Policy analysis

Effects of the COVID-19 pandemic on noise pollution in three protected areas in metropolitan Boston (USA)

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ABSTRACT

Noise pollution can reduce the ability of urban protected areas to provide a refuge for people and habitat for wildlife. Amidst an unprecedented global pandemic, it is unknown if the changes in human activity have significantly impacted noise pollution in metropolitan parks. We tested the hypothesis that reduced human activity associated with the COVID-19 pandemic lockdowns would lead to reduced sound levels in protected areas compared with non-pandemic times. We measured sound levels in three urban protected areas in metropolitan Boston, MA (USA) at three time periods: in the fall and summer before the pandemic, immediately after the government-imposed lockdown in March 2020 when the trees were leafless, and during the beginning of reopening in early June 2020 when the trees had leaves. At all time periods, sound levels were highest near major roads and demonstrated a logarithmic decrease further from roads. At the two protected areas closest to the city center, sound levels averaged 1–3 dB lower during the time of the pandemic lockdown. In contrast, at the third protected area, which is transected by a major highway, sound levels were 4–6 dB higher during the time of the pandemic, likely because reduced traffic allowed vehicles to travel faster and create more noise. This study demonstrates that altered human levels of activity, in this case associated with the COVID-19 pandemic, can have major, and in some cases unexpected, effects on the levels of noise pollution in protected areas.

1. Introduction

Protected areas in cities promote ecosystem services and biodiversity, provide economic and cultural benefits to local communities, and offer an opportunity for people living in urban settings to interact with nature (Buckley et al., 2019; Kim and Jin, 2018; Secretariat of the Convention on Biological Diversity, 2008). Protected areas also provide a home to many animal and plant species for which highly developed areas lack suitable habitat (Dudley, 2008). People seek out protected areas for relaxation and recreation, and a growing body of evidence shows that human physical and mental health is enhanced through such experiences with nature (Larson et al., 2016; Schwartz et al., 2019; Secretariat of the Convention on Biological Diversity, 2008).

Urban protected areas have taken on added importance during the current COVID-19 pandemic. Due to the difficulty of practicing social distancing in urban areas like Boston, MA (USA), large numbers of people have relied on local parks when looking for a safe place for relaxation and exercise. On March 10, 2020, the government of Massachusetts declared a State of Emergency, and on March 23 the administration ordered the closure of all non-essential businesses. The administration also limited gatherings to less than 10 people and issued a stay-at-home advisory, instructing Massachusetts residents to practice social distancing, avoid non-essential activities, and stay home as much as possible (The Commonwealth of Massachusetts, 2020). As a result of these actions, referred to collectively throughout this paper as the initial lockdown, most recreational and social facilities were closed, including schools, summer camps, children's playgrounds, indoor gyms, and swimming pools. Local news organizations Boston.com and Boston CBS reported on state parks being forced to close parking lots as these areas continued to exceed safe number of visitors during the pandemic (Dwyer, 2020; "*Massachusetts DCR makes adjustments*", 2020).

It remains to be investigated if park-goers managed to find quiet, relaxing spaces in these urban protected areas during the initial lockdown. Noise pollution has become a common problem for protected areas across the country, and it is often associated with road traffic and airplanes (Anderson et al., 2011; Buxton et al., 2017; Xing and Brimblecombe, 2020). In over half of US protected areas, anthropogenic noise creates a soundscape that is at least 3 dB higher than background

sound levels, equivalent to a doubling in acoustic energy (Buxton et al., 2017). In one fifth of US protected areas, anthropogenic noise has increased background sound levels by 10 dB or more, equivalent to a ten-fold increase in acoustic energy and perceived by humans as a doubling in sound volume. Protected areas near urban centers experience particularly high levels of noise pollution (Buxton et al., 2017).

Repeated exposure to high levels of noise pollution can have a variety of negative effects on people, including hearing damage, disruption of spoken communication, and disturbance of sleep cycles, with serious long-term health impacts (Bloemsma et al., 2019; Goines and Hagler, 2007). High sound levels in parks can negatively affect the visitor experience; for visitors to national parks, enjoyment begins to decline after sound levels exceed 37 dB (Merchan et al., 2014). In many animal populations, noise pollution impacts physiology, behavior and fitness, and interferes with communication (Rabat, 2007; Salaberria and Gil, 2010; Sun and Narins, 2005; Uran et al., 2012; Warren et al., 2006). In bird populations, the effects on communication can result in increased success for some species and reductions in fitness for others, leading to a change in community structure (Francis et al., 2009; Francis et al., 2011; Derryberry et al., 2020). While it is clear that noise pollution has significant impacts on human visitors and wildlife of protected areas, sound levels can vary greatly over time and space in ways that are difficult to model or predict, particularly when large changes in road and air traffic occur. Barber et al. (2011) developed a model to predict sound levels based on road traffic, air traffic, and the activity of industrial oil and gas operations, and found that protected areas are significantly affected by road noise even at moderate traffic levels.

In general, noise pollution from large industrial sources, such as stone quarries, has decreased significantly in the months following the COVID-19 lockdown (Mandal and Pal, 2020; Derryberry et al., 2020). However, it is largely unknown how the pandemic has affected noise pollution in protected areas from local sources such as cars and trucks on roads, construction, landscaping, and human activity. Since the start of the pandemic, vehicular road traffic has been substantially reduced in many parts of the USA (Lockwood et al., 2020). According to Massachusetts Department of Transportation (MassDOT) traffic data, volume on major roads was down 67% in March, and down 30% in July; this is particularly notable considering that 2020 traffic volume prior to the lockdowns was up 3–5% from 2019 (MassDOT, 2020). This lower traffic volume might be expected to reduce the amount of noise pollution in protected areas, as traffic is a primary source of noise in these areas (Zipf et al., 2020). At the same time, the number of people using urban protected areas has been reported to be higher than normal, and increased human activity could create more noise in these parks (Brown et al., 2012).

It is also possible that the sound levels in urban protected areas have changed over the course of the pandemic as a result of natural biological cycles. At the beginning of the lockdown in March in the northern hemisphere most deciduous trees had not yet leafed out, and only gradually produced their leaves as the lockdown continued. As leaves can have a dampening effect on noise (Maleki and Hosseini, 2011), protected areas dominated by deciduous trees may have experienced lower sound levels only after leaf out in late April and May. Overall, it remains unknown if the combination of these potential sources of noise pollution and changing effects of sound dampening by leaves impacted noise pollution in urban protected areas during the initial COVID-19 pandemic lockdown.

To better understand the effects of the COVID-19 pandemic on noise pollution created by road traffic and other human activities in urban protected areas, we address three key questions:

1. During pre-pandemic times, how do sound levels in urban protected areas vary spatially and how rapidly do sound levels decline with distance from major roads?
2. Are urban protected areas quieter during the initial COVID-19 pandemic lockdown than they were in pre-pandemic times?

3. During the initial COVID-19 pandemic lockdown, are urban protected areas quieter after the trees have leafed out than before they leafed out?

This study will provide a fine-scale understanding of noise pollution in our protected areas both before and during the COVID-19 pandemic. It will also determine if our parks and protected areas have been quieter during the pandemic, and if so, by how much. These results will strengthen our general understanding of noise pollution in protected areas as well as provide novel information on the impacts of the COVID-19 pandemic on the environment.

2. Materials and methods

2.1. Research sites

Sound measurements were collected before and during the pandemic lockdown at three different Massachusetts, USA parks that vary in size and proximity to Boston's urban center: Blue Hills Reservation (42.2142° N, 71.0933° W), Hammond Pond Reservation (42.3228° N, 71.1741° W), and Hall's Pond Sanctuary (42.3459° N, 71.1120° W; Fig. 1). Detailed descriptions of the parks and noise monitoring efforts are available in Table 1. The vegetation at these three parks is predominantly deciduous forest, with an understory of deciduous shrubs and perennial herbs.

