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Listen but do not touch: using a smartphone acoustic device to investigate bat activity, with implications for community-based monitoring

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The monitoring of bats across the world is mostly conducted using invasive mist-netting, whereby vertical nets are placed to capture bats mid-flight. Many studies have demonstrated how this approach causes sampling bias, is labor-intensive and increases the risk of white-nose syndrome fungus, *Pseudogymnoascus destructans*, transmission among bats. Increasingly, acoustic devices are being employed to collect data on bat activity and richness. Community-based monitoring is an important data collection source for bat monitoring programs in countries such as the UK (National Bat Monitoring Program), whereby walking bat transects are conducted using bat detectors. Since the launch of smartphone devices to record and auto-identify bat echolocation calls, the quality of data collection that community members can collect has increased significantly, however, this approach is seldom used to generate data in scientific studies. In our study, we have showcased how our study design paired with state of the art acoustic monitoring devices, can be applied to community-based monitoring of bats across the world. Through employing smartphone acoustic devices, we have determined how primary and secondary vegetation cover are predictors of bat species occurrence and identified the importance of riverine and deciduous swamp habitats for rare bat species in southwestern Ontario.

Key words: acoustic monitoring, Echo Meter Touch 2, bat foraging, primary vegetation, secondary vegetation, *Myotis lucifugus*, *Lasiurus borealis*

INTRODUCTION

Insectivorous bats worldwide are renowned for the ecosystem services that they provide, in the form of agricultural insect control (Boyles *et al.*, 2011; Maine and Boyles, 2015; Kemp *et al.*, 2019; Linden *et al.*, 2019; Puig-Montserrat *et al.*, 2020). Through providing biocontrol, bats in the cotton-dominated agroecosystem alone are valued at $22.9 billion/year for the pest services they provide in the US (Boyles *et al.*, 2011; Morningstar *et al.*, 2019). In addition, bats are important consumers of mosquitoes, families of which are known to transmit disease to humans (Clare *et al.*, 2011; Wray *et al.*, 2018; Morningstar *et al.*, 2019). The North American decline of particular insectivorous bat species, such as *Myotis lucifugus*, *Perimyotis subflavus* and *Myotis septentrionalis* (Ministry of Natural Resources and Forestry, 2018), combined with the negative perceptions of bats (particularly considering the current COVID-19 pandemic — Maxmen, 2017), has large implications for food sustainability, human health and the long-term persistence of bat species (Boyles *et al.*, 2011; Tuttle, 2017; Wray *et al.*, 2018).

The surveillance of bats to monitor changes in population, roost locations, population demographics and to identify key foraging areas, is currently conducted using a range of methods. The main approaches (among others) are hibernation counts, mist netting and employing ultrasonic acoustic sensors (Constantine, 1958; Kuenzi and Morrison, 1998; van Schaik *et al.*, 2015). Mist netting involves deployment of numerous nets throughout habitats of interest, designed to capture bats mid-flight (Carroll *et al.*, 2002). There are many known drawbacks of mist netting, including labor intensity (Coleman, 2013), species capture bias (Carroll *et al.*, 2002; Larsen *et al.*, 2007; Geluso and Geluso, 2012; Marques *et al.*, 2013; Loeb *et al.*, 2015), high risk of *Pseudogymnoascus destructans* transmission (Coleman, 2013) and inaccessibility to members of the general public to participate (Loeb *et al.*, 2015). Species avoidance and long-term behavioural effects of physically trapping bats in mist nets, leads to issues with repeatability and representativeness of mist net surveys (Freyman *et al.*, 2008; MacSwiney *et al.*, 2008; Marques *et al.*, 2013; Trevelin *et al.*, 2017).

