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Abstract

Understanding how systemic biases influence local ecological communities is essential for developing just and equitable environmental practices that prioritise both human and wildlife wellbeing. With over 270 million U.S. residents inhabiting urban areas, the socio-ecological consequences of racially-targeted zoning, such as redlining, need to be considered in urban planning. There is a growing body of literature documenting the relationships between redlining and the inequitable distribution of environmental harms and goods, green space cover, and pollutant exposure. However, it remains unknown whether historical redlining impacts the distribution of urban noise or whether inequitable noise drives an ecological change in urban environments. We conducted a spatial analysis of how urban noise corresponds with the distribution of redlining categories and a systematic literature review to summarize the effects of noise on wildlife in urban landscapes. We found strong evidence that noise is inequitably distributed in redlined urban communities across the United States, and that inequitable noise may drive complex biological responses across diverse urban wildlife, reinforcing the interrelatedness of socio-ecological outcomes. These findings lay a foundation for future research that advances relationships between acoustic and urban ecology through centering equity and challenging systems of oppression in wildlife studies.

Introduction

With approximately 270 million people in the United States residing in urban and suburban areas [1], humans continually shape and are themselves shaped by urban environments. Systemic biases influencing institutional policies and urban planning practices can directly impact biological patterns and processes [2–6]. One such systemic bias, residential segregation, involves the spatial separation of different demographic groups within a city [7]. Residential segregation stems from complex political and socio-economic factors, including uneven industrial development, discrimination in real estate lending, urban zoning, inequitable government financing, and workplace discrimination [8–11]. Notably, redlining is a prominent form of residential segregation in the United States, and its legacy effects, including environmental injustices, continue to impact the health and well-being of marginalized communities to this day (Box 1).

Despite extensive research on the impacts of environmental injustices on human health, our understanding of how these social inequalities affect ecological communities is just emerging [6]. Disparities in tree and green space cover, environmental pollutants, impervious surfaces, and urban heat islands along redlining gradients may explain ecological patterns and wildlife behavior [6]. These predictions are based on the 'luxury effect,' or the association between higher socio-economic status and increased urban vegetation, which may influence animal diversity patterns [12–16]. However, support for the luxury effect has been inconsistent [15], as inequalities in biodiversity distribution across socio-economic gradients can be influenced by city spatial structure, population density, social policies, climate conditions, and human preferences [17]. For instance, highly developed urban centers may have limited space for vegetation, while less developed peripheral residential areas may support greater biodiversity, with larger yards providing resources for wildlife [17–19]. Urban population density may also be a stronger predictor of biodiversity than socio-economic status in many cases [17]. Additionally, positive relationships between biodiversity and socio-economic status might be more prevalent in arid regions due to the higher costs of planting and irrigating trees and parks [17].

Noise pollution is neglected in the urban socio-ecological literature, including research on the luxury effect, despite its well-documented human health impacts [20]. Noise from industry and transportation networks propagates over significant distances, and its prevalence is predicted to increase [21]. In the United States, traffic volume has grown 8% in the past decade (2010-2021), surpassing population growth [22]. Nearly 90% of the contiguous United States experiences noise above 30 decibels (dB), with over 100 million Americans exposed to levels exceeding 70 dB, which can cause severe health effects [20]. Even 50 dB of noise can contribute to negative health impacts [23]. These effects are compounded for communities of color and low-income groups due to segregation mechanisms exposing them to greater noise pollution compared to wealthier, predominantly white communities [24–26], with potential cascading consequences for human health [27]. However, research on inequitable noise has not explicitly assessed how racially-targeted zoning policies, like redlining, affect noise distribution and its connection to urban ecology. Here, we focus on inequitable noise distribution mapped onto redlined areas to further demonstrate these impacts on both humans and wildlife.

Noise not only significantly affects human health but also has extensive and varied impacts on wildlife. Hundreds of studies since 1990 have documented responses in terrestrial and aquatic taxa to noise, with effects becoming more severe as noise levels increase from ~40 dB to over 100 dB [21]. These impacts include changes in animal communication, movement, foraging behaviors, distributions, community structure, and predator-prey interactions. Noise exposure can also lead to adverse physiological effects on reproduction and stress [21,28,29], affecting individual fitness, energy budgets, predation risk, and vital sound cues [28–31]. Moreover, noise affects most species studied [32], making it a pervasive driver of ecological change.

