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Urban Transformations

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Evaluating the efficacy of small-plot restorative management in an urban park using low-tech soundscape tools: an exploratory study

Allison Preble[1*], Liam Heneghan[1] and Christie A. Klimas[1]

*Correspondence: allipreble@gmail.com

[1] Department of Environmental Science and Studies, DePaul University, 1100 West Belden Av, Chicago, IL 60614, USA

Abstract

The role that cities play in enhancing biodiversity conservation is increasingly recognized. However, since locations for conservation within metropolitan areas are often spatially restricted, and management for biodiversity may conflict with interventions on behalf of other desirable objectives, it is important that the outcomes of urban conservation projects are carefully monitored. Such monitoring is relatively rare. In this study we explored the value of employing soundscape analysis to provide a holistic evaluation of biotic communities at urban sites undergoing different forms of vegetation management. Using readily affordable audio recorders, we evaluated soundscapes in replicated areas within a 481-hectare urban park in Chicago, Illinois. Areas within the park are managed to achieve multiple objectives including both recreational use and nature conservation. We found that relatively small areas within the park that had been subjected primarily to restorative vegetation management supported different acoustic environments with higher avian activity and more prevalent biophonic sound than was the case in managed lawn spaces. The use of a variety of acoustic indices supplemented the analysis of these soundscapes, and whereas all indices affirmed seasonal differences, the Acoustic Complexity Index (ACI) was more helpful than the other indices we employed in discriminating between management practices. We conclude that vegetation management employed even at a small spatial scale in an urban environment can enhance faunal diversity, and that these results can be evaluated using inexpensive sound monitoring equipment.

Highlights

➢Relatively small areas (<1 hectare) that are under restorative management within a large urban park supported a richer acoustic environment with both higher avian activity and a greater diversity of animal sounds than managed lawn spaces. We show that such areas have low'biophonic absence,' a term we use to highlight the diminished soundscapes of areas managed as traditional lawns.

➢ Inexpensive recording equipment can be used to detect differences in the quality of soundscapes in managed areas. The use of a variety of acoustic indices applied

to these recordings were valuable in the analysis of urban soundscapes. Although all indices we used verified seasonal differences in sites, the Acoustic Complexity Index (ACI) was more helpful than the other indices we employed in discriminating between management practices.

➢ Vegetation management employed even at a small scale in urban environments can enhance faunal diversity.

Keywords: Acoustic monitoring, Urban tinkering, Urban restoration, Bioacoustics, Ecoacoustics, Ecological restoration, Acoustic index, Biophonic absence

Policy and practice recommendations

➢ Soundscape analysis can be an effective and affordable method for evaluating the outcomes of urban ecological restoration when specific targets for monitoring are established.

➢ Soundscape differences attributed to ecological restoration can be observed in small scale urban park plots.

➢ Expanding metrics to evaluate the effectiveness of vegetation management beyond measurements of the plant community may provide a more complete understanding of the outcome of conservation efforts in multi-use urban spaces.

➢ Management for soundscape quality in urban greenspaces should emerge as a more pressing objective given the extent of scientific evidence for the role of the acoustic environment for human health and wellbeing.

Introduction

Although the globally aggregated physical footprint of cities is relatively small (estimated to be 3–5% of total land surface; see (Seto et al. 2010)), the combined ecological footprint of cities–that is, the measure of the productive land needed to furnish urban environments with their necessary resources as well as sinks for their waste—extends, in complex ways, over a very extensive terrain (Rees and Wackernagel 2008; Chen et al. 2020). For this reason, well-conceived urban environmental management not only mitigates local urban impacts while contributing valuable ecological services to city-dwellers but may also contribute positively to sustainability and biodiversity conservation at larger scales (Aronson et al. 2017; Nilon et al. 2017). Recognizing that successful urban ecological management may have benefits at ascending spatial scales has resulted in widespread adoption of urban conservation and sustainability initiatives, many of which are relatively well-funded (Bulkeley and Betsill 2005; Wu 2014; Beatley 2016; Richards and Thompson 2019).

Despite the global context that may motivate urban biodiversity conservation, management more often than not is implemented on hyper-local scales, and it is at these scales that the success of such projects in achieving their goals must be evaluated (although, as we shall see, evaluation of management outcome is sporadically undertaken). In some exceptional cases, relatively large tracts of urban land have been set aside primarily for conservation-oriented management. For example, Chicago Wilderness (a regional alliance of landowners and conservation partners in the greater

Chicago area and the location in which the current study was undertaken) collectively constitutes 150,000 hectares of protected land. This affords some opportunities to manage parcels of land that are hundreds of hectares in extent. More typically, however, areas allocated for conservation purposes in both the Chicago region and other metropolitan areas are often small and highly fragmented. Greenspaces in the urban core, in particular, are often less than 100 hectares in total (Heneghan et al. 2012). The prevalence of small land parcels dispersed across highly fragmented landscapes in the urban conservation portfolios of Chicago and across metropolitan areas worldwide poses unique challenges for both implementation of management and monitoring of outcomes (Knapp et al. 2008).

Irrespective of parcel size, the challenges confronting urban ecological management abound and are consistently found in many large cities. Much of the land designated for conservation in urban environments is ecologically disturbed and subjected to multiple ongoing stressors including altered climates, enhanced exposure to atmospheric pollutants, highly modified hydrology, and distinctively altered soils (Alberti 2008; Niemelä et al. 2011). Compounding these difficulties is the fact that greenspace under direct management must serve multiple and oftentimes potentially competing needs (for example, such lands must serve recreational, aesthetic, and educational needs while also potentially being managed to enhance biodiversity objectives) (Van Leeuwen et al. 2010). Challenges in reconciling diverse goals on small-scale plots can be exacerbated by community conflict over the management of individual sites, which can occur especially when public input into management has been curtailed (Gobster 2001; Schultz et al. 2022).

Because of problems such as these—individual agencies having to manage many modestly-sized reserves, the poor ecological condition of most urban habitat, conditions which are often more intensely disturbed nearer to the urban core, and a conflict among management goals—the oversight of parcels of urban greenspace must routinely be undertaken with an entrepreneurial spirit. This style of flexible management is one that Elmqvist and colleagues call "urban tinkering" (Elmqvist et al. 2018). A hallmark of urban tinkering is that such projects are explicitly designed simultaneously to achieve multiple ecological goals—for example carbon sequestration, soil enhancement, and watershed protection—and at least some of such projects aim to actively enhance the flourishing of native biota and protect regionally rare species. In addition to setting these complex ecological goals, management at urban sites must also, typically, fulfil social goals.