Blue Hills Reservation (hereafter "Blue Hills") is a large 2800-hectare state park located in Milton, MA, with hiking and ski trails, a recreational lake area with swimming and a beach, a weather observatory, and a nature center. The reservation is 24 km from downtown Boston and is divided by U.S. 93 (a major interstate highway), two busy state highways (Route 138 and Route 24), and other smaller roads. Sound monitoring was performed in the western half of the park (both north and south of U.S. 93), which contains the most popular hiking trails.

Hammond Pond Reservation and the adjacent Webster Woods Conservation area (hereafter "Hammond Pond") is an intermediate sized 82-hectare woodland park consisting of Massachusetts state and Newton city land, located in Newton, MA, 11 km from Boston. It has many hiking trails and a large pond, and is transected by the Hammond Pond Parkway, a moderately active four lane road, and the Riverside Green Line railroad.

Hall's Pond Sanctuary (hereafter "Hall's Pond") is a small conservation area in Brookline, MA encompassing a total area of 1.4 ha. It is 4.2 km from downtown Boston and is surrounded by apartment buildings, residential neighborhoods, and retail businesses. Beacon Street, a high-volume thoroughfare, and Amory Street are nearby but with intervening apartment buildings (in the case of Beacon Street) and 100 m of fields (in the case of Amory Street).

The most common source of noise in these areas is vehicular road traffic. Other sources include airplane traffic (especially at Blue Hills), leaf blowers and lawn mowers (especially at Hammond Pond and Hall's Pond), construction, and park visitors.

2.2. Sound level measurement

A-weighted sound levels were measured using the SPLnFFT app (Lefebvre, 2010) with iPhones 5s-8 (Apple Inc., 2013–2017). The A-weighted sound level scale measures sound energy at the frequencies to which human hearing is most sensitive (Maling, 2014). iPhone measurements taken with the SPLnFFT app give a high level of accuracy and precision, with an accuracy of −2.3 dB compared with reference levels measured by a type 1 sound level meter, indicating that SPLnFFT measures sound levels 2.3 dB lower than their actual value. The standard error around this difference is 0.25 dB, indicating that it is fairly consistent (Kardous and Shaw, 2014; Murphy and King, 2014; Nast et al., 2014). The sound levels in these urban parks range from 30 (very quiet) to 80 (very noisy) dB, so for the purposes of this study this
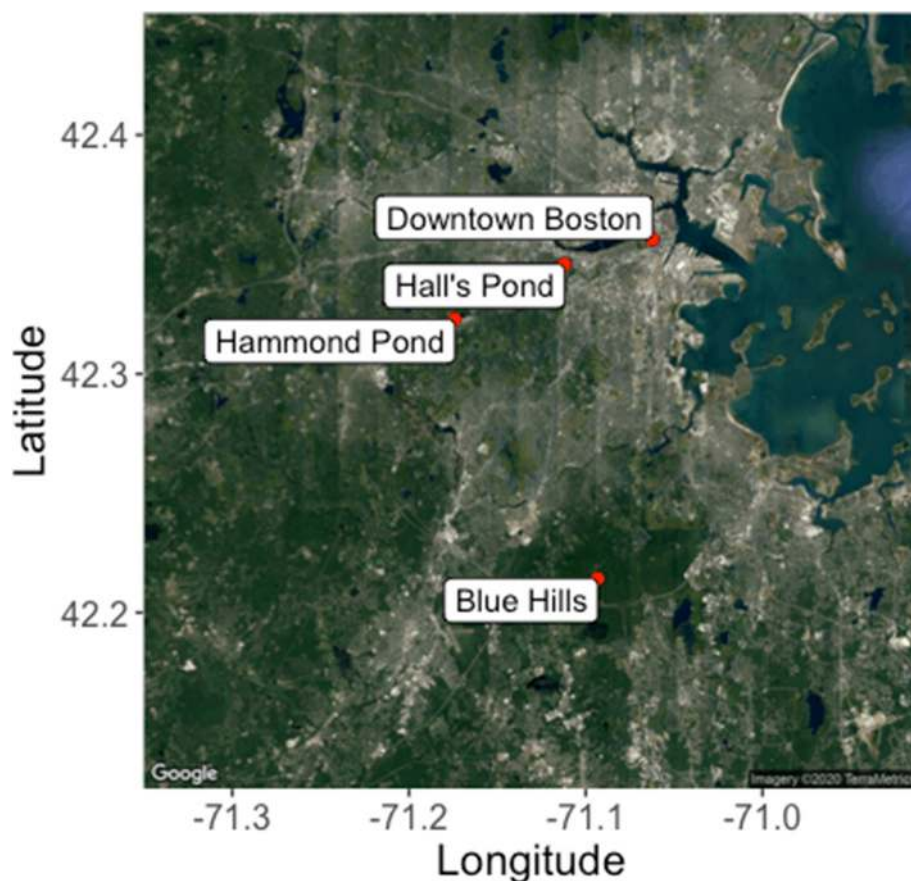
Fig. 1. The locations of the three protected areas, Blue Hills Reservation (42.2142° N, 71.0933° W), Hammond Pond Reservation (42.3228° N, 71.1741° W), and Hall's Pond Sanctuary (42.3459° N, 71.1120° W), as well as downtown Boston, MA, USA (Park Street Train Station).

Table 1
Characteristics of the three urban protected areas including size, distance from Boston's urban center (Park Street MBTA Station), roadways that border or intersect the protected area, general site characteristics, and the number of noise measurements collected during the three sampling events: before the COVID-19 lockdown (designated "pre-COVID-19"), data gathered at the beginning of lockdown in March when there were no leaves on the trees (designated "COVID-19 with non-leafed trees"), and data gathered later during lockdown when the trees had regained leaves (designated "COVID-19 with leafed trees").

Protected area	Size (ha)	Distance from Boston (km)	Major roads nearby	General characteristics	Sample size pre-COVID-19	Sample size COVID-19 with non-leafed trees	Sample size COVID-19 with leafed trees
Blue Hills	2800	24	U.S. 93 Route 138 Route 24	Recreational lake area with swimming Weather observatory Nature center Hiking and ski trails	1037	265	173
Hammond Pond	82	11	Hammond Pond Parkway	Large pond Hiking trails Nearby houses and large shopping center	637	252	219
Hall's Pond	1.4	4.2	Beacon Street	Large recreational field Nearby apartment buildings, retail businesses, and houses	84	45	45

variability in measurements is minor and acceptable. In addition, SPLnFFT is highly consistent between measurements and among iPhone versions (Zipf et al., 2020), which is an important requirement in a study with multiple people and three separate sites.

2.3. Pre-COVID-19 noise measurements

Pre-COVID-19 sound measurements were collected initially as part of a study on the value of documenting and mapping noise pollution with community scientists (Zipf et al., 2020); additional measurements were collected as part of an unpublished undergraduate thesis analyzing noise

pollution patterns in protected areas around Boston (Terry, 2020) When the initial lockdown of the COVID-19 pandemic began, these measurements created an opportunity to analyze the change in sound levels due to the pandemic.

Pre-COVID-19 measurements were collected during two time periods. The first were taken at Blue Hills and Hammond Pond during one-day community science events in September and October 2017, respectively (see: Zipf et al., 2020). Additional pre-COVID-19 measurements were taken by researchers at Blue Hills, Hammond Pond, and Hall's Pond during six, seven, and four daily trips to each park, respectively, in July and August 2019. In all cases, iPhones were

calibrated to one common standard prior to measurement (Zipf et al., 2020).

