Analysis of 7 Hidden Dangers in the New Federal Oil Tank Car Rule

1. Substandard tank cars carrying explosive crude oil will remain on the rails for a decade.

News accounts and agency statements give the impression that substandard tank cars used to carry Bakken crude oil will be off the rails soon. Not so. The Final Rule phases out DOT-111 and CPC-1232 tank cars over a **10-year period**, and crude oil can continue to be transported in these cars during that time. Final Rule, 80 Federal Register 26,648 (May 8, 2015).

Dangerous substances, like crude oil, present many hazards. In order to regulate their safe handling and transport, they are classified by "Dangerous Goods" category, then by "Packing Group" subcategory. The oil tank car phase-out is based on the type of tank car and the Packing Group classification of crude oil. Hazardous tank cars for oil in Packing Group I (described by the agency itself as "Great Danger") will be phased out of service by April 2020, Packing Group II ("Medium Danger") by July 2023, and Packing Group III ("Minor danger") by May 2025. At the May 1, 2015 press conference when the new standard was announced, Secretary Foxx suggested that crude oil—and Bakken crude in particular—would be phased out in the first 5 years, which is what Canada will require.

Nothing in the Final Rule's testing and classification scheme requires Bakken (or any other crude) to be classified as Packing Group I—the category with the quickest phase-out—and other types of crude can be classified as low as Packing Group III. The Lac Mégantic accident investigation uncovered the misclassification of Bakken crude as Packing Group III when it appropriately belonged in Packing Group II. Transportation Safety Board of Canada, *Railway Investigation Report R13D0054: Lac Mégantic, Quebec, 06 July 2013* (Aug. 2014) at §§ 1.19.2.1 & 2.8.3, at 51, 112-13; Final Rule at 26,658. The DOT Operation Safe Delivery similarly uncovered the misclassification of Bakken as Packing Group III when it belonged in Packing Group I or II. Federal Railroad Administration Letter to American Petroleum Institute (July 29, 2013). A lot of Bakken crude, including that involved in Lac Mégantic, can continue to be shipped in DOT-111s and unjacketed CPC-1232s for 8 more years.

The Final Rule is silent on when tar sands can no longer be shipped in DOT-111s and unjacketed CPC-1232s. While tar sands crude alone (also called bitumen) does not vaporize and ignite at low temperatures, tar sands crude must be diluted with other petroleum products so it can flow in and out of rail cars or pipelines. Diluted bitumen (dilbit) has a much lower temperature of vaporization and ignition. The accident and fire in Ontario on February 14, 2015 involved shipping dilbit in CPC-1232s, and the March 7, 2015 derailment and fire also in Ontario involved either dilbit or synthetic crude in CPC-1232s. Both are too explosive to be shipped in the hazardous tank cars.

2. The new tank car standards do not apply to every oil tank car.

The Final Rule has a huge loophole: It applies only to long trains carrying crude or ethanol, called, aptly, High-Hazard Flammable Trains. The new tank car and retrofit standards apply only to these High-Hazard Flammable Trains, defined as trains as having "a continuous block of

20 or more tank cars loaded with a flammable liquid or 35 or more tank cars loaded with a flammable liquid dispersed through a train." Thirty-four tank cars can carry approximately 1 million gallons of oil, meaning that the worst cars will still be on the rails, hauling explosive oil, and not bound by any of this Rule's safety provisions.

In contrast, Canada's new standards apply to every single tank car, and the National Transportation Safety Board recommended that the new U.S. standards apply to every single tank car carrying hazardous flammable liquids.

The proposed rule, released in July 2014, would have had the new tank car standards apply to trains with 20 or more cars loaded with flammable liquids. DOT increased the overall threshold to 35-car trains after the Association of American Railroads asserted that 20-60% of its 20-car blocks carrying hazardous fuels are in trains with a variety of cargoes (manifest trains), not a single product (unit trains). Final Regulatory Impact Analysis at 72.

3. Retrofitted tank cars are held to a less protective standard.

Even at the end of the phase-out, only about ¼ of the tank cars in these long unit trains will meet the new tank car standards. The rest—those cars built before October 1, 2015—will be retrofitted to a weaker standard. For these cars, the shells can remain 7/16" instead of 9/16" thick, even though DOT estimates that each additional 1/8" in shell thickness improves puncture resistance in an accident significantly. A shell 9/16" thick provides shell puncture velocity of 12.3 mph and head puncture velocity of 18.4 mph, while 7/16" thickness has a shell puncture velocity of 9.6 mph, and a head puncture velocity of 17.8 mph Regulatory Impact Analysis at 183; Final Rule at 26,672. The retrofitted cars also will not have top-fitting or rollover protections, even though top fittings represent 25% of the documented damage to tank cars in recent accidents. Proposed Rule, 79 Federal Register 45,055.

4. The oil industry will add new tank cars before old ones are taken off the rails.

Various industries claim that the phase-out of hazardous tank cars cannot happen faster because of time it takes to build new cars. The reality is that DOT is allowing the industry to build nearly 7000 new tank cars in 2016 to grow the crude-by-rail fleet rather than replace the hazardous tank cars. Regulatory Impact Analysis at 167. Approximately 20,000 jacketed CPC-1232s were added to the fleet during the rulemaking process, again to grow the fleet instead of to replace DOT-111s and unjacketed CPC-1232s.

5. Oil trains will not slow down in most populated areas.

The Final Rule codifies an existing voluntary railroad industry speed limit of 50 mph nationwide and 40 mph speed limit in high-threat urban areas for High-Hazard Flammable Trains, continuing the requirement imposed in April through an emergency order. The high-urban threat areas were designated several years ago based on the risks of terrorist attacks, not the particular risks of crude-by-rail. Only a few dozen cities around the nation are labeled high-threat urban areas, and they comprise only 7% of the nation's track. Final Rule at 26,690, 26,691.

Oil trains will continue to travel at high, risky speeds through many densely populated urban areas, rural communities, and alongside drinking water supplies, fishing streams, schools, and national parks. The new tank cars will still puncture at 12.3 mph, and the retrofit cars will puncture at 9.6 mph, and sustain head punctures at 18.4 and 17.8 mph, respectively, well below the reduced speed limit set for a small number of cities. Regulatory Impact Analysis at 183.

The National Transportation Safety Board urged using a more protective standard based on the radius of impacted people and resources near the tracks, not a pre-existing list of cities. Final Rule at 26,690.

6. The new rule leaves emergency responders and the public in the dark.

An emergency order issued in May 2014 required the railroads to disclose to State Emergency Response Centers the number and frequency of trains carrying 1 million gallons or more of Bakken crude through each county in the state, along with basic emergency response information. The proposed rule sought comment on whether the notification should extend to other hazardous fuels and to smaller volumes, something the National Transportation Safety Board supported. The Final Rule eliminates this requirement entirely and instead leaves it to emergency responders to request information from the railroads with no r requirement as to whether and how the railroads must respond. Final Rule at 26,714. A point of contact will be included in railroad security plans provided to state Homeland Security offices, which coordinate with the Department of Homeland Security on security issues, but there is no longer a requirement that even this contact information be made affirmatively available to emergency responders.

The Final Rule also takes a step back from letting the public know how many trains loaded with hazardous crude are coming through their communities. Under the May 2014 emergency order, the public obtained access to the train route and emergency preparedness information in most states under state public records laws. In place of this notification requirement, the Final Rule adds High Hazard Flammable Trains to a rule designed to address terrorist threats. Final Rule at 26,648, 26,711-12. Under that rule, railroads gather information and conduct route analysis, but the records are not required to be filed with state or federal agencies (which makes them subject to public access laws). State Homeland Security offices can obtain access on a need-to-know basis and have signed confidentiality agreements with the railroads. DOT sought to piggyback on these confidentiality agreements to preclude public access to the information that has generally been available to the public over the past year. Final Rule at 26,714.

In the Final Rule, DOT suggests that" "widespread access to security sensitive information could be used for criminal purposes when it comes to crude oil by rail transportation" and cites a vandalism incident in South Dakota. Final Rule at 26,712 & n.98. McClatchy has reported that the vandalized track held up as an example is not used for oil trains and in fact is out of service and overgrown with prairie grass.

7. DOT underestimates the harm from crude-by-rail disasters.

To determine whether new tank car standards, retrofits, speed limits, and other safety measures are cost-justified, DOT estimates the amount of harm likely to occur from crude-by-rail disasters. But in doing so, it uses data and makes assumptions that **underestimate the harm**.

- For what it calls low consequence events, DOT measures harm based on the amount of oil spilled. While called "low consequence," these events are like the rail derailments, oil spills, fires, explosions and evacuations that occurred in Lynchburg, Virginia, Casselton, North Dakota, Aliceville, Alabama, Galena, Illinois, and Carbon, West Virginia. DOT used U.S. accident data from 2006-2013 to calculate the amount of oil likely to be spilled in a mainline rail accident and come up with an average number of gallons per year. But much of this period preceded the crude-by-rail surge, which began in 2010 and resulted in a 4000% increase in car loads between 2008 and 2013. Not surprisingly, more oil spilled from the rails in the U.S. in 2013 than in 1975-2012 combined. In 2014, 6 crude and ethanol accidents occurred in the U.S. (Regulatory Impact Analysis at 301), and 3 have already taken place in 2015, in addition to two accidents that spilled ethanol. And if Canadian rail disasters were included, the number of accidents and amount of oil spilled would be even greater. As DOT itself noted, "[t]here is reason to believe that the Agency's estimate for incident severity may be low, because there is evidence that the number of flammable liquid cars per train has been growing over the past decade and may continue to grow." Regulatory Impact Analysis at 116.

- DOT appropriately recognizes that catastrophes like Lac Mégantic or worse could happen given the volume and hazards of crude-by-rail traffic and the defective tank cars used to transport crude. But then it low balls both the likelihood of such catastrophes and the potential severity of their harm. In terms of the frequency of what it calls high-consequence events, DOT uses a model that estimates between 1-5 events over 20 years. It runs the numbers using two other approaches that predict 5 and 12 high-consequence events over 20 years. Regulatory Impact Analysis at 105, 111, 118, 120. But then it uses 2 as the number for its regulatory impact analysis on the assumption that the reduced speed limits and rail routing analysis would reduce rail accidents. Regulatory Impact Analysis at 105-06. In some parts of the country, there are no alternative rail routes, but DOT never acknowledges this fact, and it overstates the degree to which speed limits will reduce the number of accidents as set out below. It also determined the incidence of high-consequence events per carload of crude and ethanol shipped in North America from 2000-2013 to develop an incidence of 1 high-consequence event per 5 million carloads, but it should have used a more recent data set that reflects higher volumes of crude shipped by rail and the higher crude-by-rail accident rates. Regulatory Impact Analysis at 118.

- In terms of its quantification of the harm from a high-consequence event, DOT uses Lac Mégantic. It acknowledges that the costs have risen dramatically from an earlier $1 billion estimate to an estimate of $2.7 billion, but then it uses $1.1 billion as its starting point with $658 million representing the non-fatality impacts. Regulatory Impact Analysis at 95, 98, 109. It then scales down this $658 million figure because the Lac Mégantic train was traveling at 65 mph, and the railroads have agreed to a 50 mph speed limit for High Hazard Flammable Trains (and 40 mph in high-threat urban areas). But

the Lac Mégantic train was intentionally traveling at any speed; it rolled down a hill and gained tremendous speed in a short period of time. Putting aside this fact, DOT scales down the $658 million to reflect a reduction in accident severity with reduced kinetic energy at lower speeds. Inexplicably, it does not use the speed limits in the Final; it uses an average speed of 41 mph from the largest 10% of U.S. accidents and a 35 mph speed in High Threat Urban Areas. Regulatory Impact Analysis at 99. Using this methodology it shrinks the $658 million non-fatality costs to between $263 million and $191 million. Regulatory Impact Analysis at 99-100.

- For fatalities from high-consequence events, DOT compares the population density in Lac Mégantic and rail segments within a square ½ kilometer along the tracks in the U.S. to predict the likelihood that a high-consequence event would occur in more heavily populated areas. It used a model that produced a range of results with a mean of $2.8 billion, $3.5 billion at the 80th percentile and $12.6 billion at the 95th percentile. Regulatory Impact Analysis at 110. Other models produced even higher estimates, such as a mean of $7.8 billion and 80th percentile of $11.1 billion. Regulatory Impact Analysis at 111. The Regulatory Impact Analysis observes that DOT's "simulation produces results that indicate the possibility of a single event that may exceed Lac Mégantic in total cost by an order of magnitude or more. Mitigating or preventing one such event might produce benefits that exceed the cost of the rule." Regulatory Impact Analysis at 118.

These projections played a pivotal role in the decisions DOT ultimately made. It decided which safety measures to require based on its cost-benefit analysis and rejected some, like a faster phase-out, the tougher tank car standards for retrofits, and lower speed limits outside high-threat urban areas on that basis. And yet its regulatory impact analysis states that "there is evidence that the true cost to society from crude oil spills may be 50 percent or even double, the Agency's central estimate" and "it is unlikely that any of these estimates capture the full comprehensive societal damages that result from these incidents." Regulatory Impact Analysis at 115.

Review

The costs of chronic noise exposure for terrestrial organisms

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Growth in transportation networks, resource extraction, motorized recreation and urban development is responsible for chronic noise exposure in most terrestrial areas, including remote wilderness sites. Increased noise levels reduce the distance and area over which acoustic signals can be perceived by animals. Here, we review a broad range of findings that indicate the potential severity of this threat to diverse taxa, and recent studies that document substantial changes in foraging and anti-predator behavior, reproductive success, density and community structure in response to noise. Effective management of protected areas must include noise assessment, and research is needed to further quantify the ecological consequences of chronic noise exposure in terrestrial environments.

Anthropogenic noise and acoustic masking

Habitat destruction and fragmentation are collectively the major cause of species extinctions [1,2]. Many current threats to ecological integrity and biodiversity transcend political and land management boundaries; climate change, altered atmospheric and hydrologic regimes and invasive species are prominent examples. Noise also knows no boundaries, and terrestrial environments are subject to substantial and largely uncontrolled degradation of opportunities to perceive natural sounds. Noise management is an emergent issue for protected lands, and a potential opportunity to improve the resilience of these areas to climate change and other forces less susceptible to immediate remediation.

Why is chronic noise exposure a significant threat to the integrity of terrestrial ecosystems? Noise inhibits perception of sounds, an effect called masking (see Glossary) [3]. Birds, primates, cetaceans and a sciurid rodent have been observed to shift their vocalizations to reduce the masking effects of noise [4–7]. However, compromised hearing affects more than acoustical communication. Comparative evolutionary patterns attest to the alerting function of hearing: (i) auditory organs evolved before the capacity to produce sounds intentionally [8], (ii) species commonly hear a broader range of sounds than they are capable of producing [9], (iii) vocal activity does not predict hearing performance across taxa [9,10], (iv) hearing continues to function in sleeping [11] and hibernating [12] animals; and (v) secondary loss of vision is more common than is loss of hearing [13].

Masking is a significant problem for the perception of adventitious sounds, such as footfalls and other byproducts of motion. These sounds are not intentionally produced and natural selection will typically favor individuals that minimize their production. The prevalence and characteristics of adventitious sounds have not been widely studied [14–16], although their role in interactions

Glossary

Alerting distance: the maximum distance at which a signal can be perceived. Alerting distance is pertinent in biological contexts where sounds are monitored to detect potential threats.
Atmospheric absorption: the part of transmission loss caused by conversion of acoustic energy into other forms of energy. Absorption coefficients increase with increasing frequency, and range from a few dB to hundreds of dB per kilometer within the spectrum of human audibility.
Audible: a signal that is perceptible to an attentive listener.
A-weighting: A method of summing sound energy across the frequency spectrum of sounds audible to humans. A-weighting approximates the inverse of a curve representing sound intensities that are perceived as equally loud (the 40 phon contour). It is a broadband index of loudness in humans in units of dB(A) or dBA. A-weighting also approximates the shapes of hearing threshold curves in birds [20].
Decibel (dB): a logarithmic measure of acoustic intensity, calculated by 10 $\log_{10}$(sound intensity/reference sound intensity). 0 dB approximates the lowest threshold of healthy human hearing, corresponding to an intensity of 10^{-12} Wm^{-2}. Example sound intensities: −20 dB, sound just audible to a bat, owl or fox; 10 dB, leaves rustling, quiet respiration; 60 dB, average human speaking voice; 80 dB, motorcycle at 15 m.
Frequency (Hz and kHz): for a periodic signal, the maximum number of times per second that a segment of the signal is duplicated. For a sinusoidal signal, the number of cycles (the number of pressure peaks) in one second (Hz). Frequency equals the speed of sound (~340 ms^{-1}) divided by wavelength.
Ground attenuation: the part of transmission loss caused by interaction of the propagating sound with the ground.
Listening area: the area of a circle whose radius is the alerting distance. Listening area is the same as the 'active space' of a vocalization, with a listener replacing the signaler as the focus, and is pertinent for organisms that are searching for sounds.
Masking: the amount or the process by which the threshold of detection for a sound is increased by the presence of the aggregate of other sounds.
Noticeable: a signal that attracts the attention of an organism whose focus is elsewhere.
Scattering loss: the part of transmission loss resulting from irregular reflection, diffraction and refraction of sound caused by physical inhomogeneities along the signal path.
Spectrum, power spectrum and spectral profile: the distribution of acoustic energy in relation to frequency. In graphical presentations, the spectrum is often plotted as sound intensity against sound frequency (Figure 1, main text).
1/3 octave spectrum: acoustic intensity measurements in a sequence of spectral bands that span 1/3 octave. The International Standards Organization defines 1/3rd octave bands used by most sound level meters (ISO 266, 1975). 1/3rd octave frequency bands approximate the auditory filter widths of the human peripheral auditory system.
Spreading loss: more rigorously termed divergence loss. The portion of transmission loss attributed to the divergence of sound energy, in accordance with the geometry of environmental sound propagation. Spherical spreading losses in dB equal $20*\log10(R/R_0)$, and result when the surface of the acoustic wavefront increases with the square of distance from the source.
White noise: noise with equal energy across the frequency spectrum.

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0169-5347/$ – see front matter © 2009 Elsevier Ltd. All rights reserved. doi:10.1016/j.tree.2009.08.002 Available online 15 September 2009

Box 1. Geographic extent of transportation noise in the USA

Transportation noise is a near ubiquitous component of the modern acoustical landscape. The method used here to estimate the geographic extent of airway (Figure Ia,b), railway (Figure Ic) and roadway (Figure Id) noise in the continental USA is calculated using the average human 'noticeability' of noise. Noise was deemed noticeable when the modeled noise intensity from transportation [in dB(A)] exceeded the expected noise intensity as predicted from population density [also dB(A)]. Although noticeability is a conservative metric of the geographic extent of transportation noise, this analysis only indicates the potential scope of the problem. How anthropogenic noise changes the temporal and spectral properties of naturally-occurring noise (Figure 1, main text) and the life histories of individual species will be crucial components of a more thorough analysis.

The maps in Figure I reflect the following calculations: (i) noise calculations are county-by-county for a typical daytime hour; (ii) county population density is transformed into background sound level using an EPA empirical formula (see Ref. [84]); higher density implies higher background sound levels; (iii) the geographic extent of transportation noise is determined by calculating the distance from the vehicle track at which the transportation noise falls below the background sound level, multiplying twice that distance by the length of the transportation corridor in the county (giving a noticeability area), and comparing that area with the total area in the county to compute the percentage land area affected. A low percentage noticeability can result if either the population density is high or the number of transportation segments is low in the county. This analysis indicates that transportation noise is audible above the background of other anthropogenic noise created by local communities in most counties in continental USA. See Ref. [84] for more details.

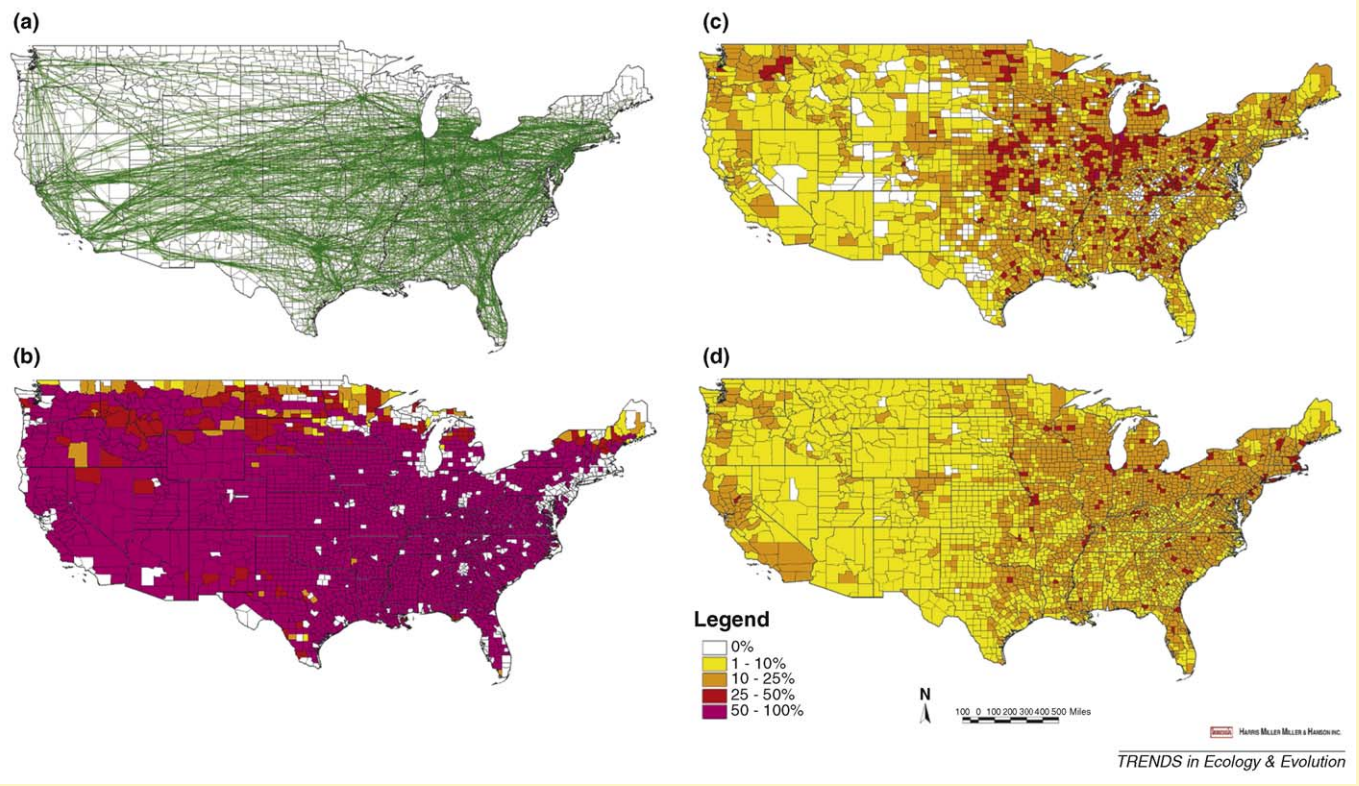

Figure I. Percent of US county areas in which transportation noise is noticeable. **(a)** Jet departures that occurred between 3 and 4 pm on Oct. 17, 2000, tracked to first destination. **(b)** Data from (a) were used to estimate the geographic extent of high altitude airway noise in the USA. The geographic extent of noise from railway and highway networks is depicted in **(c)** and **(d)**, respectively. The color-coded divisions (see legend; divisions increase in size as the percent increases) were chosen assuming that, as noticeability increases, so do estimate errors due to noticeability area overlap from different transportation segments. Adapted with permission from Ref. [84].

among predators and prey is unquestionable. In animal communication systems, both the sender and receiver can adapt to noise masking, but for adventitious sounds the burden falls on listeners.

Anthropogenic disturbance is known to alter animal behavioral patterns and lead to population declines [17,18]. However, animal responses probably depend upon the intensity of perceived threats rather than on the intensity of noise [19]. Deleterious physiological responses to noise exposure in humans and other animals include hearing loss [20], elevated stress hormone levels [21] and hypertension [22]. These responses begin to appear at exposure levels of 55–60 dB(A), levels that are restricted to relatively small areas close to noise sources [20].

The scale of potential impact

The most spatially extensive source of anthropogenic noise is transportation networks. Growth in transportation is increasing faster than is the human population. Between 1970 and 2007, the US population increased by approximately one third (http://www.census.gov/compendia/statab). Traffic on US roads nearly tripled, to almost 5 trillion vehicle kilometers per year (http://www.fhwa.dot.gov/ohim/tvtw/tvtpage.cfm). Several measures of aircraft traffic grew by a factor of three or more between 1981 and 2007 (http://www.bts.gov/programs/airline_information/air_carrier_traffic_statistics/airtraffic/annual/1981_present.html). Recent reviews of the effects of noise on marine mammals have identified similar trends in shipping noise (e.g. Refs [23,24]). In addition to transportation,

resource extraction and motorized recreation are spatially extensive sources of noise on public lands.

Systematic monitoring by the Natural Sounds Program of the US National Park Service (http://www.nature.nps.gov/naturalsounds) confirms the extent of noise intrusions. Noise is audible more than 25% of the hours between 7am and 10pm at more than half of the 55 sites in 14 National Parks that have been studied to date; more than a dozen sites have hourly noise audibility percentages exceeding 50% (NPS, unpublished). Remote wilderness areas are not immune, because air transportation noise is widespread, and high traffic corridors generate substantial noise increases on the ground (Box 1). For example, anthropogenic sound is audible at the Snow Flats site in Yosemite National Park nearly 70% of the time during peak traffic hours. Figure 1 shows that typical noise levels exceed natural ambient sound levels by an order of magnitude or more.

Roads are another pervasive source of noise: 83% of the land area of the continental US is within 1061 m of a road [25]. At this distance an average automobile [having a noise source level of 68 dB(A) measured at 15 m] will project a noise level of 20 dB(A). This exceeds the median natural levels of low frequency sound in most environments. Trucks and motorcycles will project substantially more noise: up to 40 dB(A) at 1 km. Box 2 provides a physical model of the reduced listening area that can be imposed by these louder background sound levels.

Acoustical ecology

Intentional communication, such as song, is the best studied component of the acoustical world, and these signals are often processed by multiple receivers. These communication networks enable female and male songbirds, for example, to assess multiple individuals simultaneously for mate choice, extra-pair copulations and rival assessment [26]. Acoustic masking resulting from increasing background sound levels will reduce the number of individuals that comprise these communication networks and have unknown consequences for reproductive processes [27].

Reproductive and territorial messages are not the only forms of acoustical communication that operate in a network. Social groups benefit by producing alarm calls to warn of approaching predators [28] and contact calls to maintain group cohesion [29]. A reduction in signal transmission distance created by anthropogenic noise might decrease the effectiveness of these social networks. The inability to hear just one of the alarm calling individuals can result in animals underestimating the urgency of their response [30].

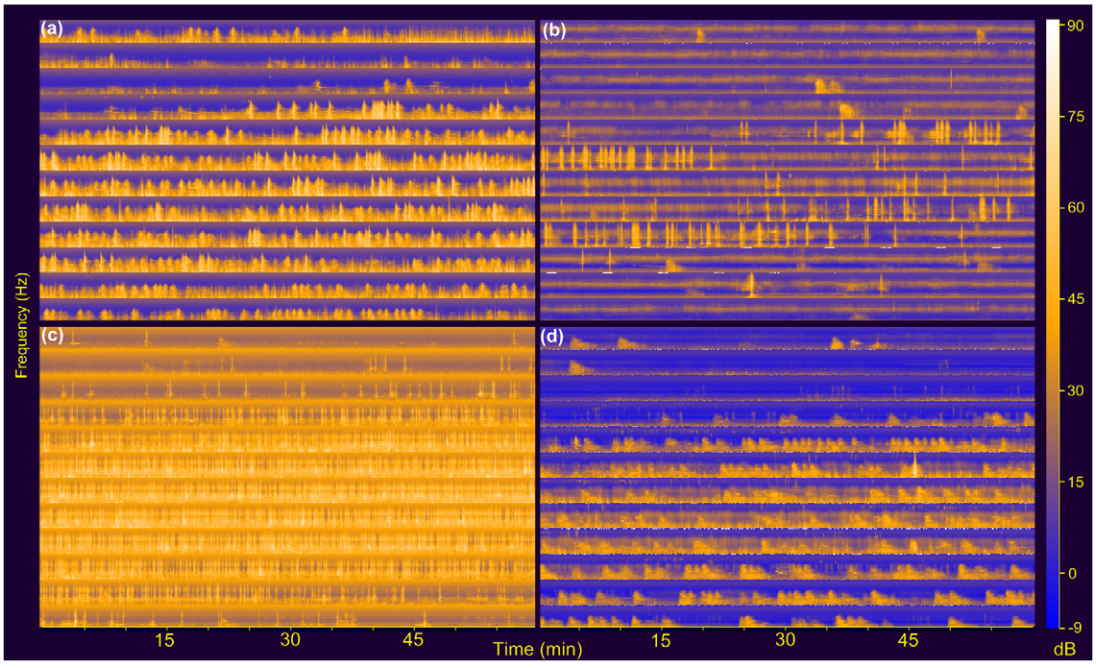

Figure 1. 24-hour spectrograms of Indian Pass in Lake Mead National Recreation Area **(a)**, Madison Junction in Yellowstone National Park **(b)**, Trail Ridge Road in Rocky Mountain National Park **(c)**, and Snow Flats in Yosemite National Park **(d)**. Each panel displays 1/3 octave spectrum sound pressure levels, with two hours represented horizontally in each of 12 rows. The first three rows in each panel represent the quietest hours of each day, from midnight to 6 am. Frequency is shown on the y axis as a logarithmic scale extending from 12.5 Hz to 20 kHz, with the vertical midpoint in each row corresponding to 500 Hz. The z axis (color) describes sound pressure levels in dB (unweighted); the color scaling used for all four panels is indicated by the color bar on the right hand edge. The lowest 1/3 octave levels are below 0 dB, the nominal threshold of human hearing. White dots at the upper edge of some rows in the panels on the right side denote missing seconds of data. Low-frequency, broadband signatures from high altitude jets are present in all four panels. Distinct examples are present just before 6 am in (a), near 12:45 am in (b) and (c), and between midnight and 12:30 am in (d). Fixed wing aircraft signatures (tonal contours with descending pitch) are present in (a) and (d), with a good example at 1:15 am in (d). Broadband signatures with very low frequency tonal components in (a) are due to low-altitude helicopters, that are prominent from ~7 am until 8 pm. Another prominent helicopter signature is at 11:30 am in (d). (b) illustrates snowmobile and snowcoach sounds recorded ~30 m from the West Entrance Road in Yellowstone. (c) illustrates traffic noise recorded 15 m from Trail Ridge Road in Rocky Mountain National Park, during a weekend event featuring high levels of motorcycle traffic. Background sound levels at the Rocky Mountain site were elevated by sounds from the nearby river.

Box 2. Physical model of reduced listening area in noise

The maximum detection distance of a signal decreases when noise elevates the masked hearing threshold. The masked detection distance: original detection distance ratio will be the same for all signals in the affected frequency band whose detection range is primarily limited by spreading losses. For an increase of N dB in background sound level, the detection distance ratio is: $k = 10^{-N/20}$. The corresponding fraction of original listening area is: $k = 10^{-N/10}$. A 1-dB increase in background sound level results in 89% of the original detection distance, and 79% of the original listening area. These formulae will overestimate the effects of masking on alerting distance and listening area for signals that travel far enough to incur significant absorptive and scattering losses. More detailed formulae would include terms that depend upon the original maximum range of detection.

Figure I illustrates the expected noise field of a road treated as a line source (equal energy generated per 10 m segment). An animal track is marked by ten circular features, that depict the listening area of a signal whose received level (expressed as a grey-scaled value for each possible source location) decreases with the inverse square of distance from the listener. The apparent shrinkage of the circles is due to masking by the increasingly dark background of sound projected from the road, just as noise would shrink the listening area. The circles span 9 dB in road noise level, in 1-dB steps from the quietest location (upper right) to the noisiest (at the crossing).

Masking effects are reduced with increasing spectral separation between noise and signal. The model presumes that the original conditions imposed masked hearing thresholds, so organisms that are limited by their hearing thresholds will not be as affected by masking. A diffuse noise source is illustrated, but the same results would be obtained if some spatial release from masking were possible, so long as the original conditions implied masked hearing thresholds (see Ref. [85] for a review of release strategies).

These measures of lost listening opportunity are most pertinent for chronic exposures. They imply substantial losses in auditory awareness for seemingly modest increases in noise exposure. Analyses of transportation noise impacts based on perceived loudness often assert that increases of up to three dB have negligible effects; this corresponds to a 50% loss of listening area.

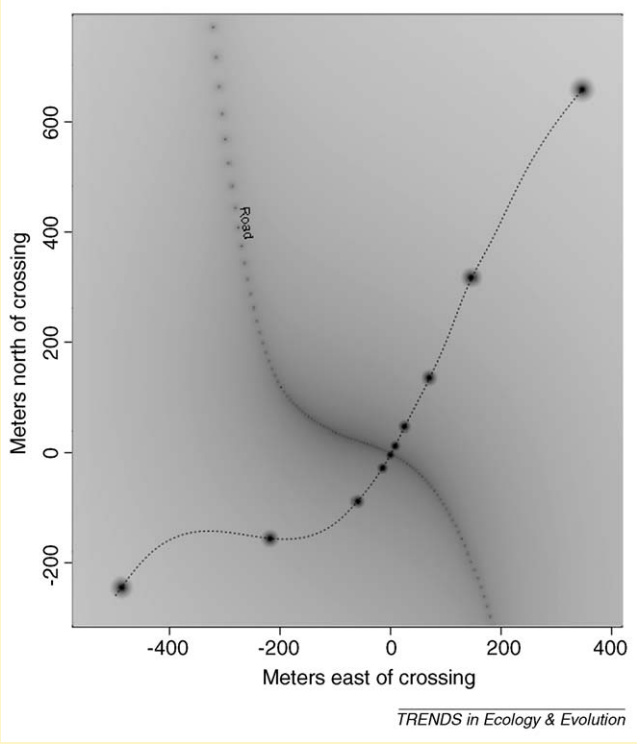

Figure I. A physical model of reduced listening area as an animal approaches a road.

Many vertebrate and invertebrate species are known to listen across species' boundaries to one another's sexual (e.g. Ref. [31]), alarm (e.g. Ref. [32]) and other vocalizations. Recent examples include gray squirrels, *Sciurus carolinensis*, listening in on the communication calls of blue jays, *Cyanocitta cristata*, to assess site-specific risks of cache pilfering [33]; and nocturnally migrating songbirds [34] and newts (Ref. [35] and Refs therein) using heterospecific calls to make habitat decisions. Reduced listening area imposed by increased sound levels is perhaps more likely to affect acoustical eavesdropping than to interfere with deliberate communication. The signaler is under no selective pressure to ensure successful communication to eavesdroppers and any masking compensation behaviors will be directed at the auditory system and position of the intended receiver rather than of the eavesdropper.

Acoustical communication and eavesdropping comprise most of the work in bioacoustics, but the parsimonious scenario for the evolution of hearing involves selection for auditory surveillance of the acoustical environment, with intentional communication evolving later [8]. Adventitious sounds are inadequately studied, in spite of their documented role in ecological interactions. Robins can use sound as the only cue to find buried worms [36]; a functional group of bats that capture prey off surfaces, gleaners, relies on prey-generated noises to localize their next meal [37]; barn owls (*Tyto alba*; [38]), marsh hawks (*Circus cyaneus*; [39]), and grey mouse lemurs (*Microcebus murinus*; [15] have been shown to use prey rustling sounds to detect and localize prey; big brown bats, *Eptesicus fuscus*, have the ability to use low-frequency insect flight sounds to identify insects and avoid protected prey [40]. In addition to prey localization, spectrally unstructured movement sounds are also used to detect predators. White-browed scrubwren (*Sericornis frontalis*) nestlings become silent when they hear the playback of footsteps of pied currawong, *Strepera graculina*, their major predator [41]; and tungara frogs, *Physalaemus pustulosus* avoid the wingbeat sounds of an approaching frog-eating bat, *Trachops cirrhosus* [42]. We are aware of only one study that has examined the role of adventitious sounds other than movement noises; African reed frogs, *Hyperolius nitidulus* flee from the sound of fire [43]. It is likely that other ecological sounds are functionally important to animals.

It is clear that the acoustical environment is not a collection of private conversations between signaler and receiver but an interconnected landscape of information networks and adventitious sounds; a landscape that we see as more connected with each year of investigation. It is for these reasons that the masking imposed by anthropogenic noise could have volatile and unpredictable consequences.

Separating anthropogenic disturbance from noise impacts

Recent research has reinforced decades of work [44,45] showing that human activities associated with high levels

of anthropogenic noise modify animal ecology: for example, the species richness of nocturnal primates, small ungulates and carnivores is significantly reduced within ~ 30 m of roads in Africa [46]; anuran species richness in Ottawa, Canada is negatively correlated with traffic density [47]; aircraft overflights disturb behavior and alter time budgets in harlequin ducks (*Histrionicus histrionicus*; [48]) and mountain goats (*Oreamnos americanus*; [49]); snowmobiles and off-road vehicles change ungulate vigilance behavior and space use, although no evidence yet links these responses to population consequences [50,51]; songbirds show greater nest desertion and abandonment, but reduced predation, within 100 m of off-road vehicle trails [52]; and both greater sage-grouse (*Centrocercus urophasianus*; [53]) and mule deer (*Odocoileus hemionus*; [54]) are significantly more likely to select habitat away from noise-producing oil and gas developments. Thus, based on these studies alone, it seems clear that activities associated with high levels of anthropogenic noise can re-structure animal communities; but, because none of these studies, nor the disturbance literature in general, isolates noise from other possible forces, the independent contribution of anthropogenic noise to these effects is ambiguous.

Other evidence also implicates quiet, human-powered activities, such as hiking and skiing, in habitat degradation. For example, a paired comparison of 28 land preserves in northern California that varied substantially in the number of non-motorized recreationists showed a fivefold decline in the density of native carnivores in heavily used sites [55]. Further evidence from the Alps indicates that outdoor winter sports reduce alpine black grouse, *Tetrao tetrix* populations [17] and data from the UK link primarily quiet, non-motorized recreation to reduced woodlark, *Lullula arborea* populations [18]. A recent meta-analysis of ungulate flight responses to human disturbance showed that humans on foot produced stronger behavioral reactions than did motorized disturbance [45]. These studies strengthen a detailed foundational literature suggesting that anthropogenic disturbance events are perceived by animals as predation risk, regardless of the associated noise levels. Disturbance evokes anti-predator behaviors, interferes with other activities that enhance fitness and, as the studies above illustrate, can lead to population decline [44]. Although increased levels of noise associated with the same disturbance type appear to accentuate some animal responses (e.g. Refs [44,48]), it is difficult to distinguish reactions that reflect increasingly compromised sensory awareness from reactions that treat greater noise intensity as an indicator of greater risk.

To understand the functional importance of intact acoustical environments for animals, experimental and statistical designs must control for the influence of other stimuli. Numerous studies implicating noise as a problem for animals have reported reduced bird densities near roadways (reviewed in Ref. [56]). An extensive study conducted in the Netherlands found that 26 of 43 (60%) woodland bird species showed reduced numbers near roads [57]. This research, similar to most road ecology work, could not isolate noise from other possible factors associated with transportation corridors (e.g. road mortality, visual disturbance, chemical pollution, habitat fragmentation, increased predation and invasive species along edges). However, these effects extended for over a mile into the forest, implicating noise as one of the most potent forces driving road effects [58]. Later work, with a smaller sample size, confirmed these results and contributed a significant finding: birds with higher frequency calls were less likely to avoid roadways than birds with lower frequency calls [59]. Coupled with the mounting evidence that several animals shift their call frequencies in anthropogenic noise [4–7], these data are suggestive of a masking mechanism.

A good first step towards disentangling disturbance from noise effects is exemplified by small mammal translocation work performed across roadways that varied greatly in traffic amount. The densities of white-footed mice, *Peromyscus leucopus* and eastern chipmunks *Tamias striatus* were not lower near roads and both species were significantly less likely to cross a road than cover the same distance away from roads, but traffic volume (and noise level) had no influence on this finding [60]. Thus, for these species, the influence of the road surface itself appears to outweigh the independent contributions of direct mortality and noise.

Recent findings on the effects of anthropogenic noise

Two research groups have used oil and gas fields as 'natural experiments' to isolate the effects of noise from other confounding variables. Researchers in Canada's boreal forest studied songbirds near noisy compressor stations [75–90 dB(A) at the source, 24 hrs a day, 365 days a year] and nearly identical (and much quieter) well pads. Both of these installations were situated in two to four ha clearings with dirt access roads that were rarely used. This design allowed for control of edge effects and other confounding factors that hinder interpretation of road impact studies. The findings from this system include reduced pairing success and significantly more first time breeders near loud compressor stations in ovenbirds (*Seiurus aurocapilla*; [61]), and a one-third reduction in overall passerine bird density [62]. Low territory quality in loud sites might explain the age structuring of this ovenbird population and, if so, implicates background sound level as an important habitat characteristic. In addition to the field data above, weakened avian pair preference in high levels of noise has been shown experimentally in the lab [63]. These data suggest masking of communication calls as a possible underlying mechanism; however the reduced effectiveness of territorial defense songs, reduced auditory awareness of approaching predators (see Box 3 for a discussion of the foraging/vigilance tradeoff in noise), or reduced capacity to detect acoustic cues in foraging, cannot be excluded as explanations of the results.

A second research group, working within natural gas fields in north-west New Mexico, US, used pinyon, *Pinus edulis*-juniper, *Juniperus osteosperma* woodlands adjacent to compressor stations as treatment sites and woodlands adjacent to gas wells lacking noise-producing compressors as quiet control sites [64]. The researchers were able to turn off the loud compressor stations to perform bird counts, relieving the need to adjust for detection differences in noise [62]. This group found reduced nesting species richness but in contrast to Ref.

Box 3. Do rising background sound levels alter vigilance behavior?

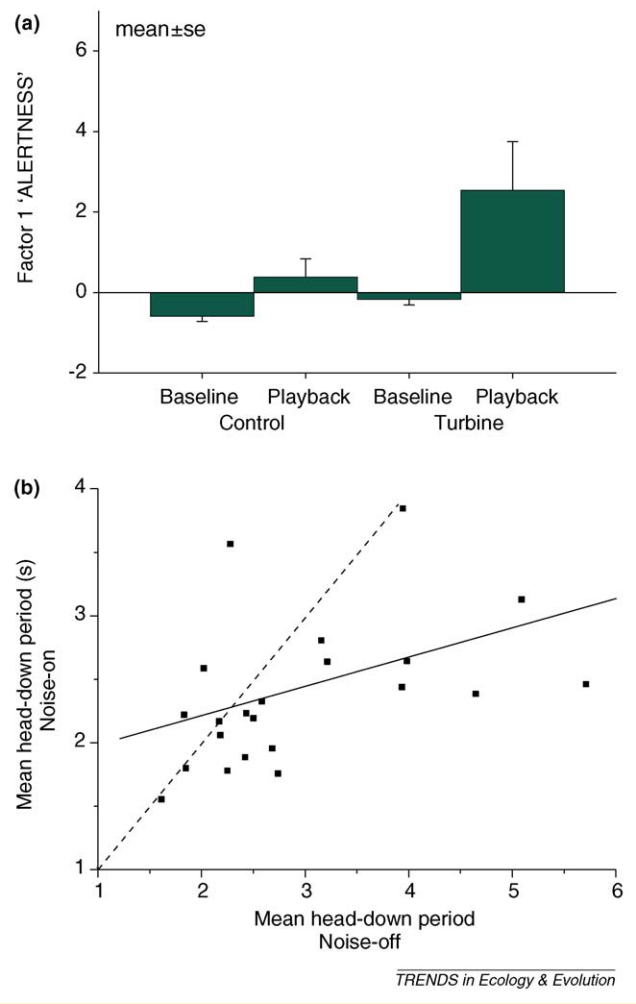

Figure I. Examples of increased vigilance behavior in noise. (**a**) When predator-elicited alarm calls are played back to California ground squirrels (*Spermophilus beecheyi*), adults show a greater increase in vigilance behavior at a site heavily impacted by anthropogenic noise, under power-generating wind turbines, than in a quiet control site [67]. (**b**) Further work on vigilance behaviors in noise comes from controlled, laboratory work with foraging chaffinches (*Fringilla coelebs*). In noise these birds decrease the interval between head-up scanning bouts, which results in fewer pecks and, thus, reduced food intake [90]. Dots depict the mean head-down period for each individual with and without white noise playback. Points below the dashed line (slope = 1) document individuals who increased scanning effort in noise. The solid regression line shows that the general trend was a more dramatic response from individuals with the lowest scanning effort. (a) adapted and (b) reproduced, with permission from Refs [67] and [90], respectively.

Predation risk and human disturbance increase vigilance behaviors (e.g. Refs [50,86]), at a cost to foraging efficiency [87,88]. Habitat features that influence predator detection, such as vegetation height, predict predation risk [88]. If background sound level interferes with the ability of an animal to detect predators, risk can increase. Do animals perceive background sound level as a habitat characteristic that predicts predation risk? Two recent studies document increased vigilance behaviors in high levels of noise (Figure I). It seems probable that these increased anti-predator behaviors are the result of attempted visual compensation for lost auditory awareness. Evidence from ungulates near roads suggests this is the case (Figure II); however, the distinct contributions of traffic as perceived threat and traffic noise as a sensory obstacle are confounded in road studies. Experimental research with birds and mammals suggests that lost visual awareness owing to habitat obstruction reduces food-searching bouts and increases vigilance (reviewed in Ref. [89]). Although no evidence exists (but see Ref. [64]), if noise shifts the spatial distribution of foraging effort, then plant growth and seed dispersal could also be altered.

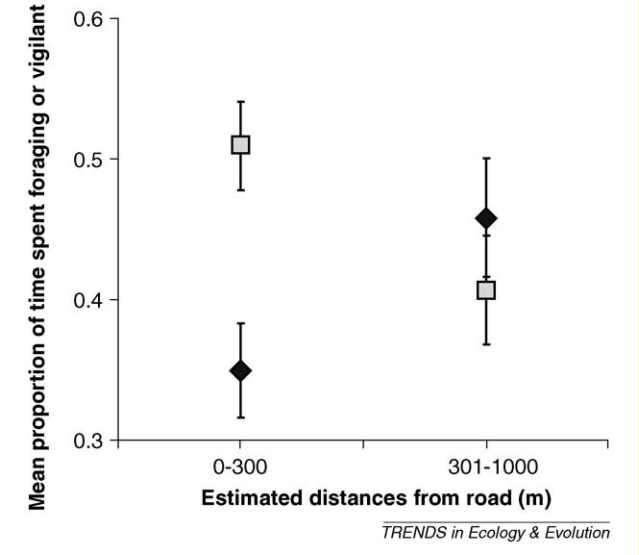

Figure II. An example of the foraging–vigilance tradeoff. Pronghorn (*Antilocapra Americana*) spend more time being vigilant (squares) and less time foraging (diamonds) within 300 meters of a road [86]. Future experiments should attempt to separate the roles of traffic as perceived threat and reduced auditory awareness on these tradeoffs. Reproduced, with permission, from Ref. [86].

[62], no reduction in overall nesting density. Unexpectedly, nest success was higher and predation levels lower in loud sites (also see Ref. [52]). The change in bird communities between loud and quiet sites appears to be driven by site preference; the response to noise ranged from positive to negative, with most responses being negative (e.g. three species nested only in loud sites and 14 species nested only in quiet, control sites). However, given the change in community structure, habitat selection based on background sound level is not the only interpretation of these data, as birds might be using cues of reduced competition pressure or predation risk to make habitat decisions [64]. The major nest predator in the study area, the western scrub jay, *Aphelocoma californica*, was significantly more likely to occupy quiet sites, which might explain the nest predation data [64]. It is probable that nest predators rely heavily on acoustic cues to find their prey. The study also found that the two bird species most strongly associated with control sites produce low-frequency communication calls. These observations suggest masking as an explanatory factor for these observed patterns. This work highlights the potential complexity of the relationship between noise exposure and the structure and function of ecological systems.

Adjusting temporal, spectral, intensity and redundancy characteristics of acoustic signals to reduce masking by noise has been demonstrated in six vertebrate orders [4–7,65]. These shifts have been documented in a variety

of signal types: begging calls of bird chicks [66], alarm signals in ground squirrels [67], contact calls of primates [68], echolocation cries of bats [65] and sexual communication signals in birds, cetaceans and anurans [4–7,69]. Vocal adjustment probably comes at a cost to both energy balance and information transfer; however, no study has addressed receivers.

Masking also affects the ability of animals to use sound for spatial orientation. When traffic noise is played back to grey treefrog, *Hyla chrysoscelis* females as they attempt to localize male calls, they take longer to do so and are significantly less successful in correctly orienting to the male signal [70]. Similar studies with the European tree frog, *Hyla arborea* show decreased calling activity in played back traffic noise [71]. *H. arborea* individuals appear to be unable to adjust the frequency or duration of their calls to increase signal transmission, even at very high noise intensities (88 dB(A), [71]); although other frogs have been shown to slightly shift call frequencies upward in response to anthropogenic noise [69]. These are particularly salient points. It is likely that some species are unable to adjust the structure of their sounds to cope with noise even within the same group of organisms. These differences in vocal adaptability could partially explain why some species do well in loud environments and others do poorly [5,7,72].

Under many conditions, animals will minimize their movement sounds. For example, mice preferentially select quieter substrates on which to move [73]. Adventitious sounds of insects walking contain appreciable energy at higher frequencies (main energy ~3–30 kHz [16]) and are thus unlikely to be fully masked by most anthropogenic noise (<2 kHz [4–7]) but the spectral profile near many noise sources contains significant energy at higher frequencies (e.g. Ref [74]). Foundational work with owls and bats has shown that frequencies between approximately three and eight kHz are crucial for passive sound localization accuracy [38,75]. In fact, a recent laboratory study demonstrated that gleaning bats avoided hunting in areas with played back road noise that contained energy within this spectral band ([74]; Box 4).

Adapting to a louder world

Animals have been under constant selective pressure to distinguish pertinent sounds from background noise. Two

Box 4. Effects of acoustic masking on acoustically specialized predators

Laboratory work has demonstrated that gleaning bats (who use prey-generated sounds to capture terrestrial prey; Figure Ia) avoid noise when foraging (Figure Ib). Interestingly, treefrogs, a favorite prey of some neotropical gleaning bats, tend to call from sites with high ambient noise levels (primarily from waterfalls) and bats prefer frog calls played back in quieter locations [91]. Extinction risk in bats correlates with low wing aspect ratios (a high cost and low wing-loading morphology), a trait that all gleaning bats share [92]. A recent analysis indicates that urbanization most strongly impacts bats with these wing shapes [93]. However, low wing aspect ratio is also correlated with habitat specialization, edge intolerance and low mobility [92,93], obscuring the links between a gleaning lifestyle, louder background sound levels and extinction risk as urbanization reduces available habitat, fragments landscapes and generates noise concomitantly.