To achieve biodiversity conservation targets in particular, ecologically restorative management techniques are commonly implemented (Gobster 2010). Although, such projects do not always seek to faithfully restore a former ecological community with historical fidelity, nonetheless they are restorative in the sense that they prioritize the recovery and ecological health of native biotic communities. Restorative management undertaken on many small urban plots often aims at achieving far-reaching conservation effects. Though diverse ecological outcomes are routinely envisioned, the installation of such projects typically prioritizes vegetation management. Alteration of vegetation is undertaken for the somewhat obvious reason that plant life is the aspect of ecological communities most readily amenable to direct manipulation. By directly controlling the vegetative community, the manager anticipates ancillary benefits for the broader ecological community (this intuition is founded on longstanding ecological truisms about the influence of vegetation on animal communities Strong et al. 1984; Begon et al. 1986)). As is the case more generally in ecological restoration, there is an expectation that manipulating the structural components of a habitat will promote the development of other elements of the biotic community (this expectation is referred to as "the field of dreams" hypothesis (Sudduth et al. 2011)). However, ensuring that these objectives are met requires timely evaluation of management outcomes in order to provide directions for future interventions (McKinney 2006; Faeth et al. 2011; Zipperer et al. 2020).

Critical to the success of "urban tinkering," therefore is a commitment to routine and systematic evaluation. Indeed, if urban environmental management is to justify its broader ambitions, locally, regionally, and globally, it will require fastidious attention to the evaluation of outcomes. However, tight budgets, lack of expertise, and conflicts among desired outcomes, can make commitment by local agencies to evaluative programs infeasible. Urban tinkering therefore suffers the same shortcoming of ecological restoration more generally, that is, the quantitative appraisal of outcomes often lags the installation of the work (Ruiz-Jaen and Mitchell Aide 2005; Zedler 2007; Wortley et al. 2013). As we have suggested above, the complexity of urban environments, and the panoply of sometimes competing desired outcomes for the management of urban space, make the implementation of conventional evaluation especially challenging (Standish et al. 2013). Nevertheless, even relatively crude and rapid assessment of management results can effectively inform adaptive management (McCarthy and Possingham 2007). Therefore, holistic, efficient, and timely approaches to monitoring of biodiversity outcomes will undoubtedly benefit urban management organizations.

One potential monitoring approach has been proposed in recent years that may provide the sort of rapid and holistic assessment needed by urban greenspace managers: acoustic monitoring (Penone et al. 2013; Newson et al. 2015). This form of monitoring is attractive for a number of reasons. It can be implemented with inexpensive equipment, and it produces data that can be relatively tractable to analyze. Furthermore, the monitoring of sound can be used to assess the broader aim of restorative management—whether the manipulation of vegetative structure in a community result in a concomitant enhancement of the broader biotic community.

In this paper, therefore, we evaluate if soundscape analysis can validate the claim that urban vegetation management promotes the occurrence of diverse groups of sound-emitting organisms at managed sites. We conducted a small-scale evaluation of novel strategies for greenspace management (that is, urban tinkering on small plots) in Lincoln Park, Chicago. This 481-ha park is situated in a generally affluent and densely populated area along the lakefront on Chicago's north side. It is maintained by both the municipal park district and private entities that engage volunteers. Lawn spaces managed for recreational and aesthetic purposes are interspersed with those managed as savanna or prairie-like habitat. Thus, recreational, aesthetic, as well as ecological service and conservation objectives, are emphasized on a suite of sites that are intermingled with busy streets, sidewalks, and structures including restaurants, museums, and even a zoo.

Vegetation management in these areas has been undertaken for various purposes and implemented over various timelines. Managers of these sites have limited capacity to holistically assess the outcomes of their work. We therefore implemented acoustic monitoring of patches under two distinctly different management styles to understand their consequences for sustaining sound emitting animals in noisy urban areas.

We ask the following questions:

1. Do restorative management efforts which are focused on the manipulation of vegetation and implemented on spatially small plots result in the production of environments with richer and more complex bioacoustic characteristics, and
2. Can soundscape analysis, employing relatively inexpensive equipment, provide insights into overall soundscape quality between plots managed for biodiversity outcomes and those managed as lawns?

We hypothesize that areas undergoing restorative management will have richer and more complex acoustic environments, and that there will be an elevated presence of sound-emitting organisms (birds, in particular) and greater soundscape activity in restored areas (managed for biodiversity) compared with open lawn spaces (managed for recreation or aesthetics). Ultimately, we hope to show that relatively low-tech acoustic monitoring—the sort that might be readily available to most land management agencies—can be a valuable means of conducting project evaluation.

Methods

Site selection and description

Our approximately 8-ha study area of Lincoln Park, Chicago, was separated for the purposes of this study into three centralized areas: North Pond, South Pond, and a section between the ponds that included both lawn and restoratively managed space. Eight site locations were identified in the study area via nonrandom assignment. These included four open grass lawns and four spaces where management for native planting had been initiated (Fig. 1). For example, one of the sites chosen near North Pond had early-stage plantings of prairie grass and forbs. The grass lawns (primarily with turf grass vegetation) resembled conditions prior to native species planting. Lawns (sites A through D, Fig. 2) are termed "open" here. Though there are some scattered trees and flower beds present in these open spaces, lawns were managed for recreational and aesthetic purposes, rather than for any conservation benefits. In contrast, the sites labeled "restored" in this study (sites E through H, Fig. 2), were managed primarily for conservation benefits and visitors were asked to remain on paths. These restored areas received ongoing management to resemble mixed prairie and savanna woodland. The proximity of all the sites that we studied ensured consistency in most environmental conditions.

All open sites (A – D) and two of the restored sites (E – F) were owned and managed by the municipal entity and its contractors. A private nonprofit organization oversees the long-term maintenance of the restored sites through contracted professionals and volunteers. Initial restoration was completed at Site E in 2001 and Site F in 2002. These aquatic contracts prioritized plant composition and water quality, including hardscape and structural maintenance for aesthetics. Trash removal was a large portion of these activities. Annual reports highlighted successes and challenges regarding vegetation and water quality, but no wildlife-specific management was routinely implemented nor were