Conversely, acoustic transects are deemed the non-invasive approach for generating data on bat occurrence, richness and activity patterns (Flaquer *et al.*, 2007; Rodhouse *et al.*, 2011; Skalak *et al.*, 2012). This approach can either be passive (i.e. stationary recording stations) or active (i.e. vehicle- or walking-based transects with acoustic detectors) (Johnson *et al.*, 2002; Coleman *et al.*, 2014; Faure-Lacroix *et al.*, 2019), and involves recording ultrasonic calls of bats and identifying species from resulting spectrograms. With the advancement of technology, companies have developed sophisticated devices and software to enable the automated identification of bat calls (Christmann, 2014). This development has resulted in an increase in studies applying acoustic devices for monitoring bat occurrence (Jung *et al.*, 2012; Banner *et al.*, 2018; Klotz *et al.*, 2019), species richness (Wickramasinghe *et al.*, 2003; Skalak *et al.*, 2012; Blackburn and Unger, 2019), roost sites (Revilla-Martín *et al.*, 2020) and foraging activity (Menzel *et al.*, 2005; Krauel and LeBuhn, 2016). With the often lower cost associated with smartphone acoustic devices versus passive recorders (Hill *et al.*, 2018), a major result of acoustic technology advancement for bat research is the possibility of public inclusion for collecting data (Buckman-Sewald *et al.*, 2014; Barlow *et al.*, 2015; Newson *et al.*, 2015, 2017; Ceccaroni *et al.*, 2019). Despite the year-round benefits of using passive acoustic recorders, using smartphone devices enables both seasonal bat data collection and local community engagement, which is important for achieving long-term community bat stewardship (Beeker *et al.*, 2013; Kellner, 2020). The National Bat Monitoring Program (NBMP) employs community-based monitoring of bats, using acoustic bat detectors in active walking transects across Great Britain (Barlow *et al.*, 2015). The NBMP is one of very few examples where community-based monitoring is being directly utilized to generate much-needed long-term data on bat population trends.

Similar to mist netting, acoustic monitoring is not without its drawbacks. Passive versus active acoustic transects can result in different patterns of bat activity (Faure-Lacroix *et al.*, 2019) and species-specific differences in detection (Johnson *et al.*, 2002; Coleman *et al.*, 2014; Torrez *et al.*, 2017; Teets *et al.*, 2019). Acoustic monitoring does have however significant benefits over mist netting, including the non-invasive nature of sampling and opportunity for non-specialists to contribute to data collection. Whilst some data can only currently be attained using mist netting, including sex and life stage of the bat (Weller and Lee, 2007), due to the previously mentioned shortcomings of mist netting, it has been suggested that invasive mist netting is unsuitable for large-scale long-term bat monitoring programs (Frick, 2013; Loeb *et al.*, 2015).

Monitoring of bat populations across Canada has become particularly important following outbreaks of white-nose syndrome (hereafter WNS), particularly across southern Ontario (Morningstar *et al.*, 2019). WNS, caused by the fungus *Pseudogymnoascus destructans*, has caused catastrophic population declines of certain bat species across Canada (Davy *et al.*, 2017). In southern Ontario, *M. lucifugus*, *P. subflavus* and *M. septentrionalis* are the three species most affected by WNS, caused by the fungus *Pseudogymnoascus destructans*, which led to the three species being listed as 'Endangered' and included on the Species at Risk in Ontario list (Ministry of Natural Resources and Forestry, 2018). As part of Ontario's White-nose Syndrome Response Plan and the specific recovery strategies for each species (Ministry of Natural Resources and Forestry, 2015), research goals including the filling of knowledge gaps on species distribution and identification of critical foraging habitat are listed as priority (Ministry of the Environment, Conservation and Parks, 2019). The cost of investigating post-WNS effects on bat populations in the US was estimated at $45 billion USD (Beeker *et al.*, 2013), therefore partnering with community-based monitoring groups to collect bat data will ultimately reduce financial costs of post-WNS monitoring.

In areas of southwestern Ontario, including Guelph, there are significant data deficiencies for the eight Ontario bat species, particularly the endangered species. The City of Guelph is situated within the Mixedwood Plains Ecozone and is characterized by diverse mixed, deciduous and coniferous forests (Rowe, 1972). Guelph is dominated by agricultural landscapes with forest fragments (woodlands) with an estimated 12 types of vegetation communities (Puric-Mladenovic and Baird, 2017). All eight Ontario bat species are considered to exist within the Guelph region, however there is next to no data on bat occurrence in this city.

In this study, our primary aim is to assess bat species occurrence and richness across four sites in Guelph using a smartphone acoustic device. Secondly, we aim to investigate species-specific associations with habitat type (riverine versus forested) and vegetation type of site transect. Overall, we aim to demonstrate the importance and potential of involvement of non-specialists in urban bat data collection.