A radio-tag study showed that a gleaning bat, *Myotis bechsteinii*, was less likely to cross a roadway (three of 34 individuals) than was a sympatric open-space foraging bat, *Barbastella barbastellus* (five out of six individuals; [94]), implicating noise as a fragmenting agent for some bats. The latter species hunts flying insects using echolocation (an auditory behavior that uses ultrasonic signals above the spectrum of anthropogenic noise) [94]. Similar findings suggest acoustically mediated foragers are at risk: terrestrial insectivores were the only avian ecological guild to avoid road construction in the Amazon [95] and human-altered landscapes limited provisioning rates of saw-whet owls [96]. That these animals plausibly rely on sound for hunting might not be coincidental.

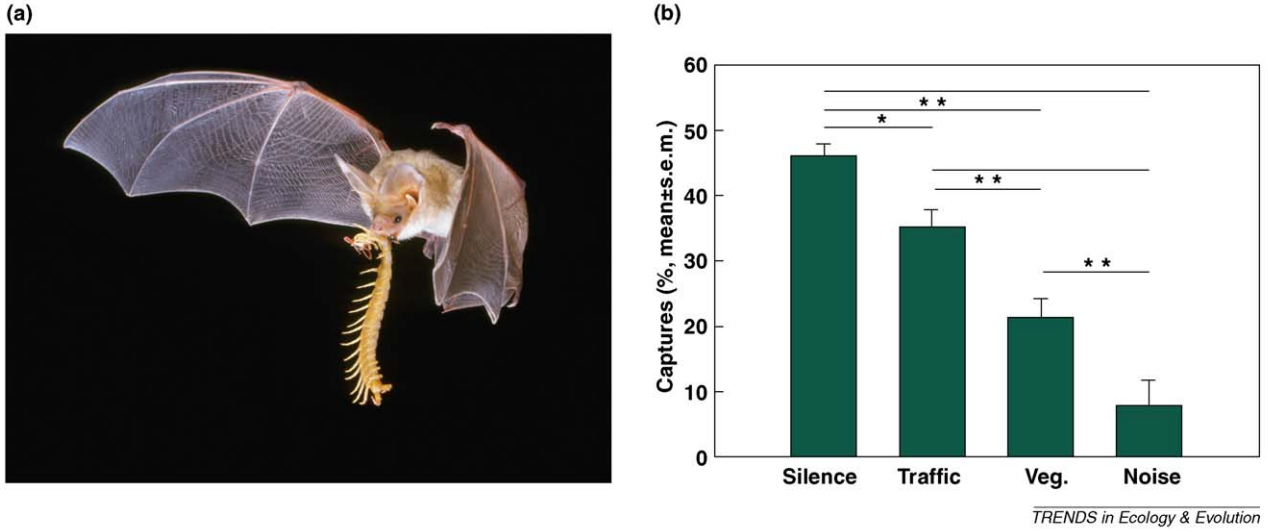

Figure I. Gleaning bats avoid hunting in noise. The pallid bat, *Antrozous pallidus* **(a)**, relies upon prey-generated movement sounds to localize its terrestrial prey. Recent work demonstrates that another gleaning bat, the greater mouse-eared bat, *Myotis myotis*, avoids foraging in noise [74]. **(b)** A laboratory two-compartment choice experiment showed that this bat preferred to forage in the compartment with played-back silence versus the compartment with played-back traffic, wind-blown vegetation or white noise. This pattern held true whether the percentage of flight time, compartment entering events, the first 25 captures per session or overall capture percentage were compared across silent and noise playback compartments. Asteriks indicate the results of post repeated-measure ANOVA, paired t-tests (**$P<0.01$, *$P<0.05$, N=7 bats). The differences between noise types (traffic, vegetation and white noise) probably reflect increased spectral overlap between prey-generated movement sounds and the spectral profile of the noise. Reproduced with permission from Scott Altenbach (a) and Ref. [74] (b).

Box 5. Outstanding questions

- Multiple studies with birds have demonstrated signal shifts in anthropogenic noise that does not substantially overlap in frequency with the birds' song [4–7,72]. To what extent does low-frequency anthropogenic noise inhibit perception of higher frequency signals? Mammals appear more prone to the 'upward spread' of masking than do birds [85,97]. Noise commonly elevates low frequency ambient sound levels by 40 dB or more, so small amounts of spectral 'leakage' can be significant. Laboratory studies should be complimented by field studies that can identify the potential for informational or attentional effects [98]. This work should use anthropogenic noise profiles and not rely on artificial white noise as a surrogate. Furthermore, we suggest that future studies measure or model sound levels (both signal and background) at the position of the animal receiver (*sensu* Ref. [23]).
- What roles do behavioral and cognitive masking release mechanisms [85] have in modifying the capacity of free-ranging animals to detect and identify significant sounds? Only one study has examined the masked hearing thresholds of natural vocal signals in anthropogenic noise [97]. This work found that thresholds for discrimination between calls of the same bird species were consistently higher than were detection thresholds for the same calls [97]. This highlights the lack of knowledge concerning top-down cognitive constraints on signal processing in noise. Can noise divide attention and reduce task accuracy by forcing the processing of multiple streams of auditory information simultaneously [99]?
- Do animals exploit the temporal patterning of anthropogenic noise pollution (see Ref. [4])? Alternatively, what constitutes a chronic exposure and how does this vary in relation to diel activity schedules?
- Does noise amplify the barrier effects of fragmenting agents, such as roads [94,100]?
- What routes (exaptation, behavioral compensation, phenotypic plasticity and/or contemporary evolution) lead to successful tolerance of loud environments?
- What role does audition have in vigilance behaviors? Are visually mediated predators at an advantage in loud environments when prey animals rely upon acoustical predator detection?
- Do animals directly perceive background sound level as a habitat characteristic related to predation risk? A noise increase of 3 dB(A) is often identified as 'just perceptible' for humans, and an increase of 10 dB(A) as a doubling of perceived loudness. These correspond to 30% and 90% reductions in alerting distance, respectively. Do organisms assess reduced alerting distance by monitoring other acoustical signals?

examples include penguin communication systems being shaped by wind and colony noise [76] and frog systems driven to ultrasonic frequencies by stream noise [77]. A meta-analysis of the acoustic adaptation hypothesis for birdsong (the idea that signals are adapted to maximize propagation through the local habitat) found only weak evidence for this claim [78]. Physiological constraints and selective forces from eavesdropping could explain this weak relationship [78], in addition to variation of noise profiles across nominally similar habitat types (e.g. insect noise, [79]).

Phenotypic plasticity enables one adaptation to anthropogenic noise. The open-ended song learning documented in great tits, *Parus major* helps explain the consistent song shifts observed in all ten comparisons between urban and rural populations [72]. Contemporary evolution (fewer than a few hundred generations) has now been quantified in several systems [80] and we might anticipate similar microevolutionary changes in many species with rapid generation times that consistently experience acoustical environments dominated by noise, particularly in increasingly fragmented landscapes.

Perhaps the greatest predictors of the ability of a given species to succeed in a louder world will be the degree of temporal and spectral overlap of biologically crucial signals with anthropogenic noise (Figure 1), and their flexibility to compensate with other sensory modalities (e.g. vision) when auditory cues are masked. Given known sensory biases in learning [81], many animals will be constrained in their ability to shift from acoustical inputs to other sensory cues for dynamic control of complex behavioral sequences.

Conclusions and recommendations

The constraints on signal reception imposed by background sound level have a long history of being researched in bioacoustics, and it is increasingly clear that these constraints underlie crucial issues for conservation biology. Questions have been raised about the value of behavioral studies for conservation practice (for a review see Ref [82]), but ethological studies of auditory awareness and the consequences of degraded listening opportunities are essential to understanding the mechanisms underlying ecological responses to anthropogenic noise (Box 5). These studies are more challenging to execute than observation of salient behavioral responses to acute noise events, but they offer opportunities to explore fundamental questions regarding auditory perception in natural and disturbed contexts.

Chronic noise exposure is widespread. Taken individually, many of the papers cited here offer suggestive but inconclusive evidence that masking is substantially altering many ecosystems. Taken collectively, the preponderance of evidence argues for immediate action to manage noise in protected natural areas. Advances in instrumentation and methods are needed to expand research and monitoring capabilities. Explicit experimental manipulations should become an integral part of future adaptive management plans to decisively identify the most effective and efficient methods that reconcile human activities with resource management objectives [83].

The costs of noise must be understood in relation to other anthropogenic forces, to ensure effective mitigation and efficient realization of environmental goals. Noise pollution exacerbates the problems posed by habitat fragmentation and wildlife responses to human presence; therefore, highly fragmented or heavily visited locations are priority candidates for noise management. Noise management might also offer a relatively rapid tool to improve the resilience of protected lands to some of the stresses imposed by climate change. Shuttle buses and other specialized mass transit systems, such as those used at Zion and Denali National Parks, offer promising alternatives for visitor access that enable resource managers to exert better control over the timing, spatial distribution, and intensity of both noise and human disturbance. Quieting protected areas is a prudent precaution in the face of sweeping environmental changes, and a powerful affirmation of the wilderness values that inspired their creation.

Acknowledgements

We thank colleagues in the Natural Sounds Program and the NPS personnel who support monitoring projects and foster the development of acoustical resource management. We thank Lisa Angeloni, Casey Brown, Amanda Hardy, Chris Burdett and Olivier Devineau for productive discussion; Charlotte Formichella, Damon Joyce and Emma Lynch for assistance in creating figures; Shan Burson for Yellowstone monitoring data and Jessie Rinella for Lake Mead monitoring data; the researchers who shared their data and three anonymous reviewers who improved this review. The preparation of this manuscript was supported by a Postdoctoral Fellowship to J.R.B. from the National Park Service/ Natural Sounds Program and Colorado State University.

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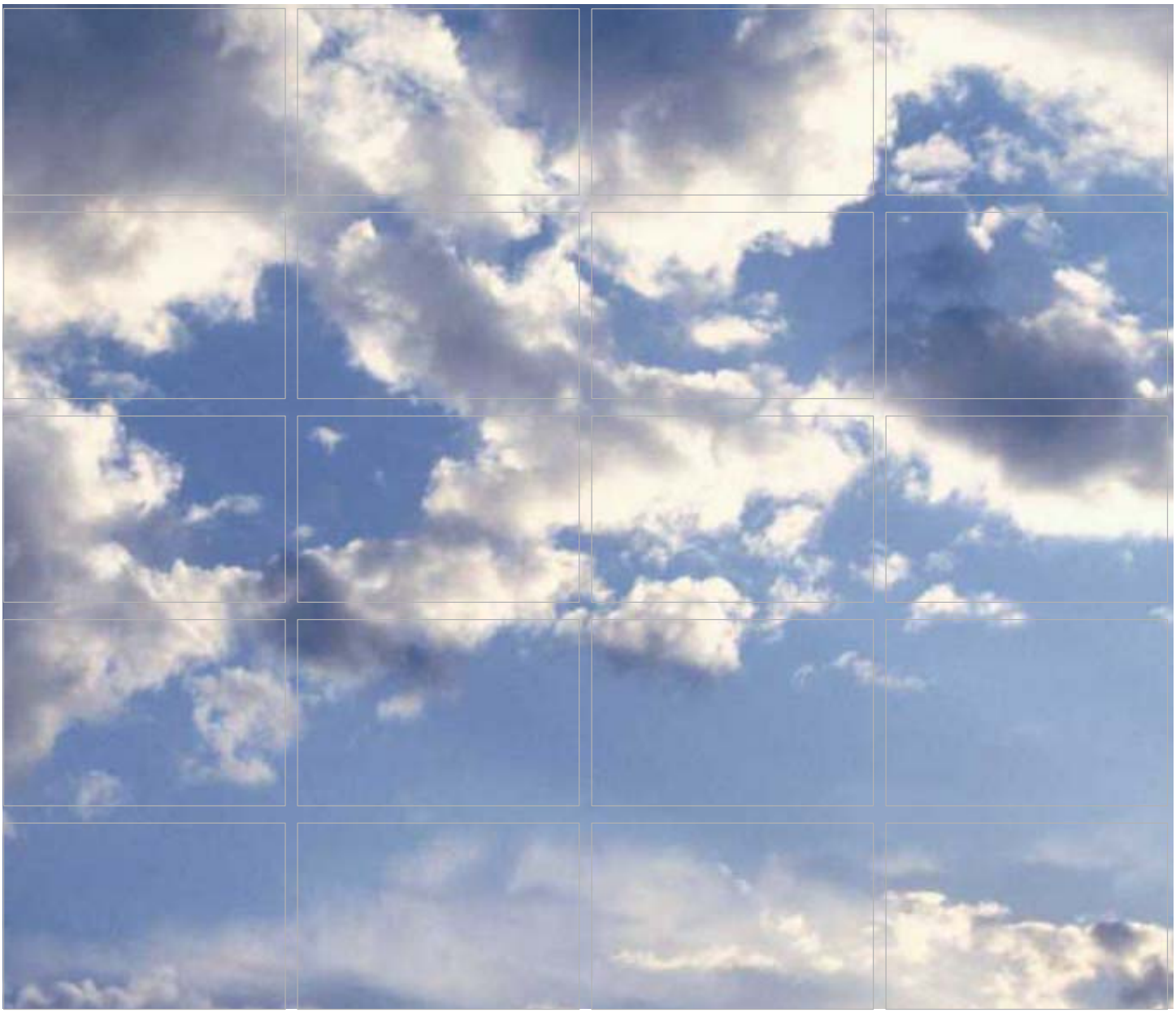

Bay Area Petroleum Refinery Mercury Air Emissions, Deposition, and Fate

Prepared for:
Western States
Petroleum Association

WSPA Member Facilities
San Francisco Bay Area, California

June 2009

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Project No. 0032209

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Acknowledgments

We gratefully acknowledge Vic Vickery and
Jason Buhlman of Brown and Caldwell for their contributions to this study. They provided sampling equipment, field services and a wealth of experience in mercury sampling methods necessary for this effort. We also thank Bob Brunette and Amy Dahl of Frontier Geosciences for their analytical support.

ERM and Flow Science

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LIST OF ACRONYMS

3-D	three-dimensional
BAAQMD	San Francisco Bay Area Air Quality Management District
CAF	Combustion-AF
CARB	California Air Resources Board
CEC	California Energy Commission
cm	centimeter
CTM	chemical transport model
CVAFS	cold vapor atomic fluorescence spectrometry
DAF	Digestion-AF
DEM	Digital Elevation Model
FCCU	fluidized catalytic cracking unit
g	gram
h	hour
Hg	mercury
HgCl2	mercury chloride
HgS	mercury sulfide
ISCST	Industrial Source Complex Short Term (Model)
kg	kilogram
km	kilometer
L	liter
LCI	Land Cover Institute (USGS)
m	meter
MDN	Mercury Deposition Network
µg	microgram

MM5	Mesoscale Meteorological 5-KM Gridded Data
Mmol	10^3 kg-mole
MMSCF	million standard cubic feet
NADP	National Atmospheric Deposition Program
ng	nanogram
NPDES	National Pollutant Discharge Elimination System
nmole	nanomole
NTN	National Trends Network
O^3	ozone
QA/QC	quality assurance and quality control
ppb	parts per billion
RFG	refinery fuel gas
RPD	relative percent difference
RWQCB	Regional Water Quality Control Board
scm	standard cubic meter
SFB	San Francisco Bay
TMDL	total maximum daily load
TRI	Toxics Release Inventory
UNEP	United Nations Environment Programme
US	United States
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
WSPA	Western States Petroleum Association
yr	year

1.0 INTRODUCTION AND SUMMARY

At the request of the Western States Petroleum Association (WSPA), ERM-West, Inc. (ERM) and Flow Science Incorporated (Flow Science) have prepared this report with mercury sampling results obtained by Brown and Caldwell, to respond to Regional Water Control Board (RWQCB) requests to the five Bay Area refineries to complete a technical report on the fate of mercury in crude oil in the San Francisco Bay (SFB) Area Petroleum Refineries.

Various analyses have been performed to comply with the RWQCB requests. These analyses are listed below and are discussed in detail throughout the report.

- Measured airborne emissions of mercury from the combustion of refinery fuel gas and from process vent stacks;
- Conducted atmospheric dispersion modeling for calculating mercury deposition;
- Prepared Synthesis of Results;
- Estimated the mercury mass balance from refinery operations; and
- Performed literature review of mercury emissions and relevant studies in the SFB Area and the United States (US) for context.

Mercury occurs naturally in the environment from enriched soil, forest fires, oceans, volcanoes, and geothermal areas. It is also released from human activities such as mining, industrial activities including cement production, municipal waste incineration, chlor-alkali production, and fuel combustion. Model-based estimates indicate that human induced recycling, natural emissions, and new point-sources each account for approximately one-third of total inputs of atmospheric mercury (Lindberg et al. 2007). Tables 1-1 and 1-2 compare mercury contributions to the SFB by source type, and through direct deposition and from watershed transport, respectively.

Table 1-1 summarizes mercury contributions from various source types in the SFB Area. The primary contributors are the erosion of buried sediments and runoff from the Central Valley watersheds. These sources account for 36% and 38%, respectively.

Table 1-1 Estimated Annual Mercury Contribution (in kg/yr) from Various Sources to the San Francisco Bay and Contributions from the Bay Area Refineries

Contributing Sources	Contribution (kg/yr)[1]	Percent of Total Mercury Contribution
Mercury Contributions – Bay Area Refineries		
SFB Area Refineries	1	< 0.1%
Mercury Contribution from Area Wide Source Types		
Wastewater (municipal & industrial) Discharges	18	2%
Non-urban Storm Water Runoff	25	2%
Direct Atmospheric Deposition into the SFB	27	2%
Guadalupe River Watershed (mining legacy)	92	8%
Urban Storm Water Runoff	160	13%
Central Valley Watershed	440	36%
Erosion of buried sediments	460	38%
Total	***1220***	***100%***

[1] Source: San Francisco Bay RWQCB total maximum daily load (TMDL), 2006.

Table 1-2 provides a comparison of mercury contributions from SFB Area refineries estimated by this study via direct atmospheric deposition to watersheds and urban runoff for both the total and from the SFB Area refineries.

Table 1-2 Estimated Annual Mercury Contribution (kg/yr) from Direct and Indirect Atmospheric Deposition into the San Francisco Bay

Source	Contribution (kg/yr)	SBF Area Refinery Percent Contribution
Direct Deposition		
Direct Atmospheric Deposition into the SFB (total)[1]	27.00	-
Direct Atmospheric Deposition into Bay Waters from SFB Area Refineries	*0.19*	***0.7%***
Indirect Mercury Contributions		
Non-Urban Storm Water Runoff (total) [1]	25.00	-
Urban Storm Water Runoff (total)[1]	160.00	-
Contribution from Surrounding Watersheds from Refineries	*0.82*	***0.44%***

[1] Source: San Francisco Bay RWQCB total maximum daily load (TMDL), 2006.

Conclusions from the mercury fate and transport analysis indicate that the SFB Area refineries contribute minimal mercury to the Bay. Modeled mercury deposition rates from SFB Area refineries, when compared with reported estimates of mercury deposition at locations within and around the SFB Area, are equivalent to approximately 0.5% to 5% of both wet and dry deposition flux estimates.

Assuming all of the mercury from SFB Area refineries that is deposited to the watershed area draining directly to the SFB reaches the Bay, the contribution from the SFB Area refineries is estimated to be equivalent to approximately 5.6% of the mercury contributed to the SFB by

municipal and industrial discharges, approximately 3.7% of the mercury deposited directly from atmospheric deposition occurring over the SFB, and approximately 0.2% of the mercury contributed to the SFB by the Central Valley Watershed (see Section 3, Table 3-3). Thus, estimated mercury loadings from atmospheric deposition of mercury emitted by SFB Area refineries are a small fraction of total mercury loadings from other sources in the SFB region.

2.0 ATMOSPHERIC DISPERSION AND DEPOSITION MODELING

An atmospheric dispersion and deposition modeling analysis was performed to simulate the downwind transport and deposition rate of mercury due to airborne emissions of mercury from the five SFB Area refineries.

2.1 Modeling Emissions

Mercury mass emission rates used in the atmospheric dispersion and deposition modeling were derived from direct measurement of mercury in refinery fuel gas and in process vent stacks. A final review of data quality on all sample results was conducted; the review confirmed the validity of the data for modeling and reporting. A discussion of the measurements and emissions calculation methodology used to determine the mercury mass emission rates is included in Appendix A.

Emissions from the combustion of fuel gas (combustible gas generated during petroleum refining) were calculated assuming that 100% of the mass of mercury contained in the refinery fuel gas burned will be emitted into the atmosphere. Emissions from each of the refineries were calculated using refinery-specific fuel gas usage from April 2007 through March 2008. Mercury emissions due to the combustion of refinery fuel gas were distributed between various stack locations based upon:

- Specific refinery operations and information provided by refinery staff;
- Combustion source size and permitted limits found in Title V Permits; and
- Source type.

Depending on this refinery-specific information, ERM minimized the number of stacks or point sources by co-locating stacks that may have similar release characteristics since this would not significantly impact total mercury deposition rates on a regional scale. Total calculated mercury emissions from combustion of fuel gas at the five SFB Area refineries are 1.14 kg/yr.

In addition, process vent stacks were directly measured for mercury content and the average stack mercury mass rate that was attributed to the process stacks at each refinery. Total calculated mercury emissions from process stacks at the five SFB Area refineries are 17.96 kg/yr.

2.2 Modeling Methodology

2.2.1 Model Selection

After the consideration of various dispersion and deposition models, it was determined that the CALPUFF modeling system would be most appropriate for the analysis of the SFB and surrounding watersheds, which encompass a large area. The modeled area, or modeling domain, was based on the drainage basins located within the SFB Area, which drain into the SFB. The modeling domain for this analysis is illustrated in Figure 2-1. CALPUFF was chosen because (1) it is a regulatory agency-approved model; (2) it can incorporate both wet and dry deposition; (3) it uses a regional meteorological data set; and (4) it is capable of predicting pollutant concentrations and deposition rates on both a local and regional scale. The United States Environmental Protection Agency (USEPA)-approved CALPUFF modeling system is the

state-0f-the art system that has been used by the CARB for modeling exercises in the Bay Area and throughout the state.

The components of the CALPUFF modeling system include:

- CALMET;
- CALPUFF; and
- CALPOST.

The CALPUFF modeling system can simulate dispersion in multiple layers with space-varying, three-dimensional (3-D) meteorological data fields, (created by CALMET) to more accurately simulate pollution dispersion. This is especially true in locations such as the SFB Area, where the terrain varies and there are many microclimates. CALPUFF also utilizes mixing height, surface characteristics such as land use and land cover, and dispersion properties that are also included as part of the CALMET output file. The CALMET processing also accounts for the land/water interface the meteorological changes that occur between water and land surfaces through the development of independent dispersive parameters of the wind and atmospheric data. It does so by using the land use data, and overwater and overland characteristics to define specific surface roughness, albedo, and bowan ratio, which are used to define dispersive conditions within the wind field.

Using ERM internal software similar to CALPOST, post-processing was performed to compile specific results tables and summary reports of the deposition values created by the CALPUFF model.

2.2.2 Meteorological Data Development

A meteorological data set was developed using CALMET. Recently, the California Air Resources Board (CARB) completed a modeling analysis to assess whether sources of air pollutants potentially contributing to regional haze may impact visibility in Federal Class I Areas (National Parks, Wilderness Areas, National Monuments, etc.). This "Regional Haze" analysis was performed using the CALPUFF modeling system and a 3-D wind field data set created by CALMET. ERM requested and received the various CALMET input and output files from CARB and has reviewed the specific characteristics, inputs, and output computer files. The primary datasets used for preprocessing included Mesoscale Meteorological 5-KM Gridded Data (MM5), United States Geological Survey (USGS) Land Cover Institute (USGS-LCI) digitized regional land-use data, and USGS Digital Elevation Model (DEM) terrain data. The MM5 data have wind vectors, speeds, temperatures, precipitation, and boundary layer heights at 5-kilometer (km) intervals. The CALMET preprocessor was first used by CARB to regrid the data to user specified grid spacing (in this case 4 km) by interpolating the MM5 data at each 4 km grid point. This "regridded" MM5 data was then incorporated with data from 279 surface stations, the digitized surface and terrain data, and the digitized land use data to modify the flow vectors (both speed and direction) based upon the angle and height of the opposing terrain.

Due to the extremely large size of the raw MM5 data sets, CARB supplied ERM with the initial "regrid" of the MM5 data at intervals of 4 km for 2002. CARB also provided ERM with the preprocessed land use and terrain data, and the preprocessed surface station data and a CALMET input file. The combined file size is over 500 gigabytes.

ERM's initial review of the final data set (files used by CARB as input to CALPUFF) revealed that the processed CALMET data set did not include the precipitation data, which are required

for calculating the wet deposition of mercury. Therefore, the data were reprocessed to include the missing precipitation data.

2.2.3 Modeling Assumptions and Input Parameters

Numerous model inputs and control parameters were used for the air dispersion and deposition modeling. Tables 2-1 through 2-3 provide information on the specific parameters and model input assumptions used in the analysis. Table 2-1 provides the general technical model inputs. Table 2-2 provides specific mercury speciation information and Table 2-3 provides chemical parameters used by the CALPUFF model for calculating deposition velocities.

Table 2-1 CALPUFF Technical Modeling Inputs

Model Input Description	Default Input used?	Model Input Used?
Length of run	No-Default	8760 hours
Technical Options		
Vertical distribution	Yes	Gaussian
Terrain adjustment method	Yes	Partial plume path adjustment
Subgrid-scale complex terrain flag	Yes	Not modeled
Near-field puffs modeled as elongated slugs	Yes	No
Transitional plume rise modeled	Yes	Yes, transitional rise computed
Stack tip downwash	Yes	Yes, use stack tip downwash
Method used to simulate building downwash	Yes	ISC method
Vertical wind shear modeled above stack top	Yes	No, vertical wind shear not modeled
Puff splitting allowed	Yes	No, puffs are not split
Chemical mechanism flag	Yes	Chemical transformation not modeled
Wet removal modeled	No	Yes
Dry deposition modeled	Yes	Yes
Gravitational settling (plume tilt) modeled	Yes	No
Method used to compute dispersion coefficients	Yes	PG dispersion coefficients for rural areas (computed using the ISCST multi-segmented approximation) and MP coefficients in urban areas
Sigma-v/sigma-theta, sigma-w measurements used	Yes	Use both sigma-(v/theta) and sigma-w from PROFILE.DAT to compute sigma-y and sigma-z (valid for METFM - 1, 2, 3, 4, 5)
Back-up method used to compute dispersion when measured turbulence data are missing	Yes	PG dispersion coefficients for rural areas (computed using ISCST multi-segment approximation) and MP coefficients in urban areas
Method for Lagrangian timescale for Sigma-y (used only if MDISP=1,2 or MDISP2=1,2)	Yes	617.284 (s)
Method used for Advective-Decay timescale for Turbulence (used only if MDISP=2 or MDISP2=2)	Yes	No turbulence advection
Method used to compute turbulence sigma-v & sigma-w using micrometeorological variables (Used only if MDISP = 2 or MDISP2=2)	Yes	Standard CALPUFF subroutines
PG sigma-y, z adj. for roughness	Yes	No
Partial plume penetration of elevated inversion	Yes	Yes
Strength of temperature inversion	Yes	No
Map Projections and Grid Control Parameters		
Projection	No	LCC: Lambert Conformal Conic
DATUM-region for output coordinates	No-Default	WGS-84, Global Coverage
Project origin (decimal degrees) latitude	No-Default	37 N
Project origin (decimal degrees) longitude	No-Default	120.5 W
Project parallels (decimal degrees) latitude	No-Default	30 N
Project parallels (decimal degrees) latitude		60 N

Model Input Description	Default Input used?	Model Input Used?
No of X grid cells (NX) (kilometers)[1]	No-Default	333
No of Y grid cells (NY) (kilometers)[1]	No-Default	333
No of vertical layers (NZ)	No-Default	12
Cell face heights	No-Default	0., 20.0, 40.0, 80.0 160.0, 300.0, 600.0, 1000.0, 1500.0, 2200.0, 3000.0, 4000.0, 5000.0
Grid origin (kilometers) (X)	No-Default	-497.2
Grid origin (kilometers) (Y)	No-Default	-544.9
Miscellaneous Dry Deposition Parameters		
Reference cuticle resistance	Yes	30.0 s/cm
Reference ground resistance	No	5.0 s/cm
Reference pollutant reactivity	Yes	8
Number of particle size intervals used to evaluate particle size deposition velocities	Yes	9
Vegetation state in un-irrigated areas	Yes	1
Miscellaneous Dispersion and Computational Parameters		
Horizontal size of puff (m) beyond which time-dependent dispersion equations (Heffter) are used to determine sigma-y and sigma z	Yes	550
Stability class used to determine plume growth rates for puffs above the boundary later	Yes	5
Vertical dispersion constant for stable conditions	Yes	0.01
Factor for determining transition-point from Schulman-Scire to Huber-Snyder Building downwash scheme	Yes	0.5
Range of land use categories for which urban dispersion is assumed	Yes	10, 19
Maximum travel distance of puff/slug (in grid units) during one sampling step	Yes	1.0
Maximum number of slugs/puffs release from one source during on time step	No	1
Maximum number of sampling steps for one puff/slug during on time step	No	1
Number of iterations using when computing the transport wind for a sampling step that includes gradual rise	Default	2
Minimum sigma y for a new puff/slug (m)	Default	1.0
Minimum sigma z for a new puff/slug (m)	Default	1.0
Minimum wind speed (m/s) allowed for non-calm conditions.	Default	0.5
Maximum mixing height (m)	Default	3000
Minimum mixing height (m)	No	20
Wind speed classes	Default	1.54, 3.09, 5.14, 8.23, 10.8
Wind speed profile power-law exponents for stability classes 1-6	Default	ISC Rural Values A, B, C, D, E, F 0.07 ,0.07, 0.10, 0.15, 0.35, 0.55

Model Input Description	Default Input used?	Model Input Used?
Potential temperature gradients for Stable Classes E and F (deg/km)	Default	0.02, 0.035
Plume path coefficients for each stability class (used when MCTADJ=3)	Default	A, B, C, D, E, F 0.5, 0.5, 0.5, 0.5, 0.35, 0.35

[1] Number of grid cells exceeds maximum number allowed in USEPA Version of the CALMET model. The executable code was revised to accommodate this large number of cells and recompiled to complete the meteorological modeling.

Mercury Speciation. Releases of Mercury to the atmosphere typically occur in three forms: elemental [Hg(0)], reactive (RGM), and particulate [Hg(p)], or any combination of these. Each mercury species exhibits different depositional characteristics. RGM deposition occurs more quickly than Hg(0) because it is more soluble and adsorbs to most surfaces. It is widely accepted that this form of mercury has the highest deposition rate (Vijayaraghavan et al. 2008). Mercury speciation data are not available specifically for the combustion of refinery fuel gas; however, there are data available for combustion emissions from coal-fired power plants. Table 2-2 summarizes Hg speciation fractions that have been compiled using emissions data from 30 coal-fired power plants located in the eastern United States (Vijayaraghavan et al. 2008). The 30 power plants referenced above represent facilities with the highest percentage of RGM emissions (Vijayaraghavan et al. 2008) and would subsequently provide a conservative basis (or upper bound) for the deposition modeling. Total mercury emissions from each of the five Bay Area Refineries were multiplied by the fractions for each of the three mercury species as indicated in Table 2-2.

Table 2-2 Mercury Speciation

Mercury Species	Speciation Description	Percent of Emitted Mercury
Hg(0)	Elemental	39%
Hg(p)	Particulate	4%
RGM	Reactive	57%

Source: Plume-in-grid modeling of atmospheric mercury (Vijayaraghavan et al. 2008)

Deposition Velocities. The CALPUFF model is capable of using site-specific atmospheric conditions and land-use data provided in the meteorological data set for calculating representative deposition velocities. In addition to land-use data, specific chemical parameters are input and used by CALPUFF to calculate site-specific deposition velocities for Hg(0), Hg(p), and RGM as discussed below. For the best representation of specific conditions in the San Francisco Bay Area, this analysis has been performed utilizing the CALPUFF-derived deposition velocities. For each of the mercury phases, both dry and wet deposition were calculated.

For the deposition of particulates, the CALPUFF modeling input parameters include mass mean diameter, the associated standard deviation, and scavenging as summarized in Table 2-3. These default values are provided by the CALPUFF model and represent default values for a non-reactive set of pollutants (nitrate-NO_3), and would provide maximum deposition rates.

For the elemental and reactive mercury phases, input parameters include diffusivity, reactivity and mesoscale resistance, and Henry's Law coefficients. The wet deposition of these more reactive mercury phases are only affected by scavenging from liquid (not frozen) precipitation

(see Table 2-3). Except for the Henry's Law coefficients, default values producing the highest deposition rates of reactive pollutants were used (nitric acid – HNO_3).

Table 2-3 Chemical Parameters

Dry Deposition Parameters (Particulate)					
Species Name		**Geometric Mass Mean Diameter (microns)**		**Geometric Standard Deviation (microns)**	
Mercury particulate (Hg(p))		0.48		2.0	
Dry Deposition Parameters (Gas)					
Species Name	**Diffusivity**	**Alpha Star**	**Reactivity**	**Meso. Resist.**	**Henry's Law Coef.**
Hg(0)	0.1628	1.0	18.0	0.0	1.00E-07
RGM	0.1628	1.0	18.0	0.0	1.00E-07
Wet Deposition Parameters					
Species Name		**Scavenging Coefficient (sec^{-1})**			
		Liquid Precipitation		**Frozen Precipitation**	
Hg(0)		6.00E-05		0.00E+00	
Hg(p)		1.00E-04		3.00E-05	
RGM		6.00E-05		0.00E+00	

Source: CALPUFF Modeling System.

Table 2-3 shows the values assumed for this analysis. As stated above, the parameters selected have the highest potential for deposition and, therefore, provide a conservative basis of deposition for this assessment. In addition, it should be noted that, for the particulate phase, the default mass mean diameter are for diameters of 10 microns or less as established by the USEPA. Particulates from the use of combustion sources are typically in the range of less than one micron, thus providing additional conservative estimates for deposition.

The CALPUFF dispersion modeling requires the input of source-specific parameters. The mercury modeling analysis was performed using a series of point sources. Point-source inputs include:

- Source location;
- Stack emissions;
- Stack gas exit temperature;
- Stack gas exit velocity;
- Stack inner diameter; and
- Stack base elevation.

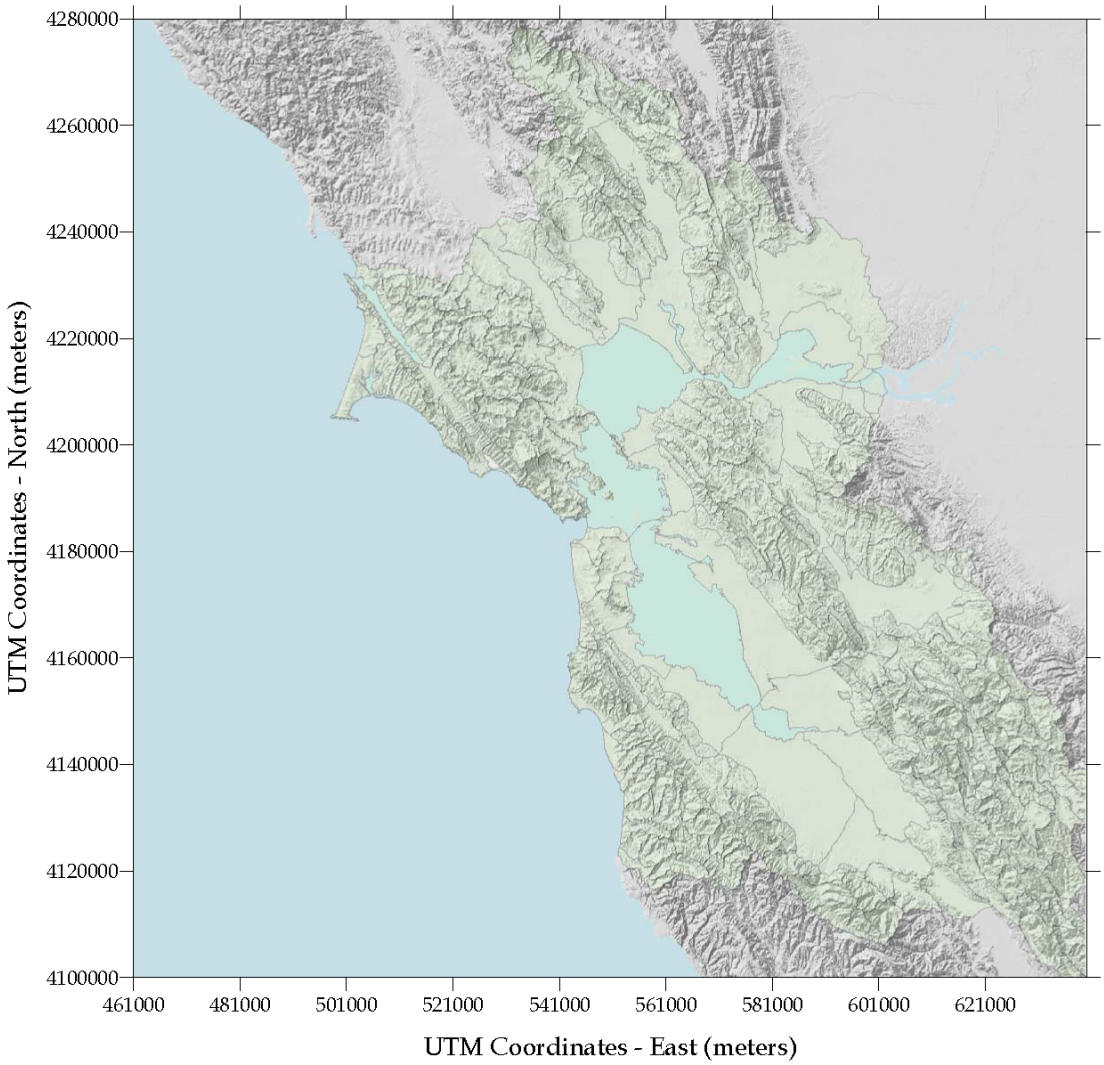
For each refinery, the modeling was performed assuming the mercury emissions are emitted from several representative stacks (between five and eight depending on the refinery). This minimized the number of modeled emission points. The stack emissions were co-located, or combined to best represent source type, size (based on Title V permits), and location. The stack release parameters were dependent on specific refinery processes and representative source-release parameters. Modeled source locations were unique for each of the refineries, in order to best represent the specific combustion sources at each site. Because this analysis is meant to calculate the transport of mercury throughout the Bay Area, the co-location of sources should

not significantly impact the overall modeled mercury concentrations and deposition within the drainage basins throughout the modeling domain.

2.2.4 Modeling Domain and Deposition Calculation Locations

The modeling analysis included the identification of numerous grid point locations for use in the calculation of deposition rates. In order to represent the regional nature of this analysis, a Cartesian grid was used, and points were placed every one and one half kilometers throughout the modeling domain. The modeling domain includes the rectangular area as illustrated in Figure 2-1 and covers the SFB and its surrounding water shed. Elevations for each of the gridded points were obtained from USGS DEMs. The CALPUFF dispersion model utilizes the model inputs, including mercury emissions, source release parameters, and regional meteorological conditions to calculate mercury deposition rates at each of the gridded point locations.

Figure 2-1 Modeling Domain



2.3 Atmospheric Dispersion Modeling Results

Air dispersion modeling was conducted for the refineries using the source parameters, which include a representative set of sources for each of the five SFB Area Refineries. Mercury deposition was calculated assuming emissions were in the particulate phase. Deposition rates within the modeling domain are dependant on many variables, including, but not limited to distance from a source, meteorology, land use and terrain features.

A review of the modeling results reveal that the majority of the deposition occurs to the north of the refinery sources, with a lesser amount depositing to the east and northwest. The lateral extent of the deposition is caused by a combination of the predominant wind characteristics in the SFB Area and local and regional terrain. CARB has illustrated seven general wind flow patterns that occur in the SFB Area as shown in Figure 2-2. Table 2-3 summarizes the percentages of directional airflow patterns (illustrated in Figure 2-2) that typically occur at four periods of the day (as well as daily average) by season. Most commonly, winds travel from the

west through the Golden Gate and from the northwest through the Cheleno and Luca Valleys. As illustrated, many are combination wind patterns, moving from one direction and changing due the interaction of local terrain. As seen in Figure 2-2 wind flow patterns labeled Northwesterly, Southerly, Bay Inflow, and Bay Outflow (flow patterns 1a, II, V, and VI) are likely to pick up mercury emissions from the refineries. These wind conditions occur approximately 51% of the time and are the most common wind patterns in this area, which are consistent with the modeling results. To a lesser extent, the modeling results show deposition occurring to the northwest of the refinery sources. This is also consistent with the wind flow patterns (III, IV, and VI) showing a frequency of 19% toward the northwest.

Upon further review of the modeling, the results showed that wet deposition dominates over dry deposition. It also reveals that a majority of the wet deposition occurs toward the north and that dry deposition occurs most often to the east. Based upon the wind flow patterns during winter months, when most of the wet weather patterns occur, southerly flows are generated by storm fronts and then move across the SFB Area. Table 2-3 shows a predominance of southerly and southeasterly winds that occur during winter months (32% during the rainy season) and would account for the dominance of the wet deposition to the north. During the summer and autumn months when rainfall is least, winds are dominated by the northwesterly wind flow regime, ranging from 78 to 54 percent, respectively.

The modeled deposition rates at each of the gridded points can also be used to calculate the total annual mercury deposition within a modeling region due to SFB Area Refineries. An analysis of the total deposition has been completed and is discussed in Section 3.

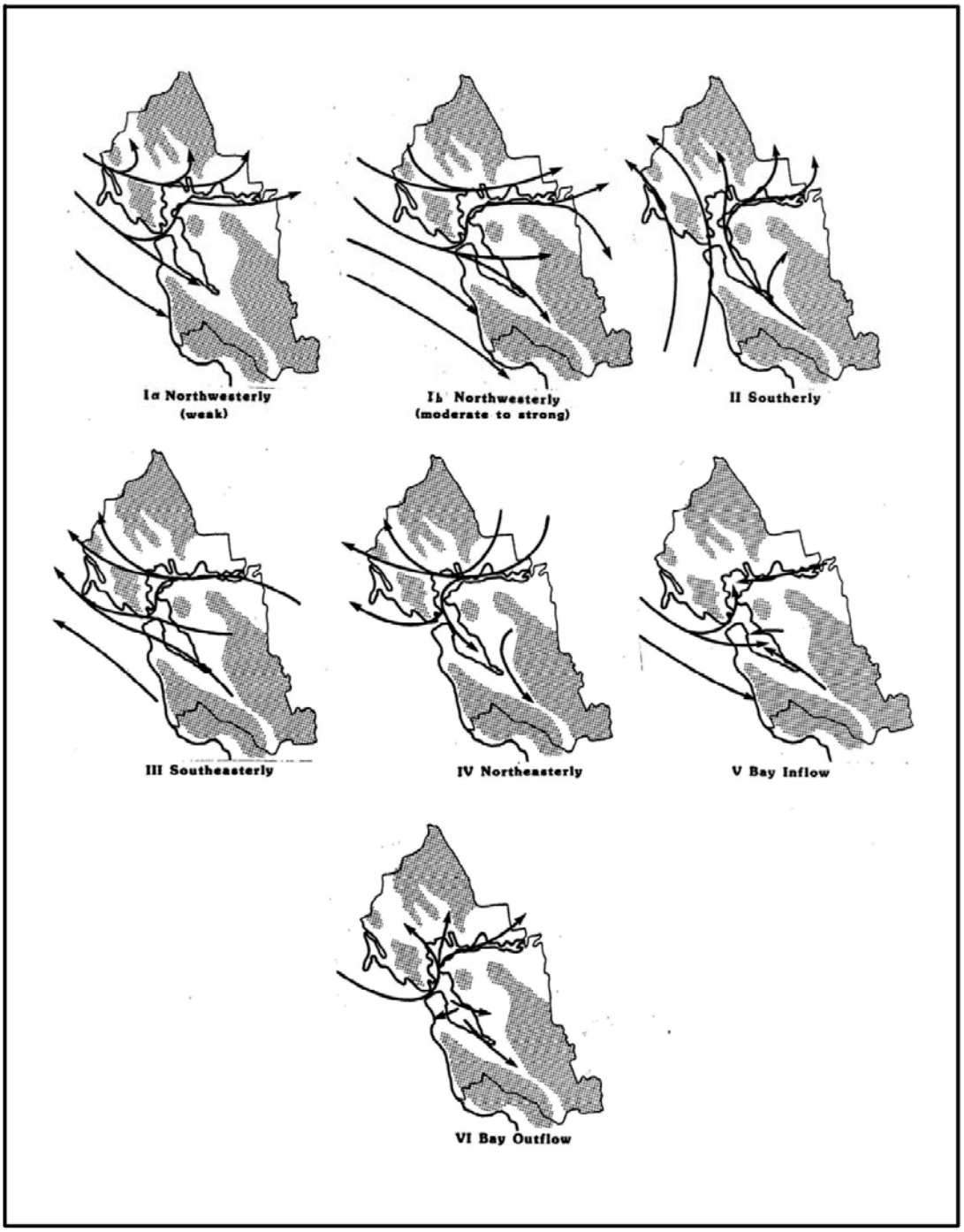
Table 2-4 San Francisco Bay Area Air Basin Surface Airflow Types Seasonal and Diurnal Percentage of Occurrence (1977-1981 Data)

Types	Ia	Ib	II	III	IV	V	VI	VII
	North-westerly (Weak)	North-westerly (Moderate to Strong)	Southerly	South-easterly	North-easterly	Bay Inflow	Bay Outflow	Calm
Time - PST	Percent of the Time							
Winter								
4 a.m.	3	4	19	14	8	21	5	24
10 a.m.	4	5	19	20	10	11	19	9
4 p.m.	16	16	16	12	13	3	22	1
10 p.m.	6	9	14	14	10	20	3	21
All Times	7	9	17	15	10	14	12	14
Spring								
4 a.m.	27	25	11	2	4	21	5	12
10 a.m.	29	25	14	6	5	3	17	1
4 p.m.	22	60	7	4	4	2	2	----[1]
10 p.m.	40	34	8	2	4	5	3	5
All Times	29	36	10	3	4	6	7	5
Summer								
4 a.m.	40	37	4	----[1]	0	6	2	10
10 a.m.	37	44	4	----[1]	1	1	13	0
4 p.m.	20	77	2	0	1	0	----[1]	0
10 p.m.	39	55	2	0	----[1]	1	1	1
All Times	34	53	3	0	1	2	4	3
Fall								
4 a.m.	25	13	7	6	3	22	3	19
10 a.m.	28	15	6	11	6	7	23	4
4 p.m.	31	46	5	2	6	2	7	----[1]
10 p.m.	37	24	6	4	3	13	1	12
All Times	30	24	6	6	4	11	9	9
Annual								
4 a.m.	24	20	10	6	4	16	4	16
10 a.m.	25	22	11	9	6	6	18	4
4 p.m.	22	50	8	5	6	2	7	----[1]
10 p.m.	31	30	8	5	4	10	2	10
All Times	26	30	9	6	5	8	8	8

[1] < 0.5 percent

Source: California Air Resources Board. Aerometric Data Division. 1984. Reprinted January 1992. California Surface Wind Climatology. June.

Figure 2-2 San Francisco Bay Area Airflow Pattern Types



Source: California Air Resources Board. Aerometric Data Division. 1984. Reprinted January 1992. California Surface Wind Climatology. June.

2.4 Analysis Limitations

Simulation of dispersion and the predictions of concentration and deposition related to mercury emissions from point sources by their very nature may include limitations in the accuracy of model predictions. Modeling for the SFB Area refineries is no different. Dispersion models calculate a wide variety of concentrations/deposition over a series of specific locations, which represent an ensemble average of specific events. Events can include "known" meteorological parameters (wind speed, wind direction, mixing height, etc.) or source-specific characteristics (point source, area source, volume source, etc.). Variations in these collective events reveal both inherent and controlled limitations of dispersion models. Inherent deviations are the variability of uncontrolled parameters in the events, such as the repeatability of identical wind speeds over numerous observations. In theory, the inherent deviations can create a difference in modeled vs. measured concentrations of ± 50% (USEPA 2005). Controlled (or reducible) variances are associated with parameters within the event that can be more easily managed or reproduced, such as a constant emission rate. Typically, these can be designed to minimize the variation. Model variations are considered reducible as opposed to inherent.

Studies for examining model accuracy have confirmed that dispersion models are more reliable in estimating long-term averaged concentrations than they are in estimating short-term averages at specific locations. Models are reasonably reliable for calculating the magnitude of highest concentration occurring within an area; however, not necessarily at a given point in time or space of that predicted concentration. Model accuracies for the highest derived concentrations typically range from ±10 to ±40%.

Studies have shown that the CALPUFF modeling system provides the technical basis and has the capabilities for addressing both long-range transport and complex wind situations. Studies have also shown that model accuracies are sufficient for use in the 50 km – 200 km range, and in some instances up to 300 km. Although scientific advancements continue to emerge, the CALPUFF model has been found to be scientifically accepted for use in regulatory applications by both state and federal agencies and for simulating long-range transport.

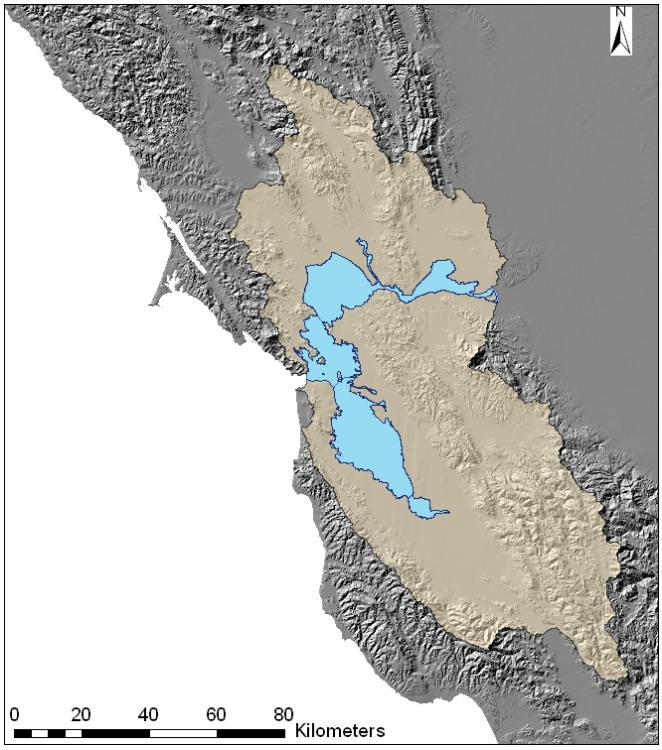
Mercury can be emitted in various phases (i.e., elemental, particulate, and reactive gas phase), or in combination. Therefore, inherent uncertainties can occur depending on the assumptions made regarding the amount of each phase being emitted. In addition, reactive gas phase mercury is highly dependent upon outside ambient conditions and, thus, would provide a greater rate of uncertainty within the dispersion and deposition in the model. However, specific mercury emissions speciation was not available for this analysis. Therefore, in order to minimize the uncertainty and increase the reliability of the modeled results, the mercury emissions were assumed to be in the particulate phase.