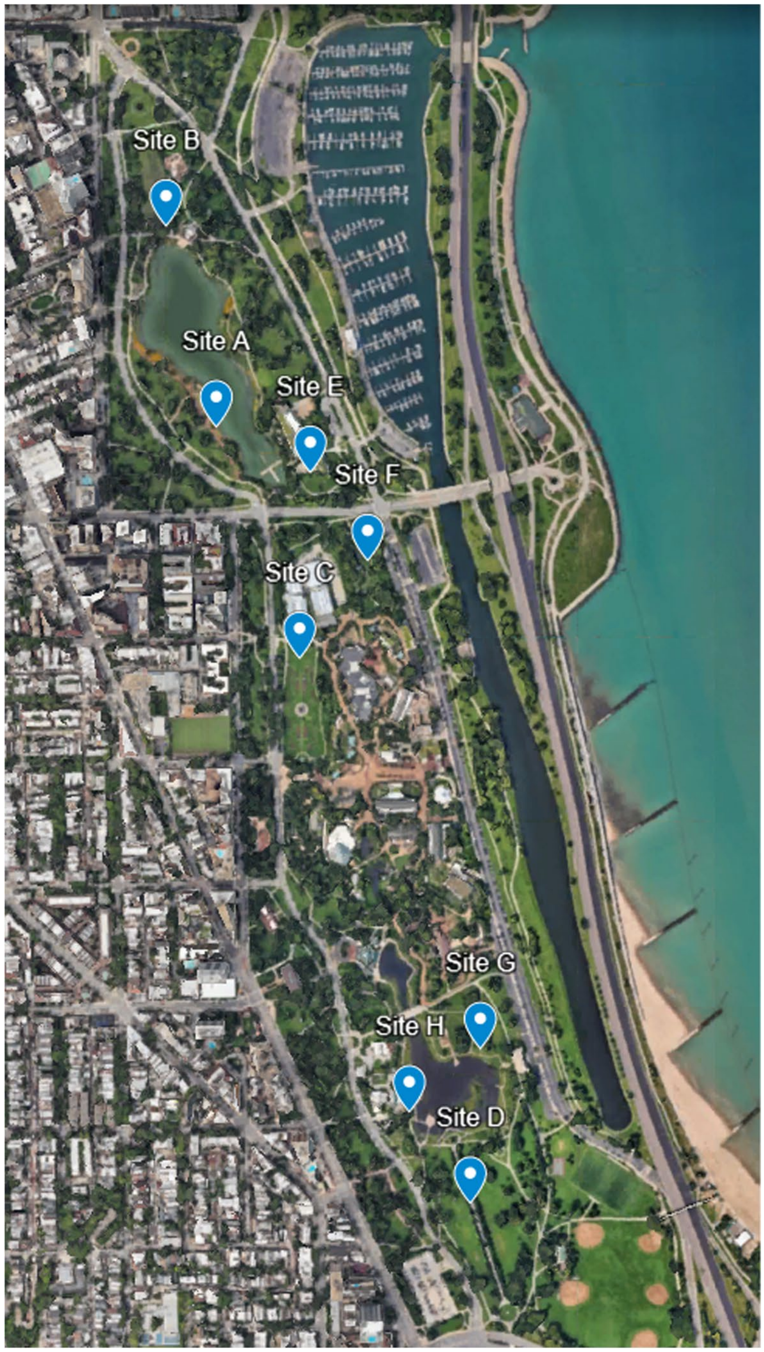

Fig. 1 Map of study sites in Lincoln Park, Chicago, USA

species of concern monitored. Rather, management was adjusted if a species of conservation value was observed on-site during a visit.

Open lawns were mowed weekly during the growing season, and tree trimming was done as needed, prioritizing ash tree removals and trimming tree hazards. Vegetation was not watered nor were any pesticides or herbicides applied in turf lawn areas (L. Umek, personal communication, February 9, 2021).

Fig. 2 Study site images: open Sites **A** and **B** at North Pond, Site **C** in the mid area, and Site **D** at South Pond; restored Site **E** at North Pond (completed 2001), Site **F** at the Lily Pool (completed 2002), and Sites **G** and **H** at South Pond (completed 2010). Photos A, B, D, E, and H from Google Maps

Sites G and H in South Pond were overseen by another nonprofit organization and involved more proactive wildlife management by staff with volunteer support. The initial restorations were completed in 2010 to serve educational purposes, improve water quality, and enhance habitat for wildlife. Semi-annual vegetation assessments had been conducted since 2016 and various fauna were routinely monitored (Huck 2019). According to a recent report, the South Pond area supported approximately 160 species of birds, seven species of bat, and various frogs, rodents, insects, and mammals (Huck 2019). Bird species present during the migratory season were reported from 2012 to 2015, but no published data exist for the last six years.

Our understanding of management objectives for the natural area sites and their implementation was enhanced through extensive discussions with site managers, as well as an evaluation of annual reports and relevant published literature. Although their objectives for natural areas are holistic, they make no explicit reference to soundscape management.

Data collection

Audio was recorded at each of the eight sites three times in 2019 on a bimonthly schedule; three rounds of data collection were coordinated in late June, August, and October. Recordings were one hour in duration, and time of day for each recording session was chosen to capture a range of anticipated biological sounds: June at sunrise, August at sundown, and October in early morning after sunrise. Recordings were made using a Zoom H1 Handy Recorder, a relatively affordable audio tool. Non-systematic field trials supported a detection radius of approximately 1-ha. The recorders were placed at or near ground level facing upwards (Fig. 3).

Data analysis

Each recording was approximately one hour in duration, then edited to a standard of 58 min. Time from 0 to 00:01:00, and from 00:59:00 to the end were removed to account for voice notes and slight discrepancies in length. This provided almost 24 h of total audio across all sites (eight) and months (June, August, October), stored in wav format. Due to the large size of the wav files, the software required the files to be broken into four clips. Acoustic indices were applied on each audio file; avian presence and biophonic "silence" analyses (see below) were confined to partial audio files due to the demands of manual listening. Each recording was subsampled in three 10-min time blocks, providing 12 h of total audio across all sites and months. Site was treated as a random effect as subsamples were repeated measures, not independent replicates.

Fig. 3 Zoom H1 Handy Recorder setup

Audio was filtered to reduce non-biophonic sound in Audacity using a high-pass filter of 1.5 kHz with a 6 dB rollover. This did not eliminate background noise but allowed animal-emitted sounds to be more prominent in recordings as these tend to be present at frequencies higher than 2 kHz while anthrophony (sounds from humans) tends to exist at lower frequencies (Fuller et al., 2015). Although the generally poor quality of urban soundscapes makes the use of acoustic indices more difficult (Fairbrass et al. 2017b), we employed them alongside other analytical methods. To increase their effectiveness, we used a background noise filter to achieve a more meaningful application of these indices.

The recording tool we used produced data for a left and right channel as the device has two adjacent microphones. In most cases these channels yielded almost identical results although we note below where there are small discrepancies.

Avian presence and species of concern

Given the dominance of bird sounds in the recordings (approximately 84% of the documented sounds), we chose to focus solely on avian species for this analysis, excluding a small set of amphibian, insect, and mammal sounds. The project lead (A.P.) and at least one other team member met in person weekly to collectively listen and manually record the presence of distinct songs and calls. This was done to increase consistency and reliability in identification since all listeners initially possessed limited auditory species identification skills. Due to COVID-19 restrictions, the process transitioned mid-way to remote listening; team members listened individually following the same procedures, then the project lead, who also listened independently, collected, and synthesized all data. Species were identified retroactively with the assistance of several local experts. Species of concern were identified from a region-specific guide from the Bird Conservation Network and Chicago Wilderness (Network 2014).

