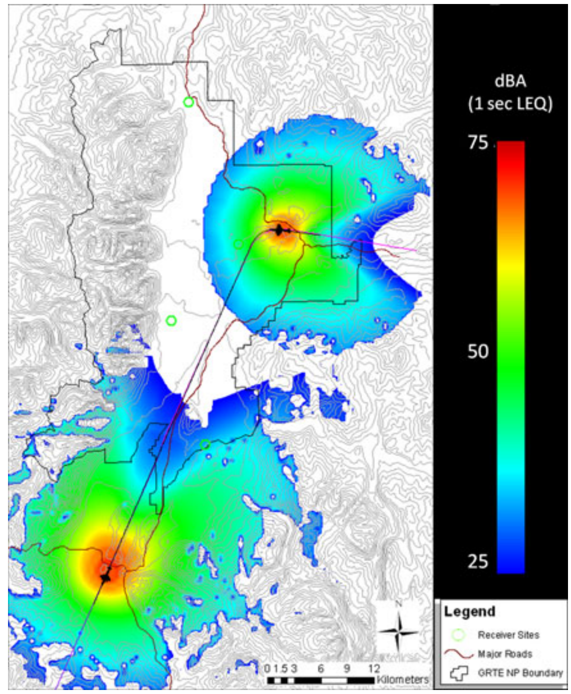

Fig. 3 Sound footprint (in dB 1 s LEQ) of two Boeing 757s landing and taking off at Jackson Hole Airport WY. See Supplementary Fig. 1 for receiver sites locations, sound levels and event exposure times. Time between flights reduced to 2 min for this figure in order to display both overflights in one panel

bats rely on passive listening to localize prey-generated sounds for hunting and this dependence on lower frequencies may explain why the same study found a non-gleaning bat much more likely to cross the road (5 of 6 individuals; Kerth and Melber 2009). Laboratory work has shown that gleaning bats prefer to hunt in quiet areas versus those with played-back road noise (Schaub et al. 2008) and these bats display reduced hunting efficiency in played back road noise (Siemers and Schaub 2010). Interestingly, the three gleaning bats that crossed the road in the previously mentioned study did so through an underpass (Kerth and Melber 2009). Sound levels were not recorded, but it is possible that a quiet 'acoustic tunnel' was audible from a distance and attracted the noise-sensitive bats. A study of forest-dependent bird movements across roads and rivers found these animals less likely to cross barriers when they were noisy (St. Clair 2003). Understanding the

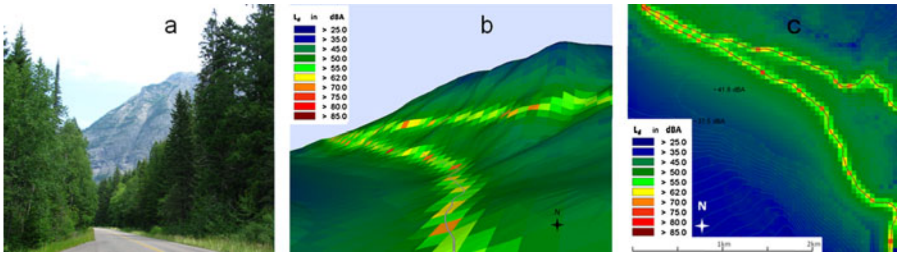

Fig. 4 One day's traffic noise (L_d in dBA) along the Going to the Sun Road in Glacier National Park (**a**), displayed in 3D (**b**) and 2D (**c**) perspectives. Two receiver locations are displayed in (**c**)

role sound plays in fragmentation will increase our ability to make underpasses and overpasses more effective at increasing landscape connectivity. Sound barriers along roadways are one way to reduce the footprint of traffic noise but these mitigation measures may reduce connectivity (Reijnen and Foppen 2006; Blickley and Paticelli 2011). Alternatively, overpasses and longer underground roadways have the potential to shelter large areas from noise exposure while concomitantly increasing connectivity. While beyond the scope of this contribution, the modeling tools we present can quantify the reduction in sound levels these structures would afford.