3.0 SYNTHESIS OF RESULTS – MERCURY TRANSPORT & FATE

Estimates of atmospheric deposition rates of mercury resulting solely from emissions from the SFB Area refineries were presented in Section 2. These deposition rates from the SFB Area refineries (at individual model grid points) were used to estimate the resulting total atmospheric deposition in the watershed area draining directly into the SFB. These model-based estimates of atmospheric deposition in the watershed area were compared with reported rates of atmospheric deposition of mercury in the SFB Area and other regions of the US.

The watersheds draining directly into the SFB (see Figure 3-1) were identified using the California Interagency Watershed Map of 1999 (updated May 2004, "calw221") (CalWater 2.2.1, http://gis.ca.gov/catalog/BrowseRecord.epl?id=22175), which is the State of California's working definition of watershed boundaries. Note that the drainage area indicated in Figure 3-1 does not include the drainage area associated with streams flowing into SFB through the Sacramento-San Joaquin Delta (such as the Sacramento and San Joaquin Rivers). Thus, the area identified in Figure 3-1 is consistent with the modeling domain chosen in Section 2. The SFB watershed area in CalWater 2.2.1 also includes watersheds that drain directly into the Pacific Ocean, but these watersheds were also excluded from this analysis.

Figure 3-1 Watershed Area Draining Directly into the San Francisco Bay



Notes:

- Identified from the California Interagency Watershed Map of 1999.
- Note that the above area does not include the (indirect) contributing area of streams flowing into the San Francisco Bay through the Sacramento-San Joaquin Delta.

The modeled atmospheric deposition at individual modeled grid points, as described in Section 2, were interpolated to obtain average total deposition rates from the SFB Area refineries over the areas identified in Figure 3-1. Spatial averaging was performed using a commercial Geographic Information System software package called ArcGIS®. For this analysis, the point-based model deposition rates were interpolated to a fine grid mesh (0.1 km x 0.1 km) and aggregated over these fine grids to obtain spatially averaged deposition rates for the entire area shown in Figure 3-1. Over the SFB Area, the average annual total deposition rate of mercury resulting from the SFB Area refineries was estimated to be 0.1 $\mu g/m^2/yr$ (see Table 3-1).

Table 3-1 Total Deposition of Mercury to the SFB Area Obtained by Interpolation of Point-Based Model Estimates Presented in Section 2

Location	Area (km^2)	Total Deposition (g/yr)	Average Deposition ($g/km^2/yr$ or $\mu g/m^2/yr$)
SF Bay Water	1121	190	0.17
SF Bay Land	9035	820	0.1
SF Bay Area Total	10156	1010	0.1

Estimated deposition fluxes of mercury from the SFB Area refineries were compared with reported estimates of atmospheric mercury deposition fluxes at locations within and around the SFB in Table 3-2. Table 3-2 illustrates that the modeled total deposition flux of mercury from the SFB Area refineries varies from 0.5% to 5% of both the wet and dry deposition flux estimates reported in the literature. The observed and model-based estimates of deposition fluxes presented in Table 3-2 are within the range of deposition fluxes reported from other parts of the US (see Table B-8).

Table 3-2 Comparison of Estimated Total Mercury Deposition Fluxes (µg/m^2/yr) From SFB Area Refineries with Measured and Modeled Atmospheric Mercury Deposition Fluxes Reported in the Literature

Geographical area	Wet/dry	Flux (µg/m^2/yr)	Estimate from this study as a fraction of literature value	Data Source
San Francisco Bay and the watersheds draining directly into it (Figure 3-1)	wet+dry	0.1	-	Model-based estimates from this study
Covelo, CA (MDN Site CA 97)	wet	3.8-4.8	2.1% - 2.6%	Measurements from MDN, see Table B-5
San Jose, CA (MDN Site CA 72)	wet	2.1-3.1	3.2% - 4.7%	Measurements from MDN, see Table B-5
Entire San Francisco Estuary	wet	4.2	2.4%	Measurements from Tsai and Hoenicke (2001), see Table B-6
Moffett Field, Central CA	wet	4.4	2.3%	Measurements from Steding and Flegal (2002), see Table B-7
Long Marine Lab, Central CA	wet	4	2.5%	Measurements from Steding and Flegal (2002), see Table B-7
San Francisco Bay Area[a]	wet	5-15	0.7% - 2%	Model-based estimates from Seigneur et al. (2004)
San Francisco Bay Area[b]	wet	2-4	2.5% - 5%	Model-based estimates from Selin and Jacob (2008)
Entire San Francisco Estuary	dry	19	0.5%	Model-based estimates from Tsai and Hoenicke (2001), see Table B-6
San Francisco Bay Area[a]	dry	2-5	2% - 5%	Model-based estimates from Seigneur et al. (2004)

Notes:
[a] Based on a visual inspection of the maps presented by Seigneur et al. (2004), the values corresponding to two grid cells (100 km x 100 km grid resolution) approximately overlapping with the San Francisco Bay Area were used.
[b] Based on a visual inspection of the maps (grid resolution of 400 km x 500 km) presented by Selin and Jacob (2008), the values associated with the area generally corresponding to the San Francisco Bay Area were used.

The model-based estimate of total annual deposition of mercury from the SFB Area refineries to the region shown in Figure 3-1 (1.0 kg/yr, see Table 3-1) was compared with the contributions of mercury from various sources to the SFB (see Table 3-3). if it is assumed that all of the mercury deposited to the watershed region shown in Figure 3-1 reaches the Bay, the modeled contribution of atmospheric deposition from the SFB Area refineries, is approximately:

- 5.6% of the mercury contributed to the SFB by municipal and industrial discharges
- 3.7% of the mercury deposited directly from atmospheric deposition to the SFB water surface, and
- 0.2% of the mercury contributed to the SFB by the Central Valley Watershed.

It is unlikely that 100% of the mercury deposited to the watershed by atmospheric deposition is transported in storm water to SFB. Therefore these results are likely overestimates.

Table 3-3 Comparison of Annual Total Mercury Deposition (kg/yr) from SFB Area Refineries to the San Francisco Bay Area (model-based) with Annual Mercury Contribution from Various Sources Listed in the SFBRWQCB TMDL (2006)

Source	contribution (kg/yr)	SFB Area Refinery atmospheric contribution (as a fraction of other Hg sources to the SFB)
Maximum WSPA member facilities (model-based estimate from this study)[a]	1	-
Wastewater (municipal & industrial) Discharges	18	< 5.6%
Non-urban Storm Water Runoff	25	< 4%
Direct Atmospheric Deposition	27	< 3.7%
Guadalupe River Watershed (mining legacy)	92	< 1.1%
Urban Storm Water Runoff	160	< 0.6%
Central Valley Watershed	440	< 0.2%
Erosion of Buried Sediments	460	< 0.2%

[a] This estimate, from Table 3-1, assumes that all mercury deposited to watersheds surrounding SFB reaches the Bay in storm water, and is thus a highly conservative estimate.

3.1 Conclusions

This analysis indicates that the SFB Area refineries contribute minimal mercury to the Bay. Modeled mercury deposition rates from SFB Area refineriesare equivalent to approximately 0.5% to 5% of both wet and dry deposition flux estimates (as presented in Table 3-2) when compared with reported estimates of mercury deposition at locations within and around the SFB Area,.

Assuming that all of the mercury from SFB Area refineries deposited to the watershed area draining directly to the SFB reaches the Bay, the contribution from the SFB Area refineries is estimated to be approximately:

- 5.6% of the mercury contributed to the SFB by municipal and industrial discharges,
- 3.7% of the mercury deposited directly from atmospheric deposition occurring over the SFB Area, and
- 0.2% of the mercury contributed to the SFB by the Central Valley Watershed.

Thus, estimated mercury loadings from atmospheric deposition of mercury emitted by SFB Area refineries are a small fraction (less than 6%) of total mercury loadings from other sources in the SFB region.

The calculation of the mercury emissions, dispersion and deposition modeling and the predictions of mercury transport and fate include component features that may over-predict or under-predict. Overall, the predicted mercury deposition and transport to the SFB is likely to be over-predicted due to the conservative emission calculation assumptions, as well as assumptions used in the transport.

3.2 Analysis Limitations

Predicting the fate of mercury deposition related to the SFB Area Refineries may be affected by laboratory and analysis methodologies, as well as the need to simplify some aspects of the dispersion and deposition process for maintaining a manageable computational effort and the conservative assumptions, and thus, the overestimation for mercury contributions to the SFB.

3.2.1 Mass Emissions

Refinery fuel gas (RFG) testing follows a rigorous QA/QC protocol during the first quarter of the study period. In general, the sampling results differed from the duplicate results by an average of 16%, which is in the expected range for this analysis. This could be either an over estimate or under estimate of mercury concentrations. However, the mercury emissions from the RFG combustion were calculated assuming that 100% are emitted into the atmosphere. This is most likely an overestimate of emissions.

The accuracy of stack testing results and the resultant emission calculations, which are based on the source testing of the Fluidized Catalytic Cracking Unit (FCCU) stacks, could also be marginally in error due to testing frequency, laboratory methodologies, and from variations in operating conditions. The scheduled stack tests were to be spread out over a one-year period to best represent annual average conditions. This again could be the estimated 10%-20% error range toward an over estimation or an underestimation of stack mercury concentrations. Again, the emissions calculated from the combustion of refinery fuel gas assumed 100% of the mercury content is emitted into the atmosphere which would lead to an over prediction of mercury concentrations and depositions rates.

3.2.2 Dispersion Modeling

In general, USEPA-approved dispersion models, including CALPUFF, have been found to have an accuracy factor of two from observations. These variations can either over-predict or under-predict pollutant concentrations and deposition rates. However, studies have shown that dispersion models are more reliable when estimating long-term average concentrations and deposition rates than short-term at specific locations. Since the modeling for this analysis only involves the predictions of annual-average concentrations, the accuracy of the results are expected to be less than a factor of two in variation, increasing the overall reliability in the assessment.

Mercury can be emitted in various phases, or in combination (i.e., elemental, particulate, and reactive gas phase.) There are no mercury speciation data available for refinery fuel combustion. However, there are data available from coal-fired power plants. The mercury speciation used in the modeling analysis was compiled using emissions data from 30 coal-fired power plants located in the eastern United States. These 30 facilities were chosen at the request of the RWQCB as conservative parameters to define the speciation due to their high percentage of RGM emissions. It is widely accepted that RGM has a higher deposition rate than the other emitted species (Vijayaraghavan et al. 2008). The use of speciation data from the coal-fired power plants would most likely result in conservative deposition modeling results.

3.2.3 Transport and Fate

The assumptions used in the transport and fate analysis were conservative and over-predictive. The analysis assumed that 100% of the mercury deposited within the San Francisco Bay Watershed would eventually drain into the Bay. This assumption will overstate transport because it does not account for soil absorption and the root and leaf uptake of plants.

Although this mercury transport and fate analysis includes both component features that overestimate and underestimate impacts, on balance, total Bay Area mercury deposition are likely overestimated.

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Appendix A
Mercury Mass Balance

APPENDIX A MERCURY MASS BALANCE

Information on levels of mercury in refinery processes and discharges include some recent compilations of mercury in crude samples, refined products, and estimates by the refineries reported to the USEPA and local agencies. WSPA member refineries conducted additional sampling and analysis as requested by the RWQCB. Sampling and analytical data collected during the course of this study, data provided by the refineries, and publicly available information were used to develop a simplified mercury mass balance estimate around the five SFB Area refineries. Sampling and analytical data were used to estimate mercury concentrations of refinery fuel gas, process vent stacks, crude oils, and petroleum coke. Information on the refined products, the refinery production rates, and material properties obtained from the individual refineries were used to determine annualized mass rates. Publicly available information reported by the refineries was used to estimate the amount of mercury being removed in waste sent to landfills outside of the Bay Area and in permitted water discharges. WSPA performed the mercury material balance to comply with the request from the RWQCB but the level of accuracy is different for each source and this makes it infeasible to close the mercury material balance. This section presents a summary of the mercury in the various refinery streams, contains a discussion regarding the methodologies used to obtain the data, and its relative accuracy. The variation in the regulatory mandated calculations of many streams restricts any statistical method to "balance" the data.

A.1 Summary of Mercury Mass Findings

The simplified mercury mass balance that was developed for this study is based on the estimated amounts of mercury entering the five SFB refineries in crude oil, the amount of mercury exiting in the combustion of refinery fuel gas, petroleum coke, process stacks, refined products, refinery waste, and in permitted water discharges. This is a simplified approach, because it does not attempt to quantify the effect of accumulation of mercury that can occur in process equipment, introducing a time-dependent fluctuation, in mercury quantities that may influence waste quantities generated in future years. The results are also dependent on the level of accuracy of the sources used to calculate mercury mass rates. These factors should be considered when evaluating the result of the mercury mass balance calculation. The calculated mass of mercury contained in the refinery streams used to develop the mass balance is provided in Table A-1 below.

Table A-1 Simplified Mass Balance Summary

	Media	Basis of Calculation	Mercury Mass Rate (kg/year)	Analytical Variability (kg/year)	Observed Variability (kg/year)
Input	Crude	Analytical Data and Crude Processing Rates.	224	+/-58	+/-62
Output	Refinery Fuel Gas	Analytical Data and Refinery Fuel Gas Consumption Rates	1.14+/-	+/-0.18	+/-0.44
	Process Stacks	Process Stack Test Results	17.96	+/-3.59	6.82
	Petroleum Coke	Analytical Data and Coke Production Rates	3.18	+/-0.22	--
	Water Discharges	NPDES Permit Limitation	0.91	--	--
	Refined Products	Literature Review	26.0	--	--
	Refinery Waste	TRI Reports for 2000 through 2007	221 to 650	--	--
	Total Output		**270 to 699**		

The result of the mass balance suggests that there is a difference between the amount of mercury entering the facilities (224 kg/yr) and the amount exiting the facility (270 to 699 kg/yr). This inherent variation was predictable due to inherent differences in the governmental mandated methods used to calculate the various waste streams, and in the differing timeframes for each of the media data reported. Statistically, the analysis necessarily uses unlike data sets, particularly the TRI data and the Water Discharge Data. These data sets were not intended for comparison nor designed to be used in a mass balance exercise. Therefore, there is inherent variation in the level of accuracy in comparing any of this source data. The sources of information and the relative level of accuracy for each data set is discussed in the following sections.

A.2 Mercury in Crude Feedstock

Mercury enters the refining process as a trace component of crude oil feedstocks. Samples of crude oils were collected and analyzed on a monthly basis. These analytical data were used along with throughput volumes to calculate the mercury mass. The 9-month sampling period for the crude was from October 2007 through June 2008.

A.2.1 Crude Mercury Sampling Results

For each of the refineries, approximately three crude samples were collected during each of the first 6 months of the sampling period and a single sample was collected during each of the final 3 months. Representative samples were obtained at the inlet to each crude unit at each refinery and represented the mix of crudes being processed at that time. The crude samples were sent to either Frontier Geosciences or CEBAM Analytical for analysis. Both laboratories analyzed for total mercury by cold vapor atomic fluorescence spectrometry (CVAFS). However, samples analyzed by CEBAM were prepared according to the Combustion-AF (CAF) Method and samples analyzed by Frontier Geosciences (Frontier) were prepared using the Digestion-AF (DAF) Method. Both are USEPA-approved methods and have adequate detection limits for the purposes of this study (0.5 parts per billion [ppb]).

Laboratory procedures were modified to improve consistency of results between the two laboratories. Frontier addressed the potential of volatile mercury loss that was reported in previous studies (Wilhelm, 2007) by withdrawing the sample from the container with a gas-

tight syringe and injecting the sample below the surface of the nitric acid used for digestion. Both laboratories used sonication to homogenize the crude samples prior to sample preparation.

To evaluate the potential impact from using different analytical methods, ERM submitted duplicate samples from the seven crude units to the two laboratories. The difference between the Frontier and CEBAM results ranged from between 0.4 ppb and 13.29 ppb. The largest differences were reported for the samples with the highest mercury content. Similar comparative studies indicate that the standard deviation of these analytical results is proportional to the mercury concentration (Hwang, 2007). That is, the largest variations were found in the samples with the highest mercury concentrations. The relative percent difference (RPD), which is the difference of the duplicate means as a percentage of the inter-laboratory mean, ranged from 2-56% with an average of 27%. A similar study conducted for these two laboratories found that the inter-laboratory results varied by an average of 24%, with a range of 0.1-94% (Wilhelm 2007).

The average mercury concentrations found in the crude at the different refineries ranged from 1.52 to 14.69 ppb. Table A-2 provides the average mercury concentration for the crudes at each of the refineries. Good agreement was found with other recent and independent studies of mercury concentrations in the crude refined in North America (Wilhelm, 2007). The quality assurance/quality control (QA/QC) results for the crude samples show that the results are within the expected range.

Table A-2 Mercury Concentrations in Crude

Refinery Crudes	Minimum Mercury Concentration (ng/g)	Maximum Mercury Concentration (ng/g)	Average Mercury Concentration (ng/g)
1	<0.36	14.30	6.19
2	2.26	9.05	5.23
3	3.32	21.63	10.30
4	5.10	41.29	19.07
5	<0.42	8.69	1.52
6	0.50	10.80	2.87
7	<0.32	2.91	1.51

A.2.2 Crude Processing Rate and Properties

Annual throughput of processed crude and the crude properties were provided by each of the refineries. For the purposes of this analysis, throughputs from July 2007 through June 2008 were used. These 12 months include the 9-month sampling period. The throughput from the refineries for this time period was approximately 278,155,288 barrels. During the sampling period, the density of the crude ranged from 308 to 342 pounds per barrel.

A.2.3 Mass of Mercury in the Crude

Based on the average mercury concentration in the crudes at each crude unit and annual crude throughputs, the estimated amount of total mercury in the crude is approximately 224 kg/year. This estimate may vary by ±58 kg/yr (24%)to account for limitations in the analytical results as discussed above in Section A.2.1. Inherent variability in the process data may also cause the estimate to vary by an additional ±62 kg/yr (26%).

Measurement "uncertainty" is intrinsic in any sampling and calculation method of this scale. Because the mercury mass in the crude was estimated as the product of the crude processing

rate and the corresponding mercury concentration in the crude, the inherent variability of these parameters will generate uncertainty in the total mercury mass estimated. A statistical analysis was conducted on the sampling data for each refinery discussed in A.2.1 and A.2.2 to estimate the standard error of the sample mean and the percent variability around the mean at the 95 percent confidence level. The variability of the mercury concentration in the various crude stocks ranged from 19.3% to 47.9% (depending upon the refinery), with a mean standard error variability of 24.5%. The variability of the crude processing rate is much less, ranging from 2.3% to 17.1% and a mean standard variability of 7.7%. The standard error variability of the product of the mercury concentrations and the crude processing rate ranged from 19.2% to 58.1%, with a mean standard error variability of 26.6%. This compares well with the average variability of the two individual parameters (26.3% standard error variability).

A.2.4 Crude Representativeness

Detailed crude representativeness data for the study period and the 5 years prior to the study was provided to the RWQCB in a Confidential Business Information binder. This information was used to assess whether the crude sampled during this study was similar to the crude that had historically been processed at these facilities. Table A-3 below shows that the average percent of SJV crude processed during the study period and the historical percent processed. In general, the average percent lies within the range of the historical data.

Table A-3 Percent of SJV Crude in Mercury

Refinery Crudes	Percent of SJV Crude					
	2003	2004	2005	2006	2007	Study Period
1	20%	24%	27%	33%	33%	25%
2	91%	78%	77%	77%	79%	76%
3/4	28%	23%	23%	23%	25%	27%
5	0%	0%	0%	0%	0%	0%
6/7	66%	65%	68%	74%	76%	69%

A.3 Refinery Fuel Mercury Content

Samples of mercury in refinery fuel gas (RFG) were collected by Brown & Caldwell from the five SFB Area refineries to estimate potential emissions resulting from its combustion. For most of the refineries, RFG mercury sample collection began in May 2007 and lasted until January 2008. Each of the refineries had unique testing dates and completion dates. Samples were obtained at seven locations (i.e., two of the refineries had two independent fuel gas systems requiring additional sets of samples).

A.3.1 Refinery Fuel Gas and Sampling Methodology and Results

The mercury sampling in RFG was performed by Brown & Caldwell using USEPA's Modified Method 30b. The sample collection method is consistent with the ASTM and ISO methods for the measurement of mercury in natural gas. The efficacy of this method was proven during a pilot test that also resulted in approved design specifications for sampling equipment handling combustible gas in a refinery environment. During the first quarter of sampling, monthly testing followed a rigorous QA/QC process, which included:

- Two simultaneous tests (using two blank sorbent traps) at two separate locations to provide critical precision measures (four measurements);
- Two sorbent traps spiked with known mercury concentrations – laboratory spikes to verify analytical methodologies are correct and consistent; and
- At least one sorbent trap spiked with a known mercury concentration – field spike to verify collection techniques are correct and consistent.

During the first quarter, seven to nine samples were obtained from each location: two duplicates for 3 months plus at least one field blank. The sampling results must meet the following QA/QC requirements:

- Field duplicate results fall <25% relative percent difference (RPD) for each monthly sampling;
- Breakthrough to "B" trap must be < 10% of "A" trap; and
- Spike recovery must fall within 75%-125% recovery.

If these criteria were met, the sampling efforts were relaxed to one sample per monthly sampling event, per sampling location, with spiked duplicates and a field blank collected once per quarter. If criteria were not met, then the sampling continued with the collection of duplicates and/or spikes.

The average mercury content for the different refineries ranged from 0.023 microgram per standard cubic meter (µg/scm) to 0.83 µg/scm. The average mercury concentration in the refinery fuel gas across all of the refineries was 0.294 µg/scm. Table A-3 provides the average, mercury concentration in the seven types of refinery fuel gas used at the refineries. The QA/QC analysis shows that most of the data were within acceptable limits. The following is a summary of the QA/QC results:

- Forty-one of the 58 samples met all of the QA/QC criteria.
- Five samples had an RPD value greater than 25%.
- Breakthrough to "B" trap was reported for one sample.
- Spike recovery did not meet the QA/QC criteria for 15 samples. Most of these samples were collected at one location. Subsequent sampling indicates that the issue at this location was addressed.

In general, the sample results were similar to the duplicate results. Even with the five samples that exceeded the RPD criteria, the average RPD was 16%, which is within the expected range for this analysis.

Table A-4 Mercury in Refinery Fuel Gas

Refinery Fuel Gases	Minimum Mercury Concentration (µg/scm)	Maximum Mercury Concentration (µg/scm)	Average Mercury Concentration (µg/scm)
1	0.064	1.119	0.398
2	0.011	0.102	0.041
3	0.027	0.499	0.144
4	0.060	0.162	0.109
5	0.041	2.386	0.786
6	0.001	0.052	0.021
7	0.002	2.404	0.812

A.3.2 Refinery Fuel Gas Combustion Rates

The volume of RFG that was combusted during the 12-month period (April 2007 through March 2008) was approximately 178,000 million standard cubic feet (MMSCF). The volume of RFG combusted by sources that vent to process stacks was not included in this total. Combustion emissions from those sources were captured by the sources tests. Information regarding the RFG usage was provided by the individual refineries.

A.3.3 Mercury Emissions Rates from Refinery Fuel Gas Combustion

The aggregate annual mercury emission rate of 1.14 kg/year was calculated using RFG combustion rates and the measured concentrations of mercury. This estimate may vary by ±0.18 kg/yr to account for the potential error in the analytical results (16%) and an additional ±0.44 kg/yr (39%) for the inherent uncertainty in the process data.

Statistical analyses were conducted to assess the inherent variability of the refinery fuel gas combustion rate and the mercury concentration data used to estimate the overall mercury emissions for each of the refineries. The 95th-percentile standard error for the mercury concentrations in fuel gas sampling at the five refineries (which contain a total of seven fuel gas systems), ranged from 16.2% to 54.1%, with a mean standard error percentage variability of 40.2%. Similar to the crude measurements, the standard error was much lower within each refinery, ranging from 2.3% to 10.2% around the individual means (averaging 3.9%). The average standard error percent variability for the product of the refinery gas combustion rate and the corresponding mercury concentration (used to calculate mass mercury emission rates) ranged from 21.5% to 52.7%, with a mean value of 39.3%. Again, this is in good agreement with the average variability of the individual refinery fuel gas combustion rates and mercury concentrations (40.2%).

A.4 Mercury Emissions from Process Stacks

A.4.1 Process Stack Source Testing Methodologies and Results

The RWQCB requested that the each of the four refineries with a fluidized catalytic cracking unit (FCCU) perform three sampling events. The sampling events were scheduled such that they would provide an adequate representation of normal annual operating conditions. This was accomplished by scheduling the stack testing events during different times of the year.

The stack testing was performed by the Avogadro Group, LLC (Avogadro). The sampling techniques employed for all but one of these stack tests were consistent with USEPA Method 101A, which tests only for mercury in the exhaust gas. For one of the tests, Avogadro used techniques consistent with USEPA Method 29, which is used to determine concentrations of multiple trace metals at the exhaust stack.

The process stack test results were reviewed for quality and validity by Avogadro through a rigorous QA program. Avogadro's review procedure includes review of each source test report by a project QA officer, including review of laboratory and fieldwork, data sheets, data inputs, calculation and averages, and the preparation of report text. In addition, the equipment used to conduct the emission measurements are maintained according to the manufacturer's instructions and calibrations are performed according to the schedule outlined by the CARB. All this work is completed to produce the most accurate results possible with the methods that were used. However, Avogadro does acknowledge that, with isokinetic tests such as these, there is an inherent error of approximately 10% (Avogadro, 2009). There is also a potential 10% error margin from the laboratory analysis for values that are above the reporting limit. Due to the conservative nature of the detection limit, values below the detection limit are not expected to have any significant error. Stack test results are summarized in Table A-5.

Table A-5 Mercury Concentration from Process Stacks (FCCUs)

Refinery Stacks	Test Date	Minimum Hg Concentration @ 7% O_2 (µg/m^3)	Maximum Hg Concentration @ 7% O_2 (µg/m^3)	Average Hg Concentration @ 7% O_2 (µg/m^3)
1	Feb. 7-8, 2008	<0.11	0.71	<0.45
	Oct. 8-10, 2008	<0.12	<0.13	<0.13
	Dec. 16-17, 2008	<0.11	<0.11	<0.11
2	Jun. 6-8, 2006	2.38	3.35	2.92
	Feb. 7, 2008[1]	2.32	2.50	2.39
	Feb. 23, 2009	1.90	2.03	1.96
3	Feb. 7-8, 2008	<0.13	<0.14	<0.13
	Aug. 27–28, 2008	<0.12	<0.13	<0.13
	Oct. 30-31 2008	<0.09	<0.12	<0.10
4	Mar. 19–20, 2008	<0.57	1.75	<1.09
	Sept. 16-17, 2008	0.104	0.286	0.186
	Nov. 11-12, 2008	0.26	<0.38	<0.32

Notes:

1. Four runs were performed for this test. The first run is not considered in these results because the test was performed while the unit was not operating at the proper load.

The inherent uncertainty of the mercury concentration from the process stacks were estimated from the standard error around the means for the mercury concentration data from the four refineries with FCCUs. The 95th-percentile standard error for the mercury concentrations ranged from 10.7% to 68.1%, with a mean standard error percentage variability of 38.0%.

A.4.2 Mercury Emissions

The stack testing results indicate that 17.96 kg Hg/yr are emitted into the atmosphere from the SFB Area refinery FCCU stacks. For each of the four refineries, the estimated annual mercury emissions exceed 0.1 kg Hg/yr, which was the minimum detection limit specified for the testing. The mass estimate may vary by plus or minus 11.28 kg/yr to account for the potential error from the sampling and analytical methods (10% for non-detect values and 20% for values reported above the detection limit) and an additional 38% from the inherent variability of the sampled data.

A.5 Mercury Content in Petroleum Coke Sample

A.5.1 Petroleum Coke Sampling and Results

Samples of the petroleum coke were collected to determine annual quantities of mercury removed from the site via coke production. Petroleum coke samples were gathered from the four refineries with a total of five coke production units. The coke samples were composites of daily samples that had been collected throughout the week. The composite samples were delivered to the laboratory (Frontier Geosciences) where they are prepared by a hydrofluoric/nitric/hydrochloric bomb digest according to Method FGS-111. The digested samples were then analyzed for total mercury by CVAFS according to Method FGS-069. The results from the five different types of coke analyzed are summarized in Table A-6.

Table A-6 Mercury in Petroleum Coke

Coke	Average Mercury Concentration (ng/g)
1	1.22
2	1.70
3	2.40
4	4.37
5	0.63
Average	2.07

Frontier indicated that the data were within the set control limits and they provided statistical information to access the relative accuracy of the coke analytical data. The data show that the average laboratory control sample RPD is 6.8% with a 2 sigma value of 18.3%.

A.5.2 Mass of Mercury

The analytical data and the amount of crude processed indicate that approximately 3.18 kg of mercury leaves the refineries in coke. According to Frontier, the precision of the mass mercury estimates may vary by about ± 0.22 kg/yr (6.8%).

Due to the limited amount of data relating to the coke production and the corresponding mercury emissions, no statistical analyses could be conducted to assess the inherent variability of the process data with any confidence in the results. Since the overall magnitude of the emissions is small relative to the overall emissions, the inherent variability would not have a significant change to the overall results of this study.

A.6 Mercury in Waste Materials Transported Off-site

The refineries provide annual Toxics Release Inventory (TRI) reports per criteria established by by USEPA. Each TRI submittal includes estimates of mercury in waste materials transported off-site to appropriate waste management facilities using these criteria. Readily available reports show that from 2000 through 2007, the five SFB area refineries generated aggregate amounts of mercury that ranged from 220.72 kg/yr in 2007 to 648.75 kg/yr in 2002 and averaged 417.27 kg annually. Table A-7 summarizes the mass of mercury removed off-site and presents the main waste stream types for each facility. The "main waste streams" reflected in Table A-7 contain at least 90% of the mercury removed off-site.

Table A-7 TRI Waste Discharge Data [table not formatted to show the boxes}

Refinery	Main Waste Stream Types	Off-Site Mercury Waste Discharge (Kg)							
		2000	2001	2002	2003	2004	2005	2006	2007
1	1. Lab Waste/Packs 2.Soil/Dirt 3. Sludge 4. Spent Caustic	11.93	3.48	17.56	4.74	9.24	31.64	61.02	29.57
2	1. Equipment Cleanout 2. Filter Waste 3. Biosolids 4. Lab Waste/Packs	388.28	567.84	299.93	306.77	351.63	158.75	165.81	31.05
3	1. Sludge 2. Soil/Dirt 3. Tank Bottoms 4. Industrial Debris	188.74	5.90	52.07	55.70	8.35	25.17	19.50	24.18
4	1. Metal Cake 2. Diesel & Catacarb Filters 3. Biosludge 4. Tank Bottoms 5. Lab Waste	0.23	14.52	0.14	3.18	0.01	0.01	1.36	26.31
5	1. Biological Sludge 2. Tank Bottoms 3. Soil/Dirt 4. Primary Sludge	2.84	57.02	99.68	32.55	40.78	66.94	64.08	109.62
Total		592.02	648.75	469.37	402.94	410.01	282.51	311.77	220.72

There are a number of issues that limit the accuracy of TRI data toward developing mass estimates that account for mercury in all refinery waste streams. The most common of these include changes in waste classification, profiles, and time. The mandated calculations might be useful in comparing year-to-year data, but cannot be used as an accurate measurement of mercury where it was not directly monitored, sampled, and measured.

A.6.1 Classification

Waste classification sampling is typically based on grab samples. It is prohibitively difficult to cost effectively conduct a rigorous and representative sampling protocol that is designed to collect representative samples from high and low volumes, solid and/or liquid, and from homogenous and combined waste streams. Waste classifications are based on concentration thresholds independently set by federal, state, and local regulations. The analytical reporting limits provided by the laboratories are typically set at these regulatory thresholds to simplify lab reporting and to provide the refinery a straight-forward (i.e., yes or no) way to classify each waste stream constituent. If the concentration of mercury in the waste stream varies significantly, the estimate of mercury has the potential to be greater than, less than, or approximately equal to the actual mercury total mass.

A.6.2 Profiles

Waste stream profiles are "recertified" by the disposal site each year. Once a profile has been established, the waste profile remains "certified" unless the process(es) that generate the waste(s) have changed. Complex and changing refinery operations (often on a day-to-day basis) such as new projects, variations in feedstock, market-driven changes in product

formulations and demand, and product additives can create the need for "recertification." In addition, multi-year maintenance turnaround schedules, unplanned upsets, and unit downtime may introduce variable waste streams that may not fully reflect the current certified waste profile.

A.6.3 Time

The timeframe chosen for assessing mercury in waste streams should account for data types, process, sampling and measurement variability, maintenance and turnaround activities, and raw material inputs, products, by-products, and waste products that occur or are consumed or produced over a specific time span. Raw material inputs vary over time based on availability, price, and production requirements. Products and by-products vary according to market, and regulatory demand and requirements. Waste streams may require 5 to 10 years or longer between scheduled maintenance activities to capture the majority of the disposed mass for a given constituent.

Considering the process and requirements for assessing refinery waste streams, the analytical profile methodology for quantifying constituent content, and the time duration of waste stream generation as compared to this quantification, it would be reasonable to find that the TRI reports reflect greater mercury mass than what might be found in the annual estimates of mercury in the crude.

A.7 Mercury Content of Refined Products

A literature search was conducted to assess published information available that would characterize the mercury content in the most commonly produced products produced by the Bay Area refineries. The most commonly produced products include gasoline (both California and non-California blends), diesel fuel, aviation fuel, and fuel oil. The annual mercury mass for these streams from the five SFB Area refineries in aggregate is estimated to be 26 kg/yr.

The literature survey produced nine studies that are detailed Section 4. Of note are two literature sources specific to the Bay Area:

- "Estimate of mercury emission from gasoline and diesel fuel consumption, San Francisco Bay area, California" (Conaway et al., 2005).
- "Trace Element and PAH of Jet Engine Fuels"(US Navy, 2000).

Table A-8 below provides a summary of the mercury content in the most commonly produced, refined products. The mercury content from the literature was combined with associated product volumes for the five SFB Area refineries obtained from the California Energy Commission (CEC) to produce mass estimates.

Table A-8 Average Mercury in Refined Products

Product	kg/year
Aviation Fuel	02.4
Diesel	02.0
Fuel Oil	13.0
Gasoline, CA	07.7
Gasoline, non-CA	00.99
Total	26.0

The most common products accounted for 85 to 92% of all product categories from the SFB Area refineries. The refineries were polled to determine categories of less or least common, including lube oils, bunker fuel, asphalt, propanes, and butanes. As much as the individual percent of each non-predominant product can vary from week to week, in aggregate they ranged from 8 to 15%.

Data were obtained from the CEC reflecting annual volumes of the five predominant products from the SFB Area refineries, and are shown in Table A-9 below.

Table A-9 CEC Product Volumes

Predominant Product	Volume (barrels/year)
Aviation Fuel	38,098,000
Diesel	42,183,000
Fuel Oil	16,627,000
Gasoline, CA	139,998,000
Gasoline, non-CA	18,043,000

To produce mercury mass estimates, most common product densities were obtained from literature references and a conversion calculation was performed. It was assumed that the density of non-California gasoline was equivalent to that of California gasoline.

Table A-10 below provides a more detailed overview of mercury in refined products, with values rounded to two significant figures here and in Table A-6. Mercury content was combined with associated product volumes to produce mercury mass estimates. Based on the range of results, the average provides a suitable estimate for the SFB Area refineries. Mercury content values for gasoline and diesel in the literature were produced from studies in the SFB region, and those for aviation fuel from a California study conducted by the US Navy.

Table A-10 Detailed Mercury in Refined Products

Predominant Product	Low Hg (kg/year)	High Hg (kg/year)	Average Hg (kg/year)
Aviation Fuel	0.01	04.9	02.4
Diesel	0.29	05.7	02.0
Fuel Oil	1.60	21.0	13.0
Gasoline, CA	1.30	22.0	07.7
Gasoline, non-CA	0.16	02.8	0.99
Total	3.40	56.0	26.0

The amount of mercury in predominantly refined products is estimated to be 26 kg/yr. These products make up approximately 85 to 92% of product streams in the refineries, where the non-predominant products mentioned above make up the remaining 8 to 15%.

A.8 Mercury Content in Water Discharges

The amount of mercury allocated to each facility by the respective National Pollutant Discharge Elimination System (NPDES) permits was used to calculate an estimate of the amount of mercury in the wastewater discharge. The aggregate amount of mercury that can be emitted according to the NPDES permits is 0.91 kg/yr. This represents a conservative estimate of the mass of mercury because the refineries typically operate below the respective limits to stay in compliance with the permit.

A.9 References

Avogadro Group. (2009). Telephone conversation between Dan Duncan of Avogadro Group and Eric Rivero-Montes of ERM. 15 January.

Conaway CH, Mason RP, Steding DJ, Russell Flegal A. (2005) Estimate of mercury emission from gasoline and diesel fuel consumption, San Francisco Bay area, California. Atmospheric Environment 39: 101-105.

Hwang, J. David, PhD. 2008. Hg Analyzer and Analysis Evaluation. 28 April.

US Navy. (2000). Trace Element and PAH of Jet Engine Fuels. December.

Appendix B
Literature Review

APPENDIX B LITERATURE REVIEW

Flow Science conducted a review of relevant literature to evaluate natural and anthropogenic emissions of mercury and rates of atmospheric deposition of mercury in the SFB Area and the US, and to assess the overall mercury budget of the SFB. In the remainder of this document, the terms SFB and San Francisco Estuary are used interchangeably. Considering the focus of this report on evaluating emissions of mercury from SFB Area refineries, the following literature review emphasizes studies dealing with atmospheric deposition and emission of mercury in the SFB Area.

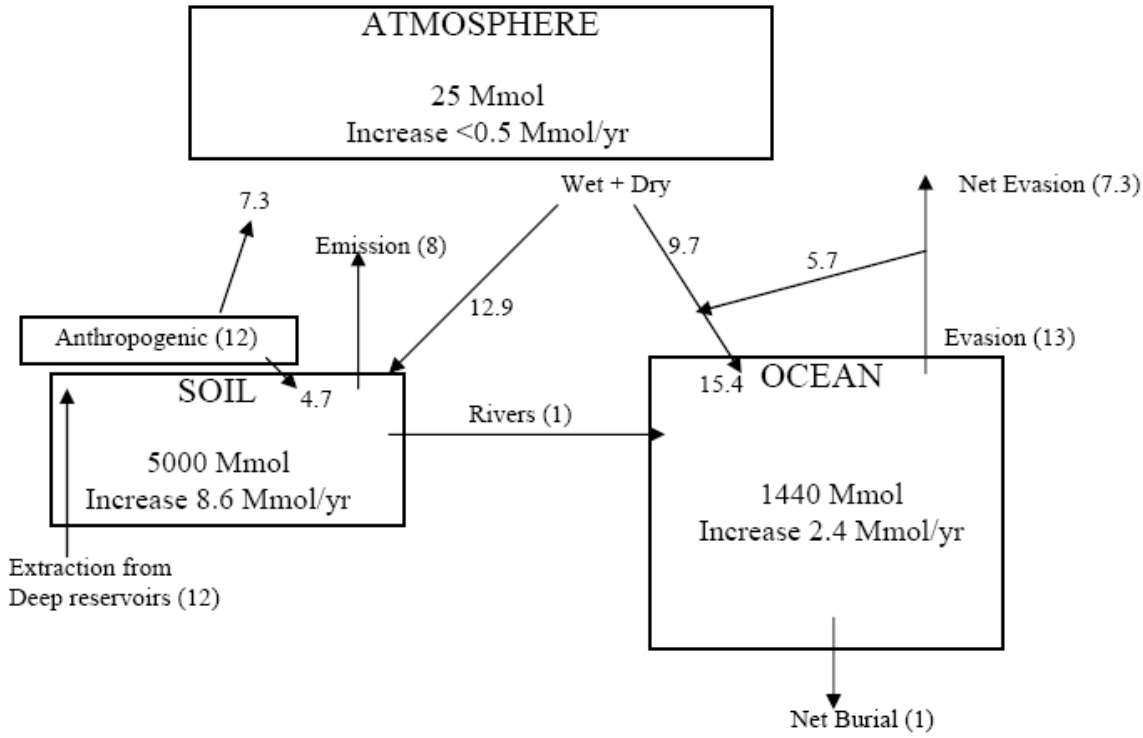
B.1 Overview of the Global Mercury Cycle

The study of Lindberg et al. (2007) and the report by United Nations Environment Programme (UNEP 2002) provide a comprehensive overview of the scientific literature on the sources, sinks, and transformations of mercury in the environment. The following material is drawn from these studies.

Mercury occurs naturally in the environment with some of the sources of atmospheric mercury being naturally enriched soils, forest fires, oceans, volcanoes, and geothermal areas. Mercury is also released to the environment due to human activities such as mining (including legacy mining) and industrial activities such as cement production, municipal waste incineration, chlor-alkali production, and fuel combustion. Mercury is rarely found in the nature as the pure, liquid metal, but rather within inorganic compounds (such as mercury sulfide [HgS] and mercury chloride [HgCl2]) and as organic mercury compounds (such as methylmercury). The most common organic mercury compound that microorganisms and natural processes generate from other forms is methylmercury, which can accumulate in aquatic animals to levels that are thousands of times greater than levels in the surrounding water. Mercury has a long retention time in soils (up to hundreds of years) and mercury accumulated in soil could be released from soils and/or sediments to surface waters and other media for prolonged periods.

Model-based estimates indicate that global mercury deposition increased by two to four times in modern times compared to the pre-industrial times. Model-based estimates also indicate that human-induced recycling, natural emissions (from land and oceans), and new point-source releases each account for about a third of the total present-day atmospheric inputs of mercury (see Figure B-1). Although current emissions from North America and Europe are decreasing, there has been no discernible net change in the size of the atmospheric pool of mercury in the Northern Hemisphere since the mid-1970s.

Figure B-1 Estimates of Present Day Global Mercury Reservoirs (Mmol) and Fluxes (Mmol/yr) from these Reservoirs



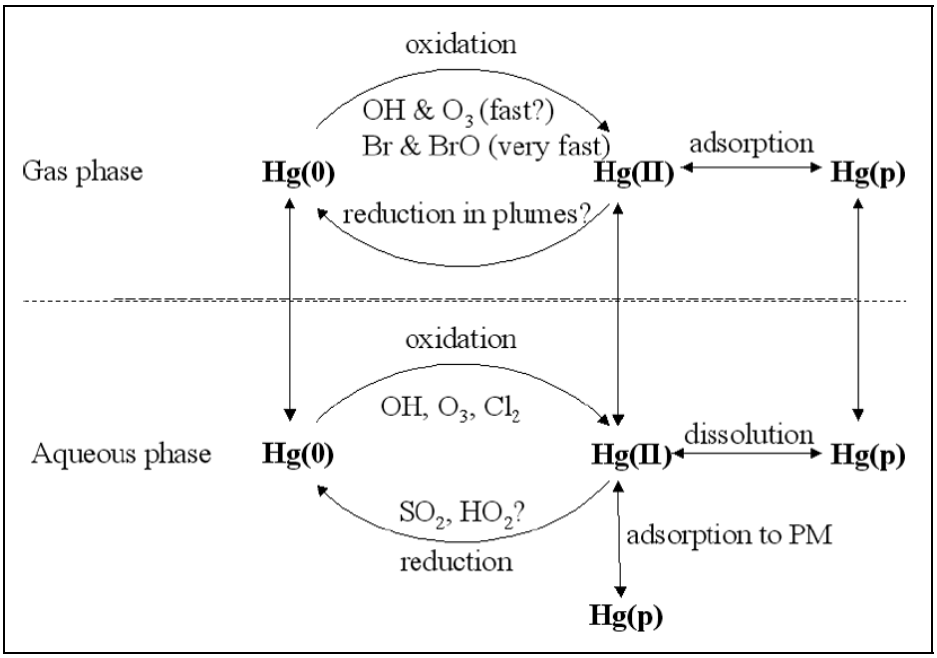
Notes:

- Flux of volatile mercury from water to air is termed "Evasion" in the above figure.
- Modified from Lindberg et al. (2007)
- Mmol = 10^3 kg-mole

Mercury can exist in the atmosphere in elemental (Hg(0)), divalent (Hg(II)), and particulate (Hg(p)) forms. Elemental mercury in the atmosphere can undergo transformation into inorganic mercury forms, providing a significant pathway for deposition of emitted elemental mercury (see Figure B-2). Among the three predominant species of mercury, Hg(0) is removed slowly from the atmosphere via wet and dry deposition, and Hg(II) species are removed rapidly because of their high solubility and reactivity. Thus, chemical transformation between Hg(0) and Hg(II) can affect the atmospheric lifetime of mercury.

The primary mechanisms for wet deposition of mercury are oxidation of Hg(0) by ozone (O_3) in clouds and the gas-phase oxidation of Hg(0) by OH and O_3 followed by an uptake of cloud droplets. Wet deposition of mercury occurs predominantly as dissolved and particulate Hg(II), with methylmercury occurring in minimal concentrations (~0.5-2.5% of total). Dry deposition of mercury mainly occurs by direct deposition of gas-phase Hg(0) and Hg(II) and, to a lesser extent, by deposition of atmospheric particles containing mercury. Measurement or model estimation of dry deposition is complicated by the fact that dry deposition processes are a complex function of meteorological factors (temperature, wind speed, etc.), surface wetness, and surface geomorphology. Estimates of dry deposition in forested catchments in the northern hemisphere were found to equal or exceed estimates of wet deposition.

Figure B-2 Schematic Illustrating the Atmospheric Chemical Transformations Between Elemental (Hg(0)), Divalent (Hg(II)) and Particulate (Hg(p)) Mercury in Gaseous and Aqueous Phases



Notes:

- "?" indicates uncertainties in these processes.
- Source: Lindberg et al. (2007).

B.2 Mercury Emissions from Anthropogenic and Natural Sources

B.2.1 Mercury Emissions Reported by the California Air Resources Board

The California Toxics Inventory, from the CARB, provides estimates of mercury emissions (and emissions of other toxic substances) from stationary, area-wide (associated with human activity and emissions taking place over a wide geographic area, such as unpaved road dust, wildfires, etc.), mobile (both off-road and on-road), and natural sources. The total emissions of mercury from the above sources for each of the nine counties comprising the San Francisco Bay Area Air Quality Management District (BAAQMD), for the year 2004 and 2006, are presented in Table B-1. Compared to the statewide average annual emission values of 44.6 and 37.1 micrograms per square meter per year ($\mu g/m^2/yr$), the average annual emissions from the SFB were 79.7 and 81.7 $\mu g/m^2/yr$, for the years 2004 and 2006, respectively.

Table B-1 Reported Total Emissions of Mercury (tons/yr) from the Nine Counties Comprising the BAAQMD for the Years 2006 and 2004

Region	area[a] (km^2)	2006 Emissions tons/yr	2006 Emissions $\mu g/m^2/yr$	2004 Emissions tons/yr	2004 Emissions $\mu g/m^2/yr$
BAAQMD					
Alameda County	1910	0.29	153.2	0.22	113.9
Contra Costa County	1865	0.32	169.3	0.30	160.1
Marin County	1346	0.05	33.5	0.04	32.9
Napa County	1952	0.04	21.3	0.04	21.6
San Francisco County	121	0.09	734.8	0.08	664.7
San Mateo County	1163	0.12	106.5	0.12	103.3
Santa Clara County	3343	0.39	115.7	0.45	133.6
Solano County	2148	0.10	45.6	0.11	50.9
Sonoma County	4081	0.07	17.7	0.07	17.3
Area-weighted average	-	-	81.7	-	79.7
Entire California	403928	15	37.1	18	44.6

Notes:

- a - Areas were obtained from http://quickfacts.census.gov/qfd/index.html
- b - Area-weighted average emissions were computed in units of $\mu g/m^2/yr$, consistent with the units of other estimates presented in this report.
- Source of the data is http://www.arb.ca.gov/toxics/cti/cti.htm.

B.2.2 Mercury Emissions from Anthropogenic Sources in the United States

In 1990, more than half of the emissions of mercury from the US were from coal-fired power plants, municipal solid waste combustors, and medical waste incinerators (USEPA 2006). Due to increased regulation, overall mercury emissions to the atmosphere from the US have decreased by about 45% from 1990 (199 tons) to 1999 (102 tons) (see Table B-2).

Table B-2 Estimated Emissions of Mercury (tons/yr)[a] in the US from Various Anthropogenic Sources

Source	1990	1994-95	1999
Utility Coal Boilers	46.3	47.2	43.4
Industrial Boilers	10.9	-	10.9
Medical Waste Incinerators	45.1	14.6	1.5
Municipal Waste Combustion	51.4	26.9	4.4
Hazardous Waste Incinerators	6.0	-	6.0
Chlorine Production	9.1	6.5	5.9
Electric Arc Furnaces	6.3	-	NA
Gold Mining	3.1	-	10.4
Other	21.3	-	19.6
Total	199.4	144.0	102.1

Notes:

- a - Original data for 1990 and 1999 were converted to metric tons per year (1000 kilograms [kg]/year [yr]) and presented here.
- Source of the data for 1990 and 1999 is USEPA (2006) and for 1994-95 is USEPA (1997).

B.2.3 Mercury Emissions from Fuel Consumption in the San Francisco Bay Area

Conaway et al. (2005) collected gasoline and diesel samples from refineries and service stations in the SFB Area and found that mercury concentrations in gasoline ranged from 0.08 to 1.4 nanograms per gram (ng/g) and those in diesel ranged from 0.05 to 0.34 ng/g. On average, combustion of these fuels in the SFB Area was estimated to contribute 5 kg/yr (range of 0.7 to 13 kg/yr) of mercury to the environment (see Table B-3). The authors estimated that the total flux from the combustion of these fuels, assuming that all of the resulting emissions reach the atmosphere, represents less than 3% of the total atmospheric emissions (including those from medical waste incinerators, cement manufacturing and geothermal power production) in the SFB Area. Considering that reported concentrations of mercury in crude oil are generally much higher than those found in the study samples, the authors suggested that there is little evidence that mercury is enriched into these fuels from the refining process.