Elevated noise may represent an invisible, but important factor shaping the distribution and behavior of urban animals [21]. Elevated noise levels in redlined neighborhoods may interact with reduced green space cover and other environmental inequities to further diminish urban biodiversity and habitat suitability, affecting ecological processes and potentially impacting both humans and wildlife. However, our current understanding of the distribution of noise across redlining grades and the potential impacts of heightened noise on urban wildlife remains incomplete. To address these knowledge gaps, we first quantified the distribution of noise across HOLC redlining grades. We focused on redlining instead of other measures of residential segregation because redlining is explicitly racially-targeted and outcomes of redlining can be directly linked to racial bias. Additionally, historical redlining patterns are strongly associated with present-day racial segregation and income inequality [33]. Next, we conducted a literature review to assess the current state of knowledge regarding the effects of noise pollution on urban wildlife, with particular emphasis on how these impacts change as noise levels escalate. Lastly, we synthesized potential impacts of inequitable noise on urban wildlife using the results from both the redlining noise analysis and the literature review.

Results

Noise distribution across HOLC redlining grades

We summarized noise levels across redlining HOLC grades (Box 1) in 83 U.S. cities using estimates of daily (averaged over 24-hours) exposure to transportation noise [34]. The mean excess noise level (N mean; an area-weighted metric of average transportation noise energy in A-weighted decibels >35 dBA averaged over a 24-hour period per 30m x 30m pixel; see Methods for calculation) across all cities ranged from 0.17-69.9 dBA/pixel with a mean of 20.6 ± 0.8 dBA/pixel. The mean noise level ranged from 47.4-58.5 dBA and had an average of 52.3 ± 0.09 dBA, without accounting for area. The best-fitted linear model found that both HOLC grade and city population size significantly predict excess noise in an area (top model has 100% of model weight and ΔAICc for the top three models was 0.00, 29.83, and 52.20). However, HOLC grades had much greater effects on excess noise (R^2 = 0.24, see Supplementary Table 1 for full model output). These results indicate that historic HOLC grades are predictive of area-corrected excess urban noise throughout the United States, being more correlated to noise than city area and population size. Excess noise was lower in grade A neighborhoods compared to all other grades (p <

0.001), and in grade B neighborhoods compared to grade D neighborhoods (Fig. 1A). Grade C and D neighborhoods had the highest maximum noise overall (maximum noise values averaged across all cities per each HOLC grade: A = 77.3 dBA; B = 83.9 dBA; C = 88.0 dBA; and D = 89.8 dBA; Fig. 1B). Notably, grade D neighborhoods experienced 17% higher maximum noise levels (12.5 dBA - a more than 10-fold increase in sound pressure level) than grade A neighborhoods (Fig. 1B). The distribution of maximum noise levels also varied by HOLC grade, where cities more frequently had maximum noise levels below the 70 dBA EPA upper noise limit in grade A neighborhoods and nearly all cities had noise levels above the 70 dBA limit in grade D neighborhoods (Fig. 1C). Cities also more frequently had maximum noise levels in grade C and D neighborhoods above the 90 dBA level (known to cause hearing loss, physical pain, and stress in humans [35,36], relative to A and B neighborhoods (Fig. 1C). These trends are also represented in spatial maps of noise distribution across cities, where C and D grade neighborhoods, relative to A and B grade neighborhoods, have a larger area covered by noise emitted from transportation networks on average over a 24-hr period, especially noise levels over 100 dBA (Fig. 2).

Impacts of noise to urban wildlife

We only included noise level data in the literature review when the original authors found significant biological responses to urban noise (Supplementary Fig. 1). We henceforth refer to any statistically significant response to urban noise in these original studies as "responses". Urban noise studies are geographically biased, with most research conducted in North America and Europe compared to the Global South (Supplementary Fig. 2). Urban noise research also shows biases in the biological responses, noise categories and sources, and taxa studied. Vocal behavior was the most studied biological response to noise, followed by population-level and physiological responses (Fig. 3A). Birds were most frequently studied (84% of papers), but recent years have seen an increase in studies on other taxa (Fig. 3B). Environmental and transportation noise, with most acoustic energy in frequencies < 2 kHz, were the most studied noise categories, with most studies examining effects on vocal behavior (Fig. 3C). Only four aquatic studies were identified in our review, indicating a strong terrestrial bias in urban environmental research. Thus, we restricted our analysis to terrestrial studies.

Across the studies reviewed, wildlife was affected by average maximum, mean, and minimum noise levels ranging from 45-113 dB, 32-112 dB, and 23-86 dB, respectively (Fig. 4A). Only 15% of the studies documented a biological response below a mean noise level of 50 dB, which is approximately the average noise level found across all HOLC grades (Fig. 1B). However, the percentage of studies reporting a response increased rapidly until noise levels exceeded 90 dB (Fig. 4A). Redlined communities experience greater spatial coverage of noise levels at 90 dB and above (Fig. 1 and Fig. 2), at which 95% of studies found a biological response. Thus, as noise increases to the maximum levels more commonly found in redlined neighborhoods, the cumulative evidence of biological effects on wildlife is likely to increase as well.