To capture frequency of presence, species were given a code based on how many samples they occurred in across the month at each site. If a species was recorded only in one subsample, for that month at that site it received a “1”; if it were present in all subsampled time blocks, it received a “3”. Count values were then totaled for each month at each site. Grouped data by site allowed for comparisons between management treatments. Count by subsample allowed us to discern how frequently a particular species was recorded during a month.

Acoustic indices (AI)

Using R version 3.5.2 (R Core Team 2018) and the RStudio interface (RStudio Team 2020), the packages *tuneR* (Ligges et al. 2018) and *soundecology* (Villanueva-Rivera and Pijanowski 2018) were used to calculate three indices: Acoustic Complexity Index (ACI) to quantify the variation and intensity of sounds (Pieretti et al. 2011), Bioacoustic Index which measures avian abundance via certain frequencies (BIO) (Boelman et al. 2007), and the Normalized Difference Soundscape Index (NDSI) which computes the ratio of anthrophony and biophony (see Remote Environmental Assessment Laboratory (REAL); (Kasten et al. 2012). Boxplots were generated to visually represent the relationship between indices, management (open or restored), and month (June, August, or October).

When data was normal (ACI and BIO), we used generalized linear mixed models (glmms) with a gaussian distribution. For NDSI, a gamma distribution within the glmm was more appropriate. We used the packages *nlme* (Pinheiro et al. 2018) and *MASS* (Venables and Ripley 2002) for model creation. Values were given on a scale from -1 to 1, which were then converted to a scale of 0 to 2. Management, month, and their interaction were included as fixed factors with site as a random factor to account for multiple clips within each site. Pairwise comparisons, to illustrate differences between the interactions by both management and by month, were completed using Tukey's adjustment and results shown with compact letter display in packages *emmeans* (Lenth 2021) and *multcompView* (Graves et al. 2019). Different letters indicate significant differences.

Biophonic absence

Biophony reflects the collective sound produced by all living organisms in a habitat. As we use the term here, biophonic absence is defined as the lack of discernable sounds emitted by non-human animals. Anthropogenic or weather-related sound was still present and may have overwhelmed biological sound. Unlike the avian-focused presence analysis, biophonic sound considered all recorded faunal sounds. The only animal sound captured in the recordings and grouped with other anthropogenic sound was dog barking as it is interwoven with the presence of people. Because recordings were captured during times with assumed high biophonic activity (e.g., sunrise in spring), higher durations of absence are assumed to represent a soundscape that is either unavailable or undesirable for the activity of sound-emitting fauna.

Filtered audio was also used here for ease of listening and to accurately capture biophonic sound. Using headphones, each clip was listened to with a stopwatch which ran for any duration of biophonic absence. If a clip had zero biophony, it would have a biophonic absence duration of 00:10:00 (hour:minute:second); if zero biophony was recorded in every clip and month for an entire site, that site would have a maximum duration of 01:30:00. Because we evaluated biophonic absence manually, the duration values listed for each clip are somewhat approximate. A non-parametric Wilcoxon-signed rank test was run in R using total seconds per site to determine whether there were statistical differences in biophonic absence with respect to management.

Results

Avian presence and occurrence of species of concern

Sixty-seven distinct bird songs and calls were described across all sites, 51 of which could be identified at the species level. Some species had multiple distinct sounds. For example, both the chirp and song of the cardinal (*C. cardinalis*) were identified. Thirty species were identified in all with more individual species identified across all restored sites (open = 26; restored = 29). Presence counts (for definition, see methods) by subsamples showed a more substantial difference (open = 174, restored = 267). The starkest contrast in presence counts between open and restored plots was in October (open = 79; restored = 168); differences between these management types were only slight in June (open = 70; restored = 74); whereas the presence counts were recorded in August were low and did not differ with management type (both management types = 25, Fig. 4).

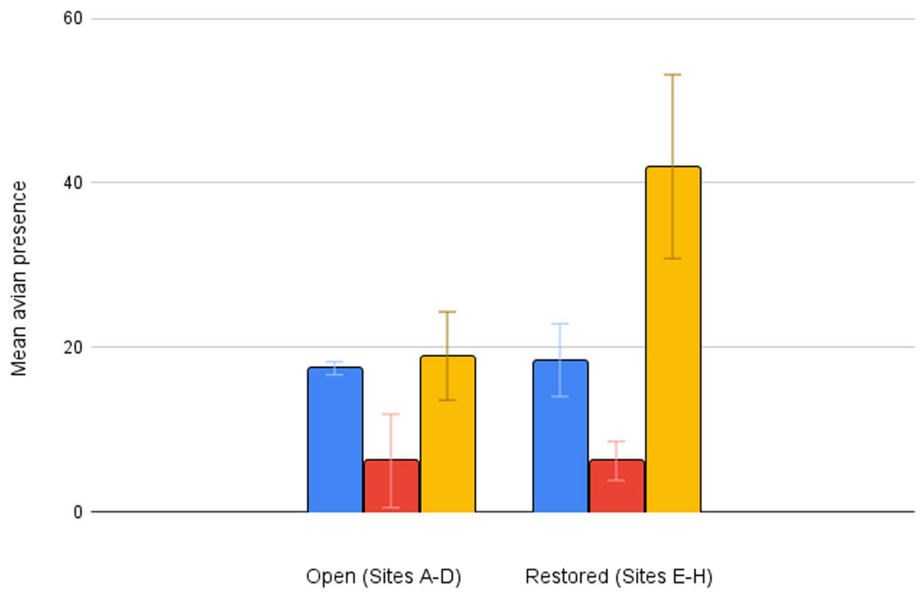

Fig. 4 Mean avian presence counts across all subsamples grouped by management treatment open (Sites **A**-**D**) and restored (Sites **E**–**H**), per month June (blue), August (red), and October (yellow) with standard deviation

Three identified species were listed as birds of concern by the Bird Conservation Network: northern flicker (*Colaptes auratus*), marsh wren (*Cistothorus palustris*), and black-throated green warbler (*Setophaga virens*). *C. auratus* was counted three times in one open site and twice at one restored site; *C. palustris* was recorded once at one open site and three times across two restored sites; *S. virens* was counted only at two restored sites, once at each. (See Table 1 for species presence across sites and managements.)

Acoustic indices

Acoustic complexity index (ACI)

ACI values were normally distributed. Results were the same for both left and right channels of the recorders. ACI was significantly higher in restored areas, and there were significant effects when months were compared (Table 2 and Fig. 5). Figure 5 illustrates pairwise comparisons by month. Boxplots with different letters indicate a significant difference between ACI values. The June and October ACIs in open lawn were significantly higher than August ACI for the same treatment. August had the lowest mean ACI in the open treatment.