Table B-3 Gasoline Consumption and the Corresponding Estimated Mercury Emissions to the Environment (kg/yr) in the San Francisco Bay Area for a Range of Mercury Concentrations Measured in Gasoline Samples

year	Gasoline consumption (x 10^9 liters)	Total emission (kg/yr)[a]			emission flux[b] ($\mu g/m^2/yr$)
		Low concentration (0.08 ng/g)	average concentration (0.50 ng/g)	High concentration (1.4 ng/g)	
1993	11	0.6	4	10	0.03-0.56
1994	11	0.6	4	10	0.03-0.56
1995	11	0.6	4	10	0.03-0.56
1996	11	0.6	4	11	0.03-0.61
1997	12	0.7	4	11	0.04-0.61
1998	12	0.7	4	12	0.04-0.67
1999	12	0.7	4	12	0.04-0.67
2000	13	0.7	5	12	0.04-0.67
2001	13	0.7	5	13	0.04-0.73

Notes:

- a - Total emission was estimated by the authors after converting the consumption volume to a mass (by multiplying with a density of 0.7 g/cm^3) and multiplying it by the corresponding concentration.
- b - Average emission flux, in units of $\mu g/m^2/yr$, consistent with the units of other estimates presented in this report, was obtained by dividing the emissions (in kg/yr) by the total area of the nine counties mentioned above (17929 kilometers $[km]^2$).
- San Francisco Bay Area comprises the counties of Alameda, Contra Costa, Marin, Napa, San Francisco, San Mateo, Santa Clara, Solano, and Sonoma.
- Source: Conaway et al. (2005).

B.2.4 Mercury Emissions from Wildfires

Although vegetation and soil are sinks for atmospheric mercury, mercury contained in vegetation and soil can be released to the environment during wildfires. Wiedinmyer and Friedli (2007) estimated that annual average emissions of mercury from fires were about 30% of the total anthropogenic emissions in the US, with the highest monthly averages in western states such as California (see Table B-4). The authors indicated that, on an annual basis, mercury emissions from forest fires dominate the inventory of fire emissions and that agricultural fire emissions are relatively insignificant.

Table B-4 Average Monthly Emission of Mercury (kg/month) from Fires in California and the US for the Period 2002-2006

Month	Emissions (kg/month)	
	California	United States
JAN	17	800
FEB	37	900
MAR	58	2600
APR	68	2800
MAY	73	1500
JUN	160	4800
JUL	513	8000
AUG	600	11100
SEP	974	4800
OCT	709	3800
NOV	173	1800
DEC	56	800
Total (kg/yr)	3438	43700

Source: Wiedinmyer and Friedli (2007).

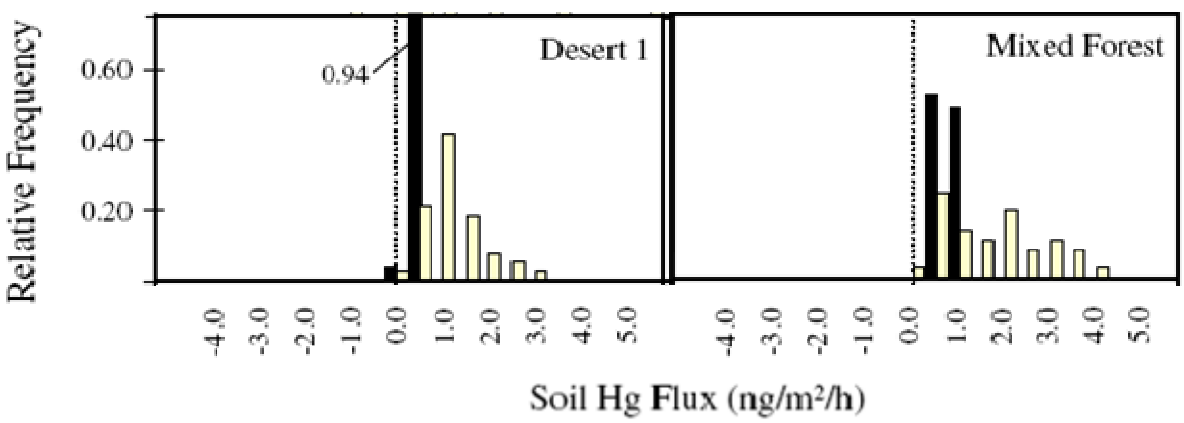
Friedli et al. (2003) measured mercury concentrations in a wildfire (177 hectares of final burn area) plume in the Ontario region of Canada and found that most of the emitted mercury was elemental mercury in the gaseous form (0.56 ng/m^3 of Hg(0), compared to less than 0.083 ng/m^3 of Hg(p)). Artaxo et al. (2000) measured mercury and trace metal concentrations in ambient air over the Amazon Basin in Brazil from August to September 1995, a time period coinciding with the peak biomass burning season. The authors found that, over areas that are heavily impacted by mining or biomass burning, mercury concentrations ranged from 5 to 14 ng/m^3, whereas the concentrations over pristine areas and areas not downwind of mining activities ranged from 0.5 to 2 ng/m^3. The authors estimated that biomass burning contributed to about 31% of the measured mercury concentration. Adsorption of gaseous mercury on existing biomass burning particles, direct release of mercury from the vegetation to the atmosphere, and evaporation of mercury from soil were suggested as some of the mechanisms by which mercury is released during forest fires.

B.2.5 Mercury Emissions from Natural and Other Sources

Nacht et al. (2004) estimated the mean flux of mercury from the Sulphur Bank Mercury Mine Superfund site located in the Sonoma-Clear Lake area, about 110 km northwest of Sacramento (total study area of 3.8 km^2), an area of extensive mercury enrichment due to historical mining and geothermal activity. The authors found that the mean flux of mercury (only the elemental form, Hg(0)) from this site ranged from 14 to 11,000 nanograms per square meter per hour (ng/m^2/h), and that the estimated total annual emission flux of mercury was 17 kg/yr. This estimate of annual flux reported by the authors is equivalent to an emission flux of 4473 µg/m^2/yr, and is two orders of magnitude higher than the reported estimates of average atmospheric deposition flux at several locations in the US (see Section B.3.4). The authors estimated that about 47% of the total emissions originated from anthropogenically disturbed material (mine waste, ore, etc.) and the remainder from the undisturbed naturally enriched substrate.

Ericksen et al. (2006) measured mercury fluxes from bare soils with low mercury concentrations (<0.1 mg/kg) at 11 sites (situated in varying ecosystem types) across the US. Two of the 11 sampling sites were in California, including a site in Northern California (in the foothills of the Sierra Nevada, about 100 km north of Sacramento) situated in a mixed forest ecosystem, and a site in Southern California (in the Mojave Desert) situated in a desert ecosystem. The authors observed that, for most of the sites, soil fluxes during dark conditions (<1 W/m^2) were significantly lower than those during light conditions (>1 W/m^2) (see Figure B-3). The authors indicated that there were significant differences in the soil fluxes between the sampling sites and that no consistent trends of soil flux as a function of the ecosystem type were identified. However, based on limited data available from one semiarid site in Nevada, the authors found significant correlation between soil moisture content and soil mercury flux. The mean soil flux at the Southern California and the Northern California sites was found to be 0.6±0.6 ng/m^2/h and 1.1±1.0 ng/m^2/h, respectively. Soil fluxes at the Southern California site (a very arid desert ecosystem) were found to be significantly different from those at a semiarid desert site in Nevada, although the two sites had similar air and soil mercury concentrations, possibly due to differences in the soil moisture content at these sites. The authors estimated that the soil flux averaged across all the sites and averaged over light and dark conditions was 0.9±0.7 ng/m^2/h (equivalent to 1.8 to 14 µg/m^2/yr). Thus, the "background" flux of mercury from bare soils is comparable to the reported estimates of average atmospheric deposition flux at several locations in the US (see Section B.3.4). By scaling up their measurements and making other simplifying assumptions, the authors estimated the background mercury flux from soils in the contiguous US to be approximately 100 tons/yr.

Figure B-3 Histograms of Relative Frequency of Soil Mercury Flux



Notes:

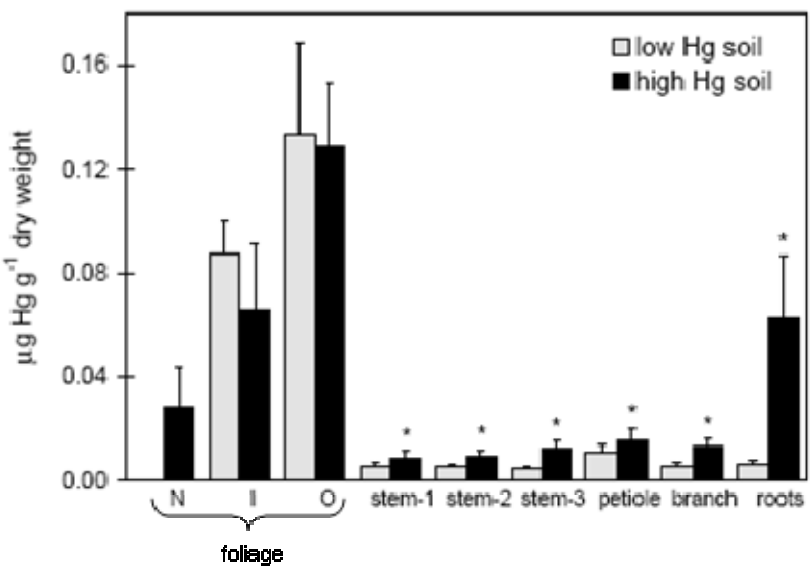
- Soil mercury conditions shown as dark (dark shaded bars) and light (light shaded bars).
- Southern California sites are labeled "Desert 1."
- Northern California sites are labeled "Mixed Forest."
- Negative flux indicates deposition.
- Modified from Ericksen et al. (2006).

B.2.6 Mercury Exchange between Vegetation and the Atmosphere

In a controlled experimental environment, Ericksen et al. (2003) measured exchange of mercury between air and vegetative surfaces of aspen stands (including foliage, stems, branches and roots) grown in soils with high (12 µg/g) and low (0.03 µg/g) mercury concentrations. The authors indicated that foliar mercury concentrations increased as a function of leaf age and that the concentration of mercury in foliage grown in low mercury soil was similar to the concentration in those grown in high mercury soil (see Figure B-4). Thus, uptake of gaseous

Hg(0) was the predominant pathway by which mercury accumulated in the foliage. However, mercury concentration in other plant compartments (stems, braches, roots, etc.) was higher in the aspen grown in high mercury soil than in the aspen grown in low mercury soil. The authors indicated that because of leaf senescence, approximately 80% of mercury in the aboveground biomass was found in the litterfall by the end of their experiment. Thus, litterfall would serve as an input of mercury to terrestrial ecosystems. Moreover, the authors suggested that litterfall measurements could be useful in estimating mercury deposition rates in deciduous forests, since incident precipitation, the other route by which mercury could be removed, removes less than 3% of the mercury deposited on the foliage.

Figure B-4 Foliar and Non-foliar Mercury Concentrations in Aspen Grown in Low Hg Soil (0.03 µg/g, grey bars) and High Hg Soil (12 µg/g, black bars) in 2001



Notes:

- Foliar Hg concentrations are shown for new ("N", <21 days old), intermediate ("I," 22-35 days old) and old ("O," >35 days old) leaves.
- Stems were sampled from the top 1/3 ("stem-1"), middle 1/3 ("stem-2") and bottom 1/3 ("stem-3") portions of the aspen stand.
- The "*" indicates that Hg content was statistically different between samples from low and high Hg soils.
- Modified from Ericksen et al. (2003).

B.3 Estimates of Atmospheric Deposition of Mercury

B.3.1 Wet Deposition Estimates from the Mercury Deposition Network

The Mercury Deposition Network (MDN) is a national database of routine mercury measurements that is maintained by National Atmospheric Deposition Program/National Trends Network (NADP/NTN) and is available online. The MDN reports weekly concentrations of total mercury in precipitation and the seasonal and annual flux of total mercury in wet deposition at selected locations throughout the US and Canada. The annual wet deposition flux for the period 1997 through 2007, estimated from event-based flux measurements within each calendar year, at two sites (CA 97, which is located in Covelo, about 270 km north of San Francisco, and CA 72, which is located in San Jose, about 70 km south of San Francisco), is presented in Table B-5. As expected, the annual wet deposition flux is correlated with the annual precipitation amount at both these sites.

Table B-5 Annual Wet Deposition Flux[a] of Mercury (µg/m^2/yr) and Corresponding Annual Precipitation Total[b] (in mm/yr) at Two MDN Sites (CA 97 in Covelo and CA 72 in San Jose) in Central California

	CA 97		CA 72	
year	Flux (µg/m^2/yr)	Precip. (mm/yr).	Flux (µg/m^2/yr)	Precip. (mm/yr).
1997	-	-	-	
1998	4.8	1211	-	
1999	4.3	837	-	
2000	3.8	639	3.1	318
2001	-	-	2.9	289
2002	-	-	2.2	242
2003	-	-	2.2	279
2004	-	-	2.3	255
2005	-	-	2.1	431
2006	-	-	2.5	310
2007	-	-	-	-

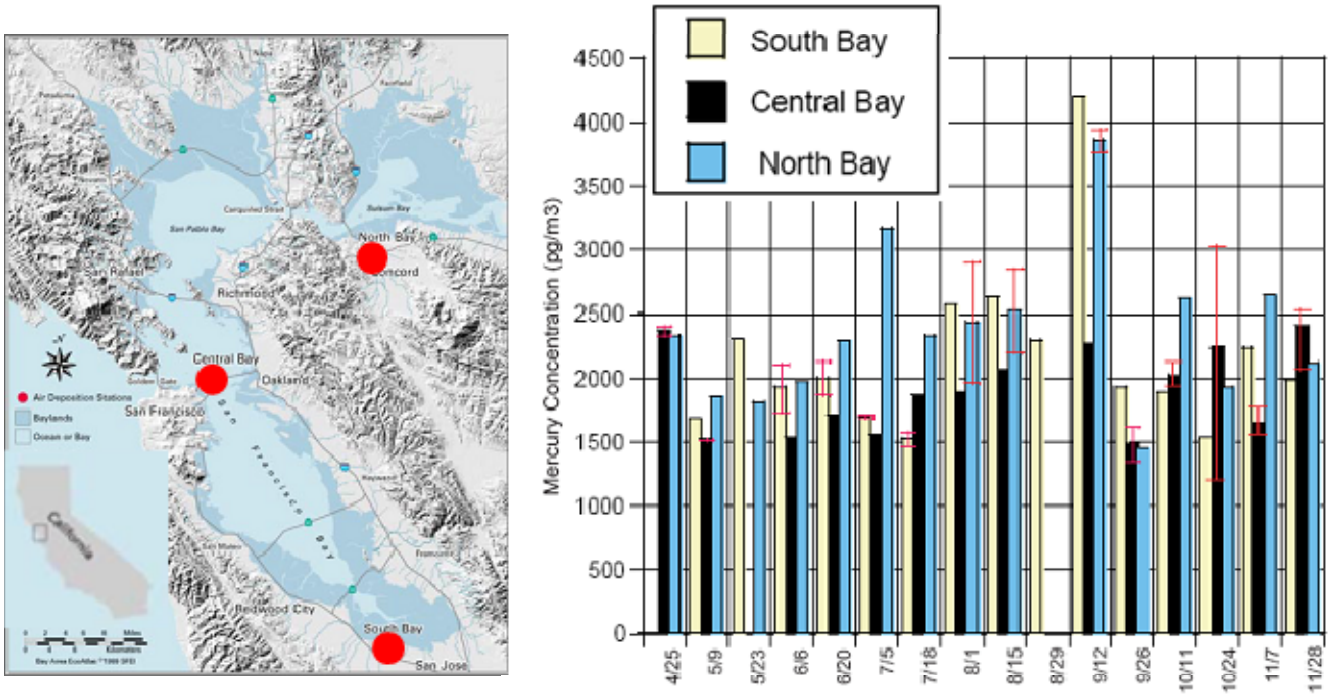
Notes:

- a - Annual wet deposition flux was assumed to be the sum of event-based estimates within the calendar year. Only those years for which event-based estimates spanned the complete calendar year were used.
- b - Event-based precipitation totals, corresponding to only those events for which flux estimates were available, were summed to obtain the annual total.
- Raw data are from http://nadp.sws.uiuc.edu/mdn/

B.3.2 San Francisco Bay Atmospheric Deposition Pilot Study (Tsai and Hoenicke 2001)

Tsai and Hoenicke (2001) used measurements of mercury concentration to estimate the deposition flux of mercury at three locations in the San Francisco Estuary (North Bay, Central Bay, and South Bay in Figure B-5) for the period August 1999 through November 2000. The North Bay site was situated in an industrial corridor, near several petroleum refineries, the Central Bay site was close to two major airports and seaports, and the South Bay site was downstream of several abandoned mines. The South Bay site also coincided with the MDN site (CA 72, see Section B.3.1) in San Jose.

Figure B-5 Modeling Sites in the San Francisco Bay Area



Notes:

- Left panel: Modeling sites (indicated by red circles) in the San Francisco Bay Area.
- Right panel: Mercury concentration (pg/m^3) in the ambient air in the San Francisco Bay Area in the year 2000 at the three study sites.
- Red whiskers on the bars indicate the range in these measurements.
- Source: Tsai and Hoenicke (2001).

The authors found that mercury concentrations in the ambient air at the North Bay site were statistically higher than those at the Central Bay site (see Figure B-5), but concentrations at the South Bay site were not statistically different from either the Central Bay or the North Bay sites. Using a flux deposition model, the authors estimated that the average annual dry deposition flux at North Bay, Central Bay and South Bay sites to be 21, 18 and 19 µg/m^2/yr, respectively (see Table B-6). From measurements of mercury concentrations in precipitation, the authors estimated that the wet deposition flux of mercury ranged from 3.5 to 4.5 µg/m^2/yr, with an overall estimated wet deposition rate of 4.2 µg/m^2/yr for the entire estuary. The authors indicated that uncertainties in some of the parameters used in the dry deposition flux model could be within 200% (of the values used within the model), and uncertainties in measurement of mercury concentration in precipitation could be within 25%. The authors suggested a 2- to 5-fold uncertainty in the estimates of overall atmospheric loading to the estuary (estimated as the sum of direct atmospheric deposition and indirect loading through inputs from upstream watersheds).

Table B-6 Estimated Flux (µg/m²/yr) of dry and Wet Deposition of Mercury and the Associated Direct Atmospheric Loading (kg/yr) at the Three Study Sites of Figure B-5

		Direct Atmospheric Loading (kg/yr)				Flux (µg/m²/yr)
		Hg(p)	Hg(II)	Hg(0)	Total	
Dry Deposition						
South Bay		2	6.6	0.57	9.2	19
Central Bay		0.8	2.6	0.38	3.8	18
North Bay		1.9	6.4	0.7	9	21
Entire Estuary		4.6	15	2.2	22	19
Wet Deposition	**Precip. (mm/yr)**					
South Bay	360	-	-	-	1.7	3.5
Central Bay	680	-	-	-	0.96	4.5
North Bay	580	-	-	-	1.8	4.3
Entire Estuary	530	-	-	-	4.8	4.2

Source: Tsai and Hoenicke (2001).

B.3.3 Mercury Concentrations in Coastal California Precipitation

Steding and Flegal (2002) used a model of air mass trajectory to identify the sources of mercury in rainwater samples collected from an urban site (Moffett Field) and a rural site (Long Marine Lab) in Central California. The authors suggested that a combination of Hg(0) emissions from Asia, a series of reduction-oxidation reactions resulting in increased production of atmospheric Hg(II), and the subsequent trans-Pacific transport of Hg(0) and Hg(II) result in higher mercury concentrations, compared to baseline concentrations from the equatorial Pacific, observed in the rainwater samples. The authors also argued that local sources of mercury result in elevated concentrations at the urban site (Moffett Field) relative to the rural site (Long Marine Lab). Estimated wet mercury deposition at these two sites is shown in Table B-7.

Table B-7 Estimated Annual Wet Deposition Flux of Mercury (µg/m²/yr) at Long Marine Lab (rural site) and Moffett Field (urban site) in Central California Based on Data Collected During 2000-2001

Site	Precip. (mm)	Flux (µg/m²/yr)
Moffett Field	350	4.4
Long Marine Lab	740	4.0

Notes:

- a - Although not all the events during the study period were sampled, the authors indicated that it did not result in biased estimates, as verified by comparisons with independent observations.
- b - Deposition values were converted from nanomoles (nmol)/m²/yr to µg/m²/yr.
- Source: Steding and Flegal (2002)b.

B.3.4 Mercury Deposition Estimates from Across the United States

A summary of reported mercury deposition estimates from locations across the US is presented below (see Table B-8). The overall range of wet deposition estimates is 1.48 to 30 µg/m²/yr and the overall range of dry deposition estimates is 0.4 to 15.8 µg/m²/yr.

Table B-8 Estimates of Wet and Dry Deposition Flux of Mercury (µg/m²/yr) at Locations Across the US

Geographical Area	Time Period	Wet/Dry	Flux (µg/m²/yr)	Source
Chesapeake Bay, MD	1997-1998	wet	14-30	Mason et al. (2000)
Lake Champlain basin, VT	1993	wet	9.26	Burke et al. (1995)
Sturgeon Point, NY[a]	Sep-Oct 1993	wet	9.7 (2.2-18.6)	Vermette et al. (1995)
Little Rock Lake, WI	----	wet	6.8±2	Fitzgerald et al. (1991)
MDN Site NV99, NV[b]	2005-2006	wet	2.28-7.2	Lyman et al. (2007)
MDN Site NV02, NV[b]	2005-2006	wet	1.48-3.32	Lyman et al. (2007)
overall range (wet)		**wet**	**1.48-30**	
Everglades, FL[d]	Feb-Mar 1999	dry	4.85±1.46	Marsik et al. (2007)
Everglades, FL[d]	June 2000	dry	2.15±1.02	Marsik et al. (2007)
MDN Site NV99, NV[c]	2005-2006	dry	1.44-10.8	Lyman et al. (2007)
MDN Site NV02, NV[b]	2005-2006	dry	0.4-15.8	Lyman et al. (2007)
Little Rock Lake, WI	----	dry	3.5±3	Fitzgerald et al. (1991)
Overall range (dry)		**dry**	**0.4-15.8**	**----**

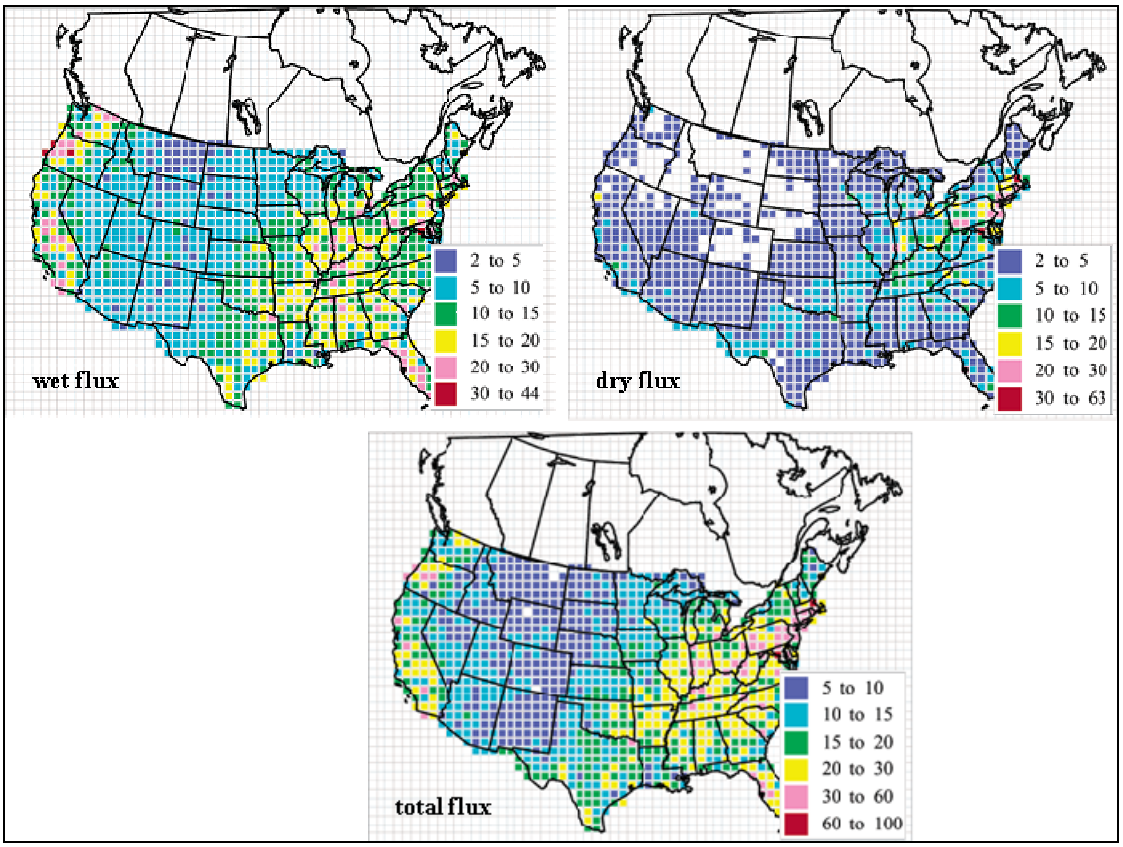
Notes:

- a - Reported weekly flux was converted to an annual flux.
- b - Seasonal estimates for spring, summer, fall and winter were converted to annual estimates, and the complete range of these annual estimates is presented.
- c - Seasonal estimates for spring, summer and fall were converted to annual estimates, and the complete range of these annual estimates is presented.
- d - Reported daily flux was converted to an annual flux.

B.3.5 Recent Large-Scale Modeling Estimates of Atmospheric Mercury Deposition

Seigneur et al. (2004) used a continental-scale, 3-D chemical transport model (CTM), nested within a global CTM, to simulate the global atmospheric fate and transport of mercury and the resulting deposition in the contiguous US. The spatial distribution of simulated total fluxes generally reflected the distribution of local/regional emission sources, precipitation patterns and prescribed upwind boundary conditions in the CTM. However, a comparison of simulated wet deposition fluxes for the year 1998 with observations from the NADP network indicated only a moderate agreement (coefficient of determination, r2, was 0.50). Simulated wet deposition fluxes (see Figure B-6) were highest in eastern US and in the coastal western US. The authors noted that the high wet deposition fluxes on the West Coast are due to the prescribed upwind boundary conditions of global Hg(II) concentrations in the continental CTM (i.e., transport of mercury across the Pacific) and the high precipitation along the mountain ranges of the Cascades and Sierra Nevada. The high dry deposition flux of 15-20 µg/m²/yr in northern California corresponds to the location of geothermal geysers, which were assumed to have relatively high emissions in the CTM. The authors also estimated that, on average, North American anthropogenic emissions contribute to 30% of the total mercury deposition in the contiguous US, while natural emissions and other regions account for 33% and 37%, respectively.

Figure B-6 Simulated Annual Wet, Dry, and Total Deposition Flux of Mercury (µg/m^2/yr) over the Contiguous US for Each 100 km by 100 km Grid Cell for the Year 1998

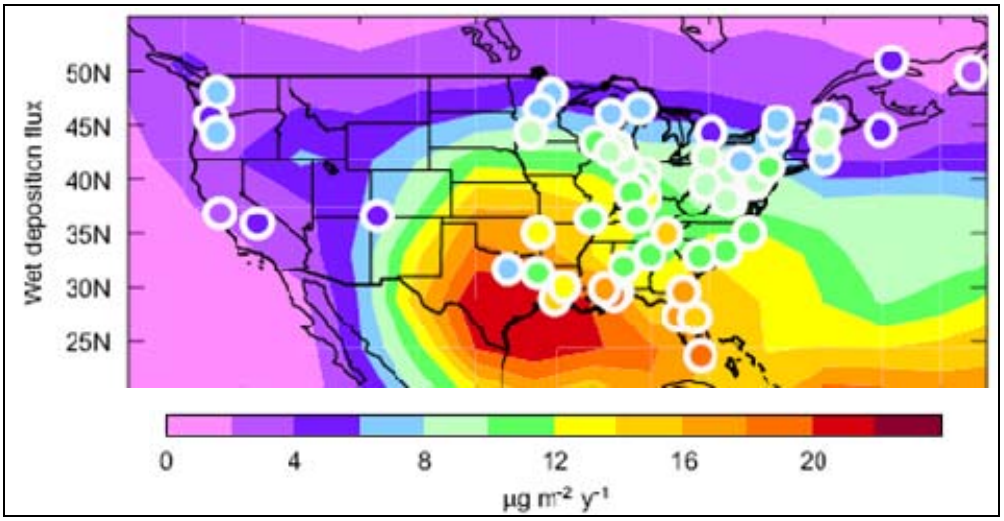


Notes:

- Simulated annual wet deposition flux of mercury shown in top left panel.
- Simulated dry deposition flux of mercury shown in top right panel.
- Simulated total deposition flux of mercury shown in bottom panel.
- Source: Seigneur et al. (2004).

Using a global 3-D CTM (GEOS-Chem), Selin and Jacob (2008) simulated wet and dry deposition fluxes of mercury for the contiguous US. Comparison of simulated wet fluxes for the period 2004-2005 with observations from the NADP network (see Figure B-7) indicated a reasonable agreement (coefficient of determination, r2, was 0.73). The authors indicated that GEOS-Chem also reasonably captured the observed latitudinal gradient and seasonal phase as well as the variation of seasonal amplitude with latitude. North American anthropogenic emissions were estimated to contribute 20% of the simulated total mercury deposition in the contiguous US, and dry deposition fluxes were estimated to account for 70% of the simulated total deposition.

Figure B-7 Simulated Annual Mean Wet Deposition Flux of Mercury (µg/m^2/yr) for the Period 2004-2005 (model grid resolution of 400 km x 500 km)



Notes:

- The circles show the 57 NADP sites where simulations were evaluated against observations.
- Source: Selin and Jacob (2008).

B.3.6 Comparison of Pre- and Post-Industrial Atmospheric Deposition

Sanders et al. (2008) measured total mercury concentration in sediment cores from four lakes in California that were considered to be devoid of any present or historical anthropogenic activity. The four lakes selected by the authors were the Castle Lake in Shasta-Trinity National Forest, Wildcat Lake in Point Reyes National Seashore, Island Lake in Tahoe National Forest and Emerald Lake in Sequoia and Kings Canyon National Park. The authors found that sediment mercury concentrations for the 1970-2004 (modern) time period are higher by a factor of 10 (Castle Lake), 5 (Emerald Lake), 4 (Island Lake), and 2 (Wildcat Lake) than those for the pre-1850 (pre-industrial) period. The authors suggested that increased atmospheric mercury deposition, increased atmospheric nitrogen inputs resulting in elevated primary productivity, and increased carbon inputs are the possible mechanisms to explain the enrichment of mercury in the modern lake sediments. The authors found statistically significant correlations between mercury concentrations and percentage organic material in only two of the four lakes. Thus, the source of the enriched mercury in modern sediments could not be determined. Nonetheless, the authors indicated that mercury enrichment in modern sediments found in this study is generally higher than the 3-time increase in modern atmospheric deposition rate found in lake sediments of northern Alaska by Fitzgerald et al. (2005) and the 2- to 4-time increase in modern atmospheric deposition reported by Lindberg et al. (2007).

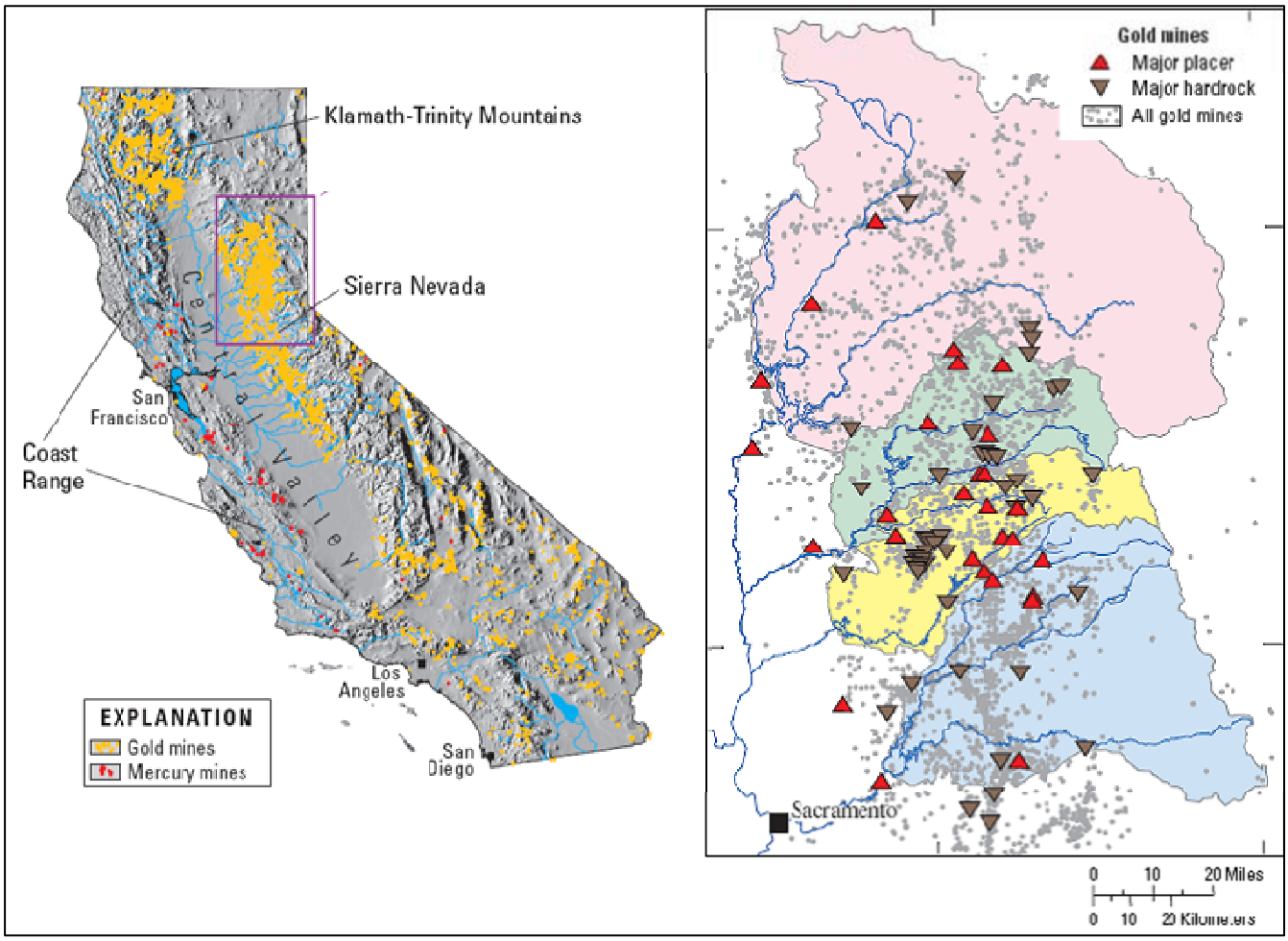
B.4 Mercury Concentrations in Sediments

Numerous studies in the scientific literature have addressed mercury contamination in the SFB and other related issues. For instance, see Conaway et al. (2007), McKee et al. (2005), Hornberger et al. (1999) and the references therein. In this section, only selected studies from the literature that are directly relevant to the broader scope of this report are discussed.

The SFB is affected not only by natural weathering of highly mercury-mineralized rocks in the upstream watersheds, but also by historical gold and mercury mining in the upstream watersheds (see Figure B-8). Between 1850 and 1900, most of the mercury mining in the world

occurred in the upstream watersheds of the SFB. Mercury was used to enhance gold recovery in the gold mining operations that occurred extensively in the mid to late 19th Century, and mercury-contaminated sediments from the Sierra Nevada were largely deposited to the northern reach of the SFB. Seasonal variability of mercury concentrations in water and sediment of the estuary are strongly dependent on the freshwater discharges from the watersheds draining into the estuary. Long-term (decadal) variability in sediment mercury concentration is a function of sediment deposition from the Central Valley, erosion of buried sediments resulting in enhanced active sediment layer, and mixing processes.

Figure B-8 Locations of Historical Gold and Mercury Mines in California



Notes:

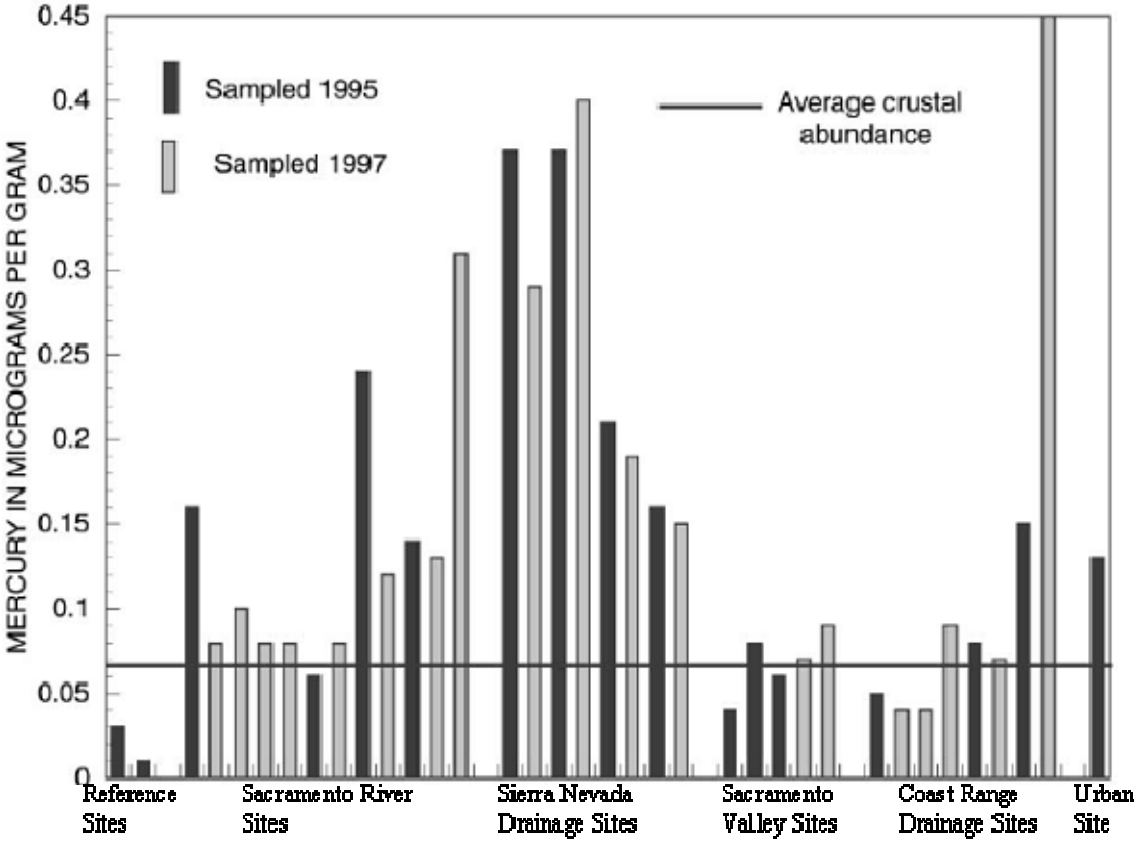
- Inset shows historical gold mines and major placer and hardrock gold mines in the northwestern Sierra Nevada.
- Source: Alpers et al. (2005).

B.4.1 Mercury in Water and Sediments of the Sacramento River Basin and the Sacramento-San Joaquin River Delta

Domagalski (2001) measured mercury and methylmercury concentrations in water and streambed sediments at 27 locations within the Sacramento River Basin. Compared to the global crustal abundance of mercury of 0.067 µg/g, the author found elevated mercury

concentrations in streambed sediments from areas downstream of historical mining activity (such as Sierra Nevada sites, see Figure B-9). Methylmercury concentrations in streambed sediments ranged from 0.27 to 2.84 ng/g, and the highest concentration measured at the Sacramento Slough was attributed to the longer water residence times (about a few months), anoxic conditions in sediments, and the availability of organic material for microbial processes, all of which are ideal conditions for mercury methylation. Mercury and suspended sediment concentrations in unfiltered water followed the seasonal rainfall patterns, with the highest concentrations in winter (wet season) and relatively lower concentrations in late spring and fall (dry season). The highest loading of mercury to the SFB was attributed to sources within the Cache Creek watershed, which are downstream of historical mines, and to a possible volcanic source. The author also indicated that although mercury concentrations in Cache Creek could be as high as 2,248 nanograms per liter (ng/L) during storm water runoff events, the transported mercury was found to have a low potential for geochemical transformations, as the transported mercury was largely not present in a free ion form or in a form that could easily be reduced to elemental mercury. The author observed that it was only during high streamflow conditions that mercury in unfiltered water exceeded the federal and state recommended criterion for protection of aquatic life (as of 2001, 50 ng/L as total Hg in unfiltered water). The author estimated that the total amount of mercury transported from the Sacramento River Basin to the SFB to be 487 kg for the winter of 1996-97 and to be 169 kg for the winter of 1997-98.

Figure B-9 Concentrations of Mercury (µg/g) in Streambed Sediment of the Sacramento River Basin



Notes:

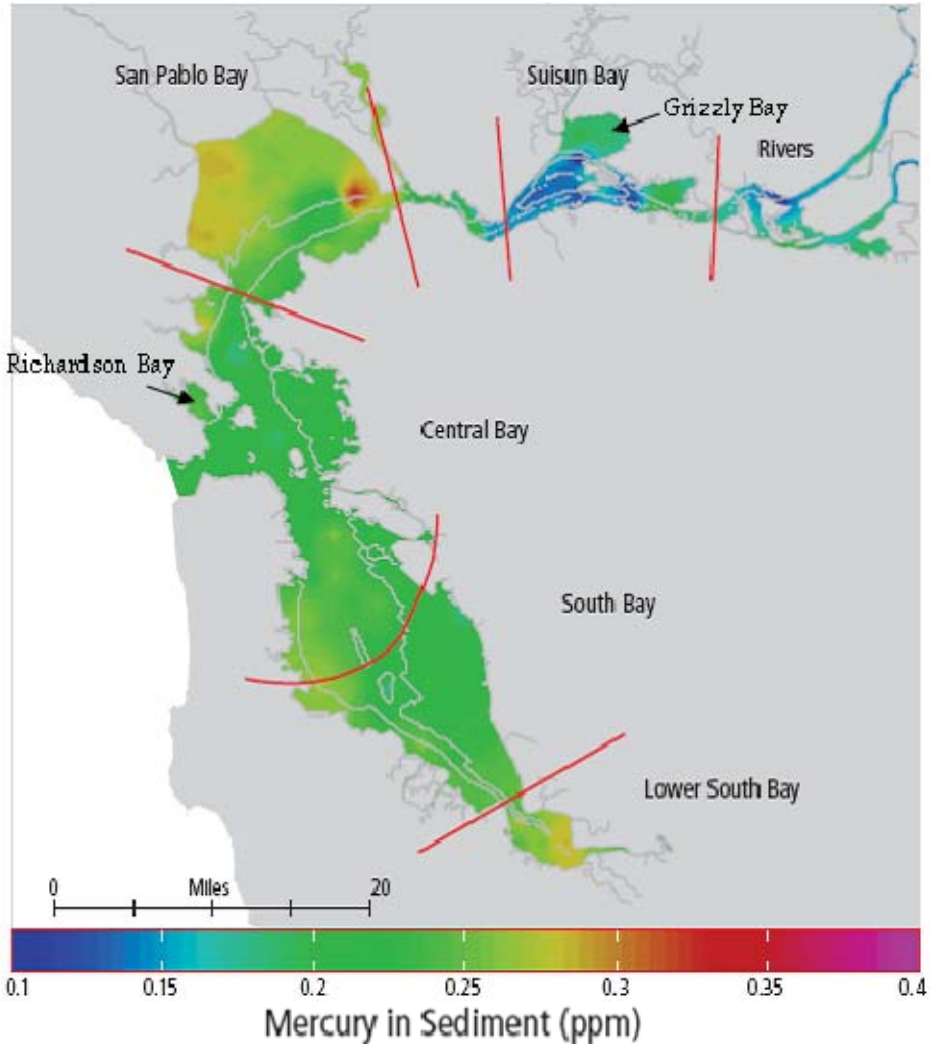
- Sampled during 1995 (black bars) and 1997 (grey bars) at 27 locations within the Basin.

- Modified from Domagalski (2001).

McKee et al. (2005) measured total mercury concentrations in water samples collected during 2002 and 2003 at Mallard Island in the Sacramento-San Joaquin Delta, about 8 km downstream of the confluence of Sacramento and San Joaquin Rivers. The authors observed that total mercury concentrations ranged from 4 to 14 ng/L and dissolved mercury concentrations ranged from 0.8 to 1.6 ng/L. The authors indicated that the Delta exhibits a "first flush" phenomenon for mercury indicated by greater mercury concentrations during the first flood of a season, despite the subsequent floods having a greater flow rates. Similar to Domagalski (2001), the authors found a relationship between suspended sediment concentration and total mercury concentration. Using this relationship and adjusting for the tidal effects on water transport (and hence mercury) within the Delta, the authors estimated that daily total mercury loads varied from 3 to 1,803 g/day and that the annual total mercury load was 58±20 and 97±33 kg/yr for water years 2002 and 2003, respectively. The authors extended their estimates to obtain a long-term average annual mercury load of 201±68 kg/yr.

B.4.2 Mercury in Sediments and Fish of the San Francisco Bay

Figure B-10 Mercury Concentration in Sediments (ppm) of the San Francisco Bay Based on Data Collected by the San Francisco Estuary Institute from 2002 to 2007

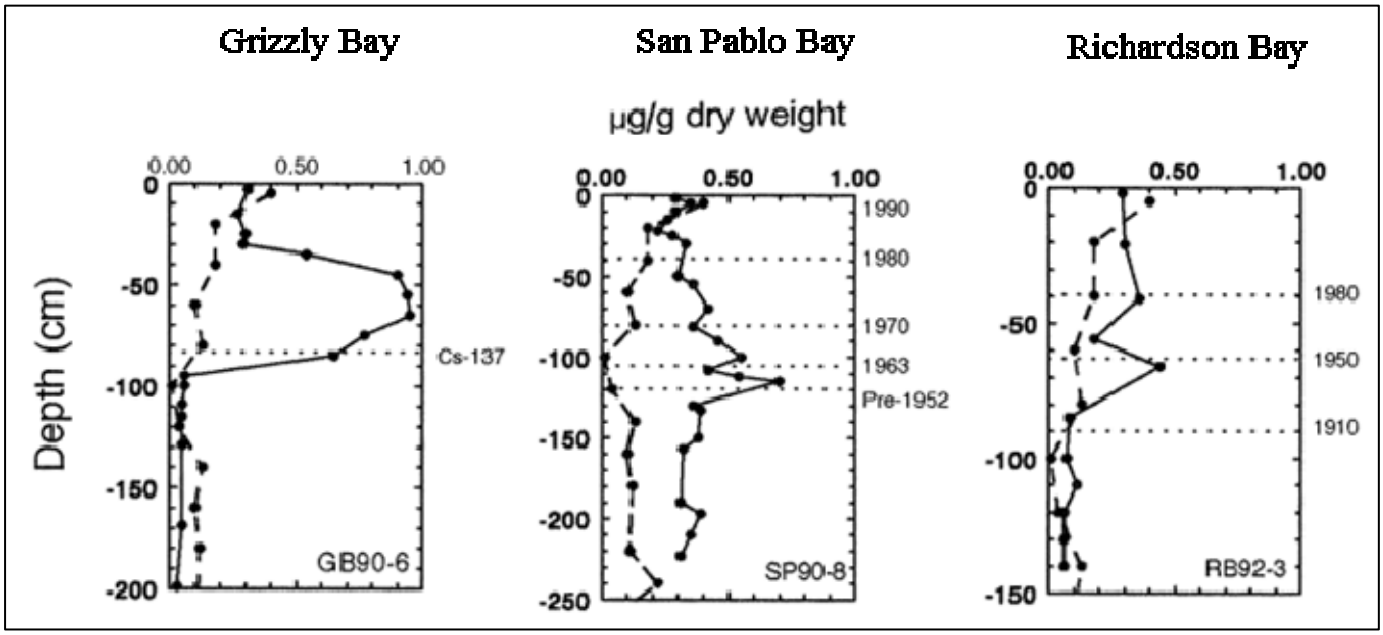


Source: SFEI (2008).

Hornberger et al. (1999) measured concentrations of mercury and other metals in six sediment cores from various parts of the SFB. These concentrations were compared to those in a sediment core from Tomales Bay (located about 45 km north of the SFB, not shown in Figure B-10), considered to be a uncontaminated reference because of a lack of industrialization and urbanization. The authors found that concentrations in the deepest horizons of the SFB cores were comparable to those in the reference core (0.06±0.01 µg/g) from Tomales Bay (see Figure B-11) and that the maximum concentrations of mercury in the cores were found to be about 20 times the reference baseline concentration. The authors also indicated that the earliest anthropogenic influence on metal concentrations appeared as mercury contamination (concentration of 0.3-0.4 µg/g) in sediments deposited at San Pablo Bay between 1850 and 1880. Due to the presence of mercury mines in the upstream watersheds, concentrations of mercury

were higher in sediments of San Pablo and Grizzly Bay (both in the northern part of the SFB, see Figure B-10 for the location of these sites), compared to those in Richardson Bay (which is affected by both the northern and the less contaminated southern portions of the SFB).

Figure B-11 Mercury Concentrations (µg/g, dry weight) in Sediment Cores from Grizzly Bay, San Pablo Bay, and Richardson Bay Compared to those in the Reference Sediment Core from Tomales Bay

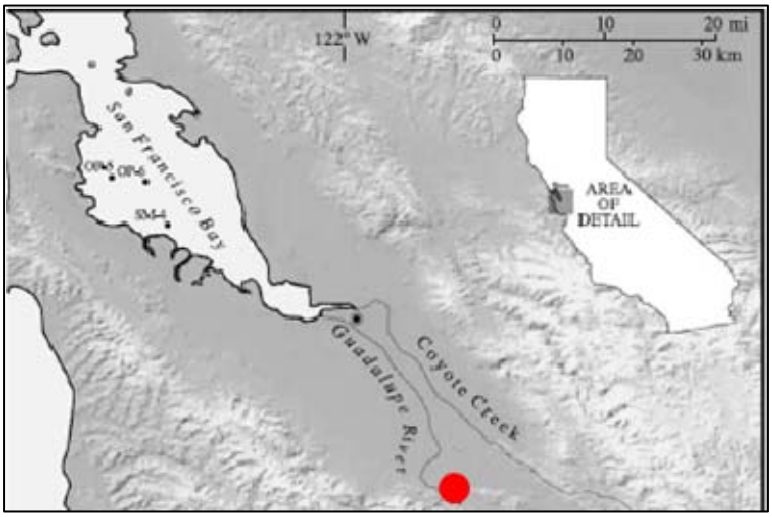


Notes:

- Sediment core from Tomales Bay shown as broken vertical line.
- The horizontal dotted lines denote the minimum age of sediment horizon as determined by an age dating model.
- Source: Hornberger et al. (1999).