Urban noise exposure resulted in diverse biological responses across multiple taxa and trophic levels, encompassing changes at the ecosystem, population, and species levels (Fig. 4B, 4C, and 4D). Noise levels ranging from 23-113 dB were associated with effects on animal physiology, fitness, and behaviors such as vocalization, vigilance, movement, mating, and foraging (Fig. 4B). Even noise levels below 50 dB, the EPA's recommended threshold for avoiding harm in humans [23], still triggered changes in vocalization, population metrics, physiology, fitness, and ecosystem metrics (Fig. 4B). Bird studies showed that noise levels from 23 to 93 dB led to changes in abundance, species richness, community composition, physiology, reproduction, mating behaviors, vocalization characteristics, vigilance, and foraging behaviors (Fig. 4B and 4D). For terrestrial mammals, changes in abundance and behaviors like vocalization, vigilance, and foraging occurred at noise levels between 38 and 80 dB (Fig. 4B and 4D). In herpetofauna, noise levels between 37 and 78 dB impacted abundance, vocalization, movement, and mating behaviors, while noise levels from 68 to 116 dB influenced reproduction, vocalization, and vigilance behaviors of terrestrial invertebrates (Fig. 4B and 4D).

Discussion

This study aimed to advance knowledge in urban ecology with a justice-centered approach, explicitly considering how racially-targeted zoning practices, like redlining, shape noise distribution and potential impacts on wildlife. We found 1) strong evidence that noise is inequitably distributed across HOLC redlining grades in 83 U.S. cities; 2) that environmental, transportation, and industrial noise drive shifts in diverse biological responses (including population- and ecosystem-level, physiological, fitness, and behavioral responses) across a broad range of urban taxa; and 3) that the cumulative evidence of biological effects on urban wildlife increases as noise exposure rises until reaching levels over 100 dBA. Below, we discuss these findings and their importance to urban and acoustic ecology and highlight a list of key future questions that integrate noise pollution, wildlife, and social inequity to advance knowledge relevant to urban conservation practitioners and planners.

Noise distribution across HOLC redlining grades

Our analysis provides clear evidence that noise is not equitably distributed in U.S. cities with historical redlined communities. Our model shows that redlining plays a stronger role in predicting noise pollution than other factors like population size. Grade D neighborhoods experienced a greater spatial extent of excess noise and over a 10-fold increase in sound pressure level than grade A neighborhoods (Fig. 1A, 1B, and Fig. 2). While not explicitly focused on redlining, other research similarly found that noise exposure is greater in residentially segregated neighborhoods [24], and in neighborhoods of lower socio-economic status and/or higher percentages of racial and ethnic minorities in the United States [25,26,37,38], South Africa [39], China [40], Canada [41], the United Kingdom [42], and Germany [43]. Our study expands these findings by providing evidence explicitly linking noise exposure to the racially-targeted urban practice of redlining.

These inequities have direct consequences for humans — on average an increase of 10 dB of noise above background sound levels equates to elevated human health risks and a 90% decrease in listening ability [44]. Moreover, grade A neighborhoods in our study experienced average maximum noise levels closer to the U.S. Environmental Protection Agency's recommended upper limit for annual average noise exposure at 70 dB (Fig. 1B and 1C), the baseline level at which damaging health effects emerge [23]. In contrast, redlined neighborhoods more frequently experienced average maximum noise levels above 90 dB, with greater coverage of maximum values up to 120 dB (Fig. 1B, 1C, and Fig. 2) — equivalent to the sound experienced when standing next to a chainsaw and above the human pain threshold [45]. Noise levels between 90 and 120 dB can cause damage to hearing, hearing loss, physical pain, and psychophysiological stress in humans [35,36]. Notably, these maximum noise levels represent the highest noise levels found in a HOLC grade averaged over a 24-hr period, indicating that some sections of grade D neighborhoods are experiencing noise levels that are both severe (over 90 dB) and chronic (consistent over a 24-hr period). Consequently, it is not surprising that mounting evidence is linking residential segregation, noise pollution, and human health disparities [27].

Emerging research suggests that there is a strong correlation between urban systemic racism and environmental health, and understanding these interconnected processes is an urgent priority for urban conservation [6]. However, existing studies addressing this issue have primarily focused on other types of environmental injustice, such as inequitable air and water pollution and disparities in green space coverage [6]. Our finding that noise pollution is also related to systemic racism can inform urban planning.

If noise is an important unseen factor shaping urban environments, then urban planning projects failing to take noise into account while addressing other environmental injustices in historically redlined communities may fall short of realizing their full beneficial potential. Our findings establish new research avenues to determine how inequitable noise pollution interacts with other forms of environmental injustice to exacerbate their impacts on marginalized communities and their wildlife neighbors. For example, noise, air, and light pollution often co-occur because they are all emitted from transportation and industrial sources. Yet, these forms of pollution are often only moderately correlated [46,47], suggesting that noise impacts may extend beyond the footprint of other forms of environmental injustice.