Bioacoustic index (BIO)

Bioacoustic values were normally distributed. Since there were some differences detected between the microphone channels on our recorders, we therefore present results from both channels. There was no significant difference in biophony between open and restored areas, though biophony varied significantly by month (Tables 3 & 4). August had a significantly higher biophony as measured by this index than both June and October in open lawn. October had the lowest biophony in open lawn. This was significantly lower than June in the right channel, but not the left channel (Fig. 6). August

Table 1 Species presence across sites

		Open				Restored			
Species (common name)	**Site:**	*A*	*B*	*C*	*D*	*E*	*F*	*G*	*H*
	American crow	5	4	2	4	7	2	4	5
	American goldfinch			3		3	1	3	3
	American robin			3	1		3	4	3
	Baltimore oriole	2	2	2	2	1		2	3
	Barn swallow	2	1		4	1	3	1	2
	Black-capped chickadee†					1	2	1	2
	Black-throated green warbler*†						1		2
	Blue jay	3	3		2			1	2
	Canada goose	7	5	2	4		3	4	4
	Chipping sparrow	2		3	2	2	3	2	2
	Common grackle*		3			1	1	5	5
	Dark-eyed junco	2	3	3	5	4	2	4	6
	Downy woodpecker†							3	2
	Eastern bluebird	2		2	3		2	3	3
	European starling	4	2	2	2	1	3	5	7
	Golden-crowned kinglet†					3			3
	Gull (Larus spp.)	3	1	3	4	2	3	3	2
	House sparrow			6	4	5	9	6	6
	House wren				1	3	1	2	
	Mallard duck	4	2	3	1	5	4	2	4
	Marsh wren*			1			1	2	
	Mourning dove □				1				
	Northern cardinal	1			1	3	2		
	Northern flicker*			3			2		
	Northern rough-winged swallow	2		5	3	1	1	6	6
	Pine siskin			1	1	2	1	3	2
	Red-winged blackbird	3	3		4	5	2	6	5
	Rock pigeon			1					1
	White-breasted nuthatch				1			1	
	White-throated sparrow		3	3	2	3	3	3	3
	Total presence counts:	**174**				**267**			

Numbers represent how many subsamples a species was counted at that site across all months (maximum = 9). Star symbol (*) represents species of conservation concern; dagger symbol (†) represents species found only at restored sites; square symbol (□) represents species found only at open sites

was also significantly different from June and October in the restored areas, but not due to higher biophony (Fig. 6).

Normalized Difference Soundscape Index (NDSI)

We found that a gamma distribution was most appropriate for representing NDSI values. Significance results were the same for both left and right channels; we present results for the left channel below. There was no significant difference in anthrophony levels between open and restored treatments (Table 5). Anthrophony in October was significantly higher than in June and August (in both restored and open treatments). August had the highest proportion of biophony; this was significantly higher than June and October in both open and restored treatments (Fig. 7).

Table 2 Best-fit general linear mixed model for estimating effects of prescribed management and month on ACI (Acoustic complexity index) at 8 study sites (left channel)

ACI model: Left channel					
	Value	**Std. Error**	**DF**	**T-val**	***P*-val**
(Intercept)	25,909.373	253.3622	84	102.26220	0.0000
MgmtRestored	953.448	358.3082	6	2.66097	0.0375*
MonthJune	755.642	236.0950	84	3.20058	0.0019*
MonthOctober	922.999	236.0950	84	3.90944	0.0002*
MgmtRestored:MonthJune	-1050.232	333.8887	84	-3.14546	0.0023*
MgmtRestored:MonthOctober	-1157.783	333.8887	84	-3.46757	0.0008*
Random effects = Site					
(Intercept)	381.1665				
Residual StdDev	667.7774				

AIC = 1468.304; log likelihood = -726.1518

Significance is shown at $p < 0.05$ (*)

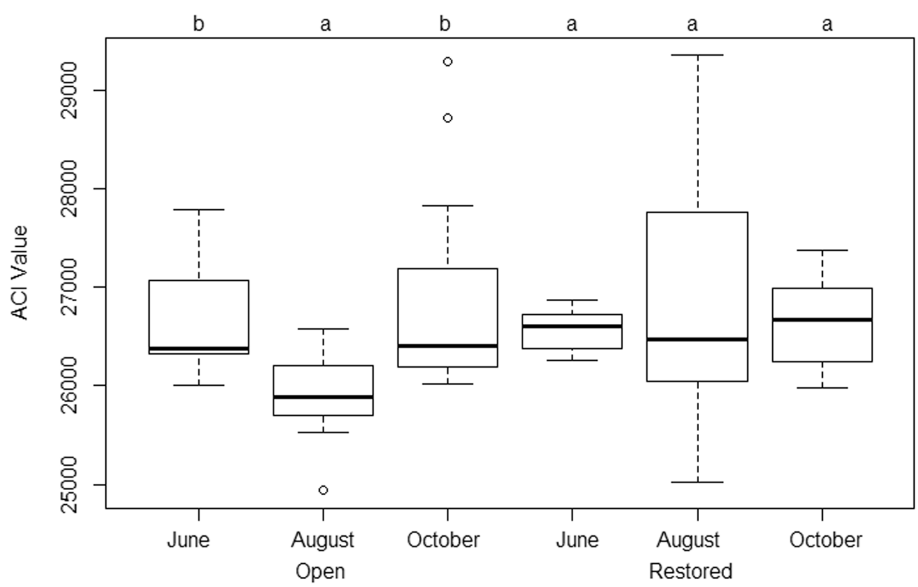

Fig. 5 Showing left channel: boxplots for ACI (Acoustic complexity index) with respect to management by month in August, June, and October with compact letter displays. Sites with dissimilar letters are significantly different; in open sites, June and October are significantly different than August

Biophonic absence

There was significantly more biophonic absence (total duration in seconds per site) in open lawn based on Wilcoxon-signed rank tests ($p = 0.02857$, Fig. 8). Biophonic absence at open sites in June (00:04:27), August (00:07:04), and October (01:12:03), were consistently higher than absence at restored sites in June (00:00:56), August (00:00:01), and October (00:18:12). Total absence across all open sites was 01:23:34 and 00:19:09 across restored sites. Site E, a North Pond restored site, had the lowest total duration of absence across all recordings from all months at 00:00:21. Site B, a North Pond open site, had the highest total duration of absence at 00:25:11.