Researchers and community scientists recorded sound levels along trails at each protected area. Noise measurements were taken approximately every 25–30 m along trails at Blue Hills and Hammond Pond and every 15–20 m at Hall's Pond to create a well-dispersed distribution of sound measurements throughout each park that could be used to make maps of noise pollution. Sound levels were measured for a period of 20–30 s at each point in accordance with the methods of Zipf et al. (2020). Sound measurements were recorded as the median sound level for each period, referred to as the L50 value. The latitude and longitude were also recorded for each measurement.

All measurements were taken between 9 am and 4 pm on days with low wind (less than 5.8 m/s) and without rain (Table S1). While measuring, iPhones were held so that the body and hand of the researcher blocked most wind, a method found to be effective by Zipf et al. (2020). Average temperatures on measurement days ranged between 19 and 27 °C (Table S1). According to the Massachusetts Department of Transportation, traffic data and our own observations, the road traffic levels were generally comparable between the different visits to any one location during pre-pandemic times (MS2 Transportation Data Management System, 2020), and researchers observed that the numbers of park visitors were generally similar as well. There were leaves on the trees during all pre-COVID-19 measurements. Researchers also noted if airplane noise was noticeable during the recording period. Airplane noise affected a small percentage of the measurements (8% at Hammond Pond and 20% at Blue Hills) at the three sites before the pandemic, and airplane noise was virtually absent after the pandemic. Due to its low occurrence and our interest in mapping the effects of road noise and other localized noise sources, in the analyses presented here we excluded noise measurements that included airplane noise.

2.4. COVID-19 pandemic noise measurements

Pandemic noise measurements were first taken at Blue Hills, Hammond Pond, and Hall's Pond in the weeks after the implementation of a strict stay at home order in Massachusetts due to COVID-19 which began on March 10, 2020. Measurements were taken on March 24 (Hammond Pond), March 31 and April 1 (Blue Hills), and April 15 (Hall's Pond). Measurements were taken using the same methods as the initial pre-COVID-19 measurements. There were no leaves on the deciduous trees that dominate these protected areas in Massachusetts in March, in contrast to the pre-COVID-19 measurements. In addition, the amount of visitor traffic to these parks was higher, according to the authors' observations, while the amount of road traffic was noticeably lower during this time than pre-COVID-19; volume on major roads was down 67% in March, and down 30% in July (MassDOT, 2020). According to Mass-DOT's Transportation Data Management System, the major roadways that surround the protected areas in this study typically receive average annual daily traffic volumes on the order of thousands or tens of thousands of cars per day. This large reduction in road volume lead to a noticeable increase in the speed of traffic on U.S. 93 at Blue Hills; although full data is not available for all roads, MassDOT data for the Massachusetts Turnpike, which is a major highway that leads into Boston, shows that, during the days and times of day that measurements took place, average traffic volume (vehicles per hour) had greatly decreased from pre-COVID to post-COVID. Additionally, the data shows that the proportion of vehicles traveling greater than 112 km/h had greatly increased (Table 2). We noted that road blockages that would normally slow speeds, such as traffic jams and construction, were nonexistent (Terry, personal observation).

Additional noise measurements were taken at each site in late May 2020, at the beginning of the Phase 1 of reopening plan, when Massachusetts went from a "stay at home" order to a "safer at home" order with certain businesses reopening, such as manufacturing, construction

Table 2

Vehicle traffic data on Massachusetts Turnpike (42.337433, −71.288790). Data from days of Blue Hills collection dates. "Percent of vehicles traveling 112+ km/h" and "average volume per hour" used only data from 10:00 AM to 3:00 PM. Data gathered from MassDOT MS2 Transportation Data Management System (Location AET10).

Sampling events	Percent of vehicles traveling 112+ km/h	Average volume per hour
Pre-COVID-19	47.5	7628
COVID-19 with non-leafed trees	66.7	2909
COVID-19 with leafed trees	57.8	4675

sites, and lab sites (The Commonwealth of Massachusetts, 2020). The dates for these measurements were May 26 (Hammond Pond), May 27 (Blue Hills), and May 26 (Hall's Pond). At this time, the deciduous trees had leafed out and the volume of traffic on Massachusetts highways was about 30–50% of normal volumes (MassDOT. 2020). As with the March measurements, the low traffic volumes on highways like the Mass Turnpike and U.S. 93 resulted in substantially increased vehicle speeds (Table 2).

During all data collection, researchers followed social distancing guidelines; wearing masks, arriving to sites in separate vehicles, and staying at least 2 m away from each other and from park visitors.

2.5. Data analysis

All statistical analyses were done in R statistical software version 3.5.2 and all results were considered significant at alpha ≤0.05 (R Core Team, 2018). We first created noise maps of the measurements taken in each protected area to map the spatial distribution of sound levels in the parks using the R packages *ggplot2* version 3.3.2 and *ggmap* version 3.0.0 (Kahle and Wickham, 2013; Wickham, 2016). Using the packages *gstat* version 2.0-6 and *sp* version 1.4-4 (Gräler et al., 2016; Pebesma, 2004; Pebesma and Bivand, 2005; Bivand et al., 2013), we fit the data to a spatial variogram, and used ggplot 2 version 3.3.2 and ggmap version 3.0.0 to create interpolated maps of the sound levels for Hammond Pond and Blue Hills (Kahle and Wickham, 2013; Wickham, 2016). To assess the spatial extent of the impact of road noise pre-COVID-19, linear regressions were performed between sound level and the log of the distance to the nearest road at Blue Hills and Hammond Pond (Kesten and Tauck, 2012). In addition, sound levels at Blue Hills and Hammond Pond were binned based on distance from the nearest road using R packages *rgeos* version 0.5-5, *tigris* version 1.0, *sp* version 1.4-4, and *ggplot2* version 3.3.2 with bins defined at every 50 m (Bivand et al., 2013; Bivand and Rundel, 2019; Pebesma and Bivand, 2005; Walker, 2019; Wickham, 2016). Using the R package *multcompView* version 0.1-8, we performed a one-way ANOVA and a post-hoc Tukey test to test for significant differences between road distance groups (Graves et al., 2019). We did not perform distance from road analyses with the Hall's Pond, as there are no data points further than 100 m from the nearest road.

To compare sound levels pre- and during COVID-19, we separated the data into three categories: data collected before the COVID-19 lockdown (designated "pre-COVID-19"), data gathered at the beginning of lockdown in March when there were no leaves on the trees (designated "COVID-19 with non-leafed trees"), and data gathered later during lockdown when the trees had regained leaves (designated "COVID-19 with leafed trees"). For each site, we employed a generalized linear model approach to assess the combined impacts of road distance and category. We created three alternative models (Table 3) to predict the sound level for each location based on category and log distance to the nearest roads. We used the AIC model comparison score to determine the best performing model for each site; the model with the lowest AIC score was selected as the best fit model (Table S4). When creating the category divisions, pre-COVID-19 data was defined as the reference

Table 3

Model equations used to predict sound levels at different locations. L50 is the median sound level over the measurement duration, dist.min is the minimum distance to any major road, and category is the time period in which measurements were taken (pre-COVID-19, COVID-19 with non-leafed trees, and COVID-19 with leafed trees).

Model number	Equation
0	L50 ~ 1
1	L50 ~ log(dist.min)
2	L50 ~ Category
3	L50 ~ log(dist.min) + Category

category in order to directly compare pre- and during-COVID-19 sound levels. We created a set of models without the pre-COVID-19 data and used COVID-19 data with non-leafed trees as the reference category, in order to directly compare sound levels at the beginning and end of the lockdown. In addition, we created individual models for each category at Blue Hills and Hammond Pond comparing sound level and log distance from roads.