Impacts of noise to urban wildlife

Our literature review reveals the extensive impact of urban noise exposure across various species, behaviors, demography, and environments. Urban transportation, environmental, and industrial noise affect animal physiology, fitness, and behaviors across trophic levels and taxonomic groups (Fig. 4B, 4C, and 4D). Noise levels as low as 23 dB have been shown to affect wildlife, and the cumulative effects of noise intensify with higher noise levels (Fig. 4A). More than 95% of studies (encompassing multiple taxonomic groups; Fig. 4D) observed a biological response at 90 dB, a noise level commonly observed in redlined communities. Similar results were found in a review by Shannon et al. (2016)[21]. The consistent evidence suggests that as noise levels rise, the biological impacts on wildlife become more widespread. Consequently, higher noise levels in redlined neighborhoods may lead to substantially greater biological effects as more species respond with a broader range of shifts at such levels.

Our literature review revealed a bias in noise impact studies on urban wildlife, with a focus on birds due to their ease of observation. Invertebrates were the least studied taxa. This bias towards birds may be beneficial since bird diversity sometimes reflects overall biodiversity in ecosystems [48], serving as an index of urban biodiversity at the community (abundance, richness, composition) and individual levels (behavior, physiology, reproduction) [49]. Furthermore, birdwatching in urban settings is associated with increased human wellbeing and a stronger connection to nature [50]. Thus, bird diversity can also indicate human engagement with the natural world.

Vocal behavior and population-level effects received most attention (Fig. 3A and 3C), which is consistent with other reviews [21,51,52]. A strong geographic bias was also evident, with 74% of studies in North

America and Europe alone (Supplementary Fig. 2). Other reviews have similarly found that the Global South is underrepresented in research on noise impacts to wildlife [52], partially due to disparities in research and funding [53]. Scientists from wealthy countries in the Global North commonly conduct research in the Global South without effectively engaging local communities, disconnecting local peoples from leading environmental initiatives and granting authority over conservation outcomes to Global North institutions [53]. Increasing locally led research in the Global South is crucial as it supports greater species richness and diversity [54,55] and is undergoing rapid urbanization [56]. With more of the global population living in urban areas [56], planning for sustainable and healthy cities becomes imperative. To understand the impact of inequitable noise on unique species, research must broaden to encompass various biological responses, geographic locations, and taxa [21], especially understudied aquatic urban species.

The majority of responses to elevated noise exposure involved reduced biodiversity and altered acoustic diversity. Seventy-two percent of population-level studies reported a decrease in wildlife abundance or occurrence, while 93% of vocalization studies noted changes in vocal behavior. Urban species, especially birds, heavily rely on acoustic communication for mate attraction, territory defense, and signaling dangers [57]. However, urban noise often masks these vital signals, particularly at lower frequencies [58,59], leading to various adjustments in vocal behavior, such as shifting song frequencies, increasing vocal amplitudes [60–62], or altering timing or complexity of vocalizations [63,64]. Noise exposure can also be perceived as a direct threat, which along with acoustical masking and distraction from other environmental stimuli, has been shown to increase physiological stress and alter foraging, vigilance, and reproductive behavior across all ontological stages [21,52]. These adjustments in behavior and physiology are likely to have considerable long-term fitness consequences that can scale up to the population-level [65,66]. In cases where species cannot adapt their acoustic signals or behavior to cope with chronic noise exposure, they may alter their movement or habitat use to avoid noisy areas [67,68], potentially leading to profound changes in species composition and interactions at the community level [69], and affecting ecological processes like predation [70], pollination [71], and seed dispersal [72].

Synthesizing socio-ecological Impacts

The finding that higher noise is linked to reduced biodiversity and altered acoustic diversity has significant implications for urban wildlife, humans, and equitable urban planning. In redlined neighborhoods, human residents face disproportionate challenges in accessing parks and green spaces

[5,73]. Even when green spaces are present, these neighborhoods may still experience lower animal biodiversity and degraded natural soundscapes. Thus, residents of redlined neighborhoods likely suffer both direct health impacts from inequitable noise [20] and indirect impacts from reduced access to nature. This decreased access to nature and natural sounds is of concern because direct or perceived exposure to natural sounds and biodiverse greenspaces has been shown to improve human health by reducing stress, anxiety, and depression, by enhancing mood, cognitive performance, psychological well-being, and improving immune system response to transmissible diseases [74–80]. The combination of increased exposure to diverse natural sounds and reduced anthropogenic noise can amplify these restorative impacts [81], potentially leading to increased biodiversity in urban greenspaces and improved physical and psychological benefits for both wildlife and humans, creating positive feedbacks between humans and biodiversity [82]. Moreover, reduced biodiversity often correlates with diminished ecosystem function, resilience, and services for humans [83,84]. The unequal distribution of noise in urban landscapes might also hinder conservation funding and opportunities for people in redlined communities, as conservation efforts often focus on areas with high biodiversity [85]. These interconnected issues emphasize the crucial need to address urban inequities for the well-being of both people and wildlife. Research is particularly needed to uncover mechanisms underlying the reciprocal relationship between enhanced soundscape quality, increased biodiversity, and the multifaceted well-being benefits for both urban residents and local wildlife populations [82]. Such work should also aim to evaluate the social equity aspects of these interactions and how they impact different demographic groups.