Table 3 Best-fit generalized linear mixed model for estimating effects of management and month on BIO (Bioacoustic Index) (left channel)

BIO model: Left channel					
	Value	**Std. Error**	**DF**	**T-val**	**P-val**
(Intercept)	8.863026	0.8077869	84	10.971985	0.0000
MgmtRestored	-0.812163	1.1423832	6	-0.710938	0.5038
MonthJune	-1.352125	0.6329096	84	-2.136363	0.0356*
MonthOctober	-4.124998	0.6329096	84	-6.517516	0.0000
MgmtRestored:MonthJune	1.278821	0.8950693	84	1.428740	0.1568
MgmtRestored:MonthOctober	1.541919	0.8950693	84	1.722681	0.0886
Random effects = Site					
(Intercept)	1.344965				
Residual StdDev	1.790139				

AIC = 405.1617; log likelihood = -194.5809

Significance shown at $p < 0.05$ (*)

Table 4 Best-fit generalized linear mixed model for estimating effects of prescribed management and month on BIO (Bioacoustic Index) (right channel)

BIO index model: Right channel					
	Value	**Std. Error**	**DF**	**T-val**	***P*-val**
(Intercept)	9.001337	0.9182473	84	9.802737	0.0000
MgmtRestored	-1.290392	1.2985978	6	-0.993681	0.3587
MonthJune	-1.750745	0.6033855	84	-2.901537	0.0047*
MonthOctober	-4.292872	0.6033855	84	-7.114642	0.0000
MgmtRestored:MonthJune	2.698249	0.8533160	84	3.162075	0.0022*
MgmtRestored:MonthOctober	2.369028	0.8533160	84	2.776261	0.0068*
Random effects = Site					
(Intercept)	1.626212				
Residual StdDev	1.706632				

AIC = 399.1155; log likelihood = -191.5577

Significance shown at $p < 0.05$ (*)

Discussion

Relatively small plots (less than 1-ha) that are managed for contrasting outcomes—recreational/aesthetic use versus ecological benefits—within a large urban park (481-ha) are shown in this paper to differ in their biotic communities as reflected in the richness of the acoustic environments. The differences revealed in this study confirm the feasibility of achieving at least some conservation goals by means of vegetation management on small spatial scales (that is, by means of urban tinkering).

The pronounced patterns in soundscapes presented in this paper were detected using relatively inexpensive hand-held acoustic devices (Zoom H1 Handy Recorders), ones that should be affordable even for land management agencies operating on quite tight budgets. Furthermore, the analysis of soundscapes that we conducted can be successfully undertaken after a small amount of training. These factors in combination should encourage a greater willingness by managers to undertake routine monitoring of conservation projects in urban environments.

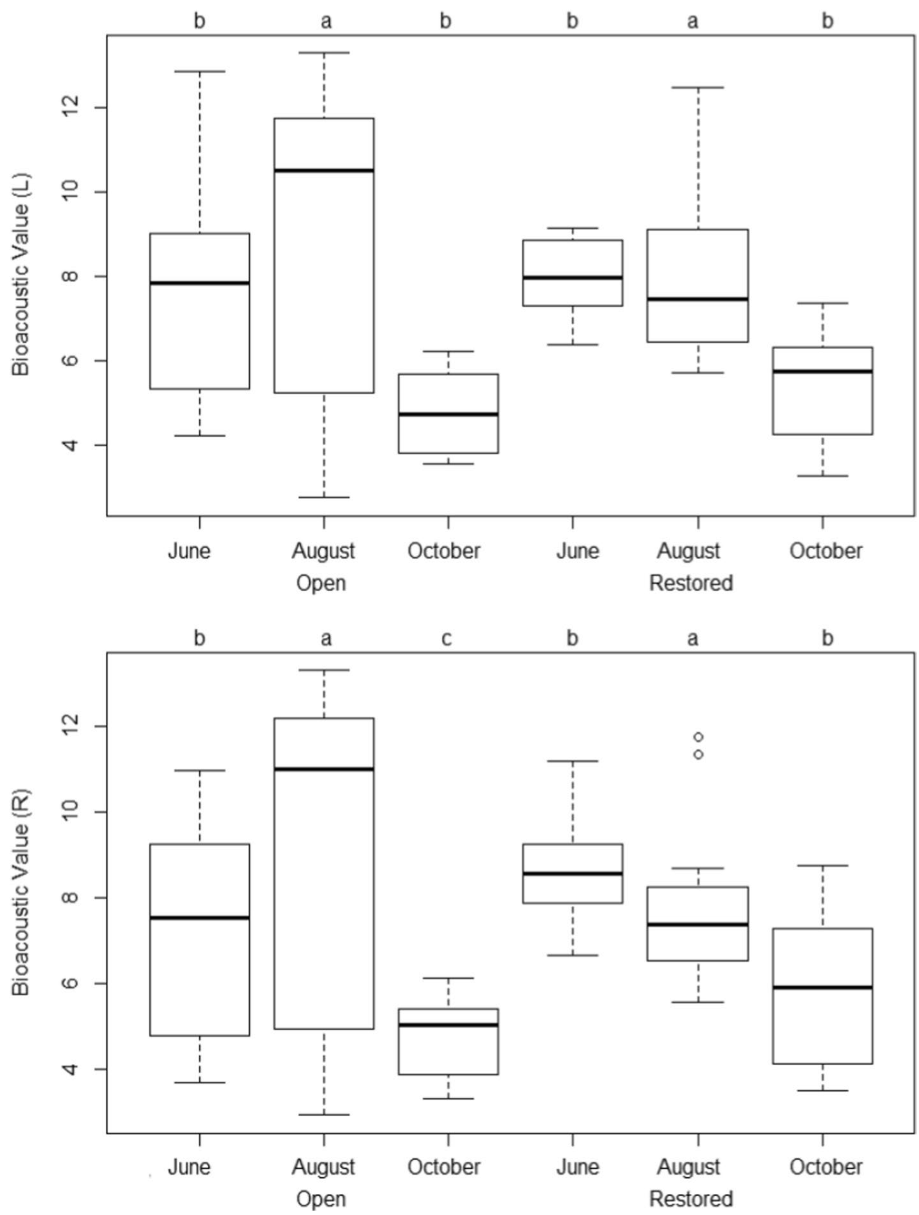

Fig. 6 Boxplots for BIO (Bioacoustic Index) with respect to management treatment open or restored (upper) and management by month in August, June, and October with compact letter displays (lower). Sites with dissimilar letters are significantly different

Monitoring of bird species

Although, as we will discuss in some detail below, we applied a variety of acoustic indices to soundscape recordings to gain a holistic insight into the composition of the entire community of sound-emitting fauna, we singled out birds for particular attention in this study. In addition to bird calls being a dominant biotic contributor to soundscapes in urban environments, we concentrated on birds because they are both sensitive to a range of stressors associated with congested urban environments but also have been shown to benefit from conservation-oriented management aimed at mitigating such stressors (Pieretti et al. 2011; Ortega 2012). Because of

Table 5 Best-fit generalized linear mixed model for estimating effects of management and month on NDSI (Normalized Difference Soundscape Index) (left channel)