3. Results

3.1. Pre-COVID-19 spatial pattern of sound levels

At Hammond Pond and Blue Hills, pre-COVID-19 sound levels were highest near the roads and lowest in the interior of the park (Fig. 2A and B). Sound levels varied greatly at both locations: Hammond Pond ranged from 35 dB to 78 dB, while Blue Hills ranged from 32 dB to 73 dB. At Blue Hills, the median sound level within 50 m of a road was 56 dB, while the median sound level in the interior of the park (more than 400 m away from a road) was 43 dB. At Hammond Pond, the median sound level within 50 m of a road was 54 dB, while the median sound level in the interior of the park (more than 400 m away from a road) was 42 dB. Hall's Pond had sound levels ranging from 44 dB to 66 dB (Fig. 2C), a much lower range of sound levels than Blue Hills or Hammond Pond due to its smaller area and lack of points close to busy roads. There was also no spatial pattern of noise at Hall's Pond.

Sound levels at Hammond Pond and Blue Hills declined significantly with logarithmic distance from the nearest road ($p < 0.001$; Table 4). When analyzed by 50 m distance intervals from the nearest road, there was a statistically significant difference in sound levels of distance intervals at Hammond Pond, determined by a one-way ANOVA (F = 46.48, $p < 0.001$) and distance intervals at Blue Hills, determined by a one-way ANOVA (F = 42.71, $p < 0.001$). At both Hammond Pond and Blue Hills, the mean sound level between 0 and 50 m from a road was significantly greater than every other distance grouping. At Hammond Pond, none of the mean sound levels past 300 m were significantly different from each other, and at Blue Hills, none of the mean sound levels past 400 m were significantly different from each other.

3.2. COVID-19 sound levels

At Hammond Pond, model 3 (distance to road + category) was the best performing of the sound models according to the AIC score (Table S5). The data continued to demonstrate a significant logarithmic relationship with distance to road. After the model accounted for distance to road, COVID-19 sound measurements taken in March when the trees did not have leaves were on average 1.5 dB lower compared with pre-COVID-19 sound levels (SE = 0.41, t = −3.63, $p < 0.001$). COVID-19 sound measurements taken in June when the trees had leaves were on average 2.8 dB lower than pre-COVID-19 sound levels (SE = 0.38, t = −7.33, $p < 0.001$; Table 4; Fig. 3A). Sound levels were lower during COVID-19 by 1.4 dB when the trees had leaves compared with non-leafed trees (SE = 0.44, t = 3.25, p = 0.0012; Table 5; Fig. 3A).

Model 3 (distance to road + category) was also the best performing model at Blue Hills according to the AIC score (Table S5). In a major departure from expectations, sound levels at Blue Hills were higher during COVID-19 than during the pre-COVID-19 era. After accounting for the logarithmic relationship with distance to road, COVID-19 sound measurements taken in March, when the trees did not have leaves, were on average 5.4 dB higher compared with pre-COVID-19 sound levels (SE = 0.4, t = 10.66, $p < 0.001$). Sound measurements taken during COVID-19 in June when the trees did have leaves were on average 4.1 dB higher than pre-COVID-19 sound levels (SE = 0.42, t = 9.61, $p < 0.001$; Table 4; Fig. 3B). Sound levels were lower during COVID-19 by 1.3 dB when the trees had leaves compared with non-leafed trees. (SE = 0.64, t = 1.98, p = 0.049; Table 5; Fig. 3B).

At Hall's Pond, the best performing model was model 2 (category only) according to the AIC score (Table S5). COVID-19 measurements taken in March when the trees did not have leaves were on average 3.1 dB lower than pre-COVID-19 sound levels (SE = 0.55, t = 5.57, p < 0.001), while COVID-19 measurements taken in June when the trees did have leaves were on average 1.5 dB lower than pre-COVID-19 sound levels (SE = 0.55, t = −2.81, p = 0.006; Table 4). Sound levels were higher by 1.5 dB during COVID-19 when the trees had leaves compared with non-leafed trees (SE = 0.46, t = −3.25, p = 0.002; Table 5).

4. Discussion

The field of soundscape ecology has demonstrated the importance of noise in environmental conservation efforts (Pijanowski et al., 2011). Soundscapes are natural resources; noise pollution has detrimental health impacts for people, and can restructure species assemblages and impact reproductive performance in wildlife (Dumyahn and Pijanowski, 2011). The primary goal of this study was to compare the extent and magnitude of noise pollution in urban protected areas during the beginning of the COVID-19 pandemic and associated lockdowns in Massachusetts to control measurements. Each of our three study sites demonstrated a change in sound levels during the pandemic lockdown; however, the directions of the changes and the details vary across sites (Fig. 4).

4.1. Impact of roads on pre-COVID-19 sound levels

Road traffic noise was the primary source of elevated sound levels at our study sites. Sound level has a logarithmic relationship with distance from the nearest road; the sound impact of traffic was greatest within the first 50 m and declined progressively with increasing distance from roads. This decline continued up to 300 m from a road at Hammond Pond and to 400 m at Blue Hills. Beyond this distance, sound levels were fairly constant and there was not a detectable change as distance increases. The lack of a significant decrease past these thresholds could indicate that road noise does not have a substantial impact past 300 or 400 m. However, it is also possible that the protected areas in our study are not large enough to observe the full impact from road noise; with greater distances away from roads, it is possible we would continue to observe a decline in sound level.

Other sources of noise were noticeable at particular parks and are also worth investigating in later studies. Most notably, airplane noise was a concern of park advocates at the Blue Hills site, and noise pollution from landscaping equipment, such as leaf blowers and lawn mowers, was particularly evident at Hall's Pond and is a contentious local issue in Newton.