Numerous cities in the United States are currently addressing environmental justice issues by increasing access to parks and green spaces for underserved populations [86], many of which are within historical redlining boundaries [33]. For instance, Denver voters approved a 0.25% sales tax increase to advance Denver's Game Plan for a Healthy City [87], aiming to provide equitable park access by identifying neighborhoods in greatest need of new or improved parks and green infrastructure. Similarly, cities like Pittsburgh, New York, San Francisco, Philadelphia, Detroit, and Minneapolis are using sociodemographic data to develop plans for increasing equitable park access [86]. Importantly, many of these initiatives are being developed with direct input from local residents and neighborhood organizations, with a goal of using affordable housing agreements or other tools to avoid green gentrification (i.e., the process in which improving green infrastructure increases local property values, displacing lower-income residents [88]).

However, it is crucial to recognize that merely adding or improving green infrastructure without addressing noise may still result in limited biodiversity and fewer opportunities for residents to experience the benefits of natural soundscapes. Therefore, equitable urban planning projects should include noise mitigation to ensure that both wildlife and people can enjoy the benefits of additional green infrastructure without the negative impacts of noise. Mitigation measures may involve adding physical barriers to limit noise from industrial and construction zones, establishing specific tree lines and border vegetation to reduce noise transmission, implementing traffic speed reductions near green spaces, and employing technological improvements to reduce noise emitted from tires and road surfaces [21].

Our review focuses on noise impacts in the urban environment, reinforcing connections between social inequities and wildlife outcomes. While the effects of urban noise on people are somewhat understood, our review highlights significant gaps in understanding on how noise influences urban wildlife. Addressing these gaps will enhance our understanding of complex urban socio-ecological systems. Here, we use our findings to outline outstanding questions that can address some key knowledge gaps on the impacts of inequitable noise for urban wildlife, people, and human-wildlife interactions (Box 2).

Conclusion

Our study found evidence that noise is inequitably distributed in U.S. cities, and that inequitable noise may drive complex biological responses across a diversity of urban wildlife. This knowledge draws attention to the often-overlooked role of inequitable noise pollution in shaping patterns of urban biodiversity, underscoring the need for further research at the intersection of noise, environmental justice, and ecology. Urban ecologists, acoustic ecologists, social scientists, and urban planners can leverage this knowledge to better understand how social processes, like redlining, can influence ecological properties, leading to implications for human-wildlife interactions. Urban ecologists are being called to reimagine a more socially just vision of conservation science and practice that centers racial and environmental justice to drive holistic and equitable policy changes in cities [6]. Here, we lay the groundwork for future research that advances acoustic and urban ecology by centering equity and challenging systems of oppression that remain embedded in our city infrastructure.

Methods

Spatial Analysis of Urban Noise Pollution

We conducted a spatial analysis of the distribution of noise pollution across HOLC grades for 83 U.S cities (Supplementary Table 2). To be included in the study, the city needed to feature in both datasets used in the analysis: 1) the Mapping Inequality Project dataset on the distribution of HOLC grades across cities [89], and 2) the National Transportation Noise Map 2018 [34]. Any cities in which the distribution of HOLC grades did not include all four grades (A- D) were excluded from the analysis, which largely excluded cities with population sizes below 100,000 people.

To evaluate noise exposure across HOLC grades for each city, we acquired spatial data on the distribution of HOLC grades across U.S. cities from the Mapping Inequality Project [89]. We also acquired data on road, rail, and aircraft noise (hereafter transportation noise models), from the National Transportation Noise Map [34] which has been used by other investigators to assess noise exposure in the United States [25,90]. The transportation noise models represent potential exposure to transportation noise reported on a decibel scale in a 30m x 30m pixel resolution. Here noise represents the average noise energy produced by road, rail, and aviation networks over a 24-hour period, measured in A-weighted decibels (dBA) (LAeq, 24h) at sampling locations deployed across a uniform grid in each city at an elevation of 1.5 m above ground level. Noise levels below 35 dBA are assumed to have minimal negative impacts to humans and the environment and thus are represented with null values in the transportation noise models.