MATERIALS AND METHODS

Site Determination

To identify suitable sites to conduct transects, we searched iNaturalist (www.iNaturalist.org) for 'Chiroptera' observations limited to Guelph (iNaturalist, 2020). Out of the 10 observations, four were our previous casual observations and the remaining six were casual observations; five of which were observations of grounded/injured bats. Based off of these observations and consultation with a local ecologist from Dougan & Associates, we identified four forested natural and parkland areas (Riverside Park; Eramosa River Trail; Hanlon Creek Park; Mitchell -Ellis Creek Park), owned by the City of Guelph to target for our transects (Supplementary Table S1). In June 2020, we visited the four sites and conducted a casual transect with a Magenta 4 heterodyne bat detector (Magenta Electronics Ltd, UK) to determine any presence of bats and the most suitable transect route.

Study Sites

Once we had confirmed that bats were present at all four sites, we identified primary and secondary land cover for each site and classified vegetation into Ecological Land Classification (ELC) classes (Puric-Mladenovic and Baird, 2017). Two of the four sites were river systems, with the other two sites being forested areas (Supplementary Fig. S1). We determined a transect route along existing paths at each site based on existing field and waterway transect protocols from Bat Conservation Trust (Bat Conservation Trust, 2020*a*, 2020*b*), with modifications including omission of stop points. We ensured each transect route duration was a minimum for 45 minutes and a maximum of 60 minutes.

Each site was surveyed ten times from 6th July 2020–24th September 2020. These months fall within the optimal sampling period identified by the North American Bat Monitoring Program (Loeb *et al*., 2015) and is the season where peak foraging occurs (Furlonger *et al*., 1987; Kunz *et al*., 2011), due to diverse prey availability and high metabolic demands of female bats (Clare *et al*., 2014*a*, 2014*b*). Throughout the sampling period, prey availability was expected to be consistent based on bat diet studies (Henry *et al*., 2002; Clare *et al*., 2011, 2014*a*, 2014*b*). Each week, we selected days of the week to conduct transects using a randomizer (https://www.getrandomthings.com/random-day-of-the-week.php) and a list randomizer (https://www.random.org/lists/) to assign transects to days of the week. We altered transect days only if storms were forecast (four occasions). All acoustic transects were conduced the same and began 10–15 minutes after sunset. To inform bat call recordings, we obtained wind speed (mph), wind direction, ambient temperature (°C), cloud cover (%) and humidity (%) from the AccuWeather website (AccuWeather) and reported moon phase (with percentage luminosity; Prugh and Golden, 2014) for each evening we conducted a transect.

Bat Species Identification

An Echo Meter Touch 2 (Wildlife Acoustics, MA) and corresponding smartphone application was used to listen, record, and identify bats while in flight at our study sites. This ultrasonic module, which can capture frequencies up to 192 kHz, is designed to transform the echolocation calls produced by bats into human-audible sounds audio that humans can hear in real time. The associated smartphone application enables the user to identify bat species by their call without the need to physically catch and handle individuals (as it conducted when using mist-netting (Kunz and Brock, 1975; Larsen *et al*., 2007). The Wildlife Acoustics smartphone application facilitates recording and automatic identification of echolocation calls, using the Auto ID Selection tab (North American database), with 8 possible species found in Ontario. All Echo Meter Touch 2 parameters remained at default settings. Recordings which resembled noise (i.e. no clear spectrogram produced) were removed automatically. All recorded calls were analyzed for number of calls per recording, pulse ratio (percentage of individual calls or pulses positively used to identify species), time, GPS location, and date of echolocation recordings. Considering the uncertainty surrounding reliability of automatic identification of bat calls (Russo and Voigt, 2016), all recorded calls were manually examined using Kaleidoscope software (v5.2.1; Wildlife Acoustics), to ensure spectrograms from identified bats were consistent with known spectrograms for the species. After confirming bat identifications, we filtered the recordings by number of calls (minimum of 5) and, following a previous study (Blackburn and Unger, 2019), we excluded any echolocation recordings with low pulse ratios (below 50%), to increase likelihood of correct echolocation call species identification.

Statistical Analyses

For this study, we conducted a variety of diversity analyses in Rstudio (version 1.1.456) using R (version 3.5.1). Firstly, we checked the normality of the data using visual methods with the ggdensity and ggqqplot functions from the ggpubr package (Kassambara, 2020) followed by testing for normality using the Shapiro-Wilk test (Shapiro and Wilk, 1965). Because our data were not normally distributed ($W = 0.915$, $P < 0.01$), we used a conservative approach (i.e. statistics that do not depend on the assumption of equal variance), for the remainder of our analysis.