Conaway et al. (2004) measured mercury concentrations in sediment cores collected from a southern SFB tidal marsh, which is downstream of the New Almaden mining district (see Figure B-12), formerly the largest mercury mining district in North America. The mercury concentrations at various depths of the sediment core and the estimated mercury accumulation rate in the sediments are presented in Table B-9. The authors observed that sediment mercury concentrations began to increase in the later part of the 19th century and reached a maximum in the mid-20th century, consistent with the temporal trends in mercury production at that mine. The authors indicated that concentrations observed in the pre-mining sediment record (prior to 1850s, 0.40±0.15 nmol/g, or 0.08±0.03 µg/g), are similar to those reported in the literature from deep-cores from the northern reach of the SFB and to river sediments from the northern California Coast Range (with no known mercury mineral deposits). Thus, the authors argued that natural weathering of mercury-rich deposits is not a substantial contribution to the southern SFB and that the principal source of mercury contamination in the southern SFB is anthropogenic activity (erosion of mercury mine wastes).

Figure B-12 Location of the New Almaden Mine



Notes:

- Location of the New Almaden mine shown with red circle, with reference to the San Francisco Bay
- Source: Conaway et al. (2004).

Table B-9 Mercury Accumulation Rate for Depth Intervals in a Sediment Core from a Southern San Francisco Bay tidal marsh

Depth (cm)	Period	Sedimentation Rate (cm/yr)	Bulk density (g/cm^3)	Hg conc. (µg/g)[a]	Hg accumulation rate ($\mu g/cm^2/yr$)[b]
0-8	1998-2001	2.70	0.8	0.46	1.00
8-35	1982-1998	1.80	0.7	0.48	0.62
35-40	1975-1982	0.71	0.7	0.54	0.26
40-125	1945-1975	2.80	0.6	0.84	1.40
125-140	1870-1945	0.18	0.6	0.52	0.06
140-280	1570-1870	0.33	0.7	0.08	0.02

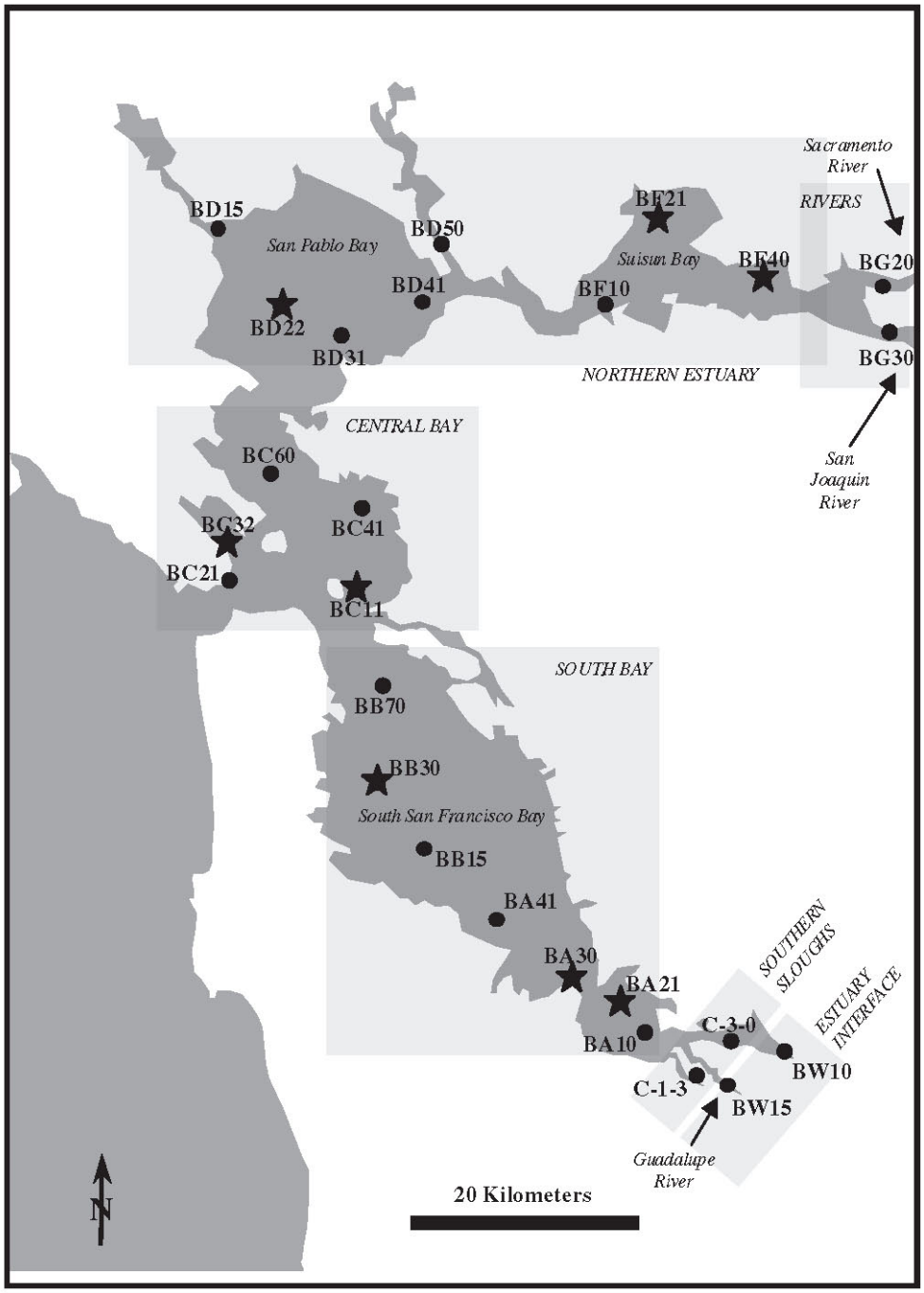
Notes:

- a - Reported concentrations were converted from nmol/g to µg/g.
- b - Reported accumulation rates were converted from $nmol/cm^2/yr$ to $\mu g/cm^2/yr$.
- Source: Conaway et al. (2004).

Conaway et al. (2007) analyzed total mercury concentrations in sediments of the San Francisco Estuary for the period 1993 to 2001 at 26 sites ranging from marine to freshwater locations (see Figures B-13A and B). For analytical purposes, the 26 sampling sites were grouped into six hydrographic regions: Estuary Interface, Southern Sloughs, South Bay, Central Bay, Northern Estuary, and Rivers. The authors found that the six hydrographic regions had significantly (statistically) different sediment mercury concentrations, with the Estuary Interface region and the Rivers region having the highest (0.35 µg/g) and lowest (0.10 µg/g) median concentrations, respectively (see Figures B-13A and B). The authors indicated that sediment concentrations during the period 1993 to 2001 have decreased at few sites in the Northern Estuary region (by 22%), Central Bay region (by 17%), and South Bay region (by 32%). This decrease in sediment mercury concentrations was attributed to the transport of relatively lower concentration mercury sediment to the estuary from the Sacramento River and San Joaquin River watersheds.

Greenfield et al. (2005) analyzed seasonal, interannual, and long-term variations in mercury contamination of sport fish in the SFB. The authors indicated that interannual variation in mercury was evident in striped bass but absent in shiner surfperch, leopard shark and white croaker. Mercury concentrations in striped bass showed no evidence of a trend between 1970 and 2000, and the authors attributed this to the continuing contribution of mercury from historical mines and erosion of buried sediments. The authors indicated that variations in mercury bioavailability to fish could be a function of fish ecology, watershed loading, contaminated sediment exposure, and net methylmercury production rates.

Figure B-13A Sampling Locations of Total Mercury Concentrations in Sediments of the San Francisco Estuary, 1993 to 2001



Notes:

- Sampling locations in the San Francisco Estuary, grouped into six hydrographic regions (from south to north): Estuary Interface, Southern Sloughs, South Bay, Central Bay, Northern Estuary and Rivers. Stars represent sites showing a statistically significant decrease in sediment mercury concentrations from 1993 to 2001.
- Source: Conaway et al. (2007).

Figure B-13B Total Mercury Concentrations in Sediments of the San Francisco Estuary, 1993 to 2001

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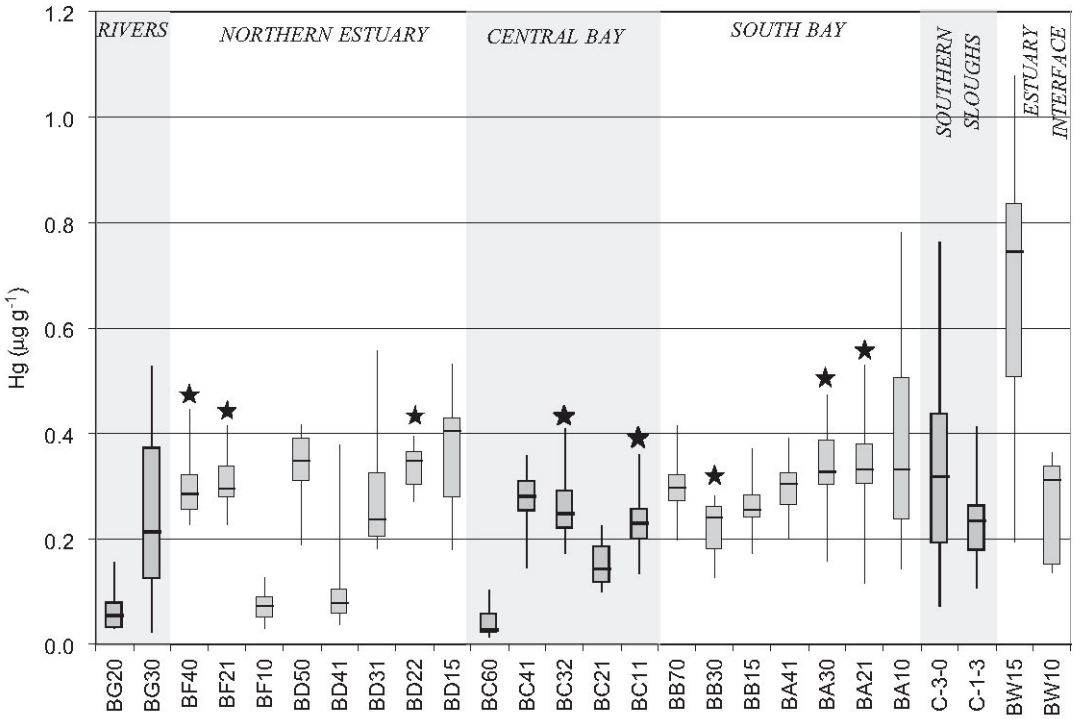


Notes:

- Box and whisker plots showing mercury concentrations (µg/g, dry weight) for the period 1993-2001. For each station (from the bottom to the top) are shown the minimum, 25 percentile, median, 75 percentile, and maximum values.
- Source: Conaway et al. (2007).

B.5 Estimates of the San Francisco Estuary Mercury Budget

B.5.1 Estimated inputs to the San Francisco Bay from Different Sources (San Francisco Bay RWQCB 2006)

The following is a summary of estimated mercury inputs to the SFB for the year 2003 from the San Francisco Bay RWQCB. The total input to the SFB was estimated to be 1,220 kg/yr (see Table B-10) and the individual sources in decreasing order of magnitude were bed erosion (460 kg/yr), the Central Valley watershed (440 kg/yr), urban storm water runoff (160 kg/yr), the Guadalupe River watershed (92 kg/yr), direct atmospheric deposition (27 kg/yr), non-urban storm water runoff (25 kg/yr), and wastewater discharges (18 kg/yr). Based on box models that account for sediment and mercury inputs and outputs to and from the SFB, this report estimated that mercury losses for 2003 (by transport to the Pacific Ocean via the Golden Gate, plus the net result of dredging and disposal and other losses), were approximately 1,700 kg/yr. The estimated contribution of the various sources of urban storm water runoff (a total load of 160 kg/yr) is presented in Table B-11 and the major sources of mercury loading to the SFB from municipal wastewater discharge are presented in Table B-12.

Table B-10 Estimated Annual Mercury Contribution (in kg/yr) from Various Sources to the San Francisco Bay in 2003

Source	contribution (kg/yr)	% of total
Wastewater (municipal & industrial) Discharges	18	1.5%
Non-urban Storm Water Runoff	25	2.0%
Direct Atmospheric Deposition	27	2.2%
Guadalupe River Watershed (mining legacy)	92	7.5%
Urban Storm Water Runoff	160	13.1%
Central Valley Watershed	440	36.0%
Erosion of Buried Sediments	460	37.6%
Total	1220	100%

Source: San Francisco Bay RWQCB total maximum daily load (TMDL) 2006).

Table B-11 Estimated Annual Mercury (in kg/yr) in urban Storm Water Flowing to the San Francisco Bay in 2003

Urban Storm Water Source	Estimated Hg Load (kg/yr)
Santa Clara Valley Urban Runoff Pollution Prevention Program	44.0
Alameda Countywide Clean Water Program	39.0
Contra Costa Clean Water Program	22.0
San Mateo County Storm Water Pollution Prevention Program	16.4
Vallejo Sanitation and Flood Control District	3.2
Fairfield-Suisun Urban Runoff Management Program	3.1
American Canyon	0.3
Sonoma County area	3.1
Napa County area	3.1
Marin County area	6.5
Solano County area	1.6
San Francisco County area	17.2
Total[a]	159.5

Notes:

- a - The total urban storm water load of 160 kg/yr, presented in Table B-10, was obtained by San Francisco Bay RWQCB TMDL (2006) by rounding off the total presented here.
- Modified from Table 5-w of San Francisco Bay RWQCB TMDL (2006).

Table B-12 Estimated Annual Mercury (in kg/yr) from Municipal Wastewater Discharge Flowing to the San Francisco Bay in 2003

Municipal Wastewater Discharger	Estimated Hg Load (kg/yr)
East Bay Dischargers Authority	3.6
San Francisco, City and County of, Southeast Plant	2.7
East Bay Municipal Utilities District	2.6
Central Contra Costa Sanitary District	2.2
San Jose/Santa Clara Water Pollution Control Plant	1.0

Notes:

- Only discharges with an estimated load of at least 1 kg/yr are presented here.
- Modified from Table 2-1 of San Francisco Bay RWQCB TMDL (2006).

B.5.2 Model-based Mercury Mass Balance in the San Francisco Bay Area

McKee and Mangarella (2006) used a combination of locally available data and estimates from other regions of the US in a simple conceptual modeling framework to estimate the mercury contribution to the SFB from storm water conveyances (see Table B-13). The authors estimated that the total mercury loading to the SFB from storm water is 176 kg/yr, with atmospheric deposition and watershed surface sediment erosion contributing 48 and 59 kg/yr, respectively.

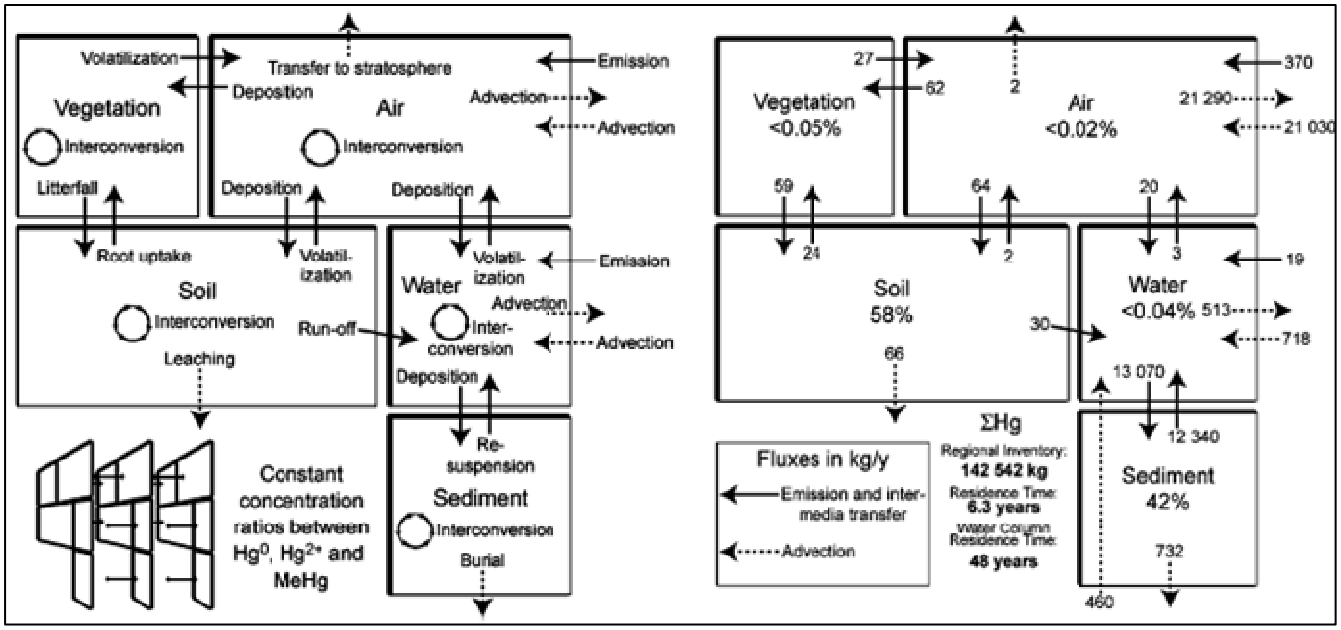
Table B-13 Estimated Annual Mercury (in kg/yr) in San Francisco Bay Area Storm Water Conveyances

Mercury Source	Estimated Hg Load (kg/yr)
Watershed Surface Sediment Erosion	59 (30-182)
Atmospheric Deposition (direct deposition to water surfaces)	48 (20-93)
Instruments	23 (8-28)
Bed and Bank Erosion	21 (4.1-160)
Switches and Thermostats	10 (9-11)
Fluorescent Lighting	4.1 (2.4-5.8)
Paint	2.6 (1-4)
Railway Lines	1.5 (0.09-3)
Industrial Hotspots	1.4 (0.25-7.4)
Landfill	1 (0.5-1.5)
Laboratory	1 (0.2-1.4)
Gasoline	1 (0.1-2)
Batteries	0.8 (0.15-1.5)
Auto-Recycling	0.7 (0-3)
Dental	0.4 (0.2-0.6)
Other Uses	0.09 (0.006-0.18)
Total	176 (76-504)

Source: McKee and Mangarella (2006).

Macleod et al. (2005) developed a model that simultaneously describes the fate and transport of elemental, divalent and methylated mercury species in the environment (including air, soil, vegetation, water and sediments) (see Figure B-14). The objective of this modeling study was to describe the long-term average dynamics of mercury in a generic regional environment, and hence, the model is not appropriate for describing episodic depletion or accumulation of mercury species.

Figure B-14 Steady-state Mass Balance of Total Mercury in the San Francisco Bay Area



Notes:

- Left panel: Model schematic showing conceptual transfer (straight arrows) and transformation (circular arrows) processes between the various reservoirs for Hg(0), Hg(II), and MeHg species.
- Right panel: Estimated fluxes (in kg/yr) and storages (as percentages of the total regional inventory).
- Source: Macleod et al. (2005).

The key assumption of this model is that temporally and spatially averaged ratios of concentrations of the individual mercury species are constant in each environmental medium (air, soil, vegetation, etc.). The authors indicated that this key assumption will be valid (however, it is not a requirement) when chemical reactions for inter-conversion of mercury species are fast relative to the rates of transport in and out of any medium or across media. The various parameters required by the model (such as physical properties of the media, transport velocity parameters, etc.) were obtained from the scientific literature, and the 95% confidence intervals of many of these parameters ranged from one third to three times their median values. The authors indicated that although their "lumped" modeling approach (which employs single best-estimate parameters) does not account for spatial and temporal heterogeneity observed within the SFB Area, it provides an approximate (and relatively simple) mass balance budget and enables the identification of key model parameters using uncertainty analysis. Comparison of model simulated and observed mercury concentrations and fluxes in the SFB Area indicated that the model is reasonably consistent with the observations and that the 95% confidence intervals on the model estimates overlapped with those on the observations. However, the fact that these confidence intervals spanned two orders of magnitude highlights the inability of a "lumped" modeling approach to capture the

spatial and temporal heterogeneity of the SFB Area. Nonetheless, the model provides useful estimates of the SFB mercury budget.

The estimated steady-state mass balance of total mercury in the SFB Area is shown in Figure B-14. The authors indicated that the mass balance of total mercury in the SFB Area is determined almost entirely by the dynamics of Hg(II) species, the most prevalent species in all environmental media except the atmosphere (where Hg(0) is predominant). It is evident from Figure B-14 that there is a net atmospheric deposition of total mercury to soils, water, and vegetation in the SFB Area. The residence time of total mercury in the atmosphere was estimated to be 6.3 years and the residence time of total mercury in the sediments was estimated to be 48 years.

Contributed Paper

Impacts of Chronic Anthropogenic Noise from Energy-Sector Activity on Abundance of Songbirds in the Boreal Forest

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Abstract: *The effects of human activities in forests are often examined in the context of habitat conversion. Changes in habitat structure and composition are also associated with increases in the activity of people with vehicles and equipment, which results in increases in anthropogenic noise. Anthropogenic noise may reduce habitat quality for many species, particularly those that rely on acoustic signals for communication. We compared the density and occupancy rate of forest passerines close to versus far from noise-generating compressor stations and noiseless well pads in the boreal forest of Alberta, Canada. Using distance-based sampling, we found that areas near noiseless energy facilities had a total passerine density 1.5 times higher than areas near noise-producing energy sites. The White-throated Sparrow (*Zonotrichia albicollis*), Yellow-rumped Warbler (*Dendroica coronata*), and Red-eyed Vireo (*Vireo olivaceus*) were less dense in noisy areas. We used repeat sampling to estimate occupancy rate for 23 additional species. Seven had lower conditional or unconditional occupancy rates near noise-generating facilities. One-third of the species examined showed patterns that supported the hypothesis that abundance is influenced by anthropogenic noise. An additional 4 species responded negatively to edge effects. To mitigate existing noise impacts on birds would require approximately $175 million. The merits of such an effort relative to other reclamation actions are discussed. Nevertheless, given the $100 billion energy-sector investment planned for the boreal forest in the next 10 years, including noise suppression technology at the outset of construction, makes noise mitigation a cost-effective best-management practice that might help conserve high-quality habitat for boreal birds.*

Keywords: Alberta, boreal forest, chronic noise, compressor stations, edge effects, energy industry, forest songbird, passerines

Impactos de Ruido Antropogénico Crónico de la Actividad del Sector Energético sobre la Abundancia de Aves Canoras en el Bosque Boreal

Resumen: *Los efectos de las actividades humanas en los bosques a menudo son examinados en el contexto de la conversión del hábitat. Sin embargo, en asociación con cambios en la estructura y composición del hábitat hay incrementos en la actividad de personas con vehículos y equipo, lo que resulta en incrementos en el ruido antropogénico. El ruido antropogénico puede reducir la calidad del hábitat para muchas especies, particularmente las que dependen de señales acústicas para su comunicación. Comparamos la densidad y la tasa de ocupación de paserinos de bosque cerca versus lejos de estaciones compresoras generadoras de ruido y pozos silenciosos en el bosque boreal de Alberta, Canadá. Utilizando muestreo basado en distancias, encontramos que las áreas cerca de instalaciones energéticas silenciosas tenían una densidad total de paserinos 1.5 veces mayor que las áreas cercanas a sitios generadores de ruido. La densidad de* Zonotrichia albicollis, Dendroica coronata *y* Vireo olivaceus *fue menor en áreas ruidosas. Utilizamos muestreos de repetición para estimar la tasa de ocupación de 23 especies adicionales. Siete tenían menores tasas de ocupación condicional o incondicional cerca de las instalaciones generadoras de ruido. Un tercio de las especies examinadas mostró patrones que soportaron la hipótesis que la abundancia esta influida por el ruido antropogénico. Cuatro*

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Paper submitted August 3, 2007; revised manuscript accepted January 7, 2008.

Conservation Biology, Volume 22, No. 5, 1186–1193
©2008 Society for Conservation Biology
DOI: 10.1111/j.1523-1739.2008.00973.x

especies adicionales respondieron negativamente a los efectos de borde. Para mitigar los impactos del ruido sobre las aves se requeriría approximately US $175 millones. Se discuten los méritos de tal esfuerzo en relación con otras acciones de reclamación. Sin embargo, dada la inversión de $100 billones planeada por el sector energético en el bosque boreal en los próximos 10 años, la inclusión de tecnología supresora de ruido al comienzo de la construcción, hace que la mitigación de ruido sea una práctica de manejo rentable que podría ayudar a conservar hábitat de buena calidad para las aves boreales.

Palabras Clave: Alberta, ave canora de bosque, bosque boreal, efectos de borde, estaciones de compresores, industria de la energía, paserinos, ruido crónico

Introduction

Human activity is increasing throughout the world's forests. Many of these activities are associated with changes in the structure and composition of native vegetation. A large body of literature addresses the effects of these footprint-related changes on wildlife. Less understood are the impacts caused by the "foot," or in other words, the presence of people and their tools on forested areas. One of the consequences of industrial activity in forested environments is increased anthropogenic noise due to vehicles, machinery, and infrastructure. Industrial noise can take many forms. Forestry and energy-sector operations can generate intense noise for periods of days to weeks in a relatively small area. Aircraft and cars moving through an area relatively infrequently create periodic but very intense noise disturbances. Responses of animals such as caribou (*Rangifer tarandus*, Bradshaw et al. 1997), whales (Myrberg 1990), and Spotted Owls (*Strix occidentalis*, Delaney et al. 1999) have been examined in this context. In general, such species seem to abandon the area when the noise is occurring and return once the noise has dissipated.

The effects on wildlife of chronic anthropogenic noise from busy highways, urban areas, and permanent industrial structures are not as well understood. Presumably, if noise levels are chronic, animals will either avoid the noisy area or become habituated to noise and remain, although remaining may have consequences for reproductive success (Habib et al. 2007). It seems logical that because of the importance of acoustic information to forest songbirds and the myriad number of ways anthropogenic noise can affect avian communication, that birds might avoid chronically noisy locations. Nevertheless, birds are plastic in their song repertoire, and many can modify song characteristics to fit their environment (Slabbekoorn & Peet 2003). If this phenomenon is generally true, then birds may be able to adapt to chronic noise, which means there would be no net loss of habitat for birds as a result of anthropogenic noise.

In much of the boreal forest of Alberta, Canada, human activity is rapidly increasing in an effort to extract oil and gas. To locate, take to the surface, and ship energy products, an intricate system of roads, pipelines, wells, and other energy facilities has been created. Compressor stations are part of the pipeline network and serve to maintain the flow of gas through pipelines. A compressor consists of 1–3 motors cooled by an equal number of large fan units housed in an insulated metal shed in a small clearing (2–4 ha). Noise levels produced by compressor stations are typically between 75 and 90 dB(A) at the source (Bolstad Engineering Associates 1978; ATCO Noise Management 2007), but can reach 105 dB(A) at large facilities (MacDonald et al. 1996). Compressor stations run 24 h a day, 365 days a year other than for periodic maintenance. Compressor noise can be heard at distances of well over 1 km in the boreal forest (E.M.B., personal observation). There are approximately 5000 compressor stations in boreal Alberta.

Our objective was to compare bird occupancy and density close to versus far from noise-generating compressor stations relative to control well pads that had similar levels of habitat disturbed but no noise. We predicted that birds would be most affected by noise if an interactive effect of noise and distance existed (i.e., bird abundance near control sites and compressor stations would differ the most). We expected sites farther from compressor stations and control sites to exhibit less of a difference in bird abundance because noise dissipates over space.

Methods

Field Methods

To study the effect of noise from compressor stations, some type of control is required. In experiments assessing the impacts of highway noise as a factor influencing birds, comparisons between highway edge and forest interior are often made. The difficulty with this design is highways are associated with other factors (i.e., edges) that make it difficult to isolate the effect of noise. Compressor stations have a significant amount of edge habitat, so comparing areas close to compressor stations to forest interior provides only a weak control. Nevertheless, across Alberta there are hundreds of thousands of well pads. Like compressor stations, well pads are clearings of forest habitat linked to the pipeline and road network.

Unlike compressor stations, natural-gas well pads produce no chronic noise; thus, we used them as our control sites.

We conducted our study in northeastern Alberta during June 2003 and 2004 in the Alberta–Pacific Forest Industries Inc. (ALPAC) Forest Management Agreement area (FMA) (see Bayne et al. [2005] for map). The landscape is a boreal mixed-wood forest (Strong & Leggat 1992), where lowland vegetation is dominated by black spruce (*Picea mariana*) bogs and fens, and upland areas are dominated by trembling aspen (*Populus tremuloides*) and white spruce (*P. glauca*).

Compressor stations and well pads were preselected in a geographic information system (GIS) with Alberta Vegetation Inventory (AVI) data, energy facility data, and road data provided by ALPAC. Bird surveys were conducted in mature, 60- to 90-year-old, aspen-dominated forest (<25% conifer). We identified all possible compressor sites identified in the GIS. Sites were then ground truthed to ensure that the energy facility was appropriate and the vegetation was of the correct type. Within the pool of ground-truthed sites with appropriate forest cover, we selected sites that were accessible by truck and close enough together to make travel between sites practical on a daily basis. Compressor sites were >3 km away from other sites to ensure noise from one site did not reach another. Well pads were occasionally <3 km away from each other. No noise was emitted from well pads, so noise contamination between sites was not a concern. Although logging occurs in the area, no sites were located within 1 km of a harvested area or within 2 km of ongoing logging at the time of study.

Ten-minute variable-radius point counts were conducted at each site to measure bird abundance. The location of each singing bird was recorded as occurring in 1 of 3 concentric circles during each point-count visit (0–50 m, 50–100 m, and 100–150 m from observer). At each site, the location where bird surveys were conducted (hereafter point-count stations) was in 1 of 2 bands around each compressor or well pad. Four point-count stations were within 100–300 m of the edge of the energy clearing (hereafter close), and 4 others were within 400–700 m (hereafter far). Each point-count station was visited 4 times. Point-count stations were a minimum of 300 m apart and at least 50 m from linear-feature edges, such as seismic lines or pipelines. The rationale for using the 2 "distance" groups was that linear features and natural vegetation patterns prevented identification of a uniform set of distances at each site. At point-count stations near compressor stations, the average noise level was 48 db(A) (SD 6), whereas the average distance from the compressor centroid was 242 m (SD 86).

Point counts were conducted from 4 through 30 June in 2003 and from 2 through 25 June in 2004 between 04:24 and 09:27. All counts were done on days with no significant precipitation and little to no wind (<3 on the Beaufort scale). Overall, we conducted counts at 104 point-count stations (52 close and 52 far) at 13 compressor stations. At 8 well pads, we surveyed 64 stations (32 close and 32 far). Non-passerines were counted but excluded from all analyses. Time of day, date, and observer (3 individuals) were fully randomized among treatments such that each station received equal effort by each observer at different times of the day and year. These nuisance variables were not included in the analysis as a result of this randomization.

At each station we recorded canopy cover, proportion of trees that were coniferous, shrub cover, canopy height, and shrub height. Cover estimates were placed in 1 of 5 ranks by 1 observer (0, none present; 1, 1–25% cover; 2, 26–50% cover; 3, 51–75% cover; 4, 76–100% cover). We used principal components analysis after variable standardization to reduce these 5 variables to 2 variables, which we used as vegetation covariates in subsequent analysis.

Estimating Density and Occupancy

Inherent in studies of bird response to ambient noise is the possibility of detectability bias. Detectability has many components, but the main concern related to anthropogenic noise is whether the observer hears a bird given that it sings within a distance detectable by the observer (Farnsworth et al. 2002). Equal detectability among habitats is assumed in most studies, but if violated as it may be in noisy versus quiet environments, estimates of density or occupancy may be biased. Absolute occurrence or true density is not particularly important for statistical inference about the effects of noise if noise has no effect on the ability of an observer to hear birds. Occupancy rate or density will simply underestimate the true value. Nevertheless, if detection error is differentially affected by the level of background noise it is plausible that a greater number of absences or the number of birds that observers can hear at noisy locations may be a function of detection error generated by noise rather than birds having reduced abundance in noisy areas. Statistical modeling of occupancy via repeat sampling (MacKenzie et al. 2002) and density estimation via distance sampling (Buckland et al. 2001) are 2 methods that allow detection error to be estimated in response to noise level and with which appropriate corrections can be made to density or occupancy estimates.

To estimate density we measured the distance to each bird and then used Program Distance (Thomas et al. 2006) to calculate detection probabilities. We fit detection functions for half-normal and uniform models with cosine and simple polynomial expansions to the second order to determine which series and key function provided the best-fitting detection curve for each distance–noise stratum.

Data from all visits and all point-count stations within each stratum were used to derive the detection function. Density was estimated for each point-count station with

$$\hat{D} = \frac{n}{\pi w^2 \hat{P}_a v}, \quad (1)$$

where n is the number of birds counted per point-count station, w^2 is the known radius of sampling (150 m), $\hat{P}_a$ is the expected proportion of birds detected with a radius w (estimated detection function), and v is the number of visits to each point-count station.

Some researchers think the ability of observers to estimate distance to singing birds on the basis of acoustic cues (even in the coarse categories we used) is not accurate enough to effectively estimate distance. In addition to a 2-week training period each year, during which we practiced estimating distance, we used a field point-count simulation with known numbers of "birds" to validate our ability to estimate distances. The simulation was done when few real birds were singing and wind was minimal. The simulation took place at 3 point-count stations at one compressor site. Background noise at the points varied from 48 to 53 dB(A). Three people moved quietly throughout the forest playing randomly selected song recordings of 6 species at appropriate volumes at distances up to 150 m. The 2 main observers stood blindfolded, standing back-to-back and counted simulated birds. A total of 200 singing events were simulated. A correct identification required the species to be identified at the correct bearing, within the correct 50-m-distance band, and at the correct time. In only one case did an error in identification occur in the simulation within 50 m of the observer, whereas 89% of simulated call playbacks at distances between 50 and 150 m were identified correctly by observers.

Using the detection-error function, we changed our counts per point-count station into estimated density of birds per point-count station. Density of birds was analyzed subsequently with a generalized linear model to determine whether the following variables were significant predictors: noise (dummy variable for compressor vs. well pad), distance (dummy variable close to vs. far from energy sector clearings), and the interaction between treatment and distance. The 2 principal-component factors that described vegetation composition were also included. If the interaction between noise and distance was not significant at $p \leq 0.05$, it was dropped from the model.

Our data were hierarchically structured, with point-count stations nested within sites, which made sites the unit of replication. To account for the potential lack of independence of point-count stations within sites, site was treated as a random effect in the model. This convention partitioned the appropriate degrees of freedom and estimated unique site-level effects, thus accounting for the multilevel structure of the data (Rabe-Hesketh & Skrondal 2005). Although distribution of density was similar to a Gaussian distribution for the common species, density of some species was not distributed normally. To ensure the p values used in statistical inference were robust to distributional assumptions, we used a permutation test run 1000 times to estimate statistical significance. All generalized linear models were run in the program Stata 9.2 (Statacorp 2005).

For many species there were insufficient data to estimate density, so we also modeled the occupancy rate. Using the program Presence (Hines 2006), we estimated detection error for each species by recording the proportion of point-count stations in which species were detected over our 4 visits and adjusted occupancy estimates accordingly. The key assumption of the MacKenzie method is that if a species is detected once during repeated visits to a site, it had to have been present but was not detected on the other visits, which generates a detection-error function (closed population assumption). If a bird is never detected at a point-count station it does not necessarily imply absence, and the approach estimates the probability that a bird would occur at each point-count station on the basis of the detection-error function. We estimated a unique detection function for each stratum and controlled for this difference when estimating occupancy rate. Likelihood-ratio tests were used to determine whether there was any evidence that the interaction between noise and distance, and the main effects of distance and noise—while controlling for vegetation covariates—influenced conditional occupancy rate (occurrence corrected for detection error within each stratum). For some species, program Presence would not converge, and in these cases we used logistic regression with a random effect to test whether occurrence differed between treatments. Occurrence was defined as whether a species was detected during any of the 4 visits.

Results

We recorded the location of 5129 passerine birds of 56 species. Of these, 23 species were detected at 2 or more point-count stations within each noise–distance stratum and were considered for analysis. We generated unique detection functions via distance estimation for all passerines combined: Ovenbird, Red-eyed Vireo, Tennessee Warbler, White-throated Sparrow, and Yellow-rumped Warbler (scientific names in Table 2). The selected functions and expansions used to estimate density were selected on the basis of model goodness of fit. For the remaining species there were insufficient detections within each stratum (60–100 detections is recommended) to estimate a suitable detection-error function with distance sampling (Buckland et al. 2001).

Table 1. Density of birds (95% CI) at the edge of natural gas well pads (noiseless sites) and compressor stations (noise-generating sites) relative to areas in the forest interior.*

Species	*Compressor near*	*Compressor far*	*Well pad near*	*Well pad far*	*Well pad vs. compressor (p)*	*Near vs. far (p)*	*Interaction (p)*
All Passerines	18.8 (17.3–20.2)	22.0 (20.4–23.6)	32.3 (30.2–34.4)	29.3 (27.3–31.2)	.	.	0.019
Ovenbird	5.5 (4.7–6.3)	7.5 (6.7–8.4)	7.2 (6.1–8.3)	7.6 (6.7–8.6)	0.359	0.009	0.181
Red-eyed Vireo	3.0 (2.3–3.7)	3.7 (3.2–4.2)	5.4 (4.6–6.2)	3.8 (3.0–3.7)	.	.	0.004
Tennessee Warbler	4.4 (3.9–4.9)	4.0 (3.4–4.6)	3.3 (2.5–4.1)	3.4 (2.7–4.1)	0.076	0.695	0.558
White-throated Sparrow	0.7 (0.5–0.9)	0.5 (0.4–0.7)	1.8 (1.5–2.2)	1.5 (1.1–1.9)	<0.001	0.211	0.750
Yellow-rumped Warbler	1.5 (1.1–2.0)	1.1 (0.8–1.5)	2.1 (1.3–3.0)	1.4 (1.0–1.8)	0.021	0.114	0.592

**Density is the number of birds per 10 ha after statistical correction with different distance-correction functions. Detection functions were derived independently for each noise-distance category. Data are reported as means adjusted to average vegetation conditions. The 95% CIs were derived from a bias-corrected and accelerated bootstrap. The* p *values were generated via Monte Carlo permutations. Main effects were not tested for significance if the interaction between noise and distance was significant at* $p \leq 0.05$.

The density of all passerines was significantly lower near compressor stations than all other strata (Table 1). The rank order of passerine bird density was: near well pads > far well pads > far compressor > near compressor. This was the pattern we expected to see. Correcting for detection error, density estimates were not significantly different between noisy versus quiet areas for Ovenbirds, although the density estimate near compressor stations was the lowest of the 4 strata. The Red-eyed Vireo had its lowest density near compressor station edges, which was significantly lower than near the edges of well pads. There was no significant difference in Tennessee Warbler density between treatments. White-throated Sparrows were 2 to 4 times more abundant at well-pad edges than at compressor-station edges (Table 1). The density of Yellow-rumped Warblers was significantly lower near compressor stations.

Occupancy rates determined from conditional estimators showed that American Redstarts, Least Flycatchers, Red-breasted Nuthatches, Red-eyed Vireos, and Yellow-rumped Warblers were less likely to occur near compressor stations (Table 2). Conditional estimators could not be calculated for the Red-breasted Nuthatch and Rose-breasted Grosbeak, but the unconditional estimator showed they were less likely to occur in noisy areas. There was no statistically significant evidence of any species being more likely to occur near compressor stations.

Discussion

A large body of literature comparing bird abundance near highways relative to forest interiors has led to the conclusion that chronic noise is a critical factor influencing habitat quality for forest birds (e.g., Reijnen & Foppen 1995; Forman & Deblinger 2000; Peris & Pescador 2004). Despite the large number of studies, the inferences drawn about the effects of road noise on birds are weak because the majority of researchers compared road edges to forest interiors, which confounds factors such as edge effects, traffic mortality, and visual cues from cars with the effects of noise. Our result that overall passerine density was significantly influenced by the interaction between noise level and distance to compressor stations provides some of the strongest support that chronic anthropogenic noise does influence habitat quality for forest birds.

Relatively few of the species we surveyed exhibited the expected interactive pattern. Nevertheless, one-third of the species supported our hypothesis that noise reduced abundance, as areas both close to and far from compressor stations had lower abundance than near well pads. This suggests that noise levels from compressor stations may be sufficiently loud to affect birds up to 700 m into the forest. Alternatively, we may have had insufficient statistical power to detect the interactive effect of noise and distance. In European road studies, where noise has been purported to cause declines in birds, lower abundances of birds near roads have been observed in 15–74% of species analyzed, with overall density typically being lower at road edges. Our results are at the low end of this range. Importantly, in none of these studies did researchers correct for detection error caused by vehicle noise.

Why some forest birds avoid noisy areas and others do not is not well understood. Birds communicate primarily through vocalizations. Males of territorial species use song to advertise their breeding status to females (Lein

Table 2. Estimates of occupancy rate (95% CI) at the edge of natural gas well pads (noiseless sites) and compressor stations (noise-generating sites) relative to matched areas in the forest interior.[a]

Species	*Est.*[b]	*Compressor near*	*Compressor far*	*Well pad near*	*Well pad far*	*Well pad vs. compressor* (p)	*Near vs. far* (p)	*Interaction* (p)
Alder Flycatcher (*Empidonax alnorum*)	U	0.18 (0.08–0.37)	0.04 (0.01–0.15)	0.09 (0.02–0.29)	0.09 (0.03–0.29)	0.533	0.032	0.137
American Redstart (*Setophaga ruticilla*)	C	0.07 (0.02–0.22)	0.20 (0.06–0.51)	0.53 (0.10–0.92)	0.32 (0.15–0.56)	.	.	0.034
American Robin (*Turdus migratorius*)	C	0.69 (0.01–0.99)	0.48 (0.09–0.89)	0.66 (0.18–0.94)	0.54 (0.03–0.98)	0.950	0.691	0.919
Black-and-white Warbler (*Mniotilta varia*)	C	0.29 (0.14–0.49)	0.46 (0.22–0.72)	0.26 (0.10–0.55)	0.71 (0.01–0.99)	0.952	0.146	0.639
Black-capped Chickadee (*Poecile atricapilla*)	U	0.18 (0.09–0.32)	0.28 (0.16–0.43)	0.09 (0.03–0.25)	0.37 (0.21–0.57)	0.873	0.011	0.131
Chipping Sparrow (*Spizella passerine*)	C	0.52 (0.31–0.72)	0.30 (0.13–0.57)	0.53 (0.33–0.73)	0.61 (0.33–0.83)	0.446	0.737	0.242
Connecticut Warbler (*Opornis agilis*)	C	0.23 (0.10–0.43)	0.32 (0.20–0.48)	0.33 (0.18–0.51)	0.28 (0.15–0.46)	0.413	0.937	0.413
Gray Jay (*Perisoreus canadensis*)	U	0.16 (0.08–0.28)	0.16 (0.08–0.29)	0.12 (0.04–0.28)	0.24 (0.12–0.42)	0.449	0.268	0.254
Hermit Thrush (*Catharus guttatus*)	C	0.30 (0.11–0.61)	0.52 (0.18–0.85)	0.59 (0.15–0.92)	0.47 (0.27–0.68)	0.703	0.694	0.375
Least Flycatcher (*E. minimus*)	C	0.20 (0.10–0.34)	0.35 (0.22–0.49)	0.52 (0.34–0.69)	0.42 (0.24–0.62)	.	.	0.049
Magnolia Warbler (*Dendroica magnolia*)	C	0.55 (0.02–0.98)	0.30 (0.13–0.57)	0.54 (0.03–0.98)	0.71 (0.01–0.99)	0.370	0.577	0.649
Mourning Warbler (*O. philadeliphia*)	C	0.31 (0.17–0.48)	0.33 (0.20–0.49)	0.50 (0.28–0.72)	0.58 (0.23–0.86)	0.209	0.868	0.890
Ovenbird (*Seiurus aurocapilla*)	U	0.98 (0.84–0.99)	0.99 (0.88–0.99)	1	1	0.999	0.685	.
Red-breasted Nuthatch (*Sitta canadensis*)	U	0.08 (0.03–0.20)	0.13 (0.06–0.27)	0.10 (0.03–0.28)	0.40 (0.22–0.61)	0.016	0.010	0.193
Red-eyed Vireo (*Vireo olivaceus*)	C	0.85 (0.71–0.93)	0.95 (0.94–0.98)	1	0.95 (0.77–0.99)	0.041	0.275	.
Rose-breasted Grosbeak (*Pheucticus ludovicanus*)	U	0.03 (0.01–0.12)	0.12 (0.05–0.27)	0.55 (0.31–0.76)	0.51 (0.28–0.73)	0.001	0.061	0.103
Ruby-crowned Kinglet (*Regulus calendula*)	U	0.13 (0.06–0.30)	0.12 (0.05–0.28)	0.13 (0.04–0.33)	0.13 (0.04–0.33)	0.908	0.942	0.842
Swainson's Thrush (*C. ustulates*)	C	0.82 (0.17–0.99)	0.43 (0.29–0.57)	0.64 (0.32–0.87)	0.78 (0.39–0.95)	0.106	0.083	0.143
Tennessee Warbler (*Vermivora peregrina*)	C	0.84 (0.72–0.92)	0.83 (0.70–0.91)	0.82 (0.64–0.92)	0.75 (0.58–0.87)	0.866	0.790	0.992
White-throated Sparrow (*Zonotrichia albicollis*)	C	0.57 (0.43–0.70)	0.61 (0.45–0.75)	0.76 (0.58–0.88)	0.84 (0.65–0.94)	0.103	0.564	0.734
Winter Wren (*Troglodytes trogloydytes*)	C	0.11 (0.03–0.30)	0.13 (0.06–0.28)	0.09 (0.02–0.36)	0.12 (0.03–0.33)	0.646	0.291	0.981
Yellow Warbler (*D. petechia*)	U	0.06 (0.01–0.23)	0.02 (0.01–0.11)	0.02 (0.01–0.19)	0.06 (0.01–0.30)	0.755	0.507	0.086
Yellow-rumped Warbler (*D. coronata*)	C	0.49 (0.36–0.63)	0.72 (0.50–0.87)	0.66 (0.42–0.84)	0.88 (0.43–0.99)	0.003	0.011	0.633

[a] *Data are reported as proportion of point-count stations occupied adjusted to average vegetation conditions. The 95% confidence intervals were derived empirically. The* p *values are from likelihood-ratio tests. Main effects were not tested for significance if the interaction between noise and distance was significant at* $p \leq 0.05$.

[b] *Conditional occupancy rate (C) was derived from an adjusted occupancy rate corrected for detection error estimated separately for each stratum. When conditional models would not converge, we used standard unconditional logistic regression (U) in which a species was considered present if it was detected during any of the 4 visits.*

1981) and to provide an indicator of their quality as potential mates (Gil & Gahr 2002; Nowicki & Searcy 2005). In noisy areas male birds may not transmit their song as clearly or over as great a distance as they otherwise could. Thus, reduced song transmission might influence the number of females that hear an individual male's song or result in females' receiving false information about a male's quality (Ballentine et al. 2004). At some of the sites examined in this study, Habib et al. (2007) found that male Ovenbirds are 17% less likely to attract a female mate if their territory was adjacent to a compressor station than if their territory was adjacent to a well pad.

Ovenbird males near compressor stations also tended to be younger (48% of birds captured were first-time breeders) than individuals near well pads (30%). This suggests that mating success may play a role in altering whether males and females chose to settle in an area with noise.

Using a naïve estimator derived from raw counts and a method of detection-error correction from Farnsworth's (2002) removal modeling approach, Habib (2006) concluded that Ovenbird abundance was somewhat lower near compressor stations than near well pads. Nevertheless, when we corrected for differences in density of Ovenbirds with distance sampling, this effect was no longer significant, which highlights the importance of the underlying detection-error function as a factor in assessing the effects of noise on birds. Whether there is a direct link between changes in pairing success and bird density remains unclear as a result. Differences in pairing success between noisy and quiet areas and among different habitats need to be looked at for forest passerines besides the Ovenbird to determine the generality of the link between pairing success, human disturbance, and bird abundance.

Song is important in identification of neighboring males, demarcation and passive defence of territory boundaries, and assessment of the physiological state of conspecifics (Lein 1981; Lemon et al. 1981; Brenowitz 1982). Noise may create difficulties in territorial defense by reducing the ability of male birds to acoustically estimate the location of known conspecifics, resulting in more physical encounters between individuals. An increase in physical encounters may increase stress in forest songbirds (Mazerolle & Hobson 2002). Campo et al. (2005) found that prerecorded mechanical noises, such as fans and vehicles, increased stress levels in laying hens and that the level of that stress was directly related to noise amplitude. In the wild, females may be less likely to hear quiet acoustic cues given by chicks in response to hunger or predation risk, which could translate into reduced fecundity. Many other acoustic communication cues used by birds may be affected as anthropogenic noise increases beyond the range of natural noise under which bird communication evolved. Further investigation of the specific communication cues and aspects of avian life history that are affected by anthropogenic noise are warranted.

Demands for energy reserves from Alberta's boreal forest continue to put stress on this ecosystem and the species in it. Our results suggest that one management action to reduce energy impacts on birds would be to suppress noise at compressor stations. If 2 ha of forest around each of the approximately 5000 compressor stations in Alberta's boreal forest are cleared, then approximately 10,000 ha of trees have been lost to compressor construction. If one assumes no forest passerines use compressor stations, then habitat for approximately 27,000 birds has been lost because of land clearing (on the basis of average density of forest passerines in forest interiors). If the effects of noise that we found are the same for birds in forests other than those dominated by aspen trees, then the effective habitat loss in boreal Alberta caused by noise within 300 m of a compressor station would result in an additional 85,000 birds lost. The cost of retroactive reduction of noise levels at existing compressor stations would vary, but a typical retrofit to reduce noise by 4 db(A) at the source would be $35,000 to $50,000 (P. Wierzba, personal communication). If a noise reduction of 4 db(A) were to return forest habitat to its original value, then the cost to recover habitat lost to noise would be approximately $175–250 million. In construction of new compressors, the latest technology in noise suppression should be used because the costs of incorporating these technologies during the construction phase are believed to be less than the cost of retrofitting.