Unlike the software platforms we demonstrate here, our community-noise model is primarily intended to be a heuristic tool for understanding past and future changes in noise-pollution levels throughout the U.S. (Fig. 5). It is important to point out that the land-use change model we used operates only across private developable land; thus, these model predictions are likely to be a substantial underestimate of the area affected by anthropogenic noise. The model does not include public lands which, as we have demonstrated here, often experience high levels of anthropogenic noise. In addition, our approach is certainly an underestimate outside of populated areas on private land. Vast areas of the conterminous US, where our community-noise model shows low sound levels, experience significant oil and gas development (Fig. 5d), aircraft overflights (Fig. 5e), and traffic exposure (Fig. 5f). The relationship between population density and noise level used in this model, derived by the EPA in 1974, may be different today. Examining road traffic alone over the period of 1970–2007 reveals that, while population increased by one-third, traffic volume more than tripled to nearly 5 trillion vehicle kilometers per year (for references see Barber et al. 2010). However, the EPA formula was verified in one Maryland county in the late 1990s (Stewart et al. 1999). One advantage of our underlying land-use model is its inclusion of vacation/second homes, which are often occupied, but are not included in traditional census estimates. These homes frequently abut protected natural areas and degrade the conservation values of these lands by reducing connectedness (Wade and Theobald 2009) and promoting cross-border effects (Shafer 1999; Mcdonald et al. 2009) although it is unclear if the noise loads these developments produce are ecologically relevant.

The strength of this approach lies in our model's development from a predictive land-change model and suggests an important future direction for modeling noise at broad spatial scales. Integrating noise-pollution models with social-economic models that simulate changes in land use, energy production, and transportation levels will not only link anthropogenic noise with their sources, but also anticipated future changes in the magnitude and extent of these sources. In other words, the noise models of the future could be completely dynamic models that are capable of not only depicting noise pollution at broad spatial scales, but also at broad temporal scales. The forecast capability of such spatially and temporally dynamic noise-pollution models will allow more effective planning to control and mitigate the ecological effects of noise pollution.

Acoustic metrics and developing thresholds

While several recommendations have been made for human exposure to noise, no guidelines exist for wildlife and the habitats we share. The US EPA considers noise a pollutant and with the Noise Control Act of 1972 established the Office of Noise