For each HOLC grade and each city, we used zonal statistics in ArcGIS Desktop v. 10.7 to summarize the median noise levels and area covered by excess noise (i.e., values > 35 dBA). We used the resulting zonal statistics estimates and the formula from Collins et al. (2019)[25] to calculate an area-corrected measure of excess noise:

$$N = (r * Md)/a$$

where N is excess noise in each HOLC grade (with units of dBA/30m x 30m pixel); r is the area covered by the 30m x 30m pixels with noise values >35 dBA across all polygons of the same HOLC grade in each city; Md is the median transportation noise value (in dBA) for those same pixels; and a is the total area of all polygons of the same HOLC grade in each city. Thus, N represents a measure of both the level of noise and the area covered by excess noise in a given HOLC grade for each city. We used the N measure of excess noise as the dependent variable in the regression model of excess noise across cities described in the Statistical Analysis section. We also produced maps of excess noise and the distribution of HOLC grades for all cities included in our study.

Statistical Analysis

We built linear regression models using standard least squares to examine the relationship between noise exceedance (N), HOLC grade, and city population size. Prior to analyses, we explored our data to assure that the assumptions of this test were met. We constructed four separate models, each with noise exceedance as the response variable and one of the following as predictive variables: HOLC grade only, city population size only, HOLC grade + city population size, and the interaction of HOLC grade and city population size. We did not incorporate city area because the exceedance value N is an area-corrected metric. Following each model, we plotted residuals against the fitted values to determine if there was non-constant error variance. As our N-mean variable displayed non-normality, we performed a log transformation on the variable and reanalyzed models using the log-transformed data. Statistical analyses were done in R version 4.2.1103 [91]. We considered models with the highest R^2 and the lowest AICc as the best predictors of noise.

Literature Review on the Impacts of Noise to Urban Wildlife

To assess the effects of noise on urban wildlife we conducted a literature review (Supplementary Fig. 1) using Thompson's ISI Web of Science and adapting the methods of Shannon et al. (2016)[21]. We adjusted of Shannon et al.'s search criteria to include urban phrases, resulting in the following search terms (TS=(WILDLIFE OR ANIMAL OR MAMMAL OR REPTILE OR AMPHIBIAN OR BIRD OR FISH OR INVERTEBRATE) AND TS=(NOISE OR SONAR) AND TS=(CITY OR *URBAN OR METROPOLITAN)). We selected papers published between 1990 and 23 June 2021 (i.e., the date we conducted our search) within the ISI Web of Science categories of 'Acoustics', 'Zoology', 'Ecology', 'Environmental Sciences', 'Ornithology', 'Biodiversity Conservation', 'Evolutionary Biology', and 'Marine Freshwater Biology'. This returned 691 peer-reviewed papers, which we filtered so only empirical studies focused on documenting the effects of anthropogenic noise on wildlife in urban or suburban ecosystems or the effects of urban noise on wildlife in rural environments were included in the final data set (n = 207). We excluded reviews, meta-analyses, methods papers, and research that took place outside of urban or suburban areas where the noise was not explicitly denoted as urban (e.g., omitted studies that measured traffic noise by parks and reserves in rural areas).

For the 241 articles previously analyzed in Shannon et al. (2016)[21], one of our authors reviewed each paper to determine which studies were focused on urban noise (n = 46). We then verified whether there were significant biological responses to a particular noise level threshold, noting each noise level if multiple biological responses were recorded. We recorded responses to noise into one of eight possible biological response categories, many of which were taken or modified from the biological response categories utilized in Shannon et al. (2016)[21]. The following were the biological response categorical values: movement behavior, vocal behavior, physiological, population, mating behavior, foraging behavior, vigilance behavior, life history / reproduction, and ecosystem. Further definitions and descriptions for each biological response category may be found in the supplemental information (Supplementary Table 3). For any new articles published since the Shannon et al. (2016)[21] dataset (n = 354) or those published between 1990 and 2013 but not reviewed by Shannon et al. (n = 96), two of our authors reviewed each paper to first determine which studies met our criteria (n = 161) and then compiled data on a number of variables of interest, including the noise levels and their resulting biological responses that were statistically significant (Supplementary Table 3). For this subset of papers, one author was randomly assigned a list of papers and then a second author was randomly assigned to assess the accuracy of the data collected by the first author. Any discrepancies were discussed as a group until an agreement was reached.