4.2. Effects of the COVID-19 pandemic on sound levels

At Hammond Pond, as expected, sound levels during the pandemic after the trees had leafed out were several decibels lower than sound levels pre-COVID-19. The sound levels in March were greater than the sound levels in June, but were still significantly lower than pre-COVID-19 levels, though to a lesser extent. The difference in sound levels
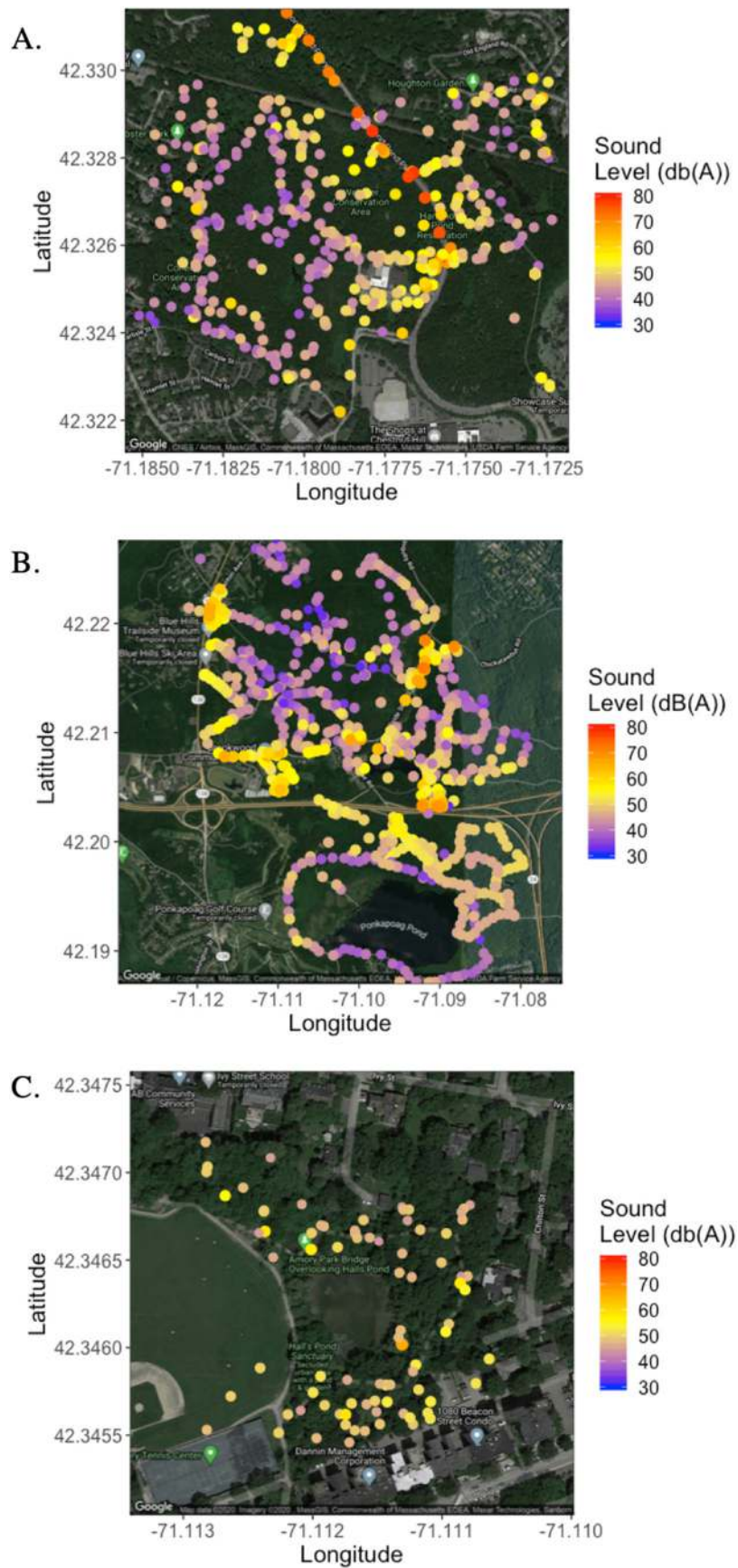
Fig. 2. Sound levels at the (A) Hammond Pond Reservation (Newton, MA), (B) Blue Hills Reservation (Milton, MA), and (B) Hall's Pond Sanctuary (Brookline, MA) taken pre-COVID-19. Each point represents the L50 median sound level of a 20–30 s noise measurement.

Table 4

Best performing models for Hammond Pond (Newton, MA), Blue Hills (Milton, MA), and Hall's Pond (Brookline, MA), with pre-COVID-19 data as the reference category.

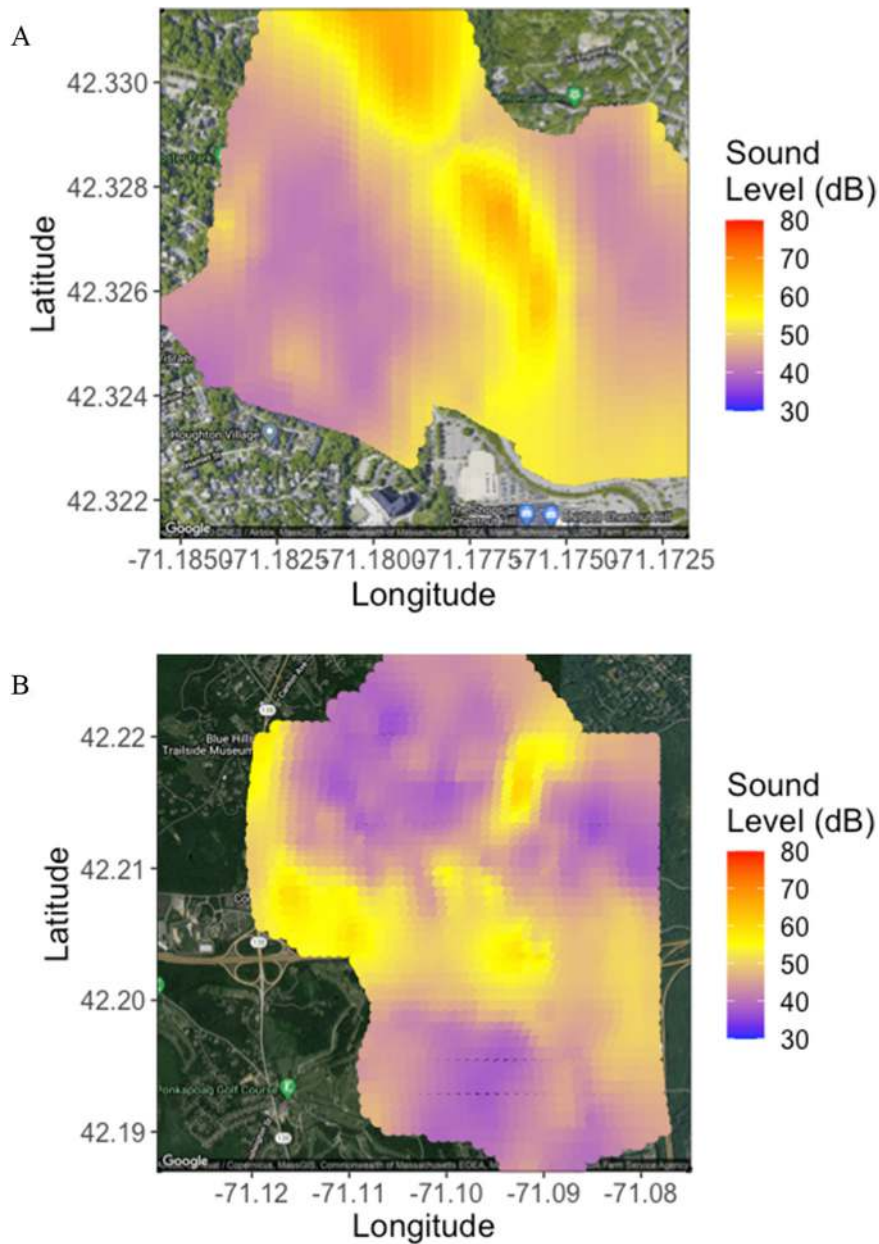| Site	R2	Explanatory variables	Coefficients	SE	t-Value	p value	Significance
Hammond Pond	0.57	Intercept	70.15	0.64	109.44	<0.001	***
		log(road distance)	−10.24	0.12	−37.74	<0.001	***
		COVID-19 with non-leafed trees	−1.49	0.41	−3.63	<0.001	***
		COVID-19 with leafed trees	−2.79	0.38	−7.33	<0.001	***
Blue Hills	0.48	Intercept	72.89	0.84	87.11	<0.001	***
		log(road distance)	−10.73	0.15	−31.88	<0.001	***
		COVID-19 with non-leafed trees	5.38	0.50	10.66	<0.001	***
		COVID-19 with leafed trees	4.06	0.42	9.61	<0.001	***
Hall's Pond	0.15	Intercept	49.40	0.32	152.90	<0.001	***
		COVID-19 with non-leafed trees	−3.05	0.55	−5.57	<0.001	***
		COVID-19 with leafed trees	−1.54	0.55	−2.81	0.006	**

Fig. 3. Interpolated sound levels at the (A) Hammond Pond Reservations (Newton, MA) and (B) Blue Hills Reservation (Milton, MA), from measurements taken pre-COVID-19.