Using the vegan package (Oksanen *et al*., 2019), we investigated bat species richness per transect across the four sites. To assess if any bat species were represented more in particular sites, we visualized the abundance of recordings for each bat species using a heatmap generated from the 'geom_tile' function in the ggplot package (Wickham, 2016).

To determine bat species assemblages from the recordings, we used non-metric multi-dimensional scaling (NMDS) on Bray-Curtis dissimilarities using the metaMDS function in the vegan package. In addition, we calculated a Shephard's plot and goodness of fit for the NMDS ordination using the stressplot and goodness functions. To assess differences among groups (site, survey month, habitat type), we used the 'vegdist' function to generate a Bray-Curtis dissimilarity matrix, the betadisper function to check for heterogeneous distribution of dissimilarities, and the adonis function to perform a permutational analysis of variance (PERMANOVA) to check for interactions between groups (site, survey month, habitat type) in the dissimilarity matrix. In addition, we conducted a fitted NMDS using the 'env.fit' function (Oksanen *et al*., 2019) in the vegan package, plotting significant variables only.

To identify the environmental and/or physical parameters that could explain the bat species diversity across the 40 transects, we used a redundancy analysis (RDA) in the vegan package (Oksanen *et al.*, 2019). We used the 'ordi.step' function for forward stepwise selection of model variables and to test the significance of the RDA axes, we ran a permutation ANOVA. To assess for multicollinearity in the explanatory variables, which can be problematic when using RDA (Liu, 1997), the variance inflation factors (VIF) of all explanatory variables were assessed using the 'vif.cca' function in the vegan package. All explanatory variables had VIF values of < 3, therefore all were retained for RDA analysis. Only explanatory variables with a *P*-value of <0.05 were plotted on the RDA (Supplementary Table S2).

RESULTS

In total, from the 40 transects conducted, we collected 1043 recordings across the four sites. Only one of the 40 transects resulted in no bat recordings (Hanlon Creek Park, 9th September). After filtering (for match ratio and number of calls), we retained 849 recordings for analysis (194 recordings removed). *Myotis septentrionalis* was not identified in any calls.

Overall, we collected the most calls from Hanlon Creek Park (*n* = 233) and the fewest calls from Mitchell-Ellis Creek Park (*n* = 175 — Table 1 and Fig. 1). In terms of bat species richness, we recorded from three bat species in Eramosa River Trail to six species at the other three transect sites (Table 1). *Eptesicus fuscus* was the most abundant bat species detected across all four sites, with a total of 522 recordings and *Myotis leibii* was by far the species with the fewest recordings (*n* = 4 — Table 2). In terms of site-specificity, we mostly detected *Lasiurus borealis* at Hanlon Creek Park, *M. lucifugus* at Riverside Park and *P. subflavus* at Mitchell-Ellis Creek park (Fig. 2). *Eptesicus fuscus* was common across all sites, with *Lasionycteris noctivagans* and *Lasiurus cinereus* present at all transect sites (Fig. 2).

Incidentally, we observed *L. borealis* emerging from the same *Acer* spp. tree roost on two consecutive occasions at the end of August at Hanlon Creek Park but failed to detect the species prior to or after these observations at this site. In addition, we detected (via bat detector and visually) both *M. lucifugus* and *P. subflavus* emerging from a roost under a road bridge each time we completed a transect at Riverside Park.

NMDS plots showed that there was overlap between species detected across sites, months and

TABLE 1. Summary table of bat transects conducted between 6th July–24th September 2020, including number of transects conducted, minimum and maximum transect lengths, total minutes surveyed, number of recordings and number of bat species

Site code	Site name	No. transects conducted	Average transect length (min)	Minimum transect length	Maximum transect length	Total minutes surveyed	Total no. recordings	Relative no. recordings per hour	Total no. bat species
HAN	Hanlon Creek Park	10	51.6	45	59	516	233	27	6
ERT	Eramosa River Trail	10	51.9	45	59	519	227	26	3
RSP	Riverside Park	10	52.9	45	60	529	214	24	6
MEC	Mitchell-Ellis Creek Park	10	53.1	45	60	531	175	20	6