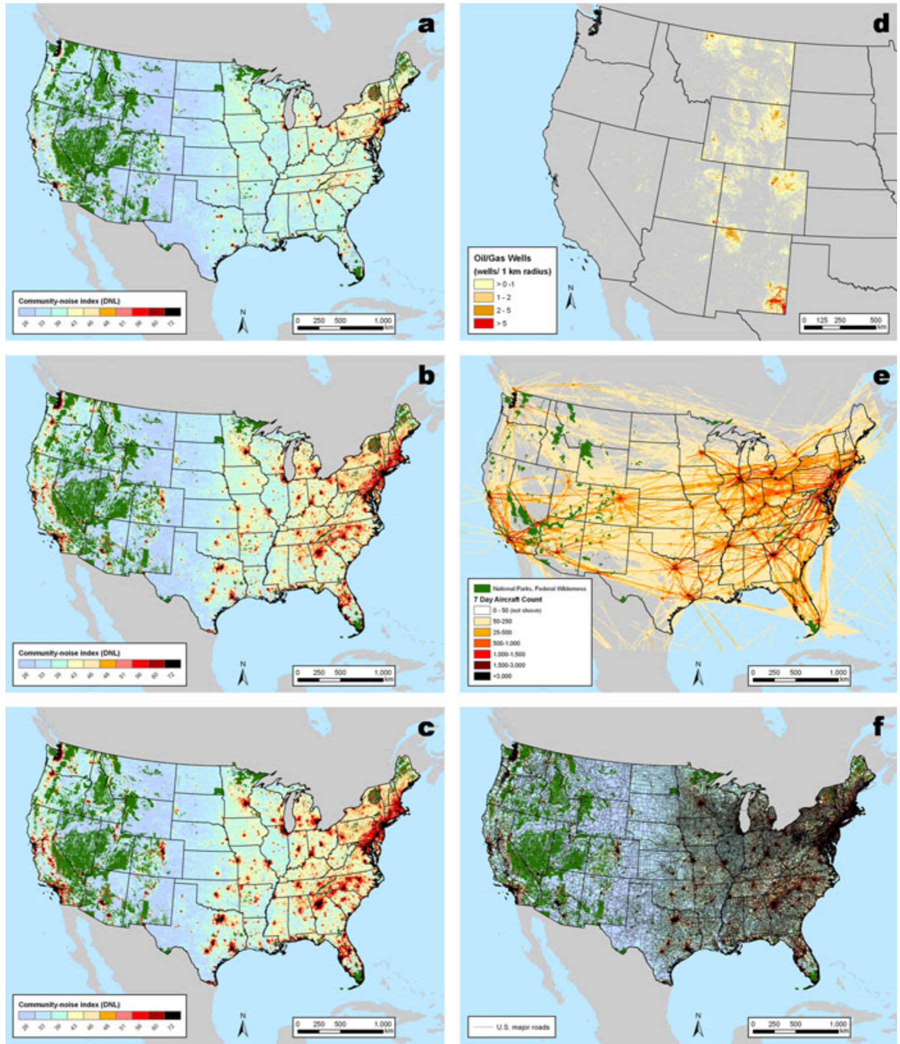

Fig. 5 A model of human community noise (DNL in dBA) for 1970 (**a**), 2010 (**b**), and 2050 (**c**). *Green* in a–c depicts protected natural areas. The extent of oil and gas wells in the western US (data from: http://sagemap.wr.usgs.gov/HumanFootprint.aspx; **d**), 7 days commercial aircraft overflights of the conterminous US (**e**), and major roads in the conterminous US (**f**) are presented to place the spatial extent of these noise sources, not included in our model of community noise, in context. For example, 83% of the land area in the US is within ~1 km of a road (Riitters and Wickham 2003). (Color figure online)

Abatement and Control. This office, defunded in 1982, recommended that to protect the human activities of speaking, working and sleeping, attempts should be made to limit exposure to a DNL of 55 dBA (1974b; see Table 1). The US Federal Interagency Committee on Noise does not recommend airport noise mitigations until a DNL of 65 dBA (FICON 1992). In 1999, the World Health Organization suggested that a sound level averaged over the daytime (L_d) of 50–55 dBA is moderately to seriously annoying and nighttime levels above 45 dBA L_n can disrupt sleep (WHO 1999). These guidelines for human exposure use metrics designed for chronic noise, while the majority of research on wildlife has focused on acute noise events. A comprehensive review of the extensive literature on wildlife response to low-altitude airplanes and helicopters reports a threshold of approximately 90–115 dBA L_{max} for a variety of animals, ranging from raptors to ungulates (Efroymson and Sutter 2001). These values tell us how loud an event needs to be before an animal flees. This can be useful information but it provides little inference for chronic exposures.

Road ecology studies near consistently busy highways provide estimates of the distances at which wildlife are affected by chronic noise exposure, although other factors clearly contribute (see "Introduction" section). None of these studies have directly measured sound levels but they do report traffic volumes and the distances where effects on animal density or relative abundance were observed. We borrow an approach first used by Reijnen and Foppen

(1995) and present distances, transformed into a 24 h LEQ using a sound propagation software platform (CadnaA) for two comprehensive studies (Table 3). We also calculated these values as DNL, for a more direct comparison with our community-noise model. The average 24 h LEQ threshold value for a frog community in Ontario was 43.6 dBA (Eigenbrod et al. 2009) and 38.3 dBA for a grassland bird community in Massachusetts (Forman et al. 2002). DNL values were about 3 dB higher. These values compare closely to similarly derived 24 h LEQ estimates for woodland birds (42–52 dBA) and grassland birds (47 dBA) in The Netherlands (Reijnen and Foppen 2006). An important caveat is these data were collected by human observers, often using auditory cues and may be biased. Interestingly a recent, controlled study that examined the ability of point counters to detect birds in compressor noise found that levels above 45 dBA (an average of three, 2 min LEQ measurements) impaired their ability to detect birds within 60 m (Ortega and Francis 2011).