Compressor stations are only one type of disturbance in a myriad of other human disturbances occurring in boreal Alberta, so it is important to put this loss of bird habitat in context. There is a large backlog of other types of energy sites that have reduced habitat quality for birds in the boreal forest. For example, in 2005 it was estimated that there were approximately 175,000 well pads in Alberta, of which approximately one-half are abandoned (Pembina Institute 2007). Current reclamation guidelines require these wells be returned "to an equivalent land capability." Although this implies well pads must be capable of growing trees, most have been reseeded to grass and will not return to a forested state without different reclamation techniques (Schneider 2002). Current estimates suggest the average well pad can be reclaimed in a way that allows forest succession to occur for approximately $2000–4000 (B. Coupal, personal communication). If planting trees on well pads restores their value for boreal birds, then the same $175–250 million used to reduce noise could restore "forested" habitat at 44,000–63,000 well pads. Replanting well pads to trees rather than retrofitting compressor stations to reduce noise could result in increased habitat for 118,000–169,000 birds when the well pads become mature forest.

In the next 10 years a $100-billion influx of energy-sector development is planned for northern Alberta. The large reduction in habitat that will occur and the reduced quality of forested areas affected by industrial noise will result in a decrease in the number of passerine birds. Add to energy-sector disturbances the change in forest structure and age caused by industrial forestry, and the threat of extirpation of some boreal forest passerines becomes quite real. To minimize this risk it is important that energy-sector companies use new mitigation approaches that address issues related to habitat conversion and the more subtle effects, such as chronic noise, created by their activities.

Acknowledgments

We thank C. Gray, D. Pueschel, E. Cameron, A. Blake, M. Lankau, M. C. Arienti, H. Clarke, M. Conboy, T. Hope, and S. Nielsen for assistance in the field and ALPAC and IHS Energy for AVI and facility GIS data layers. This research was conducted as part of the Natural Sciences and Engineering Research Council of Canada (NSERC) Industrial Research Chair in Integrated Landscape Management. Additional funding was provided by the Canadian Circumpolar Institute, Alberta Sport, Recreation, Parks, and Wildlife Foundation, and the Alberta Conservation Association Challenge Grants in Biodiversity. Personal funding to L. H. was provided by NSERC, Alberta Ingenuity Fund, and the Alberta Society of Professional Biologists.

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Conservation Biology

Review

Effects of Road Mortality and Mitigation Measures on Amphibian Populations

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Abstract: *Road mortality is a widely recognized but rarely quantified threat to the viability of amphibian populations. The global extent of the problem is substantial and factors affecting the number of animals killed on highways include life-history traits and landscape features. Secondary effects include genetic isolation due to roads acting as barriers to migration. Long-term effects of roads on population dynamics are often severe and mitigation methods include volunteer rescues and under-road tunnels. Despite the development of methods that reduce road kill in specific locations, especially under-road tunnels and culverts, there is scant evidence that such measures will protect populations over the long term. There also seems little likelihood that funding will be forthcoming to ameliorate the problem at the scale necessary to prevent further population declines.*

Keywords: mitigation, population, road mortality, wildlife crossings

Efectos de la Mortalidad en Carreteras y Medidas de Mitigación en Poblaciones de Anfibios Beebee

Resumen: *La mortalidad en carreteras es una amenaza reconocida para la viabilidad de las poblaciones de anfibios pero que raramente ha sido cuantificada. El alcance global del problema es sustancial y los factores que afectan al número de animales muertos en las carreteras incluyen características de historia de vida y rasgos del paisaje. Los efectos secundarios incluyen aislamiento genético causado por las carreteras que actúan como barreras para la migración. Los efectos a largo plazo sobre la dinámica de poblaciones a menudo son severos y los métodos de mitigación incluyen rescates por voluntarios y túneles bajo la carretera. A pesar del desarrollo de métodos que reducen los atropellamientos en localidades específicas, especialmente los túneles bajo la carretera y alcantarillas, hay evidencia escasa de que dichas medidas protegerán a las poblaciones a largo plazo. Al parecer existe también poca probabilidad de que haya financiamiento en el futuro para aminorar el problema a la escala necesaria para prevenir la disminución futura de la población.*

Palabras Clave: cruces para fauna silvestre, mitigación, mortalidad en carreteras, población

Introduction

The development and expansion of road systems facilitating rapid movement across great distances was a major triumph of 20th century engineering. Ironically, roads in many countries now constitute the most important barriers to the movement of wildlife. In Britain there was, by 2010, an average of 172 km of roads/100 km^2 of land area (World Bank 2012). By contrast major rivers average only 22 km/100 km^2 of land in Britain (Eionet 2012).

The possible effects of roads on wildlife were apparent by the 1960s. However, many of the early investigations involved mammals (e.g., Oxley et al. 1974; Goransson et al. 1976), but amphibian and reptile casualties were among the earliest noted anywhere (Savage 1935; Hodson 1966). McCaffer (1973) showed that white-tailed deer (*Odocoileus virginianus*) killed on roads correlated with independent estimates of population size. This finding raised a question that remains pertinent today: Does road mortality reflect the dynamics of local populations

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Paper submitted May 17, 2012; revised manuscript accepted December 19, 2012.

Conservation Biology, Volume 27, No. 4, 657–668
© 2013 Society for Conservation Biology
DOI: 10.1111/cobi.12063

or have a significant effect on them. I analyzed evidence accrued over more than 40 years concerning the effect of road kill on amphibian populations and the effectiveness of measures taken to help them survive. I considered how robust the quantitative assessment of road mortality is, how large the effect, in terms of individuals killed and long-term population viability, is, and how successful mitigation methods have been. Amphibians are among the most seriously declining animal groups on Earth (Stuart et al. 2004), and it is increasingly important to identify and alleviate factors affecting their mortality rates.

Quality of Estimates of Amphibian Road Mortality

Analyzing the effect of road mortality on amphibians requires preliminary consideration of data quality. Detection methods vary in efficiency and dead animals are rapidly removed by scavengers, obliterated by traffic, or swept away by heavy rain, so numbers counted are an unknown proportion of those actually killed. Of the 2 main methods used to detect road casualties, patrolling by car is much poorer than assessment on foot. In comparative studies, variation among surveyors was low for both methods, and there was a significant correlation in numbers counted between the 2 approaches, but car-based counts were consistently <5% of foot-based counts (Langen et al. 2007; Gerow et al. 2010). For absolute estimates of efficiency, dead amphibians were marked and sampled again soon afterwards by different observers. In this test, efficiency of detection even by walking was <7% for urodeles and 32–67% for anurans (Hels & Buchwald 2001). Mean carcass residency was about 1 day for anurans and urodeles and was affected by rainfall and temperature, although not by traffic density (Santos et al. 2011). Even with daily surveys on foot, extrapolating from actual counts to estimates of true road kill requires multiplication by large and highly uncertain compensation factors (Hels & Buchwald 2001; Langen et al. 2007), from 1.5- to 15-fold (anurans and urodeles, respectively). For road-based counts the corresponding multipliers would be 30- to 300-fold. Evidently, road-kill numbers cannot accurately be related to actual deaths, but relative changes over time at the same place and with the use of standard methods should provide an adequate basis for further analyses.

Risk Factors

Amphibians seem exceptionally vulnerable to death on the road. In studies on 4 continents the percentage of vertebrate kills that were amphibians ranged from 6% to >90% (Holisova & Obtel 1986; Fuellhaas et al. 1989; Ashley & Robinson 1996; Lodé 2000; Dodd et al. 2004; Gopi Sundar 2004; Glista et al. 2008; Gryz & Krauze 2008; Gerow et al. 2010; Attademo et al. 2011; Carvalho & Mira 2011; Tok et al. 2011; Vargas-Salinas et al. 2011) and averaged about 57%. Percentages estimated by surveyors on foot were generally higher (median 80%) than those assessed by surveyors in cars (median 46%). In one comparative study (Gerow et al. 2010), estimates of the percentage of amphibians killed on roads were 62.8% for data gathered on foot and 47.05% for data gathered in a car. Accounting for underestimation of this percentage in car-based investigations, it seems probable that amphibians commonly contribute more than two-thirds of all vertebrate road deaths. Despite their vulnerability, high mortality rates do not necessarily imply greater population-level effects for amphibians than for other taxa. Rarer deaths of less fecund mammals, for example, might have more serious effects on population viability. There was no significant correlation between the percentage of vertebrate kills made up of amphibians and traffic intensity on the roads studied ($r_s = -0.44$, $p > 0.65$), but adequate data for correlation were scant ($n = 6$ data sets in 6 papers [Dodd et al. 2004; Gopi Sundar 2004; Gryz & Krauze 2008; Attademo et al. 2011; Safner et al. 2011]).

Thirty-four peer-reviewed papers published since 1973 provided data on numbers of amphibians killed on roads on 5 continents (Europe, $n = 16$ [Van Gelder 1973; Holisova & Obrtel 1986; Fuellhaas et al. 1989; Herden et al. 1998; Lodé 2000; Hels & Buchwald 2001; Salio et al. 2001; Orlowski 2007; Gryz & Krauze 2008; Orlowski et al. 2008; Reshetylo & Mykitchak 2008; Sillero 2008; Hartel et al. 2009; Carvalho & Mira 2011; Duff et al. 2011; Santos et al. 2011]; North America, $n = 10$ [Fahrig et al. 1995; Ashley & Robinson 1996; Clevenger et al. 2001; Dodd et al. 2004; Gibbs & Shriver 2005; Coleman et al. 2008; Glista et al. 2008; Langen et al. 2009; Gerow et al. 2010; Sutherland et al. 2010]; South America, $n = 4$ [Cairo & Zalba 2007; Da Silva et al. 2007; Attademo et al. 2011; Vargas-Salinas et al. 2011]; Asia, $n = 3$ [Gopi Sundar 2004; Gu et al. 2011; Tok et al. 2011]; Australasia, $n = 1$ [Goldingay & Taylor 2006]). Seventy-six species, constituting more than 1% of global amphibian taxa, were reported as road casualties in these papers (Table 1). Nomenclature sometimes differs from that in original papers following the review by Frost et al. (2006) and a later revision by Speybroeck et al. (2010).

Anurans constituted 65% or more of the total records of road mortalities, probably at least in part because of recording bias against urodeles (Hels & Buchwald 2001). The European common toad (*Bufo bufo*) has received more attention than any other species with respect to road mortality. Coverage of amphibian mortality in Europe and North America was relatively extensive and probably gives a fair impression of the species most at risk on those continents. Elsewhere, data were too sparse for any generalizations. Table 2 summarizes information from papers cited about numbers of individuals killed in relation to factors that may affect vulnerability to road mortality.

Table 1. Amphibians reported as road kills and number of papers citing each species (in parentheses).

Europe	*North America*	*South America (single reports)*	*Asia (single reports)*	*Australasia (single reports)*
Bufo bufo (27)	*Ambystoma maculatum* (8)	*Hysiboas allomarginalis*	*Bufo bufo*	*Crinia tinnula*
Rana temporaria (8)	*Lithobates pipiens* (6)	*Hysiboas faber*	*Bufo minshanicus*	*Limnodynastes peroni*
Lissotriton vulgaris (4)	*Lithobates clamitans* (5)	*Hysiboas pulchellis*	*Bufo viridis*	*Limnodynastes terraereginae*
Pelobates fuscus (4)	*Hyla cinerea* (3)	*Hysiboas raniceps*	*Duttaphrynus melanosticus*	*Litoria fallax*
Triturus cristatus (4)	*Lithobates sylvatica* (3)	*Hysiboas semilineatus*	*Duttaphrynus stomaticus*	*Litoria freycineti*
Pelophylax lessonae/esculentus (3)	*Lithobates sphenocephalus* (3)	*Leptodactylus gracilis*	*Holobatrachus tigerinus*	*Litoria olongburensis*
Rana arvalis (3)	*Acris crepitans* (2)	*Leptodactylus latrans*	*Hyla arborea, Hyla savignyi*	*Pseudophryne coriacea*
Bufo calamita (2)	*Anaxyrus americanus* (2)	*Leptodactylus notoaktites*	*Nanorana pleskei*	*Rhinella marina*
Lissotriton helveticus (2)	*Anaxyrus terrestris* (2)	*Leptodactylus ocellatus*	*Pelophylax bedriagiae*	
Pleorodeles waltl (2)	*Gastrophryne carolinensis* (2)	*Melanophryniscus* sp.	*Pelophylax ridibundus, Rana kukunoris*	
Rana dalmatina (2)	*Lithobates catesbeianus* (2)	*Rhinella fernandezae*	*Rana macronemis*	
	Scaphiopus holbrookii (2)	*Rhinella schneideri*	*Triturus karelinii*	
		Scinax alter		
		Scinax nasicus		
		Scinax perececa		
Single reports	**Single reports**			
Alytes cysternasii	*Ambystoma laterale*			
Alytes obstetricians	*Ambystoma tigrinum*			
Bombina bombina	*Anaxyrus fowleri*			
Bufo viridis, Discoglossus galganoi,	*Anaxyrus velatus*			
Hyla meridionalis	*Hyla chrysocelis*			
Lissotriton boscai	*Hyla squirella*			
Ichthyosaura alpestris	*Lithobates palustris*			
Pelobates cultripes	*Lithobates sepentrionalis*			
Pelophylax perezi	*Notophthalmus viridiscens*			
Rana iberica	*Plethodon cinereus*			
Salamandra salamandra	*Pseudacris crucifer*			
Triturus marmoratus	*Pseudotriton ruber*			

Urodeles and anurans did not differ significantly in the proportion of papers reporting 10 or fewer killed compared with more than 100 deaths (Yates-corrected $\chi^2 = 1.23$, df $= 1$, $p > 0.05$). Terrestrial species that migrate to breeding ponds constituted by far the majority of records (>79%) in all mortality classes. This bias was confirmed by comparing species killed in Europe and North America with the total available species in the study areas (Conant & Collins 1998; Arnold & Ovenden 2002; Stebbins & Peterson 2003), excluding the relatively few live-bearing (thus nonmigratory) salamanders on both continents. Overall, 38.5% of 83 available species were killed on roads, and there was a significant bias in favor of terrestrial species (55% of the 42 species available in the study areas). By comparison, 31% of 16 arboreal species and 16% of 25 primarily aquatic species were killed ($\chi^2 = 6.35$, df $= 2$, $p < 0.05$). Arboreal species also migrate to breeding ponds, but their relatively low

Table 2. Patterns of amphibian road mortality on the basis of number of published reports and peer-reviewed articles.

	10s or fewer individuals	*100s of individuals*	*1000s of individuals*	*Total records*
Order				
Anura	30	21	9	60
Urodela	9	4	0	13
Autecology				
mainly aquatic	6	0	2	8
arboreal	4	2	1	7
terrestrial	29	23	6	58

mortality rates may be a consequence of more active behavior on or approaching roads. In experimental tests, many species remained still when approached by a motor vehicle, responding both to engine noise and headlights, but tree frogs (*Pseudacris crucifer*) were the least likely to remain still (Mazerolle et al. 2005). Vulnerable species such as leopard frogs (*Lithobates pipens*) do not avoid roads during migration and move more slowly near roads than when distant from them (Bouchard et al. 2009).

Landscape type may also be a significant factor. In Poland, European common toads contributed a higher proportion of suburban than rural road kill numbers, whereas the opposite was true of brown frogs (*Rana* spp.) and water frogs (*Pelophylax* spp.), probably as simple correlates of their relative abundances in the 2 environments (Elzanowski et al. 2009). For foot-based surveys ($n = 20$), there was a strong positive correlation across all publications between kills per kilometer per year and traffic density measured as mean number of vehicles per hour ($r_s = 0.705$, $p = 0.0007$).

Secondary Effects of Roads

Although the most obvious consequence of roads on amphibians is crushing by vehicles, this is not the only cause of death. In Britain, 75 great crested newts (*Triturus cristatus*) were killed during spring migration by salt applied to roads to remove ice (Duff et al. 2011). In the Adirondack Mountains of the United States salt reduced embryonic and larval survival of spotted salamanders (*Ambystoma maculatum*) in ponds close to treated roads, predisposing severe (20–40%) declines of populations within 50–100 m when applications were sustained (Karraker et al. 2008). However, a more pervasive issue is the genetic consequences of habitat fragmentation by roads.

Maintaining high levels of genetic diversity is one of several key factors that reduce extinction risk, and this requires large population sizes with sufficient connectivity to facilitate migration between them (Amos & Balmford 2001). Busy roads effectively fragment pond-breeding amphibian populations leading to smaller sizes with reduced genetic diversity. Populations of common frogs (*Rana temporaria*) separated by motorways in Germany had reduced diversity and intersite migration compared with populations separated by farmland or woodland (Rey & Seitz 1990). A similar negative effect of roads on this species was found in southern Sweden (Johansson et al. 2005). In the French Alps, results of genetic analyses suggest that new roads rapidly curtailed individual movement of common frogs but that population differentiation was much slower, probably operating over decades (Safner et al. 2011). Similar results have accrued from studies of other species in Europe (Moor frogs [*R. arvalis*] [Arens et al. 2007], agile frogs [*R. dalmatina*] [Lesbarrères et al. 2003, 2006]) and North America (wood frogs [*Lithobates sylvatica*] [Crosby et al. 2009], red-backed salamanders [*Plethodon cinereus*] [Marsh et al. 2008]). However, it is in urban environments that adverse genetic effects are most severe. Urban populations of red-backed salamanders in Quebec exhibited lower diversity and higher genetic differentiation than those in continuous high-quality habitat (Noel et al. 2007). In Britain common frogs and European common toads in town gardens persist as small populations isolated over short distances with reduced fitness due to inbreeding relative to those in surrounding countryside (Hitchings & Beebee 1997, 1998). Urban areas may provide breeding sites for amphibians in the short term, but over time urban areas are likely to constitute relatively poor-quality habitat with low species diversity (Parris 2006).

Amphibians and large mammals are the taxa worst affected genetically by roads (Holderegger & Di Giulio 2010). Because substantive genetic differentiation after barrier imposition can take many generations, current analyses may underestimate the long-term significance of this relatively novel obstruction to individual movement. Furthermore, neutral molecular markers used in these studies may not be ideal surrogates for adaptive genetic variation, the type that matters to population viability. Nevertheless strong fragmentation in urban areas (Hitchings & Beebee 1997, 1998) and long-term isolation of small populations (Rowe & Beebee 2003) can certainly adversely affect amphibian population fitness.

Effects of Roads on Populations

Two approaches have been used to determine whether roads affect amphibians at the population level. Four studies monitored road mortality at specific sites over several years to see whether numbers changed over time. However, at least 20 studies have reported on the abundance and diversity of amphibians near and far from roads and thus inferred road effects indirectly.

In a Canadian study, numbers of some species (ranid frogs and ambystomid salamanders) crossing a secondary road did not change over 8 years, whereas crossings by other species (bufonid toads and spring peepers)

Table 3. Effect of roads on amphibian species richness (ASR) and abundance.

Region	*Landscape*	*Species*	*Road effects on species richness**	*Paper*
Canada (Ontario)	mostly forest	not listed	19% decline ASR up to 2 km	Findlay & Houlahan 1997
Canada (Ontario and Quebec)	mixed, with variable amount of forest	9 anurans	up to 40% decline ASR within 500 m on the basis of traffic density	Eigenbrod et al. 2008*a*, 2009
U.S.A. (Appalachians)	forest	7 urodeles	50–60% decline ASR within 40 m	Semlitsch et al. 2007
U.S.A. (Appalachians)	forest	6 urodeles	no effect on ASR, minor decline (<20%) in abundance	Ward et al. 2008
Romania	mostly farmland and forest	9 anurans 2 urodeles	18% (low traffic) or 45% (high traffic) decline ASR	Hartel et al. 2010
Colombia	forest	8 anurans	decline in ASR and abundance of 40% within 10 m compared with ASR and abundance 150 m from road	Vargas-Salinas et al. 2011

**Distances are from the road.*

actually increased (Mazerolle 2004). The most extensive time series, however, are available for European common toads. In the Netherlands, numbers arriving at roads over 4–6 years during spring migrations remained stable at 5 locations but declined 2–99% at 9 other locations (Zuiderwijk 1989). Reduced numbers do not necessarily reflect overall population declines because fractions of populations migrating across the monitored roads are usually unknown. The ratio of road kills to spawners (*R/S*) is an important measure if road effects on populations are to be properly quantified, and the ratio can be high, often 0.23–0.66 (Elzanowski et al. 2009). In a study of European common toads near 3 ponds in England, the number of animals killed on neighboring roads showed a marked decline over 20 years that correlated with reduced numbers of animals counted in the breeding sites and inversely with increased traffic volume (Cooke 2011). By 2010 there were hardly any toads left. Such trends may erode short-term correlations between road kills and traffic volume if they are carried out after a period of population decline.

More researchers have investigated road effects indirectly by comparing amphibian numbers at various distances from highways. Species richness is frequently lower close to roads than in more distant locations (Table 3), although it can be difficult to disentangle direct effects of road mortality from covariant landscape features such as forest cover (e.g., Houlahan & Findlay 2003; Semlitsch et al. 2007; Vargas-Salinas et al. 2011) and behavioral consequences such as traffic noise interfering with male vocalization near highways (Hoskin & Goosem 2010). Irrespective of mechanism, however, roads can substantially reduce amphibian diversity within 50–2000 m of them. Furthermore, models that account for historic road densities indicate a substantial lag period, which for amphibians can extend over decades, before the full effect on species richness becomes clear (Findlay & Bourdages 2000). Eigenbrod et al. (2008*b*) analyzed combined habitat and road effects on species richness by defining *accessible habitat* as that available without crossing a road. Amphibian diversity was better predicted by this criterion than by invoking habitat or motorway proximity as separate variables.

Results of analyses at the level of individual species, often in models with other environmental variables, are summarized in Table 4. Rytwinski and Fahrig (2012) showed by meta-analysis that amphibians and reptiles are the taxa most likely to exhibit population declines associated with road mortality. Urodeles are more likely to decline than anurans and small early-maturing frogs declined more than larger species, in both cases possibly reflecting low reproductive rates. Road characteristics were not identified as important contributors to the extent of declines.

However, the proportion of studies of anurans that attributed population decline to roads (20/27, 74%) was greater than the proportion of studies of urodeles that attributed decline to roads (4/9, 38%). This may be explained by the larger median home ranges of anurans (40 m^2) relative to urodeles (4 m^2) and longer median maximum migration distances to breeding sites (anurans = 500 m, urodeles = 250m) (Wells 2007). All of the 27 species investigated except one (red-backed salamanders) regularly migrate to breeding ponds. Of 6 arboreal species, 1 (tapping green-eyed frogs [*Litoria serrata*]) was unaffected by roads. The road-effect zone (distance from road over which population size effects were detected) varied from 40 m (red-backed salamanders) to 1.5 km (leopard frogs), averaging around 0.5 km for both anurans and urodeles, and was significantly greater overall in unforested (mean 565 m) than in forested sites (mean = 233 m) (Kruskal-Wallis statistic = 4.497, p = 0.034). There was no significant correlation across studies (n = 13) between decline near roads (varying from zero to extinction) and traffic density, which varied from approximately 17 to 1284 vehicles/hour (r_s = 0.219, p = 0.459). However, including only studies that showed a decline, there was a relation between decline and effect

Table 4. Effects of roads on amphibian populations.

Region	*Landscape*	*Species*	*Measure of road effects*	*Critical variables*	*Reference*
Canada (Ottawa)	mostly forest	anurans	road kill counts ∝ 1/traffic	traffic density	Fahrig et al. 1995
Canada (Ontario)	mixed, mostly farmland and forest	*Lithobates pipiens*	abundance reduced up to 1.5 km	traffic density, species vagility	Carr and Fahrig 2001
		L. clamitans	no effect		
Canada (Ontario)	mixed, with different amounts of forest	*L. sylvaticus*	abundance reduced up to 0.6–1 km	aquatic and terrestrial land cover, distance from motorway	Eigenbrod et al. 2009
		L. pipiens			
		L. clamitans	>1 km		
		Pseudacris triseriata	>1 km		
		P. crucifer	1.1–2.4 km		
		Anaxyrus americanus	0.2–0.3 km		
		Hyla versicolor	0.2–0.3 km		
USA (Ohio)	mostly farmland	*Ambystoma texanum*	no effect	road density, forest cover	Porej et al. 2004
		A. maculatum	no effect		
		A. jeffersoniamum	no effect		
		A. tigrinum	up to 100% decline, up to 1 km		
		Notophthalmus viridiscens	no effect		
		Lithobates sylvaticus	no effect		
USA (Rhode Island)	mostly forest	*A. maculatum*	no effect	aquatic and terrestrial land cover,	Skidds et al. 2007
		L. sylvaticus	no effect		
USA (Appalachians)	forest	*Plethodon cinereus*	abundance reduced by 17% (road type) or 28% at <40 m distance	road type (gated, ungated), landscape features	Marsh 2007
USA (Vermont)	mostly forest	*N. viridiscens*	no effect at any scale	extent of habitat & type of cover	Rinehart et al. 2009
USA (New Hampshire)	forest	*A. maculatum*	up to 100% decline within 1 km of pool	road density	Veysey et al. 2011
		L. sylvaticus	up to 70% decline within 120 m		
Netherlands	moor and farmland	*Rana arvalis*	up to >70% decline near motorway	habitat fragmentation	Vos and Chardon 1998
Switzerland	mostly farmland	*Hyla arborea*	pond occupancy reduced to zero within 100 m	road and traffic density	Pellet et al. 2004
Sweden	mixed habitats	*Pelobates fuscus*	extinction within 500 m	soil type, traffic density	Nyström et al. 2007
Romania	mostly farmland and forest	*Lissotriton vulgaris*	reduced abundance within 400 m	landscape features, distance from road, traffic density	Hartel et al. 2010
		Triturus cristatus			
		Bufo bufo	800 m		
		Rana dalmatina	600 m		
		Rana temporaria	400 m		
		Hyla arborea	800 m		
		Bombina variegata	600 m		
		Pelophylax esculentus	600 m		
		Pelobates fuscus	positively affected at 400 m no effect		

continued

Table 4. continued

Region	*Landscape*	*Species*	*Measure of road effects*	*Critical variables*	*Reference*
China (Tibet)	moorland	*Rana kukunoris*	both species declined by 67–68% within 100 m	distance from road, aquatic cover	Dai and Wang 2011
Australia (Queensland)	forest	*Nanorana pleskei*			
		Litoria serrata L. rheocola	no effect	body size, call characteristics	Hoskin and Goosem 2010
		Austrochaperina pluvialis	variable for both species. No effect or 60–70% reduction within 100 m		

distance ($n = 17$, $r_s = 0.609$, $p = 0.011$) and between traffic and effect distance ($n = 11$, $r_s = 0.683$, $p = 0.023$). Overall, studies of 16 species on 4 continents confirmed a substantive effect of roads on amphibian abundance (mean 50% reduction near highways).

The consequences of road mortality on amphibian population dynamics have also been modeled on the basis of empirical data. Hels and Buchwald (2001) measured survival rates of European amphibians crossing roads and, in conjunction with traffic density, modeled probabilities of amphibians being killed (almost 1 on the busiest roads). Highly affected species such as common frogs and eastern spadefoot toads (*Pelobates fuscus*) could experience 20–25% annual mortality, although the effect on population persistence would only be high if density-independent factors dominated population dynamics. When density-dependent processes are important, as they can be at least at the larval stage, road kills are unlikely to act in a simple additive fashion to increase overall mortality rates. Using spotted salamander data from a road in New York, Gibbs and Shriver (2005) showed that an annual mortality rate of >10% can lead to population decline. They found that even when they accounted for density-dependent factors acting in early life stages and buffering populations against low traffic levels, the actual risk was in most cases well above this threshold.

Mitigation

One approach to mitigate the effects of roads on wildlife is modification of human behavior. Warning signs to make drivers aware of amphibian crossings and temporary road closures (26 in one area of Germany alone by the 1980s) have been implemented for more than 40 years (Podloucky 1989). Evidence of their value is scant. Signs are probably ineffective in most cases (Glista et al. 2009) and have little or no effect on driver speed (Coulson 1982; Jochimsen et al. 2004). Similarly, although seasonal road closures have been in place for decades, data on their effects on amphibian populations are lacking (e.g., Hilty et al. 2006). However, there is certainly scope for improving conservation prospects when designing new roads. Such developments are likely to continue, particularly in developing countries, for many years to come. Landscape-level planning to minimize future habitat fragmentation and maintain connectivity, through the use of GIS-based information technology and ecological data, is available and increasingly applied by transport authorities (Taylor & Goldingay 2010).

More promising for moderating effects on existing roads are methods that modify amphibian behavior. Volunteer groups, often combined with temporary fencing and pitfall traps along roads, catch amphibians during spring migrations and carry them across the highway. In Britain by the 1980s there were >400 such crossings in which over 500,000 animals (mainly European common toads) were transported annually (Langton 1989*a*). Twenty years later there were nearly 900 toad-crossing sites, although numbers of animals moved annually were <100,000 and the percentage killed still averaged about 10% of those arriving at the roads between 2007 and 2011 (Froglife 2012). This method and those that attempt to modify driver behavior have obvious disadvantages: animals are only protected en route to their breeding sites. It is rarely possible to help amphibians as they disperse after breeding ceases, when mortality rates can be high (Herden et al. 1998), or to assist newly metamorphosed animals as they move away from ponds. Short-term benefits (reduced number of road deaths) have been reported (Linck 2000), but in the Netherlands 9 out of 14 populations assisted in this way declined over 6 years (Zuiderwijk 1989), and over longer periods complete extirpation can occur (e.g., Cooke 2011) irrespective of toad-crossing efforts.

Another way to decrease road mortality is construction of new breeding ponds such that road crossing is no longer necessary. Such efforts include fences to prevent animals trying to reach their original pond. This proved successful for European common toads in Germany. A second breeding site was established and used by >99% of migrating toads within 5 years (Sclupp & Podloucky 1994). Creation of 8 new ponds following highway construction in France restored breeding of at least 4 out of 6 amphibian species that had used ponds destroyed during development (Lesbarrères et al. 2010). However, barrier effects with adverse consequences for metapopulation

Table 5. Characteristics of tunnels or culverts used for mitigation.[a]

Feature	*Mean*	*Median*	*Range*
Individuals using culvert (%)	50.7	42.5	4–100
Culvert length (m)	14.7	10.0	7–44
Culvert width (m)	0.48	0.20	0.1–2.5
Culvert height (m)	0.51	0.40	0.3–1.2
Culvert openness[b]	0.082	0.074	0.006–0.160
Fence length around culvert (km)	2.14	0.40	0.36–4.00
Fence height (m)	0.69	0.4	0.4–4.0
Fence mesh size (mm)	1.67	0	0–6.00

[a] *Data from Brehm (1989), Meinig (1989), Zuiderwijk (1989), Jackson & Tyning (1989), Langton (1989a), Taylor & Goldingay (2003), Dodd et al. (2004), and Lustrat (2005).*
[b] *Culvert openness, cross-sectional area per length.*

stability and genetic diversity remain unless compensated for by ecopassages below or above the roads (as created subsequently at the French site).

Wildlife underpasses (tunnels or culverts) specifically for amphibians were initiated in the 1960s in Europe (Ryser & Grossenbacher 1989). More recently overpasses, including green bridges, have worked well for some groups of animals but have not proved attractive to amphibians (Taylor & Goldingay 2010). Unlike mammals, herpetofauna lack any behavioral ability to avoid roads and must be directed to safe crossing sites by extensive barrier fencing. One-way tunnels with fencing on just one side of the road have the same problem as volunteer transport (Dexel 1989).

Placement can be problematic. For migratory species it is usually possible to identify crossing hotspots, but predictions of where these are can sometimes be difficult (e.g., spotted salamanders [Patrick et al. 2010]), and for rainforest frogs mortality may be distributed over long stretches of highway. Frequency of tunnels varies from one to several at hotspots (e.g., two, 250 m apart [Langton 1989*b*], nine about 155 m apart [Taylor & Goldingay 2003], and eight about 290 m apart [Dodd et al. 2004]).

A summary of tunnel characteristics from multiple papers is given in Table 5. Most are of approximately rectangular cross-section, 0.4 m high, and 0.3 m wide, and on average they are associated with 400 m of 0.4-m-high fencing (200 m on each side) that directs amphibians to entrances. Fences were sometimes meshed, but often they are made of uniform sheeting. However, usage measured as numbers of arriving animals that actually enter the tunnels is highly variable. The percentage at one site in Germany varied from 11–12% for smooth newts (*Lissotriton vulgaris*) to 26–30% for European common toads (Brehm 1989), but at another site >80% of amphibians used a wider tunnel (Meinig 1989). More than 75% of spotted salamander (*A. maculatum*) in Massachusetts negotiated culverts successfully (Jackson & Tyning 1989), and 98% of European common toads did so at a locality in England (Langton 1989*b*). Tree frogs (hylids) generally benefit little from tunnels because they can traverse any fence with ease. In Florida barrier and culvert systems at Payne's Prairie reduce mortality of most vertebrates by >70%, but have no effect on the large number of *Hyla* species killed on the road (Dodd et al. 2004). Evidently tunnels with fences can reduce amphibian mortality in some situations. Mortality of long-toed salamanders (*Ambystoma macrodactylum*) dropped from 10% to 2% of the population per annum at a site in Alberta, Canada (Pagnucco et al. 2012). However, results are highly variable, and for European common toads assessed at 4 separate sites, tunnel usage ranged from 4% to virtually 100%. No correlations were detected between tunnel use and any of the tunnel or fence variables listed in Table 5 (results not shown).

To investigate difference in use, tunnel and fence systems for amphibians have been investigated experimentally. "Tunnel hesitation" has been observed in many species (e.g., Jackson & Tyning 1989). For agile frogs and green frogs a soil or sand substrate increases usage 3- to 8-fold compared with bare concrete, whereas for European common toads, leopard frogs, American toads (*Anaxyrus americanus*), and spotted salamanders there is no substrate effect (Lesbarrères et al. 2004; Woltz et al. 2008; Patrick et al. 2010). Varying tunnel width from 0.3 to 0.8 m has no effect on American toads but doubles usage by green frogs and leopard frogs, whereas increasing fence height from 0.3 to 0.6 m increases constraint efficiency from 80% to almost 100% for these species (Woltz et al. 2008; Patrick et al. 2010). Optimal tunnel design should probably include widths and heights ≥0.4 m and roof slats to allow light penetrance and natural substrate rather than bare concrete. Fences should extend at least 100 m on each side of the tunnel on both sides of the road, be curved backwards at the top in the direction of arriving migration, and be at least 0.6 m high.

Can tunnels rescue populations? Remarkably, there is little published information on how they affect long-term population dynamics. In France mass mortality of European common toads occurred on a main road in 1991 (2500 dead). Volunteers sustained the population for 4 years by carrying animals across the road; thereafter, the population was sustained for at least another 10 years by a tunnel and fence system. Nearly 3000 animals passed through this system in 2005 (Lustrat 2005). At a site in Switzerland, effects of tunnel installation on the population dynamics of common frogs and common toads were monitored before tunnel construction and for 12 years after construction and were compared with a site 30 km away in a similar type of area. Numbers of both anurans increased about 4-fold in a single year, 4 seasons after tunnel construction (this did not happen at the control site), and dropped back to pretunnel levels the next year (Jolivet et al. 2008). On the basis of a mathematical model,

the authors proposed that there was initial improvement in recruitment followed quickly by density-dependent down regulation. However, this type of regulation is not well documented in postmetamorphic (as opposed to larval) stages of anuran life cycles. Furthermore, common toad numbers declined substantially starting 7 years after tunnel construction in a fashion not predicted by the model. Overall, there is little rigorous research, with appropriate replication and controls carried out over many years, on any aspect of road-mitigation work (Lesbarrères & Fahrig 2012). Tunnels are not without risk; they may focus predation on high local concentrations of migrating amphibians or cause occasional mass mortality by flooding or oil seepage. Tunnel and fence systems also require continuous maintenance to work well, and this is often not sustained.

Discussion

Roads destroy high-quality habitat and have a disproportionately large negative effect relative to the area of development involved. Amphibians constitute the highest proportion of wildlife casualties on roads because of their limited behavioral responses and in many cases a need to migrate for reproduction. In the United Kingdom, European common toads declined recently in much of England, whereas common frogs fared much better (Carrier & Beebee 2003). Roads are likely to most strongly affect species that regularly migrate long distances (Puky 2005), a characteristic of European common toads but less so of common frogs. Road density in England averages >260 km per 100 km^2 and few breeding ponds are further than 0.5 km from a road of some kind.

None of the available mitigation measures have a proven track record at sustaining populations over the long term as opposed to reducing kill rates at specific times and places. Tunnels are probably the best method, but together with essential guidance fencing they are expensive to create after road construction (in excess of $US20,000 each in the United Kingdom), which will always be the commonest situation. Vast networks of tunnel and fence systems would be needed to have an effect at large (regional or national) extents. Expense does not end with construction. Regular maintenance of fencing (repair, vegetation removal, etc.) would be a huge task over the thousands of kilometers needed, and although there is potential for volunteer labor, the number of people required implies a need for complex and expensive organization. Perhaps the best hope is to focus mitigation on hotspots where road mortality is most acute. Such hotspots often correspond to amphibian migration routes and can be identified empirically and by model predictions that are based on wetland and road proximities (Langen et al. 2009). Much more work is needed to determine whether and how well mitigation methods work and what factors affect success or failure as measured by long-term population maintenance.

Acknowledgments

I thank D. Lesbarrères, 2 anonymous referees, and handling editor K. French for constructive comments on earlier drafts of this paper.

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Experimental Chronic Noise Is Related to Elevated Fecal Corticosteroid Metabolites in Lekking Male Greater Sage-Grouse (*Centrocercus urophasianus*)

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Abstract

There is increasing evidence that individuals in many species avoid areas exposed to chronic anthropogenic noise, but the impact of noise on those who remain in these habitats is unclear. One potential impact is chronic physiological stress, which can affect disease resistance, survival and reproductive success. Previous studies have found evidence of elevated stress-related hormones (glucocorticoids) in wildlife exposed to human activities, but the impacts of noise alone are difficult to separate from confounding factors. Here we used an experimental playback study to isolate the impacts of noise from industrial activity (natural gas drilling and road noise) on glucocorticoid levels in greater sage-grouse (*Centrocercus urophasianus*), a species of conservation concern. We non-invasively measured immunoreactive corticosterone metabolites from fecal samples (FCMs) of males on both noise-treated and control leks (display grounds) in two breeding seasons. We found strong support for an impact of noise playback on stress levels, with 16.7% higher mean FCM levels in samples from noise leks compared with samples from paired control leks. Taken together with results from a previous study finding declines in male lek attendance in response to noise playbacks, these results suggest that chronic noise pollution can cause greater sage-grouse to avoid otherwise suitable habitat, and can cause elevated stress levels in the birds who remain in noisy areas.

Citation: Blickley JL, Word KR, Krakauer AH, Phillips JL, Sells SN, et al. (2012) Experimental Chronic Noise Is Related to Elevated Fecal Corticosteroid Metabolites in Lekking Male Greater Sage-Grouse (*Centrocercus urophasianus*). PLoS ONE 7(11): e50462. doi:10.1371/journal.pone.0050462

Editor: Nicola Saino, University of Milan, Italy

Received July 18, 2012; **Accepted** October 22, 2012; **Published** November 20, 2012

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Funding: Funding was provided by the National Parks Service, US Bureau of Land Management, Wyoming Sage-Grouse Conservation Fund (State of Wyoming), National Fish and Wildlife Foundation, the Wyoming Community Foundation Tom Thorne Sage-Grouse Conservation Fund, UC Davis, and the National Science Foundation (IOS-0925038). The funders had no role in study design, data collection and analysis, decision to publish, or preparation of the manuscript.

Competing Interests: The authors have declared that no competing interests exist.

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Introduction

Anthropogenic noise is becoming ubiquitous as natural landscapes are increasingly dominated by humans, but we still have much to learn about the impacts of chronic noise exposure on wildlife [1–3]. Recent studies have shown that some species avoid developed areas with high noise levels, reducing available habitat and potentially leading to reduced populations [4–6]. However, there is variation among species and individuals in the tendency to avoid noise [4,5,7], which raises the question of whether animals that remain suffer detrimental effects, or if these individuals are better able to habituate to noise or are less susceptible to its effects. It has been suggested that animals remaining in (or unable to leave) noisy areas may have lower survival and reproductive success [8–10]; indeed, recent studies have demonstrated complex effects of noise on community structure and on breeding and pairing success [4–6,11]. Given the ubiquity of noise in the environment, it is critical that we understand noise impacts on animals whether they remain in or avoid disturbed areas.

One possible impact of introduced noise on animals is the induction of stress, which may be defined broadly as nonspecific adverse effects in vertebrates but is most often characterized by its influence on neuroendocrine physiology. The duration of noise exposure affects the stress response of animals exposed to it [12]. Exposure to a brief but loud noise event, such as a single sonic boom, will result in an acute stress response. An acute stress response is characterized by a rapid release of epinephrine and norepinephrine (the "fight or flight" response) followed by a hypothalamic-pituitary-adrenal (HPA) cascade. The HPA cascade results in increased secretion of glucocorticoid hormones, cortisol or corticosterone, in the blood. Long-term exposure to a chronic noise stressor, such as a high-traffic freeway, can lead to chronic stress, defined as long-term overstimulation of coping mechanisms. This in turn can lead to less predictable changes in the HPA axis. Acclimation or exhaustion may result in reduced glucocorticoid release to the same or novel stressors; facilitation, conversely, can lead to elevated glucocorticoid release in response to novel stressors, and even in cases of reduced peak glucocorticoid response, deficits in negative feedback may develop that result in greater overall exposure to glucocorticoids due to prolonged elevation [12,13].

Glucocorticoid hormones and their metabolites are commonly used to measure a stress response [14–16]. Glucocorticoid hormones can be measured from blood samples or their metabolites may be measured non-invasively from fecal samples

as an index of the relative physiological stress of animals [17–19]. Glucocorticoid hormones play a major role in allocating energy, and prolonged exposure due to chronic stress can affect fitness by inhibiting resource allocation to reproductive or immune activities, a condition known as allostatic overload [12,20–24].

Studies in captive animals have found that noise can increase HPA activity and glucocorticoid levels [25,26]; indeed studies of stress physiology often use noise exposure as a method to induce a stress response [27,28]. Previous observational and experimental studies on the impacts of anthropogenic noise on glucocorticoid levels in wild animals have yielded mixed results. Snowmobile and wheeled-vehicle traffic was associated with elevated fecal glucocorticoid metabolites in wolves and elk [14]. Noise is one potential mechanism of this impact, but visual and other types of disturbance may also contribute to these responses; indeed, the quieter activity of Nordic skiing also correlates with FCMs in capercaillie (*Tetrao urogallus*) [29]. Delaney et al. [30] found behavioral responses in spotted owls to loud noise from visually hidden chainsaws and helicopters, but subsequent studies found no evidence of change in FCMs with exposure to quieter chainsaw noise (below behavioral response threshold) or road proximity to nesting sites [31]. Results from chronic noise studies on humans have also been mixed [32]. Studies of children in areas with high road noise have found increased overnight glucocorticoid levels in urine, as well as impaired circadian rhythms, sleep, memory and concentration, [33] and increased heart-rate responsiveness to acute stressors [34]. However, a study in children living in communities near airports found increases in some measures of stress (blood pressure, epinephrine and norepinephrine) but no similar elevation in overnight urinary cortisol [35]. These results indicate that noise may have a significant effect on glucocorticoids and other stress-related variables in many species, but that further study is needed to determine the degree and extent of these effects and how the effects may vary with different types of noise.

In this study, we test the hypothesis that chronic noise causes an increase in stress levels of lekking greater sage-grouse. We used fecal levels of immunoreactive corticosteroid metabolites (FCMs) as an index of physiological stress and compared FCMs for breeding males on display grounds (leks) with and without experimentally introduced noise. The greater sage-grouse, an iconic species once widespread in western North America, is now declining throughout its range, leading to its listing as an endangered species in Canada and its recent designation as "warranted but precluded" for listing under the Endangered Species Act in the USA [36,37]. Over the last decade, natural gas development has expanded rapidly across much of the sage-grouse range and has been implicated in reduced lek attendance and abandonment of long-occupied (often for decades) lek sites by males [e.g. 38,39–41]. Males typically gather on lekking grounds for several hours in the early morning when conditions are quiet and still, a time when they may be particularly vulnerable to disturbance from noise pollution from natural gas development and other sources [42]. To investigate whether noise exposure may have contributed to declines in lek attendance, Blickley et al. [43] experimentally introduced noise from natural gas development activities (drilling and road noise) on leks over three breeding seasons (2006–2008). This noise playback caused immediate and sustained declines in sage-grouse lek attendance. Further, different types of noise had different degrees of impact, with drilling noise and road noise causing an average 29% and 73% decline in lek attendance, respectively, compared to their paired controls. That study provides evidence that anthropogenic noise from energy development causes some males to avoid attending leks with introduced noise, but we do not yet know whether noise also has a negative impact on the individuals that remain on noisy leks. The lekking season is a time of high metabolic demand [44] and stress [45] for males, so exposure to noise during this period may have a greater fitness cost.

Here we compare the FCM levels of male sage-grouse on control leks and leks with experimentally introduced noise in the second and third seasons of experimental noise playback (2007 and 2008) [43]. We predict that if noise exposure leads to chronic stress, male sage-grouse on experimental leks will have higher FCMs than males on control leks. Such differences in observed FCM levels may also be observed if males with low glucocorticoid levels are more likely to disperse from noise-treated leks, so we compared the variance in FCM levels on noise and control leks. We also investigated whether elevated FCM levels were associated with declines in peak male attendance on leks to determine the value of this metric as a tool for predicting lek declines.

Materials and Methods

Study Area & Experimental Design

Study sites were located on federal land relatively undisturbed by human development in Fremont County, Wyoming (42° 50′, 108° 29′30). We monitored a total of 16 leks that were divided into 8 pairs, with the leks of a pair matched according to size and location (6 pairs near the town of Hudson and 2 pairs near the town of Riverton) (Figure 1). Of the 8 lek pairs, 4 pairs were randomly assigned to each noise type, such that there were 4 "drilling pairs", each including one lek exposed to drilling noise and a similar lek as its control, and 4 "road pairs," each with one road noise and a matched control. For 3 of the pairs, one lek within a pair was randomly assigned to the treatment (noise) group and the other assigned as control. For the fourth pair, the treatment and control leks were deliberately assigned due to another study that was in progress. During sample collection periods, both leks in a pair were normally visited on the same day.

Noise and playback methods have been previously described [43] and are summarized here. Noise was played beginning in mid-February to early March and continuing through the end of April of each year. Noise was recorded from drilling and main road sites at the Pinedale Anticline natural gas fields and played back using a commercial car amplifier and 3–4 rock-shaped outdoor speakers placed along one edge of the lek. On leks with road-noise playback, recordings of semi-trailer trucks and pickup trucks were combined with 30- and 60-second files of silence at a ratio reflecting the average number of each truck type found on a main energy field access road; these files were then played using the "random shuffle" feature on an MP3 player. Most shift changes occur at 8 am, so our playback may underestimate actual traffic levels during the lekking time. On leks with drilling noise, a 14-minute recording of a drilling rig was played on continuous loop. Natural gas development activities occur 24 hours a day, so noise was broadcast continuously day and night at playback levels that approximate the noise level at 0.25 mile (402 m) from a typical drilling site (JLB and GLP unpublished data). Drilling-noise recordings were broadcast on experimental leks at an equivalent sound level (L_{eq}) of 71.4±1.7 dBF (unweighted decibels) SPL re 20 μPa (56.1±0.5 dBA [A-weighted decibels]) as measured at 16 meters; on road-noise leks, where the amplitude of the noise varied with the simulated passing of vehicles, noise was broadcast at an L_{max} (maximum RMS amplitude) of 67.6±2.0 dBF SPL (51.7±0.8 dBA) (see Blickley, et al. [43], for detailed noise-exposure measurements). Noise from playback was localized to each lek due to the small size of our speakers. To control for visual disturbance of the speaker system and researcher presence, control

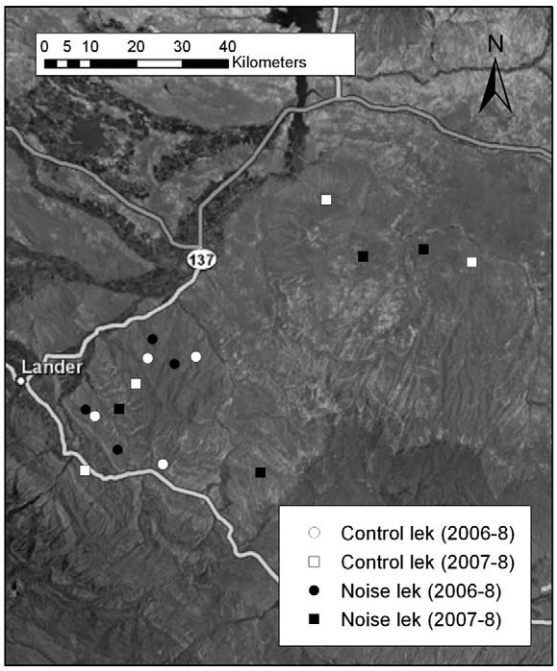

Figure 1. Noise playback study area in Fremont County, Wyoming, USA, 2006–2009. Experimental and control leks were paired on the basis of size and geographic location (the four leks in the upper right are part of the Riverton region, whereas the rest of the leks are in the Lander region).
doi:10.1371/journal.pone.0050462.g001

leks had dummy speakers placed in the same arrangement and were also visited to simulate the periodic battery changes on noise leks. This experimental protocol was reviewed and approved by the Animal Care and Use Committee at UC Davis (Protocol # 16435) and the Wyoming Game and Fish Department (Permit # 33–405).

In the first year of the experiment (2006), we played noise on only 4 of the 8 lek pairs (2 experimental leks with introduced drilling noise, 2 with introduced road noise). Therefore, some leks had been exposed to noise the breeding season prior to the first year of FCM measurement; however, we detected no significant impact of duration of noise exposure on lek attendance [43], so years of noise exposure was not included as a potential explanatory variable in candidate model sets.