Noise categories (environmental, transportation, industrial, multiple, other) were chosen for each paper by noting the explicitly stated source or description of urban noise in the methodology. Noise levels and their units were reported for each paper, with only noise levels reported in decibels (dB) being used in data analysis. All terrestrial papers used a reference pressure of 20 microPascals (µPa). Due to the low sample size of aquatic studies (n = 4), differences in reference pressures, and varying sound intensities amongst aquatic studies, we only included terrestrial studies in statistical analyses and figures. We recorded the sound metric used (i.e., SPL, SPL Max, Leq) for each paper, but were unable to convert the various sound metrics given to a single sound metric for standardization during analysis. Thus, there were various sound metrics used in the analysis of the data extracted from the literature search, in particular for the cumulative weight-of-evidence curve, which poses a limitation in the comparison of noise levels amongst papers. Additionally, we recorded the weightings for each noise level, with many of the papers being A-weighted (dBA; n = 100) and Z-weighted (dBZ; n = 4). These weightings relate to typical characteristics of sounds as observed by humans. Many papers, however, did not record the weighting and/or the exact sound metric used, leading to some unavoidable uncertainty in the comparison of noise measurements. We used the extracted noise levels to develop a cumulative weight-of-evidence curve as a function of the noise level at which a biological response was documented. This curve summarizes the cumulative percent of studies that reported a biological response at or below a given noise level across a wide range of taxa, biological responses, and acoustic metrics, with some taxa, responses, and metrics being more represented than others.

Study Scope

While our results highlight important consequences of inequitable noise for wildlife and humans, there are certain limitations to the noise analysis and the scope of our literature review that should be considered when interpreting our results. Our noise analysis used a model of transportation noise that did not include other major noise sources in cities (e.g., construction noise, generators, humans) and thus our analysis does not fully capture the diversity of noise in urban soundscapes. Moreover, the noise model represented the average noise energy produced over a 24-hour period, and likely underestimated extreme values associated with diurnal patterns of noise in cities (e.g., more frequent road and aircraft noise in the day).

Further limitations that warrant consideration include the variation in noise metrics, study designs, and geographic and sampling biases represented across the studies included in our review. First, a variety of acoustic metrics with different frequency weighting and bandwidths were synthesized together in our review and analysis (Fig. 4), as we were unable to adjust all values to a common acoustic metric that could be compared across studies (a lack of accurate reporting of acoustic metrics is a key concern noted by McKenna et al. 2016[92]). As a result, we have avoided making comparisons of how noise levels differentially affected taxonomic groups, trophic levels, or biological responses because researchers may have explored different noise levels for different groups, and thus any inter-group differences may be related to study design rather than noise levels. Given our urban focus, people and animals likely are exposed to chronic low frequency noise [93], suggesting that our findings can be more directly compared. However, the variation in metrics used across studies warrants caution in making such direct comparisons. We also caution against using our findings to conclude that low-decibel urban noise has no effect on wildlife. Although the cumulative effects on wildlife increase with noise, animals may still respond to very low noise levels [21], and the lack of evidence of effects at lower noise levels may be partially driven by biases in study design, with fewer researchers choosing to study low noise exposure levels. Similarly, redlined neighborhoods are underrepresented in citizen science projects that are used to study urban ecology [94], which likely explains why noise levels above 100 dB - more common to redlined neighborhoods - are not well represented in the urban acoustic ecology literature that we reviewed (Fig. 4A). Thus, our findings likely underestimate the full impact of inequitable noise on urban wildlife and future research should prioritize evaluating noise impacts to wildlife at levels of 100 dB and above. Our use of Web of Science for the literature review also likely missed relevant publications in the non-peer-reviewed gray literature and government reports [95], which likely contributed to the lack of publications in our review from the Global South.

Data availability

All analyzed data are available on Dryad licensed under a CC0 license, which allows future users to distribute, remix, adapt, and build upon the material in any medium or format, with no conditions. Data can be accessed at: https://doi.org/doi:10.5061/dryad.s4mw6m998 [96]

Code availability

All analyzed code is publicly available on Zenodo at: https://doi.org/10.5281/zenodo.7843664 [97]

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Author Contributions

T.M.L. and S.P.B. conceived the initial study. J.R.N.-O., T.J.L., E.A., A.L., M.L., T.M.L., K.A.S., S.S., A.K.V., and S.P.B. collected review data, participated in data analyses, and wrote the initial draft of the article. K.A.S. and S.P.B. conducted spatial analyses and modeling approaches. All authors contributed to revisions.

Competing Interest Statement

The authors declare that they have no known competing interests.

Figure Captions

Fig. 1: Noise pollution levels across the four Home Owners' Loan Corporation (HOLC) redlining grades for 83 U.S. cities (N). Panel A) depicts mean noise exceedance levels (N mean; an area-weighted measure of noise > 35 dBA) for each HOLC grade. Significant differences among group means at the alpha level = 0.01 are illustrated using a * following a Tukey post-hoc test. Boxes encompass the interquartile range and notches represent the 95% confidence interval of the median (central lines). Panel B) depicts the distribution of mean and maximum noise levels (dBA) for each HOLC grade, across 83 cities (not corrected for the area coverage of excess noise). Minimum noise levels, not shown, were similar for all HOLC grades due to the 35 dBA lower limit in the dataset. Boxplots indicate median (center line) 25th and 75th percentile (box) and min and max values (whiskers). Panel C) depicts the frequency distribution of different noise levels across the four HOLC grades and 83 U.S. cities.