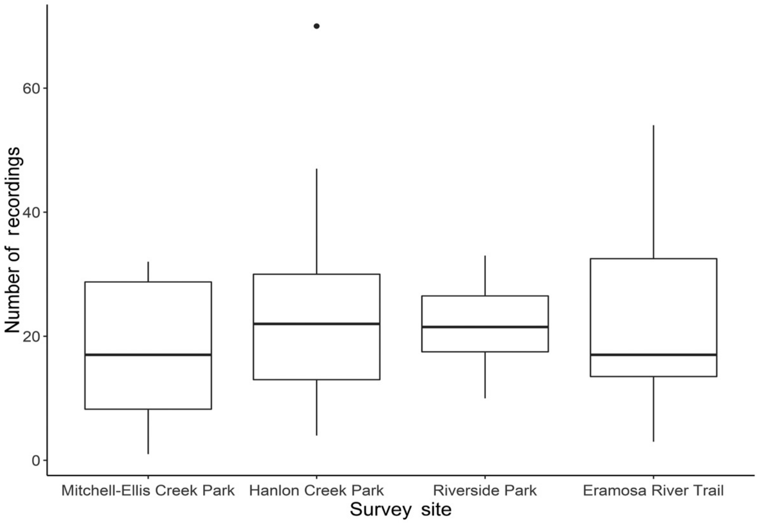

FIG. 1. Richness of bat recordings for each transect across the four sites

habitats (Fig. 3). The most clustering was observed at the site level (Fig. 3A). PERMANOVA of the bat species recording matrix, shows that habitat type explains 10.5% of the variation in Bray-Curtis dissimilarities ($P = 1.000$), site explains 31.7% of the variation ($P < 0.01$) and month sampled explains 10.9% of the variation ($P < 0.05$; Supplementary Table S3). Primary ($P < 0.01$) and secondary vegetation ($P < 0.01$) were the only significant explanatory variables in the fitted NMDS (Fig. 3D).

The first gradient in environmental parameters (RDA1 — Fig. 4), which was positively loaded with primary vegetation (biplot score = 0.861), explained variation in *M. lucifugus* and *P. subflavus* recordings (loadings = 0.676 and 0.177, respectively). The second gradient in environmental parameters (RDA2 — Fig. 4) was negatively loaded with secondary vegetation (biplot score = -0.542) and explained variation in *E. fuscus* and *L. borealis* recordings (loadings = -0.371 and -0.160, respectively — Fig. 4).

DISCUSSION

We expected to observe different bat species assemblages across the four transect sites, based on knowledge that North American bat species exhibit habitat preferences for both foraging and roosting (Furlonger *et al.*, 1987). *E. fuscus* are foraging generalists which have been observed feeding in a mix of urban and rural settings in Ontario (Geggie and Fenton, 1985), which corresponds with the high recordings abundance across the four sites in our study. Based on a recent study (Morningstar *et al.*, 2019), the home range for this species in southwestern Ontario is considered to be ca. 3 km, meaning it is not likely that the same individuals are foraging in more than one of the sites transected. This species is strongly associated with residential areas, particularly near water and open fields, due to the reliance on manmade structures for roosting and optimal foraging, which could account for the hotspot of activity at the Eramosa River Trail site (Geggie and

TABLE 2. Bat species detected in this study for each of the four sites. Included here are species common name, Latin name, Echo Meter Touch 2 species code, total number of recordings, average call duration (in seconds) and pulse ratio percentages

Species code	Common name	Latin name	Total no. recordings	Average call duration	Average pulse ratio
EPTFUS	Big brown bat	*Eptesicus fuscus*	552	5.77	0.80
LASBOR	Eastern red bat	*Lasiurus borealis*	74	6.42	0.75
LASCIN	Hoary bat	*Lasiurus cinereus*	39	7.98	0.77
LASNOC	Silver-haired bat	*Lasionycteris noctivagans*	47	7.55	0.76
MYOLEI	Eastern small-footed bat	*Myotis leibii*	4	2.54	0.76
MYOLUC	Little brown bat	*Myotis lucifugus*	84	3.59	0.70
PERSUB	Tricolored bat	*Perimyotis subflavus*	49	6.25	0.80