Table 5
Best performing models for Hammond Pond (Newton, MA), Blue Hills (Milton, MA), and Hall's Pond (Brookline, MA), using only COVID-19 data, with measurements taken when trees were non-leafed as the reference category.

| Site	R2	Explanatory variables	Coefficients	SE	t-Value	p value	Significance
Hammond Pond	0.66	Intercept	67.41	0.68	90.58	<0.001	***
		log(road distance)	−9.56	0.33	−29.11	<0.001	***
		COVID-19 with leafed trees	−1.43	0.44	3.25	0.0012	**
Blue Hills	0.48	Intercept	81.95	1.47	55.08	<0.001	***
		log(road distance)	−12.37	0.62	−19.98	<0.001	***
		COVID-19 with leafed trees	−1.26	0.64	1.98	0.049	*
Hall's Pond	0.10	Intercept	46.36	0.33	145.68	<0.001	***
		COVID-19 with leafed trees	1.51	0.46	−3.25	0.002	**
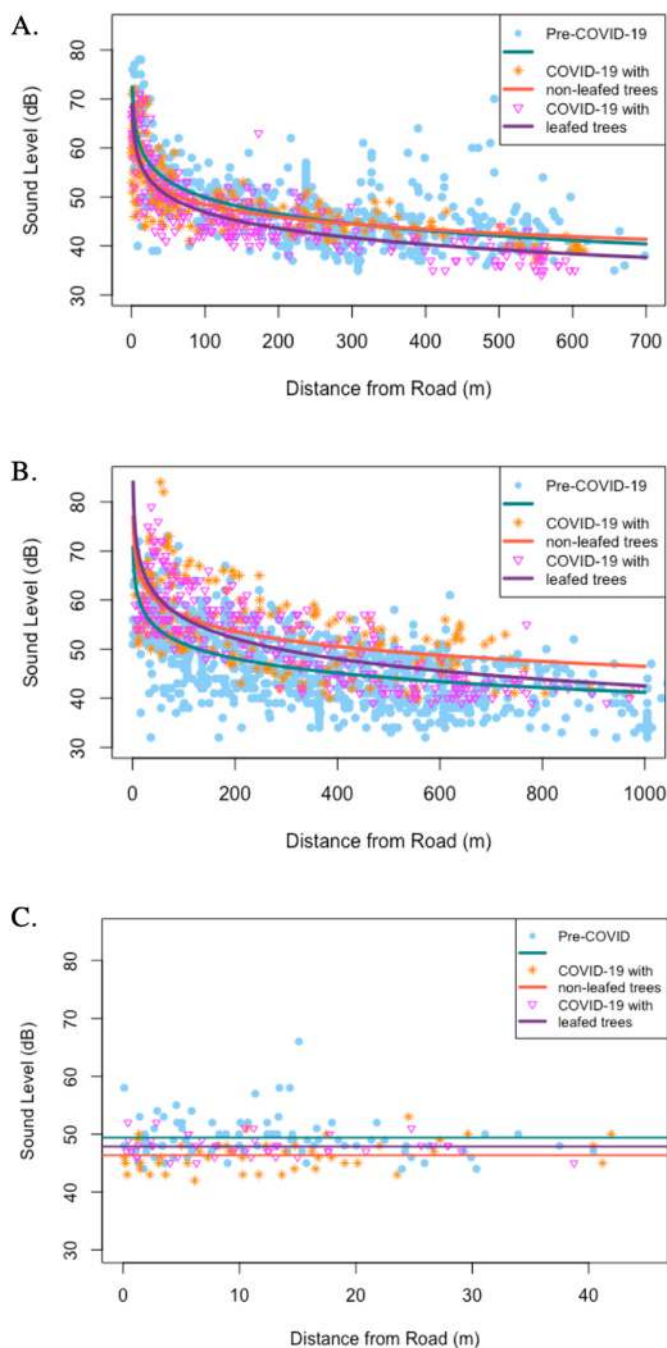
Fig. 4. Sound level vs. distance from nearest road at (A) Hammond Pond Reservation (Newton, MA), (B) Blue Hills Reservation (Milton, MA), and (C) Hall's Pond Reservation (Brookline, MA). For Hammond and Blue Hills (A and B) lines indicate a significant regression.

between March and June is likely due to the fact that the noise from roads was able to penetrate further into the forest due to the lack of tree leaves (Maleki and Hosseini, 2011).

At Hall's Pond, as hypothesized, sound levels were lower in March and June during the pandemic, both before and after trees leafed out, than pre-COVID-19 sound levels. This decrease in noise is most likely due to lower urban activity levels during the pandemic, including both reduced volume of urban traffic and less construction and landscaping activity in the surrounding area. Sound levels were somewhat higher after trees leafed out in June, presumably due to increasing activity in the surrounding area and on the roads as the pandemic lockdown began to be relaxed. Most notably, construction activities resumed during Phase 1 of reopening.

The large reduction in noise pollution in these two protected areas indicates reduction in traffic and human activity dramatically affects the soundscape of urban protected areas. Three of the key sources of noise present in urban areas are traffic, human voices, and sirens (Dumyahn and Pijanowski, 2011). By limiting human activity, the early COVID-19 lockdown in Boston reduced at least two of the primary sources of noise. Noise pollution in protected areas has been associated with changes in bird species composition and reproductive performance (Francis et al., 2009; Newport et al., 2014; Derryberry et al., 2020). In particular, road noise can mask bird song (Nemeth and Brumm, 2010). This masking is associated with declines in bird densities. The short-term reductions in road noise at Hammond Pond documented in this study may have benefited local bird populations, in that bird call adjustments to noise can be energetically costly (Lowry et al., 2012). However, it is unclear if reductions in noise during a single breeding season will have measurable ecosystem impacts. These short-term wildlife impacts warrant further exploration.

Maintaining urban biodiversity can increase human enjoyment and ecosystem services provided by protected areas (Larson et al., 2016; Schwartz et al., 2019; Secretariat of the Convention on Biological Diversity, 2008). Decreasing the road volume or speeds of nearby city streets could reduce noise pollution in this park in the long-term. Continued monitoring of noise in urban protected areas is needed to better understand the seasonal changes in noise pollution and long-term impacts of the pandemic on urban noise. The marked and sustained reduction in noise pollution during the pandemic lockdown indicates this environmental pollutant can be better managed in Hall's Pond and Hammond Pond, which will have positive ecosystem outcomes for wildlife and human visitors.

At Blue Hills, in contrast, sound levels were 4–6 dB higher during the pandemic lockdown in both March and June than during the pre-COVID-19 measurements. This increase in sound level in March and June, which occurred both near the road and hundreds of meters into the park, is most likely due to higher vehicle speeds during the time of the pandemic on U.S. 93, the interstate highway that runs through the park. Traffic noise comes from two major sources: the running of the vehicle systems, such as the engine and the exhaust system, and interaction between the tires and the road. With technological advancements reducing much of the vehicle running noise, tire/road interaction becomes the dominant noise source. This noise becomes more prominent at greater speeds

(Maling, 2014). Before the pandemic, traffic congestion throughout the day frequently resulted in traffic traveling under 65 km per hour and sometimes even completely stopped car movement (MassDOT Transportation Data Management System; Terry, personal observation). In contrast, during the pandemic, lower numbers of cars and trucks would allow cars to travel at speeds of 100–130 km per hour. At these greater speeds, the tire/road interaction would become louder and generate higher sound levels than were caused by the previously slower-moving vehicles (Cowan, 1993; Maling, 2014). It is also possible that while there was a large reduction in vehicle volume overall, commercial vehicles, such as large trucks, may not have had much of a reduction in volume compared to passenger vehicles; in fact, national data indicates that commercial traffic saw hardly any reduction at all (PrePass, 2020). Given that commercial vehicles produce more noise than other vehicles, and that commercial vehicles are far more likely to be found on major freeways such as U.S. 93 than smaller roads, it is likely that they made up a much larger proportion of traffic volume overall during COVID than before, and therefore a large portion of the increased volume traveling over 112 km/h (Table 2). In addition, similar to Hammond Pond, sound levels were slightly lower at Blue Hills in June after leaf out than in March, again likely due to the dampening effect of leaves (Maleki and Hosseini, 2011). This finding provides evidence that vehicular traffic volume is not a linear predictor of noise pollution in protected areas and changes to the primary sources of urban noise pollution do not always manifest as reductions in noise levels.