Collection of Fecal Samples

Fecal samples were collected from leks soon after all sage-grouse had left the lek for the morning. Samples were collected twice per year from each lek (once during the mid season [April 4–6 in 2007, April 6–8 in 2008] and once during the late season [April 23–26 in 2007, April 22–24 in 2008]) and were collected from paired leks on the same day. Samples were collected using a sweep-search method in which the entire lek was systematically searched and fresh fecal samples were collected individually in Whirl-Pak bags and labeled with a location on the lek relative to the speakers (or dummy speakers). To minimize the chance of collecting multiple fecal samples from the same individual, we collected samples that were a minimum of 5 meters apart, roughly the minimum territory size of a male sage-grouse. Jankowski [45] found lower FCM levels in female sage-grouse than in breeding male sage-grouse. Therefore to avoid collecting samples from females, we collected samples on dates when female visitation is rare; if there were more than 1–2 females on the lek on a potential collection day, sampling for that lek pair was postponed until the next day. Time to collect samples varied among leks from 20–80 minutes. Samples were frozen at −20°C within a few hours of collection until processing. Jankowski et al. [45] found no difference in FCM levels for greater sage-grouse samples held for variable times up to 16 hours prior to freezing.

Extraction & Radioimmunoassay of Cort

We used extraction and assay procedures, with minor modifications, that were previously validated for application to greater sage-grouse by Jankowski et al. [46]. Individual fecal pellets were kept on ice while uric acid (often present in a discrete cap on the pellet) was removed and discarded. Samples were then lyophilized and returned to storage at −20°C. On the day of extraction, individual fecal pellets were weighed to the nearest 0.0001 g, then manually homogenized, vortexed, and shaken in 5 mL of 80% methanol for at least 30 minutes. Longer incubation in methanol often occurred due to the large number of tubes in each assay, but experimentation with overnight extraction produced no substantial change in detected metabolites. Samples were centrifuged at 5000 rpm for 30 minutes, then 1.5mL of supernatant was drawn off, placed in a separate tube, dried under streaming air in a 70°C water bath and reconstituted in 1.0 mL of steroid diluent provided in the RIA kit (see below). For some very large samples, it was not possible to remove 1.5 mL; in these cases, 500 μL of supernatant was drawn off and reconstitution volume was adjusted accordingly after drying. Extracts were covered with Parafilm and stored at 4°C until assayed.

A pooled sample was made by homogenizing a collection of multiple samples from one control lek (Monument lek) in a blender prior to lyophilization. From this pooled sample, 0.5 g was assayed initially to determine parallelism with the RIA standard curve, and one or more pooled samples were included in each extraction and assay.

Radioimmunoassays were conducted according to the manufacturer's instructions (catalog # 07-120103, MP Biomedicals, Costa Mesa, CA) using 1:16 dilution of reconstituted extract. This RIA kit utilizes a rabbit-produced BSA IgG polyclonal antibody against corticosterone-3-carboxymethyloxime. This antibody has been widely used for fecal assays due to its ability to bind a broad spectrum of corticosteroid metabolites [47]. Samples were randomly distributed among assays with respect to year and treatment to minimize any impacts of inter-assay variation.

FCM measures were adjusted for the mass of the fecal sample (ng ICM/g sample) to account for differences among leks in fecal pellet mass. In dividing ICM by sample mass, we effectively assume that the relationship between sample mass and fecal transit time (during which corticosteroid metabolites are secreted into the lumen of the gut) is positive and linear. To guard against faults in this assumption, we ran the same statistical analyses using "per sample" FCM data and found no difference in the main effects as reported.

Statistical Analysis

Fecal glucocorticoid metabolites levels were natural log-transformed to meet assumptions of normality and homoscedasticity prior to analysis. We used an information theoretic approach to evaluate the support for alternative candidate models using Akaike's Information Criterion for small sample sizes (AIC_c) [48]. Candidate models for the overall effect of noise (Noise effect models) were linear mixed-effect models that assessed the relationship between explanatory variables and the concentration of FCMs collected from experimental and control leks. Potential

explanatory variables included pair type (NoiseType, drilling or road noise), control status (Treatment, noise or control), pellet/collection distance from speakers (SpeakerDist), maximum lek size for that year (MaxSize), location (Hudson or Riverton), season (early or late April), and relevant interactions (see Table 1 for full set of candidate models). All models contained lek pair ID, and year (2007 or 2008) as random effects.

We also evaluated a set of candidate models that assessed the relationship between the concentration of FCMs on experimental leks and the declines in peak male attendance from the previous year (attendance models). Models contained lek ID and year (2007 or 2008) as random effects. Models were ranked on the basis of differences in AICc scores (ΔAICc) and were assigned Akaike weights (w_i) corresponding to the degree of support. We calculated model-averaged coefficients and variable importance (sum of variable weights for all models in which the variable was included) for variables contained in all models that received strong support (ΔAICc <2). We also compared the variance in FCM concentrations measured on noise and control leks using a Levene's test. All statistical analyses were performed in R (version 2.12.1, R Development Team 2010).

Results

We measured baseline fecal immunoreactive corticosterone metabolites of 103.2 and 119.9 ng/g for control and treatment groups, respectively (Table 2). These values are lower than baseline measures of approximately 149 ng/g obtained previously for breeding male greater sage-grouse in Nevada, from which fecal samples were collected after capture [45].

Males on leks exposed to noise had higher (16.7% on average) FCM levels compared with controls ($w_i = 0.96$, Table 1, 2; Figure 2). While models that included the effect of Treatment (noise versus control) were highly supported by the data, there was little support for an interaction of Treatment with NoiseType variable ($w_i = 0.01$, Table 1), indicating that while noise exposure was associated with increased cort, there was little difference in FCM levels between leks with drilling versus road-noise playback. Candidate models containing other possible explanatory variables, including distance from the nearest speaker (SpeakerDist), maximum size of the lek (MaxSize), the regional location of the lek in the Hudson area or Riverton area (Location) and time of the season (Season), received little support relative to the null model (Table 1, Figure 2B), indicating that none of these factors had a strong influence on FCM levels.

To determine whether noise-playback leks with a higher stress response were associated with larger declines in lek attendance, we compared candidate models for the relationship between FCM level and change in lek attendance from the previous year. Only the null model received support (Table 3), indicating that fecal FCM level was not associated with the magnitude of changes in lek attendance on noise leks.

Finally, we examined whether there was a difference in variance among samples on noise leks and control leks. We found no significant differences in variance between treatment types in 2007 (variance on noise leks = 7729.94, control leks = 6168.28, Levene's

Table 1. Mixed-effect candidate models for the effect of noise playback on mass-dependent FCM concentrations (natural log-transformed).

Model[a,b]	K[c]	ΔAIC_c [d]	w_i [e]
Treatment[f]	5	0	0.66
Treatment + Location	6	2.4	0.20
Treatment + Location + Treatment:Location	7	4.7	0.06
Null- random effects only	4	5.5	0.04
Treatment + Season	6	6.5	0.03
Treatment + Season + Treatment:Season	7	10.0	<0.01
Treatment + NoiseType + Treatment:NoiseType	7	10.8	<0.01
Treatment + Location + NoiseType + Treatment:Location + Treatment:NoiseType	9	11.2	<0.01
Treatment + NoiseType + Season + Treatment:Season + Treatment:NoiseType	9	20.7	<0.01
Treatment + MaxSize + Treatment:MaxSize	7	25.3	<0.01
Treatment + NoiseType + Season + Treatment:NoiseType + Treatment:Season + Treatment:NoiseType:Season	11	27.3	<0.01
Treatment + SpeakerDistance + Treatment:SpeakerDistance	7	27.5	<0.01
Treatment + NoiseType + MaxSize + Treatment:NoiseType + Treatment:MaxSize	10	35.4	<0.01
Treatment + NoiseType + SpeakerDistance + Treatment:NoiseType + Treatment:SpeakerDistance	9	38.2	<0.01
Treatment + NoiseType + MaxSize + Treatment:NoiseType + Treatment:MaxSize + Treatment:NoiseType:MaxSize	12	45.1	<0.01
Treatment + NoiseType + SpeakerDistance + Treatment:NoiseType + Treatment:SpeakerDistance + Treatment:NoiseType:SpeakerDistance	11	60.4	<0.01

[a]Abbreviations of predictor variables in methods.
[b]All models contain lek pairing and year as a random effect.
[c]Number of parameters in the model.
[d]Difference in AIC_c (Akaike's Information criteria for small sample size) values from the top ranking model.
[e]Akaike weight (Probability that the model is the best fit model giving the data and model candidate set).
[f]Model with substantial support (ΔAIC_c <2).
doi:10.1371/journal.pone.0050462.t001

Table 2. Parameter estimates (± SE) and relative variable importance for variables in highly supported models ($\Delta AIC_c < 3$).

Variable	Parameter estimates[a]	Parameter estimates (back-transformed)[b]	Relative variable importance[c]
Intercept	4.63 (.06)	103.2[d]	-
Treatment:Noise	.15 (.04)	16.7[d]	0.96
Location: Hudson	0.02(.01)	2.9[d]	0.26

[a]Parameter estimates are natural-log transformed.
[b]SE not included due to back-transformation.
[c]Relative variable importance is the summed total of the model weights for models containing that variable.
[d]Intercept value was added to parameter estimates prior to back-transformation and then subtracted.
doi:10.1371/journal.pone.0050462.t002

W = 0.6327, p = 0.427). Variance on noise leks was significantly higher than on control leks in 2008 (variance on noise leks = 4462.28, control leks = 2758.69, Levene's W = 6.6064, p = 0.01).

Discussion

We found higher (16.7%) FCM levels on noise-treated leks compared to controls, supporting the hypothesis that chronic noise pollution increases stress levels in male greater sage-grouse. Combined with results from monitoring of lek attendance in the same experiment [43], these results suggest that noise from natural gas development activities can dramatically decrease male attendance on leks and cause physiological impacts on males that remain on noisy leks. The mean level of FCMs in remaining birds was not a good predictor of the degree of decline in peak male attendance on a lek compared with the previous year, indicating that the FCM level measured on a lek is not diagnostic of an effect of noise on peak male attendance (Table 3). Further, we did not find support for an effect of distance from the speakers on FCM levels. Male sage-grouse typically maintain a fixed territory on a lek throughout the season. Within a noise-treated lek, each individual's exposure to noise varied, depending on the location of their territory relative to the speakers. Since noise levels decline exponentially with distance from the speakers, the lack of a distance effect suggests that stress is not exclusively dependent on the noise exposure of individuals. Instead, noise impacted FCM levels on a lek-wide basis.

Blickley et al. [43] found a decline in lek attendance on road-noise leks more than twofold larger than the decline in lek attendance on drilling-noise leks, yet we found no difference in FCM levels between noise-playback types (Table 1, Figure 1). Both noise sources have most of their sound energy ≤2 kHz, but road noise is less predictable than drilling noise and more intermittent,

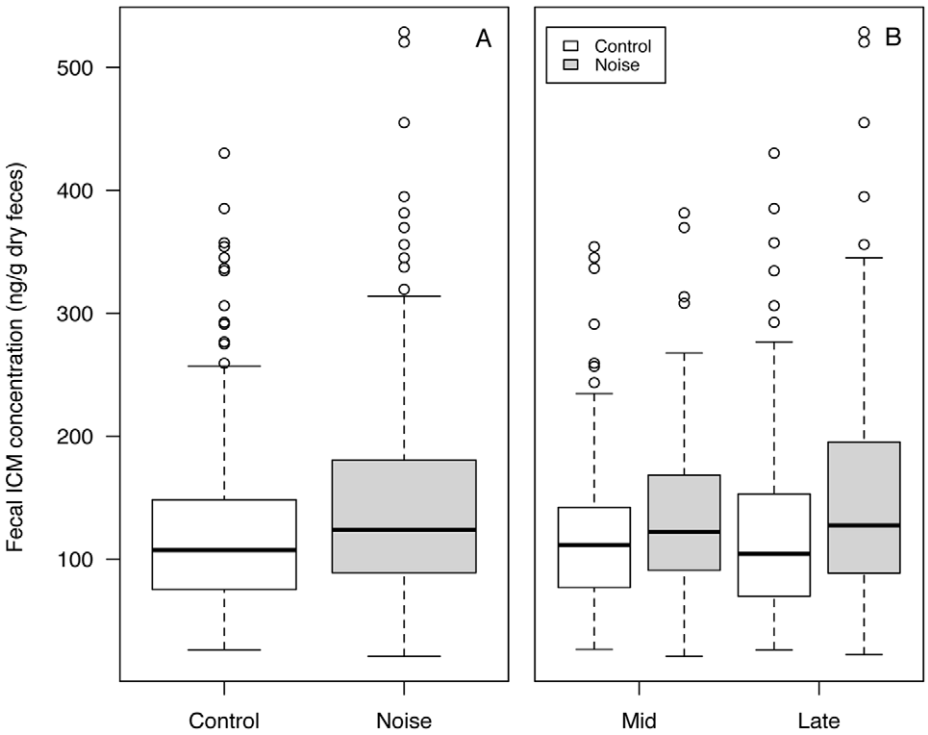

Figure 2. FCM concentrations from control and noise-treated groups. Data shown (A) pooled by season and (B) for mid and late season samples. Horizontal line represents the median value, box ends represent upper and lower quartiles, whiskers represent maximum and minimum values and open circles represent outliers. Plots present measured FCM values, not model output, which is presented in Table 2.
doi:10.1371/journal.pone.0050462.g002

Table 3. Mixed-effect candidate models assessing the relationship of FCM concentrations and changes in lek attendance from the previous year on noise-playback leks.

Model[a,b]	K[c]	ΔAIC_c [d]	w_i [e]
Null- random effects only[f]	5	0	0.90
Fecal cort	6	4.6	0.10

[a]Abbreviations of predictor variables in methods.
[b]All models contain lek pairing and year as a random effect.
[c]Number of parameters in the model.
[d]Difference in AIC_c (Akaike's Information criteria for small sample size) values from the top ranking model.
[e]Akaike weight.
[f]Model with substantial support (ΔAIC_c <3).
doi:10.1371/journal.pone.0050462.t003

leading to a lower average noise exposure across road-noise leks (43.2±0.89 dBA L_{eq}) than drilling-noise leks (56.1±0.45 dBA L_{eq}) [43]. Studies on physiological stress in rodents indicate that stressors administered at unpredictable intervals result in greater elevations in plasma corticosterone [49]. Since cort levels may also be implicated in decisions to escape from deleterious conditions [50], we cannot say with certainty that noise type has no differential impact on FCM levels, only that there was no difference observed among males that chose to remain. If road noise did result in a greater cort response in some birds, but the most susceptible birds were also the most likely to disperse, differences would not necessarily be expected among remaining birds. In this scenario, it is likely that variance would be reduced in leks with high losses, reflecting disappearance of individuals with higher FCM levels. Levene's tests did not identify any such difference in variance (indeed, there was a significant difference in one year of the study, but in the opposite direction to predictions). However, the possibility that dispersal is linked to FCM levels cannot be ruled out. Regardless of whether the stress levels of birds on noise leks increased, or whether only high-stress-level individuals remained on noisy leks, these results indicate that chronic noise at leks creates less desirable habitat for greater sage-grouse.

The unknown status of dispersed grouse – and their unknown destinations – leaves several other possible scenarios that should be considered. It is possible that the individuals most likely to disperse could have had different cort profiles at the outset compared with those more prone to remain. If noise playback caused individuals with lower integrated cort to disperse away from noisy leks, that coupled with the possible addition of those birds to control leks could cause trends similar to those observed here. Two possible sources of variation in pre-experiment cort levels among individuals are age and social status [51–53]. Reduced juvenile recruitment may have contributed to the observed declines in lek attendance on noise leks, potentially leading to a difference in age structure on noise and control leks [43]; however, this is unlikely to explain the results of this study. Studies of altricial and semi-altricial birds have found lower stress responsiveness shortly after hatching, but responses resemble those of adults by the age of fledging or first molt [54–57]. Since young male sage-grouse attending leks are likely to be at least 10 months old and after their first molt, it is unlikely that they would have lower stress response than adults. Social status can also be related to corticosteroid levels [58], therefore social upheaval caused by dispersal between noise and control leks may have contributed to observed FCM levels. Further studies are needed determine whether age-class- and social-status-dependent dispersal in response to noise contributed to the observed results.

Unlike noise sources in most energy development sites, our noise introduction in this study was localized to the immediate lek area, so birds were exposed to noise for only a few hours a day, and only during the breeding season. Therefore, we cannot quantify the effects of noise on FCMs for wintering, nesting or foraging males. Noise at energy development sites is less seasonal and more widespread and may thus affect birds at all life stages, with a potentially greater impact on stress levels. In addition, we looked only at male stress levels in this study, but males and females may respond differently to stress. For example, Jankowski et al. [45] measured FCM levels in sage-grouse in habitats with and without cattle grazing; they found no difference in male FCM levels in response to grazing regime, however, breeding females showed elevated stress response in grazed areas. This suggests that females may be more vulnerable to some types of disturbance; further studies are needed to assess whether female stress levels are influenced by noise.

Why might noise be stressful?

Increased adrenocortical activity occurs in response to circumstances perceived as threatening by an animal. Although we cannot determine from this study the extent to which noise itself is a threat to sage-grouse, noise may affect social dynamics and increase the perception of threat. Noise may have social impacts on sage-grouse by masking acoustic communication on the lekking grounds [42]. Masking occurs when the perception of a sound is decreased by the presence of background noise, which may reduce the efficacy of acoustic communication. Acoustic signals play an important role in many social interactions, including mate attraction and assessment, territorial interactions, recognition of conspecifics and alarm calling in response to environmental threats [9,10,59]. Masking of these acoustic signals may alter or interfere with social interactions and mate choice behaviors [60,61].

For prey species such as sage-grouse, noise may also increase stress levels by masking the sounds of approaching predators and increasing the perception of risk from predation [62,63]. The degree to which noise directly affects mortality through changes in predation is largely unknown, as few studies have compared predation rates or hunting success in noisy and quiet areas while controlling for other confounding factors. Francis et al. [4] did so and found that nest predation rates in some songbirds decline in noise-impacted areas, as the dominant nest predator avoided noise. This suggests that noise may cause complicated changes in predator-prey dynamics. Noise may also cause stress due to short-term disruptions in behavior, such as startling or frightening animals away from food or other resources [2,64]. Further, if individuals associate a particular type of noise, such as road noise, with a danger, such as vehicular traffic, this may provoke a stress response [43].

The impacts of chronic stress

Glucocorticoid release under challenging conditions is an adaptation to life in an unpredictable and threatening world [20]; individuals benefit from curtailing reproduction, altering behavioral patterns, and redirecting metabolic substrates to maximize glucose availability for action in response to genuine threats. Glucocorticoid levels alone are not directly or inversely correlated with fitness measures under all conditions [65], however, chronic adrenal activation has many known trade-offs that result in vulnerability to disease and death [22]. Unlike threats from predators, food shortages and inclement weather, noise typically does not directly threaten the survival of an individual or

its offspring (though there may be exceptions, as discussed below). Therefore, the cost of chronic adrenal activation in response to noise pollution is unlikely to be outweighed by the benefits in most cases, and thus the net result may be adverse.

One important trade-off is the effect of corticosterone on immune response. Chickens infected with West Nile Virus (WNV) and administered corticosterone had increased oral shedding and lengthened duration of viremia compared to those without elevated cort [66]. For sage-grouse, which are highly susceptible to WNV [67,68], reduced immune response due to elevated glucocorticoid levels could have a significant effect on survival in areas where they are exposed to WNV. Therefore, despite the adaptive nature of the stress response under natural conditions, elevated glucocorticoid levels due to human disturbance may have detrimental long-term impacts on welfare and survival of sage-grouse and other wildlife.

Stress as an indicator of human impacts on sage-grouse

Measurement of FCMs may provide a non-invasive monitoring tool to assess the impact of human development (e.g. oil and gas drilling, wind farms, highways, off-road vehicle traffic) on stress levels of greater sage-grouse and other species. However comparisons between disturbed and undisturbed areas would need to account for differences in age, sex, and breeding condition of individuals sampled as well as for differences in the environmental conditions between sites in order to isolate stress as the likely cause of change [15,18,69]. We controlled for such differences by using an experimental presentation of noise that minimized effect on other habitat variables, limiting our collection to lekking birds, collecting only on days with limited female attendance and collecting samples from all leks within a short 2–3 day window. We did not find support for differences in FCM levels from samples collected in early versus late April within each season (~20 days apart in a 2–3 month breeding season), and only limited evidence for an effect of location (Hudson vs. Riverton, ~32 kilometers apart), suggesting that these temporal and spatial differences did not affect FCM levels in our study. However with a larger sample of leks or in another region or time period, it is possible that such differences might emerge.

Conclusions

Taken together, results from Blickley et al. [43] and this study suggest that noise alone can cause greater sage-grouse to avoid otherwise suitable habitat and increase the stress responses of birds that remain in noisy areas. Thus, noise mitigation may be a fruitful conservation measure for this species of concern. In this study, we focused on the effects of noise from roads and drilling rigs in natural gas development areas; other natural gas development infrastructure, including compressor stations and generators, produces noise similar to drilling rigs, with the potential for similar effects on FCM levels. Likewise, other types of energy development produce noise similar in frequency, timing, and amplitude to the noise sources used here, including shale gas, coal-bed methane, oil, and geothermal development. The noise sources used in this study also share some characteristics with other anthropogenic noise sources that are increasing across the landscape, like wind turbines, off-road vehicles, highways and urban development; this suggests that the impacts on greater sage-grouse observed here may be widespread. More generally, populations of many species of birds [4,70–74] and mammals [75–78] decline with proximity to noisy human activities, such as roads, urban and industrial developments. While further study is needed to determine whether chronic noise exposure contributes to the impacts of these human activities by activating the chronic stress response, this study adds to a growing body of evidence that such noise pollution is a threat to wildlife [1,2], significantly increasing our estimates of the footprint of human development beyond the boundaries of visible disturbance.

Acknowledgments

For help and advice during the conception and execution of this project we thank Kurt Fristrup, Tom Rinkes, Sue Oberlie, Stan Harter, Tom Christiansen, Diane Blackwood and our many excellent field assistants. For statistical advice and helpful comments on the manuscript, we thank Teresa L. Iglesias, the Patricelli Lab at UC Davis and two anonymous reviewers.

Author Contributions

Conceived and designed the experiments: JLB GLP AHK SNS. Performed the experiments: JLB AHK GLP SNS KRW JLP JCW. Analyzed the data: JLB KRW GLP AHK JLP CCT JCW. Contributed reagents/materials/analysis tools: GLP JCW. Wrote the paper: JLB KRW GLP.

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Conservation Biology

Contributed Paper

Experimental Evidence for the Effects of Chronic Anthropogenic Noise on Abundance of Greater Sage-Grouse at Leks

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Abstract: *Increasing evidence suggests that chronic noise from human activities negatively affects wild animals, but most studies have failed to separate the effects of chronic noise from confounding factors, such as habitat fragmentation. We played back recorded continuous and intermittent anthropogenic sounds associated with natural gas drilling and roads at leks of Greater Sage-Grouse (*Centrocercus urophasianus*). For 3 breeding seasons, we monitored sage grouse abundance at leks with and without noise. Peak male attendance (i.e., abundance) at leks experimentally treated with noise from natural gas drilling and roads decreased 29% and 73%, respectively, relative to paired controls. Decreases in abundance at leks treated with noise occurred in the first year of the study and continued throughout the experiment. Noise playback did not have a cumulative effect over time on peak male attendance. There was limited evidence for an effect of noise playback on peak female attendance at leks or male attendance the year after the experiment ended. Our results suggest that sage-grouse avoid leks with anthropogenic noise and that intermittent noise has a greater effect on attendance than continuous noise. Our results highlight the threat of anthropogenic noise to population viability for this and other sensitive species.*

Keywords: chronic noise, energy development, *Centrocercus urophasianus*, roads

Evidencia Experimental de los Efectos de Ruido Antropogénico Crónico sobre la Abundancia de *Centrocercus urophasianus* en Leks

Resumen: *El incremento de evidencias sugiere que el ruido crónico de actividades humanas afecta negativamente a los animales silvestres, pero la mayoría de los estudios no separan los efectos del ruido crónico de los factores de confusión, como la fragmentación del hábitat. Reprodujimos sonidos antropogénicos intermitentes y continuos asociados con la perforación de pozos de gas natural y caminos en leks de* Centrocercus urophasianus. *Durante 3 épocas reproductivas, monitoreamos la abundancia de* C. urophasianus *e leks con y sin ruido. La abundancia máxima de machos (i.e., abundancia) en leks tratados con ruido de la perforación de pozos de gas natural y caminos decreció 29% y 73% respectivamente en relación con los controles pareados. La disminución en abundancia en leks tratados con ruido ocurrió en el primer año del estudio y continuó a lo largo del experimento. La reproducción de ruido no tuvo efecto acumulativo en el tiempo sobre la abundancia máxima de machos. Hubo evidencia limitada para un efecto de la reproducción de ruido sobre la abundancia máxima de hembras en los leks o sobre la asistencia de machos el año después de que concluyó el experimento. Nuestros resultados sugieren que* C. urophasianus *evita leks con ruido anropogénico y que el ruido intermitente tiene un mayor efecto sobre la asistencia que el ruido continuo. Nuestros*

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Paper submitted October 19, 2010; revised manuscript accepted November 20, 2011.

Conservation Biology, Volume 26, No. 3, 461–471
©2012 Society for Conservation Biology
DOI: 10.1111/j.1523-1739.2012.01840.x

resultados resaltan amenaza del ruido antropogénico para la viabilidad poblacional de esta y otras especies sensibles.

Palabras Clave: *Centrocercus urophasianus*, desarrollo energético, ruido crónico, caminos

Introduction

Noise associated with human activity is widespread and expanding rapidly in aquatic and terrestrial environments, even across areas that are otherwise relatively unaffected by humans, but there is still much to learn about its effects on animals (Barber et al. 2009). Effects of noise on behavior of some marine organisms are well-documented (Richardson 1995). In terrestrial systems, the effects of noise have been studied less, but include behavioral change, physiological stress, and the masking of communication signals and predator sounds (Slabbekoorn & Ripmeester 2008; Barber et al. 2009). These effects of noise on individual animals may lead to population decreases if survival and reproduction of individuals in noisy habitats are lower than survival and reproduction of individuals in similar but quiet habitats (Patricelli & Blickley 2006; Warren et al. 2006; Slabbekoorn & Ripmeester 2008). Population declines may also result if animals avoid noisy areas, which may cause a decrease in the area available for foraging and reproduction.

There is evidence of variation among species in their sensitivity to noise. Noise sensitivity may also differ with the type of noise, which varies in amplitude, frequency, temporal pattern, and duration (Barber et al. 2009). Duration may be particularly critical; most anthropogenic noise is chronic and the effects of chronic noise may differ substantially from those of short-term noise in both severity and response type. For example, brief noise exposure may cause elevated heart rate and a startle response, whereas chronic noise may induce physiological stress and alter social interactions. Therefore, when assessing habitat quality for a given species, it is critical to understand the potential effects of the full spectrum of anthropogenic noise present in the species' range.

The effects of noise on wild animals are difficult to study because noise is typically accompanied by other environmental changes. Infrastructure that produces noise may be associated with fragmentation of land cover, visual disturbance, discharge of chemicals, or increased human activity. Each of these factors may affect the physiology, behavior, and spatial distribution of animals, which increases the difficulty of isolating the effects of the noise.

Controlled studies of noise effects on wild animals in terrestrial systems thus far have focused largely on birds. Recent studies have compared avian species richness, occupancy, and nesting success near natural gas wells operating with and without noise-producing compressors. In these studies, spatial variation in noise was used to control for confounding visual changes due to infrastructure (Habib et al. 2007; Bayne et al. 2008; Francis et al. 2009). Results of these studies show that continuous noise affects density and occupancy of a range of bird species and leads to decreases or increases in abundance of some species and has no effect on other species (Bayne et al. 2008; Francis et al. 2009; Francis et al. 2011). Results of these studies also show that noise affects demographic processes, such as reproduction, by reducing the pairing or nesting success of individuals (Habib et al. 2007; Francis et al. 2009).

Although these studies in areas near natural gas wells controlled for the effects of most types of disturbance besides noise, they could not address the effect of noise on naïve individuals in areas without natural gas wells and compressors. Furthermore, there have been no controlled experiments that address the effects of chronic but intermittent noise, such as traffic, which may be more difficult for species to habituate. Road noise may have large negative effects because it is widespread (affecting an estimated 20% of the United States) (Forman 2000) and observational studies indicate that noise may contribute to decreases in abundance of many species near roads (e.g., Forman & Deblinger 2000).

Noise playback experiments offer a way to isolate noise effects on populations from effects of other disturbances and to compare directly the effects of noise from different sources. Playback experiments have been used to study short-term behavioral responses to noise, such as effects of noise on calling rate of amphibians (Sun & Narins 2005; Lengagne 2008), heart rate of ungulates (Weisenberger et al. 1996), diving and foraging behavior of cetaceans (Tyack et al. 2011), and song structure of birds (Leonard & Horn 2008), but have not been used to study effects of chronic noise on wild animals because producing long-term noise over extensive areas is challenging. We conducted a playback experiment intended to isolate and quantify the effects of chronic noise on wild animals. We focused on the effects of noise from natural gas drilling on Greater Sage-Grouse (*Centrocercus urophasianus*).

Greater Sage-Grouse occur in the western United States and Canada and have long been a focus of sexual selection studies (Wiley 1973; Gibson 1989; Gibson 1996). Greater Sage-Grouse populations are decreasing in density and number across the species' range, largely due to extensive habitat loss (Connelly et al. 2004; Garton et al. 2010). The species is listed as endangered under Canada's

Species at Risk Act and is a candidate species for listing under the U.S. Endangered Species Act. Deep natural gas and coal-bed methane development have been expanded rapidly across the species' range since 2000 and substantial evidence suggests that these processes may contribute to observed decreases in the number of Greater Sage-Grouse (Holloran 2005; Walker et al. 2007; Holloran et al. 2010). Many factors associated with deep natural gas and coal-bed methane development are thought to lead to these decreases, including habitat loss, increased occurrence of West Nile Virus, and altered fire regimes due to the expansion of nonnative invasive species (Naugle et al. 2004; Walker et al. 2007; Copeland et al. 2009).

The noise created by energy development may also affect sage grouse by disrupting behavior, causing physiological stress, or masking biologically important sounds. During the breeding season (February–May), male sage grouse gather on communal breeding grounds called leks. Male attendance (number of male birds on the lek) at sage grouse leks downwind of deep natural gas development decreases up to 50% per year compared with attendance at other leks, which suggests noise or aerial spread of chemical pollution as factors contributing to these decreases (Holloran 2005).

We sought to test the hypothesis that lek attendance by male and female sage grouse is negatively affected by both chronic intermittent and continuous noise from energy development. To do so, we conducted a noise playback experiment in a population that is relatively unaffected by human activity. Over 3 breeding seasons (late February to early May), we played noise recorded from natural gas drilling rigs and traffic on gas-field access roads at sage grouse leks and compared attendance patterns on these leks to those on nearby control leks.

We conducted our experiment at leks because lekking sage grouse are highly concentrated in a predictable area, which makes them good subjects for a playback experiment. More importantly, sage grouse may be particularly responsive to noise during the breeding season, when energetic demands and predation risk are high (Vehrencamp et al. 1989; Boyko et al. 2004). Additionally, noise may mask sexual communication on the lek. Lekking males produce a complex visual and acoustic display (Supporting Information) and females use the acoustic component of the display to find lekking males and select a mate (Gibson 1989; Gibson 1996; Patricelli & Krakauer 2010). Furthermore, lek attendance is commonly used as a metric of relative abundance of sage grouse at the local and population level (Connelly et al. 2003; Holloran 2005; Walker et al. 2007). We used counts of lek attendance (lek counts) to assess local abundance relative to noise versus control treatments.

Methods

Study Site and Lek Monitoring

Our study area included 16 leks (Table 1 & Supporting Information) on public land in Fremont County, Wyoming, U.S.A. (42° 50′, 108° 29′). Dominant vegetation in this region is big sagebrush (*Artemisia tridentata wyomingensis*) with a grass and forb understory. The primary land use is cattle ranching, and there are low levels of recreation and natural gas development.

We paired leks on the basis of similarity in previous male attendance and geographic location (Table 2 & Supporting Information). Within a pair, one lek was

Table 1. Pairing, treatment type, location, and baseline attendance for leks used in noise playback experiment.

Lek	*Pair*	*Pair noise type*	*Noise or control*	*Years of playback*	*Baseline attendance**
Gustin	A	drilling	control	3	26
Preacher Reservoir	A	drilling	noise	3	49
North Sand Gulch	B	road	control	3	32
Lander Valley	B	road	noise	3	67
East Twin Creek	C	drilling	control	3	44
Coal Mine Gulch	C	drilling	noise	3	83
East Carr Springs	D	road	control	3	67
Carr Springs	D	road	noise	3	92
Powerline	E	drilling	control	2	49
Conant Creek North	E	drilling	noise	2	44
Monument	F	road	control	2	53
Government Slide Draw	F	road	noise	2	55
Nebo	G	drilling	control	2	18
Arrowhead West	G	drilling	noise	2	24
Onion Flats 1	H	road	control	2	41
Ballenger Draw	H	road	noise	2	38

**Baseline attendance is the average peak male attendance value (annual maximum number of males observed averaged across years) for that lek from 2002 to 2005.*

Table 2. Mixed-effect candidate models used to assess change in peak attendance of male Greater Sage-Grouse at leks from pre-experiment baseline attendance during the natural gas drilling noise playback (2006–2008) and after the experiment (2009).

Model (year)[a]	K[b]	ΔAIC_c[c]	w_i[d]
Male experiment (2006–2008)			
treatment×type+season[e]	9	0	0.64
treatment×type[e]	7	1.8	0.26
treatment+experiment year	6	6.1	0.03
treatment+season	7	6.8	0.02
treatment	5	7.3	0.02
treatment×experiment year	7	8.0	0.01
treatment×type+treatment×season+experiment year	12	8.6	<0.01
treatment×type+treatment×season	11	9.9	<0.01
treatment×type+treatment×season+treatment×experiment year	13	10.0	<0.01
treatment+type	6	10.4	<0.01
treatment×season	9	16.2	<0.01
null- random effects only	4	57.0	<0.01
Male after experiment (2009)			
null, random effects only[e]	3	0.0	0.84
treatment	4	3.3	0.16

[a]*All models contain pair as a random effect, and experiment (2006–2008) models also include year as a random effect. Covariates: treatment, lek treatment (noise or control) assigned to individual leks within a pair; type, pair noise treatment type (road or drilling assigned to pair); season, time of year (early [late February to 1 week prior to peak female attendance for that lek; female peak ranged from 15 March to 6 April], mid [1 week before and after female peak], and late [starting 1 week after female peak]); experiment year, years of experimental noise exposure.*
[b]*Number of parameters in the model.*
[c]*Difference in AIC_c (Akaike's information criterion for small sample size) values from the model with lowest AIC_c.*
[d]*Akaike weight.*
[e]*Model with substantial support ($\Delta AIC_c < 2$).*

randomly assigned to receive experimental noise treatment and the other lek was designated a control. We randomly assigned the experimental leks to receive playback of either drilling or road noise. In 2006, we counted attendance at 8 leks (2 treated with drilling noise, 2 treated with road noise, and 4 control). In both 2007 and 2008, we included an additional 8 leks for a total of 16 leks (4 treated with drilling noise, 4 treated with road noise, and 8 controls).

Throughout the breeding season, we counted males and females on leks with a spotting scope from a nearby point selected to maximize our visibility of the lek. We visited paired leks sequentially on the same days between 05:00 and 09:00, alternating the order in which each member of the pair was visited. We visited lek pairs every day during the breeding season in 2006 and, after expanding our sample size in 2007, every 2–4 days in 2007 and 2008. Peak estimates of male attendance from >4 visits are a highly repeatable measure of abundance at individual leks (Garton et al. 2010), so the lower frequency of visits in 2007 and 2008 was unlikely to have a substantial effect on estimates of peak male attendance. At a minimum, we conducted 2 counts per visit at 10- to 15-min intervals. The annual peak attendance was the highest daily attendance value at each lek for the season for males or females. For males we also calculated the peak attendance in 3 nonoverlapping date ranges: early (late February to 1 week prior to peak female attendance for that lek; female peak ranged from 15 March to 6 April), mid (1 week before and after female peak), and late (starting 1 week after female peak).

Noise Introduction

We recorded noise used for playback near natural gas drilling sites and gas-field access roads in a region of extensive deep natural gas development in Sublette County, Wyoming (Pinedale Anticline Gas Field and Jonah Gas Field). We recorded drilling noise in 2006 within 50 m of the source on a digital recorder (model PMD670, 44.1 kHz/16 bit; Marantz, Mahwah, New Jersey) with a shotgun microphone (model K6 with an ME60 capsule; Sennheiser, Old Lyme, Connecticut). We recorded road noise in 2005 with a handheld computer (iPAQ h5550 Pocket PC, 44.1 KHz/16 bit; Hewlett Packard, Palo Alto, California) and omnidirectional microphone (model K6 with an ME62 capsule; Sennheiser). Drilling noise is relatively continuous and road noise is intermittent (Supporting Information). Both types of noise are predominantly low frequency (<2 kHz).

We played noise on experimental leks from 2 to 4 rock-shaped outdoor speakers (300 W Outdoor Rock Speakers; TIC Corporation, City of Industry, California) hooked to a car amplifier (Xtant1.1; Xtant Technologies, Phoenix, Arizona) and an MP3 player (Sansa m240; SanDisk, Milpitas, California). The playback system was powered with 12 V batteries that we changed every 1–3 days when no birds were present. We placed the speakers

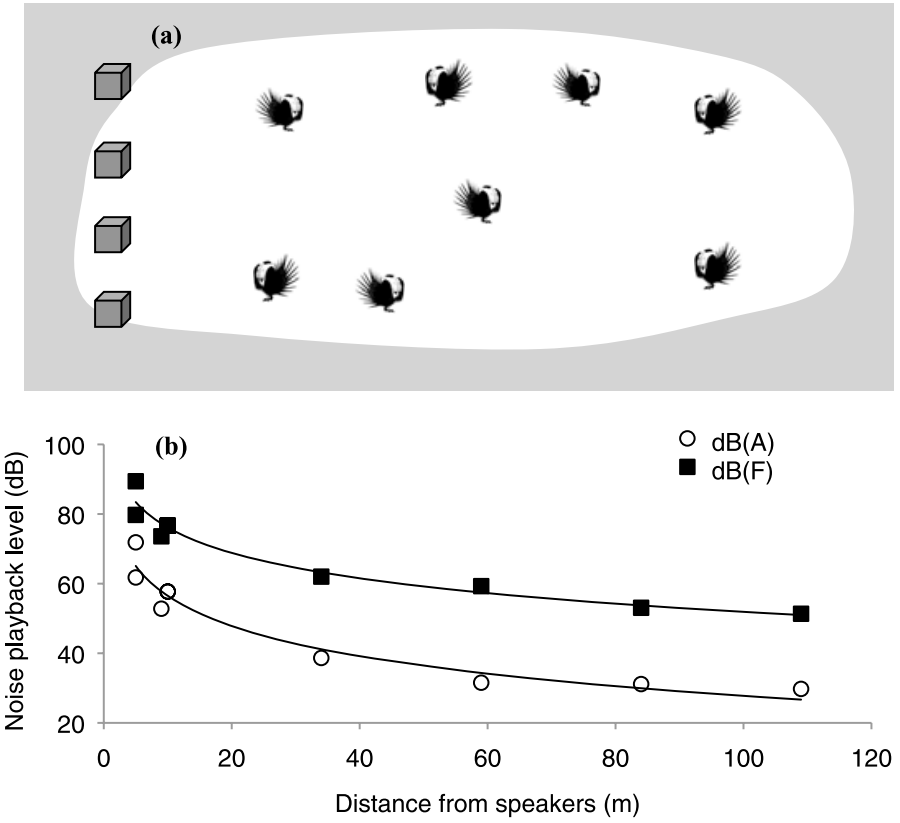

Figure 1. (a) Placement of speakers (on noise-treated leks) or dummy speakers (on control leks) (boxes) at Greater Sage-Grouse leks. (b) Mean maximum noise level (unweighted decibels, dB[F], and A-weighted decibels, dB[A], measured in Lmax [highest root-mean-square sound pressure level within the measurement period]) at Greater Sage-Grouse leks measured on transects at 25-m intervals from the line of speakers on a typical lek treated with road noise. Playback levels of natural gas drilling noise (measured in Leq) followed the same pattern. Ambient levels of noise at control leks ranged from 30 to 35 dB(A).

in a straight line across one end of the lek (Fig. 1a). In 2006 we placed 3 speakers at leks treated with drilling noise and 2 speakers at leks treated with road noise. In 2007 and 2008, we increased the number of speakers, placing 4 at each noise-treated lek to increase the area in which noise was present on the lek. At control leks, we placed dummy speakers of similar size and color to playback speakers (68-L plastic tubs). Within each lek pair, dummy and real speakers were placed in similar configurations. To control for playback-related disturbance, the leks in each pair were visited an equal number of times during the morning for counts of birds and in the afternoon for battery changes.

We played drilling noise and road noise on leks at 70 dB(F) sound pressure level (unweighted decibels) measured 16 m directly in front of the speakers (Fig. 1 & Supporting Information). This is similar to noise levels measured approximately 400 m from drilling rigs and main access roads in Pinedale (J. L. Blickley and G. L. Patricelli, unpublished data). Four hundred meters (0.25 miles) is the minimum surface disturbance buffer around leks at this location (BLM 2008). We calibrated and measured noise playback levels with a hand-held meter that provides sound-pressure levels (System 824; Larson-Davis, Depew, New York) when wind was <9.65 k/h. On drilling-noise-treated leks, where noise was continuous, we calibrated the noise playback level by measuring the average sound level (Leq [equivalent continuous sound level]) over 30 s. On leks treated with road noise, where the amplitude of the noise varied during playback to simulate the passing of vehicles, we calibrated the playback level by measuring the maximum sound level (Lmax [highest root-mean-square sound pressure level within the measurement period]).

For leks treated with drilling noise, recordings from 3 drilling sites were spliced into a 13-min mp3 file that played on continuous repeat. On leks treated with road noise, we randomly interspersed mp3 recordings of 56 semitrailers and 61 light trucks with 170 thirty-second silent files to simulate average levels of traffic on an access road (Holloran 2005). Noise playback on experimental leks continued throughout April in 2006, from mid February or early March through late April in 2007, and from late February through late April in 2008. We played back noise on leks 24 hours/day because noise from deep natural gas drilling and vehicular traffic is present at all times. This experimental protocol was reviewed and approved by the Animal Care and Use Committee at University of California, Davis (protocol 16435).

To measure noise levels across experimental leks, we measured the average amplitude (15 s Leq) of white-noise played at 1–5 points along transects that extended across the lek at 25-m intervals roughly parallel to the line of speakers. We calibrated white-noise measurements by measuring the noise level of both the white noise and either a representative clip of drilling noise or a semitrailer

10 m directly in front of each speaker. To minimize disturbance, we took propagation measurements during the day. Daytime ambient noise levels are typically 5–10 dBA higher than those in the early morning (J. L. Blickley and G. L. Patricelli, unpublished data) and are likely higher than those heard by birds at a lek.

After the experiment, we counted individuals on all leks 2–6 times from 1 March through 30 April 2009. In 2009 we continued to play noise on 2 experimental leks as part of a related experiment, so we did not include these lek pairs in our analysis of postexperiment male attendance at a lek.

Response Variables and Baseline Attendance Levels

Sage grouse leks are highly variable in size and, even within pairs, our leks varied up to 50% in size. To facilitate comparison of changes in attendance on leks of different sizes, we calculated the attendance relative to attendance levels before treatment (i.e., baseline attendance levels). We obtained male baseline abundance from the Wyoming Game and Fish Department. We used the standard lek-count protocol (Connelly et al. 2003) to count birds at leks approximately 3 times/breeding season. Due to the small number of counts in pre-experiment years, we calculated male baseline attendance by averaging the annual peak male attendance at each individual lek over 4 years (2002–2005). We assessed changes in early-, mid-, and late-season peak male attendance from this 4-year baseline attendance. Female attendance was highly variable throughout the season with a short (1–3 day) peak in attendance at each lek. Due to the limited number of annual counts, female counts from 2002 to 2005 were not reliable estimates of peak female attendance and could not be used as baseline attendance levels. Because we introduced noise to experimental leks after the peak in female attendance in 2006, we used maximum female counts from 2006 as a baseline for each of the 8 leks monitored that year. We assessed changes in annual peak female attendance from this 1-year baseline attendance. The 8 leks added to the experiment in 2007 were not included in statistical analyses of female attendance due to the lack of a baseline.

Statistical Analyses

We used an information-theoretic approach to evaluate the support for alternative candidate models (Table 2). All candidate models were linear mixed-effect models that assessed the relation between covariates and the proportional difference in annual and within-season peak attendance and baseline attendance (both males and female) (Tables 2 & 3). We ranked models on the basis of differences in Akaike's information criterion for small sample sizes (ΔAIC_c) (Burnham & Anderson 2002). Akaike weights (w_i) were computed for each model on the basis of ΔAIC_c scores. We calculated model-averaged variable coefficients, unconditional 95% CI, and variable importance (weight across models) for variables contained in models that were strongly supported ($\Delta AIC_c < 2$). All statistical analyses were performed in R (version 2.12.1) (R Development Team 2010).

Table 3. Mixed-effect candidate models used to assess change in peak annual attendance of female Greater Sage-Grouse at leks from pre-experiment baseline attendance in 2006 during noise playback.

Model[a]	K[b]	ΔAIC_c[c]	w_i[d]
Null, random effects only[e]	4	0	0.71
Treatment[e]	5	1.9	0.27
Treatment+experiment year	6	8	0.01
Treatment×experiment year	7	14	<0.001

[a]*All models contained pair and year as random effects. Due to the small sample size (4 pairs), pair type variable (road versus drilling) was not included in the model set. Covariates: treatment, lek treatment (noise or control assigned to individual leks within a pair); experiment year, years of experimental noise exposure.*
[b]*Number of parameters in the model.*
[c]*Difference in AIC_c (Akaike's information criterion for small sample size) values from the most strongly supported (lowest AICc) model.*
[d]*Akaike weight.*
[e]*Model with substantial support ($\Delta AIC_c < 2$).*

The detection probability for males and females is likely to vary across a season and among leks (Walsh et al. 2004). We sought to minimize sources of error and maximize detection by conducting frequent counts from locations with a clear view of the lek and by implementing a paired treatment design (each noise lek is compared with a similar control lek, monitored by the same observer on the same days). To ensure that detection probability did not differ among noise and control leks, we corrected our data for detection probability. First, we used detection error rates, estimated as difference between the maximum count and the count immediately before or after the maximum count within a day (for both males and females), and then we applied the bounded-count method (for males only; Walsh et al. 2004). With the multiple-count estimator, estimates of detection between noise and control leks did not differ (males: $t = 1.02$, df $= 6$, $p = 0.35$; females: $t = 0.21$, df $= 3$, $p = 0.84$). We analyzed both corrected and uncorrected counts and found that neither correction qualitatively changed our results; therefore, results are presented for uncorrected counts.

Results

Male Attendance

Peak male attendance at both types of noise leks decreased more than attendance at paired control leks, but the decreases varied by noise type. In the most strongly supported models of the candidate set ($w_i = 0.90$, all

Table 4. Model-averaged parameter direction and effect sizes and variable importance for all variables present in strongly supported models ($\Delta AIC_c < 2$ in Table 2) of changes in peak attendance of male greater sage-grouse at leks from baseline attendance during experimental noise playback.

Variable	*Percent effect size (SE)*	*Variable importance**
Intercept	31 (22)	1.0
Treatment, noise	−29 (7)	0.91
Type, road	33 (22)	0.91
Treatment, noise∗type, road	−40 (10)	0.91
Season, mid	18 (6)	0.66
Season, late	23 (6)	

**Variable importance is the summed weight of all models containing that variable.*

other models $\Delta AIC_c > 6.1$) (Table 2), there was an interaction of the effects of experimental treatment (control versus noise) and noise type (drilling versus road) on annual peak male attendance. At leks treated with road noise, decreases in annual peak male attendance were greater (73%), relative to paired controls, than at drilling noise leks (29%). As indicated by the effect size for the main effect of pair type, attendance at control leks paired with road noise leks was 33% greater relative to the baseline than control leks paired with drilling noise leks (Table 4). However, changes in attendance were compared within a pair to control for such differences. Male attendance increased over the course of a season, with 18% and 23% increases in peak male attendance in mid and late season from the early-season peaks, but seasonal increases were similar across noise and control leks (Table 4 & Fig. 2b).

There was no evidence that the effect of noise on attendance changed as years of exposure to noise increased. The models with substantial support did not contain a main effect of years of exposure or an interaction of years of exposure and treatment type (control versus noise) (Table 2). In spite of decreases in attendance throughout the experiment, peak male attendance exceeded baseline attendance on all leks in 2006, 13 leks in 2007, and 11 leks in 2008 (Table 4 & Fig. 2c). There was an increase in sage grouse abundance regionally in 2006 (Fig. 3).

After the experiment (2009), attendance at leks we experimentally exposed to drilling and road noise was lower relative to paired controls (Table 2). The model that included the treatment variable showed an effect size of −30% (across road and drilling noise leks) but had only moderate support ($\Delta AIC_c = 3.3$) relative to the null model.

Female Attendance

Peak female attendance at leks treated with noise in 2007 and 2008 decreased from the 2006 baseline, relative to control leks (Table 3). The most strongly supported model in the set was the null model; however, the model that included noise treatment was highly supported ($\Delta AIC_c < 2$). The effect size of noise treatment on female attendance was −48% (10% SE), which is similar to the effect of noise on male attendance averaged across both noise types (51%).

Discussion

Results of previous studies show abundance of Greater Sage-Grouse decreases when natural gas and coal-bed methane fields are developed (Holloran 2005; Walker et al. 2007; Doherty et al. 2008). Our results suggest that chronic noise may contribute to these decreases. Peak male attendance relative to the baseline was lower on noise leks than paired control leks, and the decrease was larger at road noise leks (73% decrease in abundance compared with paired controls) than drilling noise leks (29%; Fig. 3). These decreases were immediate and sustained. The effects of noise occurred in the first year of the study and were observed throughout the experiment, although patterns of male attendance within a season were similar at noise and control leks. Differences in male attendance between noise and control leks in the year after the experiment were not supported in the top models, which suggests attendance rebounded after noise ceased. However, the sample size for this analysis was small, and the effect size (30% average decreases in male attendance for both noise types) suggests a residual effect of noise.