NDSI model: Left channel					
	Value	**Std. Error**	**DF**	**T-val**	**P-val**
(Intercept)	0.5819522	0.1511626	14	3.849843	0.0018*
MgmtRestored	0.1260186	0.2240545	6	0.562446	0.5942
MonthJune	0.86893632	0.3126274	14	2.780829	0.0147*
MonthOctober	2.9179694	0.6898153	14	4.230073	0.0008*
Random effects = Site					
(Intercept)	2.675902e-05				
Residual StdDev	0.4921838				

Significance shown at $p < 0.05$ (*)

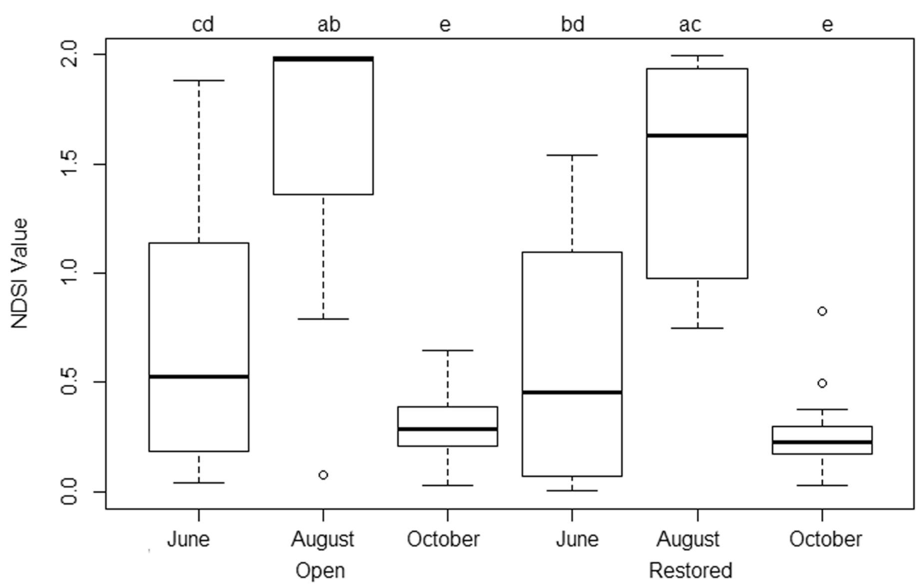

Fig. 7 Showing left channel: boxplots for NDSI (Normalized Difference Soundscape Index) (0 = all anthrophony; 2 = all biophony) with respect to management by month in August, June, and October with compact letter displays. Sites with dissimilar letters are significantly different; management treatments were not significantly different but there were differences in biophony levels by month

these factors, birds serve as good indicators in evaluating the success of conservation efforts (Padoa-Schioppa et al. 2006; Mekonen 2017). For example, both Black-capped (*Poecile atricapillus*) and Mountain Chickadees (*Poecile gambeli*) are known to be sensitive to ambient urban noise (see LaZerte et al. (2015), and yet Black-capped Chickadees were identified in our study at all four sites where restorative management has been implemented but at none of the sites that are managed otherwise. This illustrates that restorative management even on the very small scale that we evaluated can benefit certain faunal elements. Black-capped Chickadees are one of four species detected only at restored sites (see Table 1). In contrast, only one species, Mourning Dove (*Zenaida macroura*), is found exclusively at an open plot and not at all in restored plots.

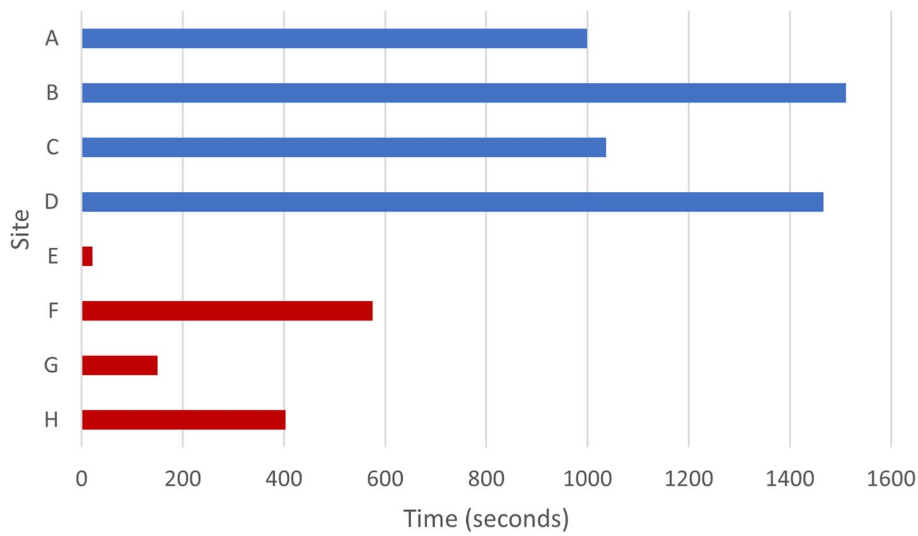

Fig. 8 Biophonic absence for open (blue; **A**-**D**) and restored (red; **E**–**H**) sites per each month of recording. Each bar represents one 10-min clip

In general, our monitoring revealed that bird activity was greater than fifty percent more prevalent in restored plots than in open ones. Though these results do not indicate which aspects of vegetation manipulation in particular promoted enhanced avian activity at these sites, it affirms the long-standing observations concerning a correlation between the structure of a habitat and its bird communities (Karr and Roth 1971; Erdelen 1984). Since vegetation management in urban areas is often aimed at promoting the broader biotic community, we have shown that not only is bird activity enhanced by this management but also that monitoring using rudimentary soundscape analysis can be used to support such claims.

If the sites under restorative management that we monitored in Lincoln Park support state or federally threatened species, these were not captured in our recordings. It is worth noting that a small population of Black-crowned Night-herons (*Nycticorax nycticorax*)—a species considered endangered in Illinois—is actively monitored here but calls of this species were not captured in our recordings even though these birds inhabit plots within several hundred meters of some of our monitoring areas. The fact that the hand-held recorders that we employed did not detect calls from this species may confirm that the spatial range of our recordings are limited to the spatial extent of the areas we targeted for monitoring. Alternatively, it may have been that the frequency of our recordings was too limited to capture their calls. If the monitoring of rarer species is an evaluative priority, soundscape sampling protocols can be designed to explicitly capture their activity (Acevedo and Villanueva-Rivera 2006; Darras et al. 2019).

While only a small number of the identified species in this study are of notable conservation concern, it is possible that any of the sixteen calls left undetermined by the knowledgeable listeners that we recruited to assist us in this study may represent calls from species considered rare in the Chicagoland area. The presence of such calls was more prevalent in samples at restored areas ($n = 52$) than open ones ($n = 33$).