Based on these results, reducing speed limits and restricting the size and types of vehicles allowed to travel on roads near protected areas may prove more effective at reducing noise pollution than attempts to reduce the overall traffic volume. In addition, while visitation during COVID-19 was not specifically measured at our 3 sites, it is likely that changing numbers of visitors played a role in the differences observed. The U.S. National Park Service has found that in more remote protected areas, visitor conversations often play a large role in noise pollution. The Park Service has seen some success reducing sound levels through the institution of "quiet zones" at selected areas (Abad et al., 2017). Establishing "quiet zones" would likely be effective at reducing noise at our sites, particularly in the interior of Blue Hills where road noise is less prominent.

4.3. Conclusions

Overall, our results demonstrate that the three urban protected areas in our study have considerable noise pollution concentrated near the roads, as well as lower-level noise pollution throughout the entire park area. Noise pollution from roads diminishes only at 300–400 m from roads, meaning only a small fraction of each of these parks is truly quiet.

As predicted, we found decreases in sound levels during the time of the pandemic lockdown at Hammond Pond and Hall's Pond, but these improvements were only 1–3 dB in magnitude. In contrast, Blue Hills became 4–6 dB noisier during the time of the pandemic, contrary to our predictions, likely due to the increased road speed. A 3 dB increase is equivalent to a doubling in sound energy; however, it takes a 10 dB increase for people to perceive sound as twice as loud (Buxton et al., 2017). The small 1–6 dB changes observed in our study are likely noticeable by people walking through the parks, and make a difference in the aesthetic qualities of a park experience, but these changes may not have a major impact on the experiences of human visitors and the behavior of wildlife.

Although the changes that we found in sound level due to the pandemic are small relative to the range of human hearing, they are significant, and can be used to inform our approach to noise management. Most importantly, this study confirms that roads are large sources of noise pollution in urban protected areas. It also shows that traffic volume is not necessarily the best indicator of noise pollution, as fewer vehicles going faster can create more noise than more vehicles going at slower speeds. This finding suggests that strict speed limits on roads

going through protected areas may be an effective and inexpensive way to reduce noise pollution, though this will have other consequences for humans including a greater travel time along roads. Finally, this study demonstrates that noise pollution can penetrate hundreds of meters into protected areas and has the potential to diminish the value of natural spaces for wildlife habitat and human recreation at great distances from roads. While specific remediation efforts can help to reduce road noise, such as walls, berms, and evergreen plantings (Renterghem and Botteldooren, 2012), the construction of any roads through protected areas will invariably increase noise pollution.

CRediT authorship contribution statement

Carina Terry: all project aspects; **Matt Rothendler**: methodology, validation, investigation, writing – review and editing; **Lucy Zipf**: methodology, software, validation, formal analysis, investigation, writing – review and editing; **Mike Dietze**: methodology, software, validation, writing – review and editing; **Richard Primack**: all project aspects.

Declaration of competing interest

The authors of this paper certify that they have no conflicts of interests associated with the creation and publication of this manuscript.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2021.109039.

References

Abad, B., Dings, A., Farnitano, D., Weissman, Z., 2017. Preserving the Soundscape: Exploring Ways to Mitigate Sound Pollution in Acadia National Park. Worcester Polytechnic Institute (unpublished bachelor's thesis).

Anderson, G.S., Rapoza, A.S., Fleming, G.G., Miller, N.P., 2011. Aircraft noise dose-response relations for national parks. Noise Control Eng. J. 59 (5), 519–540.

Barber, J.R., Burdett, C.L., Reed, S.E., Warner, K.A., Formichella, C., Crooks, K.R., Theobald, D.M., Fristrup, K.M., 2011. Anthropogenic noise exposure in protected natural areas: estimating the scale of ecological consequences. Landsc. Ecol. 26 (9), 1281–1295. https://doi.org/10.1007/s10980-011-9646-7.

Bivand, R., Rundel, C., 2019. rgeos: interface to Geometry Engine - Open Source ('GEOS'). R package version 0.4-3. CRAN.R-project.org/package=rgeos.

Bivand, R., Pebesma, E., Gomez-Rubio, V., 2013. Applied Spatial Data Analysis With R, Second edition. Springer, New York, New York.

Bloemsma, L.D., Wijga, A.H., Klompmaker, J.O., Hanssen, N.A.H., Smit, H.A., Koppelman, G.H., Brunekreef, B., Lebret, E., Hoek, G., Gehring, U., 2019. The associations of air pollution, traffic noise and green space with overweight throughout childhood: the PIAMA birth cohort study. Environ. Res. 169, 348–356. https://doi.org/10.1016/j.envres.2018.11.026.

Brown, C.L., et al., 2012. The effect of human activities and their associated noise on ungulate behavior. PLoS One 7 (7). https://doi.org/10.1371/journal.pone.0040505.

Buckley, R., Brough, P., Hague, L., Chauvenet, A., Fleming, C., Roche, E., Sofija, E., Harris, N., 2019. Economic value of protected areas via visitor mental health. Nat. Commun. 10 (1) https://doi.org/10.1038/s41467-019-12631-6.

Buxton, R.T., McKenna, M.F., Mennitt, D., Fristrup, K., Crooks, K., Angeloni, L., Wittemyer, G., 2017. Noise pollution is pervasive in U.S. protected areas. Science 356 (6337), 531–533.

Massachusetts DCR makes adjustments at parks to prevent overcrowding during coronavirus, CBS Boston. , 3 May. boston.cbslocal.com/2020/05/03/coronavirus-massachusetts-parks-dcr/.

Cowan, J.P., 1993. Handbook of Environmental Acoustics. Wiley, Hoboken, New Jersey.

Derryberry, E.P., Phillips, J.N., Derryberry, G.E., Blum, M.J., Luther, D., 2020. Singing in a silent spring: birds respond to a half-century soundscape reversion during the COVID-19 shutdown. Science 370 (6516), 575–579. https://doi.org/10.1126/science.abd5777.

Dudley, N., 2008. Guidelines for Applying Protected Area Management Categories. IUCN, Gland, Switzerland.

Dumyahn, S.L., Pijanowski, B.C., 2011. Soundscape conservation. Landsc. Ecol. 26, 1327. https://doi.org/10.1007/s10980-011-9635-x.

Dwyer, D., 2020. Parking areas at multiple state parks closed after capacities reached for "maximum safe number of visitors". May 4, Boston.com. www.boston.com/news/co ronavirus/2020/05/02/state-parks-two-hour-closures-coronavirus-capacity.

Francis, C.D., Ortega, C.P., Cruz, A., 2009. Noise pollution changes avian communities and species interactions. Curr. Biol. 19 (16), 1415–1419.

Francis, C.D., Ortega, C.P., Cruz, A., 2011. Noise pollution filters bird communities based on vocal frequency. PLoS One 6 (11), 1–8.