These numbers are a starting point for a discussion on ecological thresholds for chronic noise exposure. Our community-noise model shows that between 1970 and 2010 the area of the US we modeled, under a DNL load of 46 or higher, grew from 7 to 18% (Fig. 6). However, based on our underlying housing density model, we forecast a modest increase in area at this sound level threshold for 2050: 21%. As mentioned earlier, this model does not take into account increasing transportation, energy extraction and other anthropogenic noise sources that occur either outside of developed areas or on public lands.

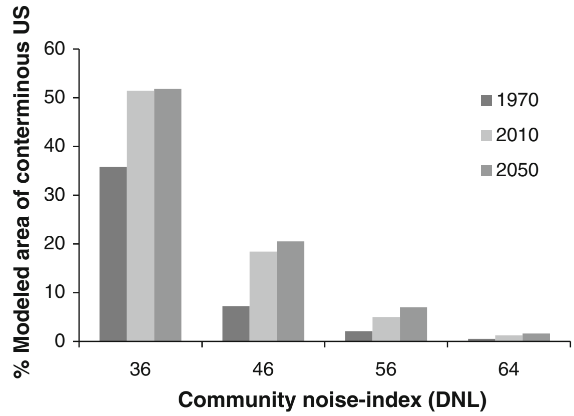

Fig. 6 The percentage of modeled land area in the conterminous US at various noise thresholds (DNL) for three decadal time points. Bin values of community noise levels reflect the categories of the underlying land-use change model, transformed into deciles

We feel it is critical that disturbance studies begin to quantify noise levels to inform management (also see Pater et al. 2009). Guidelines should be developed for both acute and chronic noise exposure. Sound levels reported should include the weighting filter and integration time used to compare across studies. Long term averages such as 24 h LEQ can be useful for chronic sources but intermittent sources should also be characterized by number and individual characteristics of disturbance events (e.g., SEL, duration) and by exceedance statistics (e.g., L_{10} and L_{90}; Table 1). DNL, which averages sound over a 24 h period with a 10 dB penalty for noise during between 2200 and 0700 h, is designed to represent disturbances to human populations (e.g., sleep) and is

Table 3 Road ecology threshold effect distances transformed into estimates of sound exposure thresholds

Study	Species	Traffic volume (vehicles/h ×10)	Effect/receiver distance (m)	DNL	24 h LEQ
Eigenbrod et al. (2009)[a,b]	Wood frog	18.3	679	43.6	40.6
	Western chorus frog	18.3	1,417	39.7	36.8
	Spring peeper	18.3	243	49.6	46.7
	American toad	18.3	198	51.1	48.2
	Gray treefrog	18.3	281	48.7	45.7
Forman et al. (2002)[b]	Average for grassland birds	8–15	400	38.5–43.8	36.8–40.5
		15–30	700	40.6–43.6	37.3–40.3
		30+	1,200	40.8	37.5

[a] Sound levels are calculated from the average of road effect threshold distances for 2006 and 2007

[b] Receiver height 0.25 m for frogs and 2 m for grassland birds

not the best metric for wildlife impacts. In fact, any metric that averages over long periods does not represent how animals process sound. The vertebrate auditory system integrates over milliseconds, not hours and when acute disturbance events are of concern, appropriate metrics should be selected (SEL or L_{max}). Even for disturbance events that are clearly multi-modal, like intermittent ORV activity, sound can provide a rigorous quantification. Ideally, acoustic data should be collected by calibrated sound level meters in 1 s, 1/3 octave bands (unweighted) paired with long-term audio recordings (such as MP3) archived with broad accessibility for computation of all metrics and comparative analyses. An excellent, low-budget alternative to include sound measurements in as many studies as possible is to use an MP3 recorder with a known relationship to a calibrated sound level meter. This approach can provide ±1 dB accuracy when acoustic energy is aggregated over time and bandwidth (e.g., an A-weighted, L_{50}; see Mennitt and Fristrup 2011).

Modeling sound in ecological studies

To understand the impact of ecological noise pollution we must begin to model soundscapes at population-relevant scales. The software tools we have demonstrated here point the way forward. Noise propagation models can facilitate understanding of complex management scenarios, and have great potential to inform decision-making processes (for an example process see Dumyahn and Pijanowski 2011). As demonstrated by the examples in this paper, however, no single modeling platform is currently available to simulate all types of anthropogenic noise sources in all types of environments.