Soundscape indices

In this study we applied the ACI to holistically evaluate soundscapes at our plots. The ACI summarizes the biotic complexity of soundscapes while attempting to minimize anthropogenic intrusions in recordings (Bateman and Uzal 2022). In our study the AIC analysis revealed differences associated with management. Generally, there was a greater variation in the AIC scores of restored areas than in open ones. Although it was clear that there are differences between the soundscapes in different seasons, comparisons across our sampling dates must be interpreted with caution since we recorded at different times of the day in each season. We chose these times deliberately, considering what we already knew to be peak biophony at different times of the year, but also to get some insights into daily fluctuations of soundscapes in Lincoln Park to serve as a basis for future studies. Had we access to a larger crew and more equipment we would have increased the frequency of our recordings to provide additional comparisons—something we'd recommend for future work.

Spring soundscapes in Lincoln Park were influenced by avian migration earlier in the season, whereas August soundscapes are dominated at most sites by insects. The significantly lower AIC scores in open areas in August is worth noting: lawn spaces with few interspersed shrubs or trees reduced the intensity of the soundscape. It is also worth noting that since the August and October sampling was done later in the day, anthropogenic noise was more prevalent, and this affected the calculation of the ACI. No recordings were entirely free from anthrophony. Applying the high-pass filter allowed for reduction of some noise in the recordings to accentuate sounds of biogenic origin. An even stricter filter would have reduced these influences further, but this would have reduced calls from some animals, especially those at greater distance from our recorders (Fairbrass et al. 2017a). Though Joo et al. (2011) reported a correlation between diversity and the acoustic index they used, we did not find a relationship between the vegetation composition and ACI. Incorporation of additional indices such as those measuring soundscape intensity (Sueur et al., 2014) can be helpful in urban studies.

Interestingly, neither the BIO, nor the NDSI differed between the management types, though both confirmed the pronounced differences between recording from different months. Since each of these indices reveal aspects of the soundscape (frequencies used by avifauna in the case of the BIO, and ratio of the energy in the anthrophony and biophony frequency bands in the case of NDSI), we urge circumspection to those analyzing this sort of data within urban environments.

Biophonic absence

One of the more pronounced differences between open and restored management treatments was revealed through the analysis of the absence of biophony (what we are calling "biophonic absence"). In August, much of the soundscape across Lincoln Park was dominated by the insect chorus—cicadas being the most prevalent component—but open sites were less likely to support these communities than restored sites, and this was detected in our analysis. In June and October, bird songs and calls constituted most of the biophony, though there was overall less biophony in June when the recordings were captured during dawn chorus. Though October had a higher diversity of sound, there was less calling from each species and substantially more anthropogenic sound from surrounding traffic on nearby minor and major roadways, helicopters, lawn mowers, and park visitors. Measuring the *absence* of biophony seems relatively rare in soundscapes studies, though our work reveals how meaningful it may be.

Methodological limitations and prospects for future work

Despite the strong patterns revealed in this study, some methodological limitations need to be underscored. Though soundscape analysis is appealing as a tool for relatively rapid analysis of restoration outcomes, nevertheless, the retroactive identification of bird species proved to be both challenging and time-intensive. These difficulties were accentuated as we did at least some of this analysis during a period of restricted social contact because of the COVID-19 pandemic. Acoustic monitoring of individual sites might have been improved by prior designation of those target species most aligned with the goals of restoration projects. Auto-detection software exists for quantifying unique calls, and technology and methodologies are advancing within the field at a rapid pace. Guidelines for improving usage with statistical indices are also accessible (Eldridge et al. 2016, Bradfer-Lawrence et al. 2019) as well as protocols for relating avian species identification to soundscape metrics using large datasets (Gage et al. 2017). Audio recording confers the ability to establish long-term data collection; this is a clear benefit for using recorded sounds as a longitudinally applied indicator of ecosystem health (Pijanowski et al. 2011; Tucker et al. 2014).

In this exploratory study, the work was both intensive and time-consuming, but setting specific targets for analysis—which we are recommending for future studies—will reduce time commitments. Use of citizen science programs for gathering and analyzing acoustic data has been shown to be highly productive (Penone et al. 2013; Newson et al. 2015; Pauwels et al. 2019; Gili et al. 2020). Affordability can be further enhanced through volunteer collaboration with land managers of municipal spaces. Our study confirms that some of this work can be conducted inexpensively and without extensive prior identification skills.

It is worth noting that the management goals of the sites we monitored did not explicitly prioritize the soundscape, but it is apparent that restored areas within an urban park space can achieve higher biodiversity and a richer acoustic environment than is found in adjacent unrestored areas dominated by lawn. The availability of low cost bioacoustic monitoring should encourage an increasing number of such projects. There are certainly many areas in urban environments that might prove to be exceptional candidates for further conservation-oriented management.

Acoustic environments impact the well-being of both wildlife and people, so restoring urban spaces intentionally for healthy soundscapes can deliver benefits to all city dwellers (Francis et al. 2017). Spaces where park visitors can experience more natural soundscapes have been shown to enhance conservation ethics by promoting a connection to nature, and that ultimately benefits ecological health (Pijanowski et al. 2011). By evaluating soundscapes under management, urban stewards can better understand the outcomes of interventions and can adaptively respond to ecosystems as they develop.

Conclusions

The insight that cities collectively exert a planet-wide impact (a phenomenon reflected in what sociologist Henri Lefebvre referred to as the "urban society," and that approximates what urbanist Constantinos Doxiadis later called an eucumenopolis—in effect, one global interconnected city, see Doxiadis 1975; Lefebvre 2003)) means that urban conservation and sustainability strategies may be key to the flourishing of human communities in the transition to the "good anthropocene" (McPhearson et al. 2021). The prospect of steering humanity away from unsustainable global trajectories and along a path to achieving sustainability goals should incentivize the work of municipal authorities charged with environmental governance, as well as motivating urban landscape managers (Hess and McKane 2021).

Opportunities for improving global environmental quality and enhancing biodiversity conservation accompanies the growing recognition that human well-being is improved by exposure to high-quality urban nature. Combined, these factors can provide an additional impetus for prioritizing access to appropriately managed urban green space (Standish et al. 2013; Kuo 2015). However, the degree to which urban projects, typically deployed on small plots, actually achieve their objectives on a variety of scales depends upon a willingness to monitor the success of projects and to use the results of such monitoring to direct future management.

We have shown that soundscape monitoring, even at a fairly rudimentary level, can provide reliable, replicable, and interesting results that can supplement more extensive and expensive project evaluation.

Abbreviations

ACI Acoustic Complexity Index
BIO Bioacoustic Index
NDSI Normalized Difference Soundscape Index

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Authors' contributions

The research was conceived by Liam Heneghan, (LH) and Allison Preble (AP). The research was conducted by AP with support of LH. Data analysis conducted by AP in collaboration with Christie Klimas (CK). First draft completed by AP. LH and CK edited and prepared the review for publication.

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Data availability

Data available upon request.

Declarations

Competing interests

None.

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