Goines, L., Hagler, L., 2007. Noise pollution: a modern plague. South. Med. J. 100 (3), 287–294.

Gräler, B., Pebesma, E., Heuvelink, G., 2016. Spatio-temporal interpolation using gstat. R J. 8, 204–218. https://journal.r-project.org/archive/2016/RJ-2016-014/index. html.

Graves, S., Piepho, H.P., Selzer, L., Dorai-Raj, S., 2019. multcompView: visualizations of paired comparisons. CRAN.R-project.org/package=multcompView.

Kahle, D., Wickham, H., 2013. ggmap: spatial visualization with ggplot2. R J. 5 (1), 144–161.

Kardous, C.A., Shaw, P.B., 2014. Evaluation of smartphone sound measurement applications. J. Acoust. Soc. Am. 135 (4), 186–192.

Kesten, P.R., Tauck, D.L., 2012. University Physics for the Physical and Life Sciences, Volume 1, pp. In: 493–558. W.H. Freeman and Co., New York, New York.

Kim, D., Jin, J., 2018. Does happiness data say urban parks are worth it? Landsc. Urban Plan. 178, 1–11. https://doi.org/10.1016/j.landurbplan.2018.05.010.

Larson, L.R., Jennings, V., Cloutier, S.A., 2016. Public parks and wellbeing in urban areas of the United States. PLoS One 11 (4). https://doi.org/10.1371/journal. pone.0153211.

Lefebvre, F., 2010. SPLnFFT Noise Meter, Version 7.0, App Store. https://apps.apple.com /us/app/splnfft-noise-meter/id355396114 (Mobile app).

Lockwood, M., Lahiri, S., Babiceanu, S., 2020. Transportation Research Board (TRB) Webinar: traffic trends and safety in a COVID-19 world. What is happening in Virginia?, Virginia Department of Transportation (VDOT). www.trb.org/Electron icSessions/Blurbs/180648.aspx. (Accessed 3 September 2020).

Lowry, H., Lill, A., Wong, B.B.M., 2012. Behavioural responses of wildlife to urban environments. Biol. Rev. 88 (3), 537–549.

Maleki, K., Hosseini, S.M., 2011. Investigation of the effects of leaves, branches and canopies of trees on noise pollution reduction. Ann. Environ. Sci. 5, 13–21.

Maling Jr., G.C., 2014. Noise, 2014. In: Rossing, T.D. (Ed.), Springer Handbook of Acoustics. Springer, New York, pp. 1001–1057.

Mandal, I., Pal, S., 2020. COVID-19 pandemic persuaded lockdown effects on environment over stone quarrying and crushing areas. Sci. Total Environ. 732, 139281.

MassDOT Mobility Dashboard. Massachusetts Department of Transportation, Massachusetts. mobility-massdot.hub.arcgis.com. accessed December 8, 2020.

Merchan, C.I., Diaz-Balteiro, L., Soliño, M., 2014. Noise pollution in national parks: soundscape and economic valuation. Landsc. Urban Plan. 123, 1–9.

MS2 Transportation Data Management System. Massachusetts Department of Transportation, Massachusetts. mhd.ms2soft.com/tcds/tsearch.asp?loc=Mhd&mo d=. accessed September 4, 2020.

Murphy, E., King, E.A., 2014. Environmental Noise Pollution: Noise Mapping, Public Health, and Policy. Elsevier Inc.

Nast, D.R., Speer, W.S., Le Prell, C.G., 2014. Sound level measurements using smartphone "apps": useful or inaccurate? Noise Health 16 (72), 251–256.

Nemeth, Erwin, & Brumm, H., 2010. Birds and anthropogenic noise: are urban songs adaptive? The American Naturalist, vol. 176, no. 4, 2010, pp. 465–475, https://doi. org/10.1086/656275. accessed December 17, 2020 (JSTOR).

Newport, J., Shorthouse, D.J., Manning, A.D., 2014. The effects of light and noise from urban development on biodiversity: implications for protected areas in Australia. Ecol. Restor. Manag. 15 (3), 204–214.

Pebesma, E.J., 2004. Multivariable geostatistics in S: the gstat package. Comput. Geosci. 30, 683–691.

Pebesma, E.J., Bivand, R.S., 2005. Classes and methods for spatial data in R. R News 5 (2), 9–13.

Pijanowski, B.C., Farina, A., Gage, S.H., Dumyahn, S.L., Krause, B.L., 2011. What is soundscape ecology? An introduction and overview of an emerging new science. Landscape Ecol 26, 1213–1232. https://doi.org/10.1007/s10980-011-9600-8.

Prepass, 2020. Truck traffic volumes remain strong despite COVID-19 pandemic. April 6. prepass.com/2020/04/06/truck-traffic-volumes-remain-strong-despite-covid-19-p andemic/. (Accessed 8 December 2020).

R Development Core Team, 2018. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. www.R-pro ject.org/.

Rabat, A., 2007. Extra-auditory effects of noise in laboratory animals: the relationship between noise and sleep. J. Am. Assoc. Lab. Anim. Sci. 46 (1), 35–41.

Renterghem, T.V., Botteldooren, D., 2012. On the choice between walls and berms for road traffic noise shielding including wind effects. Landsc. Urban Plan. 105 (3), 199–210. https://doi.org/10.1016/j.landurbplan.2011.12.017.

Salaberria, C., Gil, D., 2010. Increase in song frequency in response to urban noise in the great tit Parus major as shown by data from the Madrid (Spain) city noise map. Ardeola 57 (1), 3–11.

Schwartz, A.J., Dodds, P.S., O'Neil-Dunne, J.P.M., Danforth, C.M., Ricketts, T.H., 2019. Visitors to urban greenspace have higher sentiment and lower negativity on Twitter. People Nat. 1 (4), 476–485. https://doi.org/10.1002/pan3.10045.

Secretariat of the Convention on Biological Diversity, 2008. Protected Areas in Today's World: Their Values and Benefits for the Welfare of the Planet. Secretariat of the Convention on Biological Diversity. Montreal, Quebec.

Sun, J.W.C., Narins, P.M., 2005. Anthropogenic sounds differentially affect amphibian call rate. Biol. Conserv. 121 (3), 419–427.

Terry, C., 2020. Monitoring and Modeling the Extent of Noise Pollution in Massachusetts Protected Areas. Boston University (unpublished bachelor's thesis).

The Commonwealth of Massachusetts, 2020. Reopening Massachusetts. May 18. https://www.mass.gov/doc/reopening-massachusetts-may-18-2020/download.

Uran, S.L., Aon-Bertolino, M.L., Caceres, L.G., Capani, F., Guelman, L.R., 2012. Rat hippocampal alterations could underlie behavioral abnormalities induced by exposure to moderate noise levels. Brain Res. 1471, 1–12.

Walker, K., 2019. tigris: Load Census TIGER/Line Shapefiles. cran.r-project.org/web/pac kages/tigris/index.html.

Warren, P.S., Katti, M., Ermann, M., Brazel, A., 2006. Urban bioacoustics: it's not just noise. Anim. Behav. 71 (3), 491–502.

Wickham, H., 2016. ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag, New York, New York.

Xing, Y., Brimblecombe, P., 2020. Traffic-derived noise, air pollution and urban park design. J. Urban Des. 25 (5), 609–625. https://doi.org/10.1080/ 13574809.2020.1720503.

Zipf, L., Primack, R.B., Rothendler, M., 2020. Citizen scientists and university students monitor noise pollution in cities and protected areas with smartphones. PLoS One 15 (9). https://doi.org/10.1371/journal.pone.0236785.