There are 2 mechanisms by which noise may reduce male attendance. First, males on noise leks may have had higher mortality than males on control leks. Noise playback was not loud enough to cause direct injury to individuals, but mortality could be increased indirectly by noise playback if the sounds of predators (coyotes [*Canis latrans*] or Golden Eagles [*Aquila chrysaetos*]) were masked by noise. However, on-lek predation events were rare. We observed ≤1 predation event per lek per season during the experiment (observations of sage-grouse carcasses or feathers at a lek [J. L. Blickley, personal observation]). The cumulative effect of rare predation events would lead to a gradual decrease in attendance, rather than the rapid and sustained decrease we observed. Furthermore, experimental noise was likely too localized to substantially affect off-lek predation because noise levels decreased exponentially as distance to the speakers increased (Fig. 1b). To date, increased predation risk of adults due to anthropogenic noise has not been demonstrated in any species, but some species increase vigilance when exposed to noise, leaving less time for feeding, displaying, and other important behaviors (Quinn et al. 2006; Rabin et al. 2006). Noise may also affect off-lek mortality indirectly. For example, noise-stressed males may be more susceptible to disease due to a suppressed

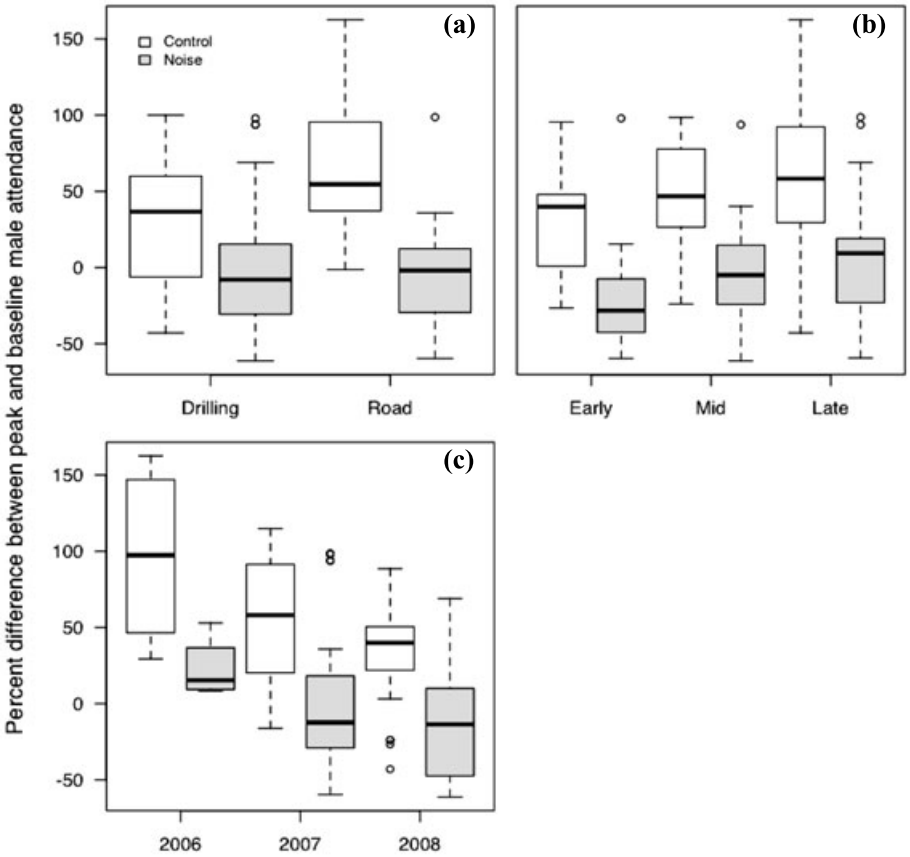

Figure 2. Percent difference between baseline attendance (i.e., abundance before experiments) of male Greater Sage-Grouse and (a) peak male attendance on control leks and leks treated with noise from natural gas drilling and road noise, (b) peak male attendance in the early (late February to 1 week prior to peak female attendance for that lek), mid (1 week before and after female peak [female peak ranged from 15 March to 6 April]), and late (starting 1 week after female peak) breeding season; on control leks and leks treated with noise, and (c) peak male attendance at control leks and leks treated with noise in experimental years 2006, 2007, and 2008 in Fremont County, Wyoming (U.S.A.) (horizontal lines, median value; box ends, upper and lower quartiles, whiskers, maximum and minimum values). Data are observed values, not model output.

immune response (Jankowski et al. 2010). Although long-term stress from noise is unlikely to be the primary cause of the rapid decreases in attendance we observed here, it may have been a contributing factor over the course of the experiment. Furthermore, in areas of dense industrial development, where noise is widespread, noise effects on mortality may be more likely.

Alternatively, noise may lower male attendance through displacement, which would occur if adult or juvenile males avoid leks with anthropogenic noise. Such behavioral shifts are consistent with the rapid decreases in attendance we observed. Adult male sage grouse typically exhibit high lek fidelity (Schroeder & Robb 2003) and visit leks regularly throughout the season, whereas juvenile males visit multiple leks and their attendance peaks late in the season (Kaiser 2006). If juveniles or adults avoid noise by visiting noisy leks less frequently or moving to quieter leks, overall attendance on noisy leks could be reduced. We could not reliably differentiate between juveniles and adults, so we do not know the relative proportion of adults and juveniles observed. Consistent with displacement due to noise avoidance, radio-collared juvenile males avoid leks near deep natural gas developments in Pinedale, Wyoming, which has resulted in decreases in attendance at leks in close proximity to development and increased attendance at nearby leks with less human activity (Kaiser 2006; Holloran et al. 2010). Reduced recruitment of juvenile males is unlikely to be the only driver of the patterns we observed because we did not observe larger decreases in lek attendance on noise-treated leks later in the season, when juvenile attendance peaks. Rather, we found immediate decreases in attendance early in the season when playback began (Fig. 2b), at which time there are few juveniles on the lek. This

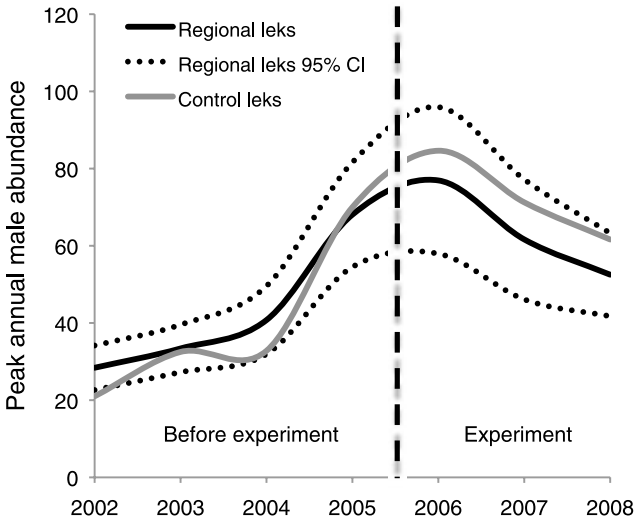

*Figure 3. Maximum abundance of male Greater Sage-Grouse from 2002 to 2008 at control leks (*n = *8) (no anthropogenic sound played) and other leks in the region that were not part of the experiment (regional leks) (*n = *38).*

is consistent with both adult and juvenile noise avoidance. We did not find evidence for a cumulative negative effect of noise on lek attendance, although cumulative effects may have been masked by regional population declines after 2006, a year of unusually high abundance (Fig. 3).

Female attendance at leks treated with noise was lower than that on control leks; however, the null model and the model that included noise treatment were both highly supported, providing only moderate support for the effects on noise on attendance. For this model, the overall estimated effect of noise on female attendance (−48%) was similar to that of the effect of noise on male attendance. Due to the high variability of female daily maximum attendance throughout the season and small sample size for this analysis (female attendance data available for only 4 of the 8 lek pairs), our statistical power to detect differences in female attendance was limited and effect sizes may not be representative of actual noise effects.

Our results suggest that males and possibly females avoid leks exposed to anthropogenic noise. A potential cause of avoidance is the masking of communication. Masked communication is hypothesized to cause decreases in abundance of some animal species in urban and other noisy areas. For example, bird species with low-frequency vocalizations are more likely to have low abundance or be absent from natural gas developments, roads, and urban areas than species with high-frequency vocalizations, which suggests that masking is the mechanism associated with differences in abundance (Rheindt 2003; Francis et al. 2009; Hu & Cardoso 2010). Sage-grouse may be particularly vulnerable to masked communication because their low-frequency vocalizations are likely to be masked by most sources of anthropogenic noise, including the noises we played in our experiment (Supporting Information). This may be particularly important for females if they cannot use acoustic cues to find leks or assess displaying males in noisy areas.

Alternatively, individuals may avoid noisy sites if noise is annoying or stressful, particularly if this noise is associated with danger (Wright et al. 2007). Intermittent road noise was associated with lower relative lek attendance than continuous drilling noise, in spite of the overall higher mean noise levels and greater masking potential at leks treated with drilling noise (Supporting Information). Due to the presence of roads in our study area, sage grouse may have associated road noise with potentially dangerous vehicular traffic and thus avoided traffic-noise leks more than drilling-noise leks. Alternatively, the pattern of decrease may indicate that an irregular noise is more disturbing to sage grouse than a relatively continuous noise. Regardless, our results suggest that average noise level alone is not a good predictor of the effects of noise (Slabbekoorn & Ripmeester 2008) and that species can respond differently to different types of noise.

Our results cannot be used to estimate the quantitative contribution of noise alone to observed decreases in Greater Sage-Grouse abundance at energy development sites because our experimental design may have led us to underestimate or overestimate the magnitude of these effects. Decreases in abundance due to noise could be overestimated in our study if adults and juveniles are displaced from noise leks and move to nearby control leks, which would have increased the difference in abundance between paired leks. Similar displacement occurs in areas of energy development, but over a much larger extent than is likely to have occurred in response to localized playbacks in our experiment (Holloran et al. 2010).

In contrast, we could have underestimated noise effects if there were synergistic effects of noise and other disturbances associated with energy development. For example, birds with increased stress levels due to poor forage quality may have lower tolerance for noise-induced stress, or vice versa. Noise in our experiment was localized to the immediate lek area and only played during the breeding season, so we cannot quantify the effects of noise on wintering, nesting, or foraging birds. Noise at energy development sites is less seasonal and more widespread than noise introduced in this study and may thus affect birds at all life stages and have a potentially greater effect on lek attendance. Leks do not represent discrete populations; therefore, local decreases in lek attendance do not necessarily reflect population-level decreases in abundance. However, at large energy development sites, similar displacement of Greater Sage-Grouse away from the ubiquitous noise may result in population-level declines due to spatially exten-

sive changes in land use or increases in dispersal-related and density-dependent sources of mortality (Aldridge & Boyce 2007). Enforcement and refinement of existing seasonal restrictions on human activity could potentially reduce these effects.

We focused on the effect of noise associated with deep natural gas and coal-bed methane development on sage grouse, but our results may increase broader understanding of the effects of noise on animals. Both intermittent and constant noise from energy development affected sage grouse. Other noise sources with similar frequency range and temporal pattern, such as wind turbines, oil-drilling rigs, and mines, may have comparable effects. Similar effects may also be associated with highways, off-road vehicles, and urbanization so that the potential for noise to have an effect is large.

We believe that noise should be investigated as one potential cause of population declines in other lekking North American grouse species that are exposed to similar anthropogenic development. Populations of many bird (van der Zande et al. 1980; Rheindt 2003; Ingelfinger & Anderson 2004) and mammal (Forman & Deblinger 2000; Sawyer et al. 2009) species have been shown to decrease in abundance in response to road, urban, and energy development, and noise produced by these activities may contribute to these decreases. Our results also demonstrate that wild animals may respond differently to chronic intermittent and continuous noise, a comparison that should be expanded to other species. Additionally, we think these results highlight that experimental noise playbacks may be useful in assessing the response of wild animals to chronic noise (Blickley & Patricelli 2010).

Acknowledgments

We thank A. Krakauer, T. Rinkes, S. Oberlie, S. Harter, T. Christiansen, W. Elsberry, our many excellent field assistants, and reviewers. Funding was provided by the University of California, Davis, U.S. Bureau of Land Management, U.S. National Fish & Wildlife Foundation, Wyoming Sage-Grouse Conservation Fund, and the Wyoming Community Foundation Tom Thorne Sage-Grouse Conservation Fund.

Supporting Information

Spectrograms and power spectrums of drilling noise, road noise and male sage-grouse vocal display (Appendix S1), map of experimental and control leks (Appendix S2), and noise playback levels on experimental leks (Appendix S3) are available online. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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Global Ecology and Conservation 3 (2015) 62–71

Contents lists available at ScienceDirect

Global Ecology and Conservation

journal homepage: www.elsevier.com/locate/gecco

Original research article

Anthropogenic noise alters bat activity levels and echolocation calls

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ARTICLE INFO

Article history:
Received 4 November 2014
Accepted 4 November 2014
Available online 8 November 2014

Keywords:
Bats
Anthropogenic noise
Echolocation
Activity level
Gas compressor station

ABSTRACT

Negative impacts from anthropogenic noise are well documented for many wildlife taxa. Investigations of the effects of noise on bats however, have not been conducted outside of the laboratory. Bats that hunt arthropods rely on auditory information to forage. Part of this acoustic information can fall within the spectrum of anthropogenic noise, which can potentially interfere with signal reception and processing. Compressor stations associated with natural gas extraction produce broadband noise 24 hours a day, 365 days a year. With over half a million producing gas wells in the U.S. this infrastructure is a major source of noise pollution across the landscape. We conducted a 'natural experiment' in the second largest gas extraction field in the U.S. to investigate the potential effects of gas compressor station noise on the activity levels of the local bat assemblage. We used acoustic monitoring to compare the activity level (number of minutes in a night with a bat call) of the bat assemblage at sites with compressor stations to sites lacking this infrastructure. We found that activity levels for the Brazilian free-tailed bat (*Tadarida brasiliensis*) were 40% lower at loud compressor sites compared to quieter well pads, whereas the activity levels of four other species (*Myotis californicus, M. cillolabrum, M. lucifugus, Parastrellus hesperus*) were not affected by noise. Furthermore, our results reveal that the assemblage of bat species emitting low frequency (<35 kHz) echolocation calls also showed a response, with a 70% reduction in activity levels at loud sites compared to quieter well pad sites whereas the assemblage using high frequency (>35 kHz) echolocation did not exhibit altered activity levels in noise. Lower activity levels of Brazilian free-tailed bats at loud sites indicate a potential reduction in habitat for this species. Additionally, a comparison of echolocation search calls produced by free-tailed bats at sites with and without compressor stations reveal that this species modifies its echolocation search calls in noise—producing longer calls with a narrower bandwidth. Call alterations might affect prey detection. These preliminary findings highlight the important need for further research of how anthropogenic noise affects bats on a landscape scale.
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http://dx.doi.org/10.1016/j.gecco.2014.11.002
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1. Introduction

Human infrastructure removes wildlife habitat, alters connectivity, causes mortality, and introduces ecological pollutants (Sih et al., 2011). A meta-analytical study found that bird densities decline within 1 km of infrastructure and non-volant mammal densities are reduced within 5 km (Benitez-Lopez et al., 2010). Less is known about bats' response to human alteration of habitat but recent work has shown that bat diversity and activity decline with increasing human development (Jung and Kalko, 2011). For instance, roads cause direct mortality (Medinas et al., 2013) and appear to function as barriers and filters to bat assemblages by restricting movement (Abbott et al., 2012; Bennett and Zurcher, 2012) and changing space use (Bennett et al., 2013). Because the disturbance created by roads and other human infrastructure is comprised of multiple components it is difficult to estimate the strength of any single influence (Forman et al., 2003).

Anthropogenic noise associated with infrastructure is an overlooked stimulus that could affect bat habitat use (Barber et al., 2010; Francis and Barber, 2013). Laboratory experiments have shown that gleaning bats exposed to traffic and gas compressor station noise exhibit reduced foraging efficiency (Siemers and Schaub, 2011; Bunkley and Barber, in preparation) and, when presented with the option, avoid hunting in noise (Schaub et al., 2008). Gleaning bats hunt using prey-generated sounds, which could be masked by anthropogenic noise. The potential effects of noise on these bats as wells on bats that utilize echolocation for prey detection are poorly understood at the landscape scale.

Two approaches have been used to tease apart the independent role of noise from other stimuli associated with anthropogenic disturbance of wildlife. Large-scale playback experiments have shown that traffic noise caused over a one-quarter decline in migrating songbird abundance (McClure et al., 2013) and gas drilling and truck noise reduced male Greater Sage-Grouse (*Centrocercus urophasianus*) lek attendance (Blickley et al., 2012). Other investigators have taken advantage of 'natural experiments' in natural gas extraction fields, comparing quiet sites to loud sites with compressor stations. These studies have found that breeding bird densities decrease by one-third at loud sites (Bayne et al., 2008) and avian communities are substantially altered (Francis et al., 2009), resulting in the disruption of key ecological services such as seed dispersal (Francis et al., 2012). Despite strong evidence illustrating habitat degradation and loss for birds, no study has attempted to identify the effects of noise on habitat use by bats.

Here we present data from our investigation of bat activity levels in the second largest gas extraction field in the U.S., in northwest New Mexico. Compressor stations used for the extraction and transportation of natural gas produce spectrally broadband noise 24 h a day, 365 days a year. We compared bat activity levels and sonar call structure at control well pad sites to nearby sites with compressor stations and predicted that this noise source would influence bat activity levels and shape echolocation characteristics. In this 'natural experiment', control sites lacked a compressor station with its characteristic broadband, high intensity noise, but were otherwise similar to treatment sites with compressors. We quantified the background sound level of sites using continuously deployed Acoustic Recording Units while simultaneously acoustically monitoring bat activity. We hypothesized two potential responses of bat activity level to noise: (1) a continuous response with activity decreasing as background dB levels increased, indicating that bats respond to noise exposure in a dose–response fashion, or (2) a binary response to compressor noise with more activity at well pads than compressor sites and with equal activity between treatment sites, regardless of differences in background dB level, indicating a step-function response to noise exposure.

1.1. Methods

We conducted this work in May and June of 2013 in and near Rattlesnake Canyon Habitat Management Area in the San Juan Basin in northwestern New Mexico (Figs. 1 and 2). The San Juan Basin is the second largest natural gas basin in the United States and the largest coal-bed methane reservoir in the world, producing about one trillion cubic feet of gas per year from more than 150 gas fields (Fassett, 2010).

Piñon pine (*Pinus edulis*) and juniper (*Juniperus osteosperma*) trees are the dominant plants in this arid region (Francis et al., 2012). Rock outcroppings and a nearby canyon and reservoir are also prominent features on the landscape. The San Juan Basin is within the range of 17 bat species (Nowak, 1994), 14 of which were detected during this study. All identified species are in the family Vespertilionidae except for *T. brasiliensis*, which is in the Molossidae family (Nowak, 1994). Piñon–juniper woodlands in New Mexico support an abundant and diverse bat community (Jones, 1965; Chung-MacCoubrey, 1996, 2005). In part this is because piñon–juniper forest is a transition zone between the arid grass/shrubland community at lower elevations and mesic forests at higher elevations, thus, providing habitat for species from both regions (Jones, 1965).

Piñon–juniper habitat in general is considered suitable in terms of food availability, water, and roosts for *E. fuscus*, *M. ciliolabrum*, *M. evotis*, *M. thysanodes*, and *M. volans* (Chung-MacCoubrey, 2005). Large bodies of water, such as the Navajo reservoir, are also important resources for *T. brasiliensis* (Chung-MacCoubrey, 2005). Thus, we consider the piñon–juniper habitat to be generally suitable to the range of bats we sampled at our sites. Treatment and control sites can be considered similar in terms of roost habitat and potential foraging opportunities. We sampled during a time of high bat activity, which coincides closely with pregnancy and parturition of many bat species and thus represents a time of higher energy requirements for females (Altringham, 2011; Fenton, 1997; Kunz et al., 1982). Reproductive females appear to prefer piñon–juniper habitat to the grass/shrubland and mesic forest communities, both for maternity roosts and for rearing young (Chung-MacCoubrey, 2005).

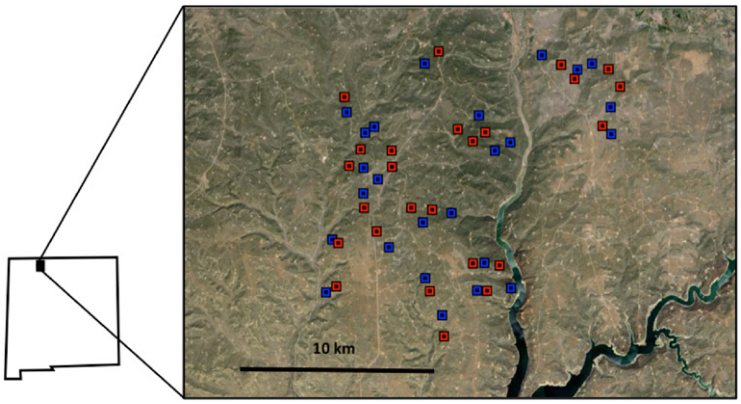

Fig. 1. Study area in the San Juan Basin New Mexico. Red/gray markers indicate sites with a compressor station (treatment) and blue/white markers identify well pads that lack a compressor station (control). Sites were matched by proximity and sampled simultaneously. Bare spaces on the aerial photos illustrate the physical footprints of the gas extraction sites. The Navajo reservoir is visible in the Southeast corner of the map.

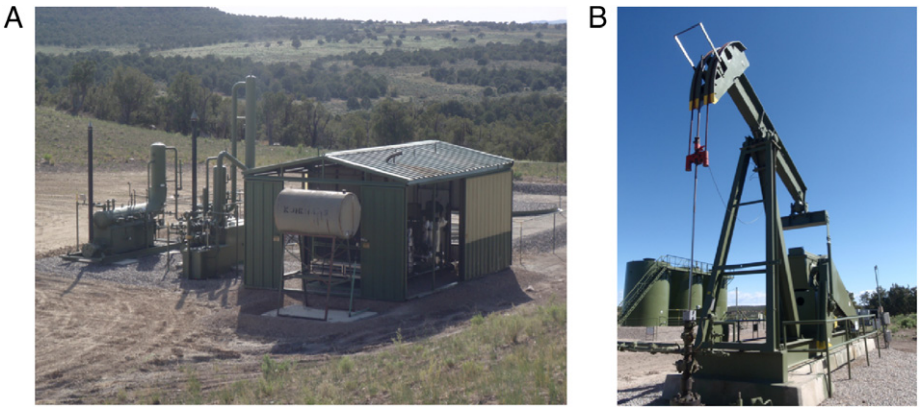

Fig. 2. A. A typical compressor station associated with a well pad. B. A derrick present on a site without a compressor station.

Twenty-five sites with gas compressor stations (treatment) were spatially matched with 25 well pad sites without compressors (control) that were within 500–1000 m (Figs. 1 and 2). Importantly, we sampled control and treatment sites matched by proximity simultaneously, controlling for the effects of moon phase, time, weather, and other factors that can influence bat activity. None of the sites were artificially illuminated during the night, excluding potential affects of anthropogenic light pollution. Using Wildlife Acoustics SM2 or SM2BAT+ Bat Detectors with SMX-US omni-directional, ultrasonic microphones, we conducted passive, acoustic surveys of bat activity levels (the number of minutes in a sampling night with an identified bat call; Kuenzi and Morrison, 1998).

We rotated 12 detectors through the 50 sites for one to seven nights per site using our simultaneous sampling regime of matched control and treatment pairs throughout. Sampling two to five nights typically detects 40%–60% of species richness at a site (Skalak et al., 2012). Five sites were sampled for only a single night as a result of battery failure; however, these data still proved to be useful and were included in analyses. Although it is unlikely that 100% of species were detected for every site, by having detectors deployed simultaneously at both the matched treatment and control sites, the detection rate should be comparable between site types.

Acoustic sampling of bats is regarded as an effective tool of assessing bat species and activity level and is a widely used method (Kunz and Brock, 1975; Kuenzi and Morrison, 1998; O'Farrell and Gannon, 1999). As with all data collection techniques, there are limitations to acoustic monitoring. The detection area of bat detectors varies with the temperature, humidity, wind, and the frequency and directionality of bat sonar, with an estimated 30–90 m detection distance (Wildlife Acoustics). Fortunately, with our spatially matched sampling design, monitors at control and treatment sites likely varied similarly in their detection range due to comparable microclimates.

Acoustic Recording Units (ARUs; Roland R05 or R09; MP3 128 kbps) were simultaneously deployed with bat detectors to measure the sound level at each site (Mennitt and Fristrup, 2012; Figs. 3 and 4). We used custom programs (Damon Joyce, NPS, AUDIO2NVSPL) to convert the MP3 recordings into hourly sound pressure levels and then to hourly LEQ (equivalent continuous sound level) values in dB(A) (Damon Joyce, NPS, Acoustic Monitoring Toolbox). These hourly sound levels were averaged over the duration of the bat detector deployment, which allowed us to use the continuous variable of decibel level (dB(A)) or the discrete variable of treatment (control vs. treatment) for the noise condition of a site in statistical analyses.

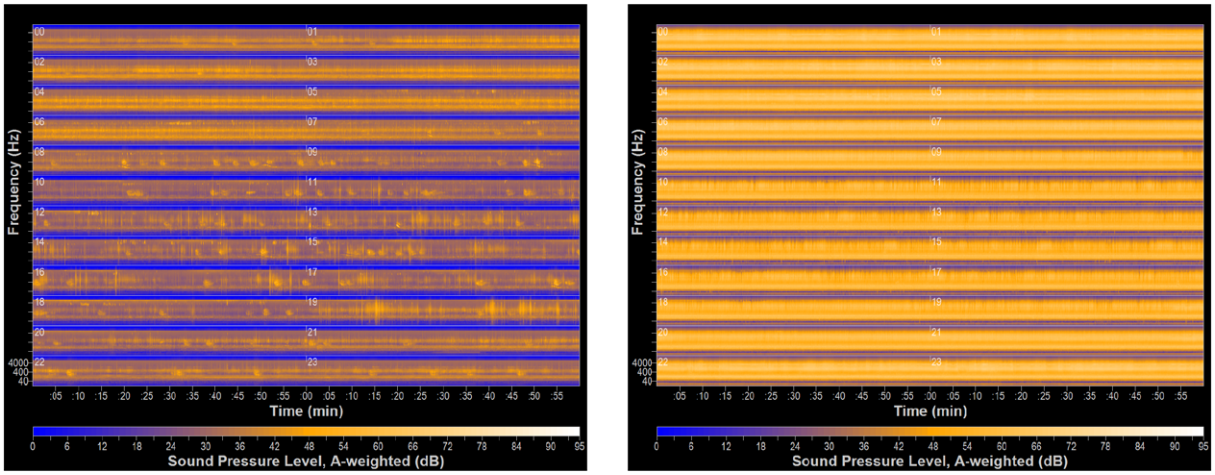

Fig. 3. Twenty-four hour spectrograms of control site 7 (left) and treatment site 7 (right) made using Roland R05 recording units, MP3 128 kbps. Frequency (kHz) is on the *y*-axis and time (minutes) is on the *x*-axis. Each line represents a two-hour block of time. The intensity of the color ramp illustrates amplitude or volume.

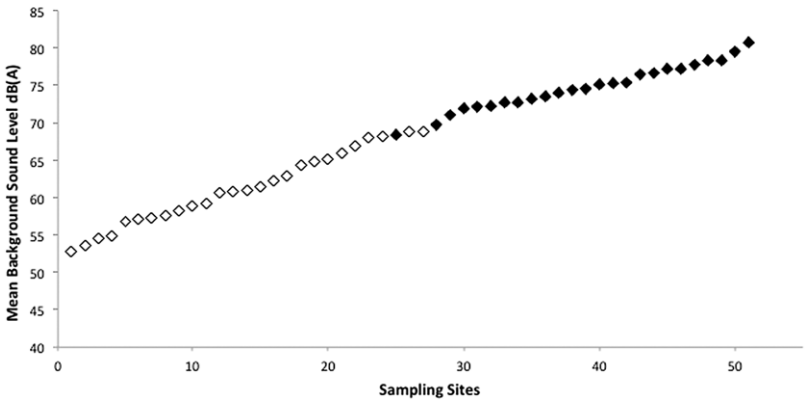

Fig. 4. Mean background sound level dB(A), which includes compressor station noise for treatment sites, varies as a continuous variable by site. White diamonds are sites that did not have compressor stations (control) and black diamonds represent sites that did have operating compressor stations (treatment).

We placed the bat detectors and ARUs approximately 1.5 m above the ground and positioned them along a woodland well pad edge approximately 50 m away from the center of the site. The triggering program for all bat detectors was designed to exclude any background noise below 24 kHz (the frequency band of compressor noise) to ensure that compressor noise did not bias triggering for the treatment sites. The triggering programs were identical for control and compressor sites, which avoided any triggering bias based on treatment. This was confirmed in a laboratory test by playing bat calls in quiet and compressor noise played back at the same sound level as field recordings. Some species produce calls below the 24 kHz triggering threshold such as *Lasiurus cinereus*. Therefore, calls used in analysis are only those calls that were produced by these species above the 24 kHz triggering threshold.

To assess and classify bat echolocation calls, we used the analysis program, Sonobat 3.2.0. This software identifies bat calls and their frequency range (high: >35 kHz; low: <35 kHz) among other call variables. Sonobat uses call classification algorithms of acoustic features to determine species identification by referencing a known call library. For analyses that required species level classification we only used calls that had a 90% or greater discrimination probability of falling within a multivariate parameter space of species call traits (Szewczak, 2013). Additionally, calls used for analysis also had a majority of individual calls in a recording sequence identified as the same species (Szewczak, 2013). We used the high (>35 kHz) and low (<35 kHz) acoustic groupings established by Sonobat in our analysis because of the natural clustering of bat calls above and below the 35 kHz delineating point (Szewczak, 2013). Other researchers have used these same frequency groups due to the functional differentiation of most migratory bats producing low frequency (<35 kHz) echolocation calls (Weller and Baldwin, 2012).

We also examined if bats change the structure of their calls in response to compressor noise. We focused on *T. brasiliensis* because it was the only individual species documented to avoid compressor sites in our models (see Section 1.2). Seventy-two *T. brasiliensis* calls from separate recordings taken at control sites and 48 calls from unique recordings taken at treatment sites were hand-analyzed to extract the call parameters of peak frequency (F_{peak}), maximum frequency (F_{max}), minimum

frequency (F_{min}) taken from −15 dB of F_{peak} (Surlykee and Moss, 2000), overall bandwidth (Band), maximum frequency of harmonic one ($H1_{max}$), minimum frequency of harmonic one ($H1_{min}$), harmonic one bandwidth ($H1_{band}$), and lowest apparent frequency (measured from the spectrogram). The frequency response curves of our microphones were ±3 dB across the relevant spectrum.

Only calls visually identified from spectral and temporal characteristics as search phase calls (Griffin et al., 1960; Surlykee and Moss, 2000) were compared to control for behavioral variation in calls. Calls used for analysis were taking from separate recordings and are not part of the same call sequence. In limited instances calls from multiple recording from the same site and night were used in analysis. Thus, it is possible that some of these calls could be from the same individual. However, because we did not analyze multiple calls from a single recording our results are likely conservative. We used Avisoft SASLab Pro Version 5.2.07 (Specht, Avisoft Bioacoustics) for sound analyses. We created spectrograms of the call using a Hanning window Fast Fourier Transformation (FFT) of 1024 to increase resolution of spectral features. Call duration was measured from the oscillogram and guided by the spectrogram.

1.2. Analysis

Using the number of echolocation calls recorded at a site, we assessed how noise affects bat activity level of species and assemblages (Weller and Baldwin, 2012). The number of minutes in a night that had an identified bat call was used as a metric of bat activity (Miller, 2001). We treated each sampling night separately and controlled for repeated sampling of sites using a random intercept. Recordings of bat calls were grouped by echolocation frequency. For these "call frequency assemblages" we used all calls identified as high and low frequency (9701 calls), not the more conservative species classification. Species with 45 or more classified calls were analyzed individually.

We analyzed data using R (R Development Core Team, 2011). We first identified and calculated habitat variables considered important to bat activity from the literature (Ciechanowski et al., 2007; Kanuch and Kristin, 2005; Korine and Pinshow, 2004). These included: percent of forest cover (the area with trees present) within 500 m of the center of each site, distance to the nearest large body of water, which was the Navajo Reservoir (the only other water present was in small, tanks set out for cattle) and a lunar metric calculated by multiplying the percent of the moon that was full by the amount of time the moon was visible. We also considered linear and quadratic effects of date in the event that there were unknown changes that might have occurred throughout the season, such as an insect hatch or seasonal movements of bat species, which might alter bat activity over time. We used a two-step procedure (Steen et al., 2013) that allowed us to test the effects of noise on bat activity level while also determining – and controlling for – other environmental factors that may drive activity levels at different sites. First, we built Poisson-distributed generalized linear mixed-effects models containing varying combinations of environmental covariates thought to affect bat activity and a random intercept for survey site. Not all combinations of environmental variables were tested because several, such as date and the moon metric, were highly correlated ($r > 0.5$) and consequently were not included in the same model. Assumptions of normality were assessed for each variable. We used Akaike's Information Criterion (AIC, Akaike, 1974) to rank models and then incorporated the covariates within the highest-ranked model into the next step.

In the second step, we tested our a priori hypothesis that noise affects activity levels of different bat assemblages. We hypothesized that noise might affect activity levels of bats in one of two ways: (1) Bats may respond to noise in a linear fashion with activity decreasing linearly with increasing background dB levels—meaning that well pad and compressor sites that are inherently noisier will experience less bat activity. Or, (2) bats may respond to noise in a binary fashion with more activity at well pads than compressor sites, but activity being equal among treatments regardless of differences in background sound levels. Each hypothesized response was represented by a model—one that added a covariate indicating the dB level of a site to the best model from step one, and another that added a binary factor indicating treatment (gas compressor = 1, well pad = 0) to the best model from step one. We ranked and compared these two models – dB and treatment – as well as the best habitat model from the first step using AIC. We considered our hypothesis of noise affecting site-use by bats as supported if either the dB or treatment models received a lower AIC value than the best model from step one, and as strong evidence if the 95% confidence interval for either the dB or treatment variable excluded zero and marginal evidence if the 85% confidence intervals excluded zero. Eight-five percent confidence intervals are more consistent with an AIC approach than are 95% confidence intervals (Arnold, 2010). All predictor variables were z-transformed before analysis to put them on the same scale and thus make them comparable during analyses.

For the analysis of the echolocation call data we used linear mixed effect models with a random intercept for point. We built three models, one containing a variable for decibel (dB) level, one with a noise treatment parameter, and a null model. Models were ranked by AIC and evaluated.

2. Results

Site activity levels were measured at the point of the detector (approximately 50 m from the center of the well pad or compressor station) with the addition of the microphone range (30–90 m). The detectors logged 9701 recordings identified as bats, of which 2278 were classified to species (Szewczak, 2013). Those calls classified to the species level met both the 90% minimum discrimination probability and the classification agreement of multiple calls in a sequence, making this a more conservative measure (Szewczak, 2013). Species identified included: low frequency (<35 kHz) – *Antrozous pallidus*

Table 1
AIC tables of the Habitat, Habitat + Treatment, and Habitat + dB for all bats, low frequency echolocating assemblage (*Antrozous pallidus*, *Eptesicus fuscus*, *Lasionycteris noctivagans*, *Lasiurus cinereus*, *Myotis thysanodes*, and *Tadarida brasiliensis),* the high frequency echolocating assemblage (*Lasiurus blossevillii*, *M. californicus*, *M. cillolabrum*, *M. evotis*, *M. lucifugus*, *M. volans, M. yumanensis*, and *Parastrellus hesperus*), and individual species for bat activity levels of 50 sites sampled in the San Juan Basin in northwest New Mexico. See Section 1.1 for call parameter information.

Model	K	AIC	ΔAIC	W_i
All bats				
Habitat	4	2500.94	0.00	0.49
Habitat + Treatment	5	2502.08	1.14	0.28
Habitat + dB	5	2502.50	1.56	0.23
Low frequency				
Habitat + dB	6	425.78	0.00	0.44
Habitat + Treatment	6	426.41	0.64	0.32
Habitat	5	426.95	1.17	0.24
High frequency				
Habitat	4	1231.62	0.00	0.52
Habitat + Treatment	5	1232.88	1.26	0.28
Habitat + dB	5	1233.45	1.83	0.21
Myotis californicus				
Habitat	4	217.86	0.00	0.57
Habitat + Treatment	5	219.71	1.85	0.22
Habitat + dB	5	219.83	1.97	0.21
Myotis cillolabrum				
Habitat	4	1142.95	0.00	0.51
Habitat + Treatment	5	1144.15	1.21	0.28
Habitat + dB	5	1144.71	1.76	0.21
Myotis lucifugus				
Habitat	4	226.33	0.00	0.56
Habitat + dB	5	228.17	1.84	0.22
Habitat + Treatment	5	228.28	1.95	0.21
Parastrellus hesperus				
Habitat	4	313.91	0.00	0.57
Habitat+Treatment	5	315.77	1.85	0.22
Habitat + dB	5	315.88	1.97	0.21
Tadarida brasiliensis				
Habitat + Treatment	6	368.46	0.00	0.37
Habitat	5	368.50	0.04	0.36
Habitat + dB	6	369.08	0.62	0.27

(gleaning), *Eptesicus fuscus*, *Lasionycteris noctivagans*, *Lasiurus cinereus*, *Myotis thysanodes*, and *Tadarida brasillensis* and high frequency (<35 kHz) – *Lasiurus blossevillii*, *M. californicus*, *M. cillolabrum*, *M. evotis*, *M. lucifugus*, *M. volans, M. yumanensis*, and *Parastrellus hesperus*. Twelve species were observed at both compressor and well sites. *M. thysanodes* was recorded only at sites with compressors and *M. volans* was detected only at well pad sites. Both species were rare and low overall levels of activity (few recordings) might be reflective of lower abundance or detectability (Table A1). Discrimination probabilities of species identification did not differ between calls recorded at treatment and control sites, indicating that background noise did not affect call classification (Linear mixed-effects model fit by maximum likelihood, All bats: $p = 0.46$; *Tadarida brasiliensis*: $p = 0.47$).

All bat species and assemblages were associated with one or more of the habitat variables (Table A2). Covariates of either noise treatment or level were then added to the best model containing habitat covariates (Table 1). Adding these noise covariates improved the fit of models for *T. brasiliensis* and the assemblage of species with low-frequency echolocation calls—supporting our hypothesis that noise affects activity levels of some bat species and assemblages. *T. brasiliensis* was negatively associated with the binary treatment factor with activity at compressor sites being 60% (85% CI = 36%–100%) that of quieter well pad sites. Species with low frequency echolocation calls were negatively associated with site dB level, with activity at the loudest sites being 33% (85% CI = 20%–55%) that of quieter sites (Fig. 5). Models with noise covariates have a lower AIC score than the habitat only models, indicating that these models overcame the 2 AIC penalty for an additional parameter and are more parsimonious. The exclusion of zero with 85% confidence intervals indicates marginal evidence that these groups are affected by noise.

The assemblage of species with high frequency echolocation calls did not have either noise variable in the top model. Additionally, the treatment and dB factors did not improve the fit of the models for *Myotis californicus, M. cillolabrum, M. lucifugus*, or *Parastrellus hesperus*, indicating that noise level is not an informative parameter for the activity levels of these species.

For the 120 *T. brasiliensis* search calls analyzed for spectral characteristics the best model for call duration included a factor for treatment indicating that calls were 0.90 ms (85% CI = 0.35–1.44, 95% CI =0.16–1.64) longer at compressor

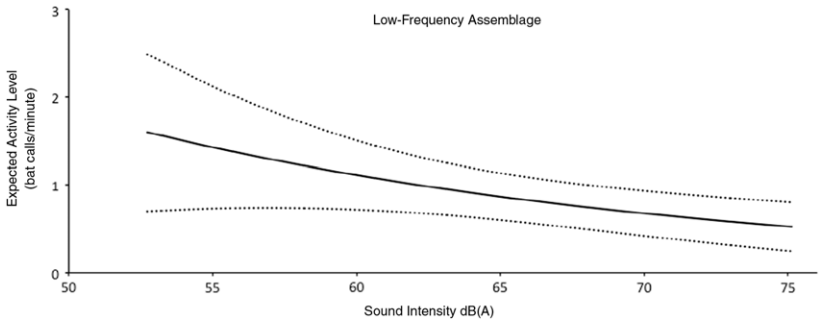

Fig. 5. Modeled activity level of the low frequency (<35 kHz) echolocating bat assemblage corresponding to background noise level (dB(A)), with standard error lines.

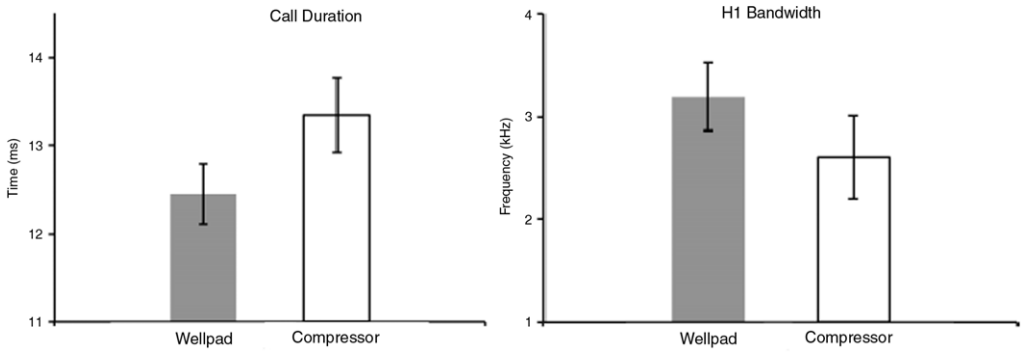

Fig. 6. *Tadarida brasiliensis* mean call duration and first harmonic bandwidth of 48 calls from sites with compressor stations (white/treatment) and 72 calls from sites without compressor noise (gray/control), with standard error bars.

Table 2
AIC tables for analyzed parameters of 120 *Tadarida brasiliensis* calls from sites with gas compressor stations (treatment) and sites lacking stations (control). See Section 1.2 for definitions of call parameters.

Model	K	AIC	ΔAIC	Wi
Call duration				
Treatment	4	497.32	0	0.674
dB	4	499.78	2.46	0.20
Null	3	500.64	3.32	0.13
$H1_{band}$				
Treatment	4	468.45	0	0.44
Null	3	468.56	0.11	0.41
dB	4	470.54	2.09	0.15
F_{peak}				
Null	3	432.12	0	0.55
Treatment	4	433.87	1.75	0.23
dB	4	433.98	1.87	0.22
$H1_{min}$				
Null	3	409.25	0	0.55
Treatment	4	410.96	1.70	0.23
dB	4	411.10	1.85	0.22
Lowest apparent frequency				
Null	3	415.53	0	0.57
Treatment	4	417.36	1.83	0.23
dB	4	417.53	2.00	0.21

sites compared to well pad sites (Fig. 6). The best model for bandwidth also included a factor for treatment (Table 2), indicating bandwidth was 600 Hz (85% CI = 70–1100) narrower at locations with compressor stations (Fig. 6). The null model (intercept only) was the best model for all other characteristics of *T. brasiliensis* search calls.

3. Discussion

Our results indicate that anthropogenic noise from natural gas compressor stations affects some bat species. Our findings offer marginal evidence that *T. brasiliensis* show lower activity levels in gas compressor noise specifically and low frequency (<35 kHz) echolocating bat species reduce activity levels in increased noise in general. Additionally, we found strong evidence that *T. brasiliensis* lengthens echolocation call duration and marginal evidence that bandwidth is narrowed in response to noise. These multiple lines of evidence reveal that some bat species are affected by noise and highlights the need for continued research in this field. Our use of spatially matched sites during data collection and modeling of habitat variables in analysis isolated noise as a primary factor driving these patterns, however, other factors such as predator abundance and prey distributions might also play a role in altered activity levels.

T. brasiliensis showed reduced activity levels at sites with compressor noise, lending support for the hypothesis that broadband, high intensity compressor noise reduces species' activity. One of the dominant species in our study site, *T. brasiliensis,* likely drove the observed effect of a negative association with dB(A) level for the assemblage of low frequency echolocating bats. However, noise had an apparently stronger effect (higher model weight (Table 1) and narrower 85% confidence intervals for the low frequency group (CI = 20%–55%) as compared to only *T. brasiliensis* (CI = 36%–100%)) indicating a general response among the low frequency group to noise. Our model shows a near linear relationship between activity levels of this assemblage and background noise level (Fig. 5). The low frequency group of bats exhibits a continuous response of decreasing activity with increasing background dB levels, lending support to the dose–response hypothesis. Interestingly, three other species that produce high frequency (>35 kHz) echolocation calls did not show a difference in activity level in loud versus quiet sites. This indicates that echolocation frequency is important to a bat species' response to noise exposure.

The effect of noise on the activity levels of low frequency echolocating bats indicates potential masking of some echolocation call frequencies (Fenton and Bell, 1981) by lower frequency anthropogenic noise (<24 kHz). Several of the species in the low frequency assemblage (*A. pallidus, E. fuscus, L. cinereus, L.s noctivagans, M. thysanode, T. brasillensis*) have echolocation calls that contain components that could be susceptible to masking by anthropogenic noise (Fenton and Bell, 1981; Szewczak, 2013). Even calls that do not directly overlap the frequency range of noise may still experience negative effects from noise. Recently, Hage et al. (2013) showed echolocation frequency shifts in the constant frequency (CF) horseshoe bat, which has a dominant resting frequency at 75 kHz (Hage et al., 2013). These bats shifted frequency upwards when presented with noise from a variety of spectra, including in the range of anthropogenic noise (10–30 kHz; Hage and Metzner, 2013), indicating that perhaps even noise that does not spectrally overlap echolocation signals can cause interference with echolocation processes.

Our analysis of *T. brasiliensis* echolocation search calls produced at sites with compressor noise indicates a reduction in call bandwidth at the 85% confidence level and an increase in duration at the 95% confidence level (Fig. 6). Concentrating energy spectrally likely improves signal-to-noise ratio of the call and increasing call duration increases signal redundancy and thus detectability in noisy background conditions (Schnitzler and Kalko, 2001; Brumm et al., 2004). *T. brasiliensis* is often considered an urban survivor because it commonly uses anthropogenic structures, such as bridges, for roosting (Davis et al., 1962; Allen et al., 2008). However, in a landscape where sites vary in their noise exposure, *T. brasiliensis* exhibits greater levels of nocturnal activity in quieter areas during periods when bats are likely foraging. Differences in noise tolerance at roosts versus foraging sites might be attributable to an alteration in prey detection as a result of noise induced changes in echolocation calls. Additionally, the narrower bandwidth and increased temporal variation of highway traffic noise as compared to the spectrally broad and temporally constant nature of compressor noise may also affect behavioral responses in *T. brasiliensis*.

We recognize that our data are limited in scope, due to the single sampling season, and the marginal evidence of several of our results. However, because multiple lines of evidence, lowered activity levels of low frequency echolocating bats and *T. brasiliensis,* as well as the altered echolocation signals of *T. brasiliensis,* indicate that some species of bats are likely negatively affected by noise, emphasizing the need for continued investigations into the response of bats to noise at the landscape level. The scale of anthropogenic noise across the world, including, but not limited to noise produced by energy extraction infrastructure, is immense. Given this reality in addition to the sensitive status of many bat species makes the question of bat responses to noise worthy of continued investigation.

We estimate that compressor noise affects about 356 km^2 of potential *T. brasiliensis* habitat in the San Juan Basin. Detectors were set approximately 50 m from the center of a compressor site and given the conservative measure of a 30 m detection rate our analysis of activity level represents a 80 m detection radius around a compressor. In 2010 there were 35,431 productive wells in the San Juan Basin (Fassett, 2010) and approximately half had compressor stations, resulting in an estimated 356 km^2 of habitat directly affected by noise. However, because wells are spread throughout the region, a much larger area likely experiences habitat degradation and possibly fragmentation. A map illustrating the range of *T. brasiliensis* overlaid with known compressor stations illustrates the potential scale of the problem (Fig. 7).

Bats are important indicators of ecosystem health and providers of a variety of ecosystem services (Jones et al., 2009) making them an essential group to conserve in order to maintain functioning habitats. North American bat populations are currently experiencing unprecedented declines as a result of the white nose syndrome epidemic and mortality from wind power development (Foley et al., 2011; Kunz et al., 2007). As a consequence, preventable habitat degradation should be prioritized by management agencies to reduce anthropogenic pressures on these sensitive populations. Our findings that

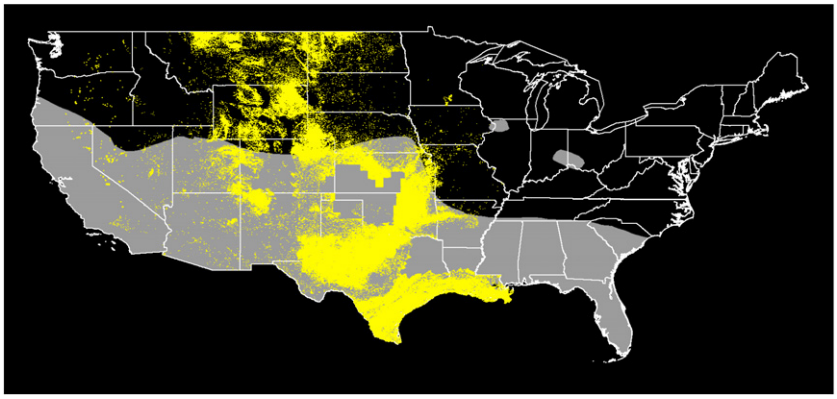

Fig. 7. The geographic overlap of *T. brasiliensis* range (gray area; Bat Conservation International, NationalAtlas.gov, 2013) with available data for compressor stations in the Western United States (yellow/white; U.S. Geological Survey, 2012).

some bat species and assemblages experience reduced activity at sites with noise suggest that anthropogenic noise is another potential threat to bat populations and should be considered when managing habitat (Francis and Barber, 2013). Fortunately, anthropogenic noise can be mitigated, for instance by building sound-damping walls around compressor stations (Francis et al., 2012). These important preliminary findings indicate that further research is promptly needed to better understand the responses of bats to anthropogenic noise on the landscape scale.

Acknowledgments

We thank Bat Conservation International for funding and the Natural Sound and Night Skies Division of the National Park Service for equipment; P. Myers, H. Nelson, A. Bunkley, S. Allred, D. Nelson, for their help with fieldwork; Z. Mroz for conducting acoustic analyses; R. Nuxoll and O. Bigelow of the BSU Scientific Instrumentation Shop for assistance with materials; J. Belthoff, I. Robertson, and three anonymous reviewers for useful manuscript suggestions; and the Farmington, NM and Tres Rios, CO BLM personnel for site information.

Role of the funding source

Funding was provided by Bat Conservation International (BCI). BCI did not play a role in study design, collection, analysis and interpretation of data, writing, or in the decision to submit the paper for publication.

Appendix A. Supplementary data

Supplementary material related to this article can be found online at http://dx.doi.org/10.1016/j.gecco.2014.11.002.

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