Fig. 2: Home Owners' Loan Corporation (HOLC) redlining grades (A-D) and noise pollution levels (dBA) across four U.S. urban areas. Depicted regions include a) the Bay Area, CA (San Francisco and Oakland), b) Dallas, TX, c) Louisville, KY, and d) Philadelphia, PA. Noise pollution data was derived from the U.S. Department of Transportation Rail, Road, and Aviation Noise 2018 dataset (USDOT 2020). Here noise represents the average noise energy produced by road, rail, and aviation networks over a 24-hour period, measured in A-weighted decibels (dBA) (LAeq, 24h). HOLC redlining data derived from the Mapping Inequality Project dataset (Nelson et al. 2021).

Fig. 3: Scope of literature review findings. Shown are the number of papers documenting a) different types of biological responses of wildlife to urban noise, b) the biological responses of different taxonomic groups to urban noise from 2003-2021, and c) the number of studies that analyzed noise effects on different types of biological responses, symbolized by the type of noise studied (environmental, transportation, industrial, multiple, or other noises).

Fig. 4: Scope of the effects of noise on urban wildlife (N = 304, total studies). Panel A depicts the cumulative percentage of terrestrial studies demonstrating biological responses (the point at which there was a significant response from the species of interest) at a particular noise level (dB). The minimum, mean, and maximum noise levels of biological responses are plotted independently. Also shown are boxplots representing distributions of the average minimum, mean, and maximum noise

values at which significant biological responses were found across B) biological responses, C) trophic levels, and D) taxa. All boxplots indicate the median noise level (center line), with the box bounding the interquartile range between the 25th and 75th percentile, and the min and max values (whiskers).

Box 1. Background information on the history of redlining and its legacy effects in U.S. cities.

The practice of redlining is one of the most explicit forms of residential segregation in the United States. From 1933 to 1968, the Home Owner's Loan Corporation (HOLC) assigned neighborhoods in U.S. cities into ordinal grades from A to D, largely based on race. Neighborhoods with populations considered "non-white" at the time, such as people of color and immigrants, were designated C and D grades which received limited investment from governments and banks, leading to reduced opportunities to obtain loans [98]. These communities were also excluded from purchasing homes in predominantly white A and B grade neighborhoods [98].

The legacy effects of these Home Owners' Loan Corporation (HOLC) grades and other mechanisms of residential segregation continue to persist today, resulting in landscape-level trends and pervasive environmental inequities. Segregated communities experience the highest incidences of urban poverty in the United States and are disproportionately affected by environmental injustices [99–104]. Redlined neighborhoods have higher exposure to environmental hazards like air pollution, toxic waste, and flood risks, while facing reduced access to environmental necessities such as parks and tree cover [5,73,105]. Reduced greenspace access compounds the issues related to increased pollutant exposure, as greenspaces play a vital role in ecosystem services, like air pollution removal, carbon sequestration, and heat island reduction [106–108].

Historical and present-day city policies contribute to these observed patterns. We focus on redlining as a historical practice, but other practices like formal segregation, intentional hazardous pollutant placement, weak regulatory enforcement in marginalized communities, and limited opportunities for communities of color in decision-making [10,73,109,110] also exacerbate urban planning inequities, leading to increased toxin exposure and health risks, including asthma, cancer, and higher mortality rates [105,111–117].

Box 2. Knowledge gaps on the impacts of inequitable noise on urban wildlife and people.

1) How does noise pollution or the presence of natural sounds interact with tree cover, building density, and other environmental gradients that are inequitably distributed across cities to alter wildlife distributions and population connectivity?
2) How does inequitable noise and inequitable natural soundscape exposure affect human health and well-being? Is legislation effectively tackling the health impacts of noise in urban environments?

3) How does exposure to inequitable noise pollution affect community perceptions of wildlife and human-wildlife relationships? How might these perceptions affect urban wildlife management and conservation priorities?
4) What mitigation techniques, such as noise barriers or green walls, and infrastructure improvements (e.g., building spatial orientation and green space) yield the most benefits for urban wildlife and people in areas of higher noise pollution?
5) Do elevated noise levels drive shifts in acoustic traits of urban animal populations? How might these shifts in traits vary spatially and temporally (rate of change), and how might these drive evolutionary outcomes and fitness consequences?
6) Do established hypotheses (e.g., acoustic adaptation hypothesis, Lombard effect, luxury effect hypothesis) accurately predict the sensitivity or tolerance potential of urban species in light of inequitable noise? Does inequitable noise contribute to or exacerbate these hypotheses?
7) Does inequitable noise have cascading consequences for ecosystem function, ecosystem resilience, and the ecological services provided to humans in urban environments?

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