The majority of software packages for modeling noise propagation were developed in response to European and U.S. regulations regarding human noise exposure in urban areas (e.g., CadnaA, LIMA, NoiseMap, and SoundPlan) or near highways and airports (e.g., INM, NMSim, and TNM). Each of these software packages has its strengths and limitations. For example, although commercially-available CadnaA supports a variety of input and output data formats and allows 3-dimensional and dynamic visual rendering of model results (Fig. 4; Supplemental Animation 2), modeling large areas with complex topography can be computationally intensive, even when multi-threading and 64-bit capabilities are enabled. On the other hand, NMSim is available in a free, public version, runs more efficiently, and has 3-dimensional modeling capability for aerial sources (Fig. 3; Supplemental Animation 1). However, NMSim does not account for the attenuation effects of vegetation (nor does the TNM version of CadnaA we used) and is not readily compatible with common geographic information system (GIS) data formats, although a pending release increases GIS compatibility. SPreAD-GIS is freely available, incorporates custom source spectra and ambient conditions, and calculates cumulative noise propagated from multiple simultaneous sources (Fig. 4), but at present simulating dynamic sources in SPreAD-GIS requires custom programming and modeling is limited to an upper limit of 2 kHz (Reed et al. 2011). A lack of field validation, calibration, and comparison of model predictions is a key limitation of all noise modeling packages (Kaliski et al. 2007).

Thoroughly examining the extent and magnitude of anthropogenic noise impacts on ecosystems will require accounting for a range of source types and ambient conditions and integrating noise propagated from multiple sources, at multiple scales. An integrated modeling platform should allow simulation of noise propagated from point and line sources, ground and aerial sources, and sources that are dynamic in space and time. To link more effectively with empirical field studies, models should accommodate custom source spectra and ambient sound conditions. To be useful for estimating the effects of noise on species other than humans, models should account for frequency-dependent attenuation effects and allow for alternate frequency-weighting and time-integration of model results.

Due to the emphasis on developing models of human noise exposure in urban ecosystems, a detailed understanding of factors affecting noise propagation in natural ecosystems (e.g., effects of vegetation composition and structure; Fang and Ling 2003) is a particularly important gap in current software capacity. Moreover, the high cost of many commercial packages may render them prohibitively expensive for ecological research and land management applications, and computational limitations prevent the development of models for large spatial extents. Ideally, an integrated anthropogenic noise modeling platform would be developed in an open source environment, encouraging diverse applications and allowing noise propagation algorithms to be refined over time with additional empirical information on source spectra and attenuation effects.

Conclusions

While landscape scale investigations of noise pollution are urgently needed, soundscape ecology must continue to simultaneously operate at small scales to determine the mechanisms through which noise exerts its ecological effects. It is clear that masking is a significant problem in elevated background sound levels (Barber et al. 2010) and continued research on hearing abilities in noise (e.g., critical ratios and upward spread of masking) is important. However, other forces besides masking appear to also play dominate roles. The finding that played back intermittent road noise elicits a much stronger avoidance reaction in sage grouse than continuous oil drilling noise (a better masking stimulus) is compelling evidence that other factors, such as stress, are critically important (Blickley et al. 2011). Furthermore, attentional and informational masking effects (Kidd et al. 2008) can impact information transfer even when classical masking paradigms do not apply (e.g., Chan et al. 2010).

Francis et al.'s (2009) finding that noise displaces most birds in a community but amplifies the reproductive success of others, likely through the absence of a major nest predator, highlights the importance of working at the community scale. Studying communities will prevent researchers from missing emergent properties of ecological systems. Sound propagation modeling has the potential to extend soundscape investigations to the largest scale; a scale where substantive arguments can be made for mitigation measures. Mitigation should be a priority in protected natural areas but developed areas should not be dismissed; as conservation depends on individual experience with biodiversity (Dunn et al. 2006).

Landscape ecologists have long understood the hierarchical nature of most ecological relationships and the value of linking mechanistic studies conducted at finer scales with contextual studies conducted at broader scales (O'Neill et al. 1986). The new field of soundscape ecology must embrace this approach.

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