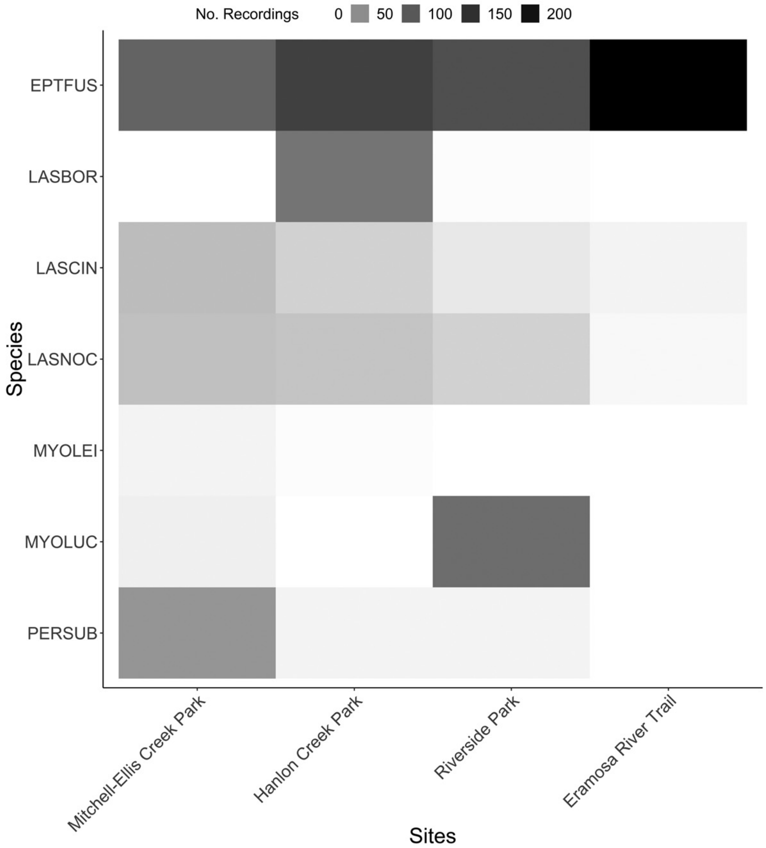

FIG. 2. Total number of recordings for each bat species varies between sites. EPTFUS = *E. fuscus*; LASBOR = *L. borealis*; LASCIN = *L. cinereus*; LASNOC = *L. noctivagans*; MYOLEI = *M. leibii*; MYOLUC = *M. lucifugus*; PERSUB = *P. subflavus*

Fenton, 1985; Soper and Fenton, 2007; Layng *et al.*, 2019). In contrast, *M. lucifugus* relies heavily on aquatic habitats such as rivers, river edges and wetlands in which to forage and roost (Fenton and Bell, 1979; Fenton and Barclay, 1980). A significant portion of their diet is aquatic insects (Clare *et al.*, 2011; Morningstar *et al.*, 2019) and it is suggested this species uses rivers for navigation also (Shively and Barboza, 2017). Female little brown bats are known to roost-switch during summer months, with some female bats occupying home ranges of ~6 km (Slough and Jung, 2020), meaning some of the same females may have been recorded in both Riverside Park and Mitchell-Ellis Creek Park. Primary vegetation cover was a strong explanatory variable for occurrence of *M. lucifugus*. At both sites where this species was detected (Riverside Park and Mitchell-Ellis Creek Park), deciduous swamp was the dominant vegetation class, suggesting this type of ecosystem is pivotal for the occurrence of this species. In addition, the detection and foraging recordings of *M. lucifugus* at Riverside Park, suggest the Speed River system is a key habitat for this species in Guelph. Incidental discovery of a *M. lucifugus* and *P. subflavus* roost at this site highlights the benefit of conducting walking-based active acoustic transects, as it is likely this roost would not have been identified via passive and/or vehicle-based transects (Froidevaux *et al.*, 2020).

Tree-roosting bat species, consisting of *L. cinereus*, *L. noctivagans* and *L. borealis*, were mostly detected across the two forested sites in our study. *L. cinereus* and *L. borealis* are both typically associated with habitats where there is a high density of foliated trees for roosting (Jung *et al.*, 1999). For all three species, open canopies with multiple layers or habitat edges are likely most important for foraging (Jung *et al.*, 1999; Walters *et al.*, 2007; Ethier and Fahrig, 2011; Mills *et al.*, 2013). Forest fragments, similar to those observed in Guelph, are key habitat for *L. cinereus*, *L. noctivagans* and *L. borealis* (Ethier and Fahrig, 2011; Jantzen and Fenton, 2013). *L. borealis* in particular is known to associate with open park areas on the edges of woodland, where they commonly roost in *Quercus* spp. or *Acer* spp. trees (Mager and Nelson, 2001). There is lack of information on the summer home ranges of tree-dwelling bat species, *L. borealis*, *L. cinereus* and

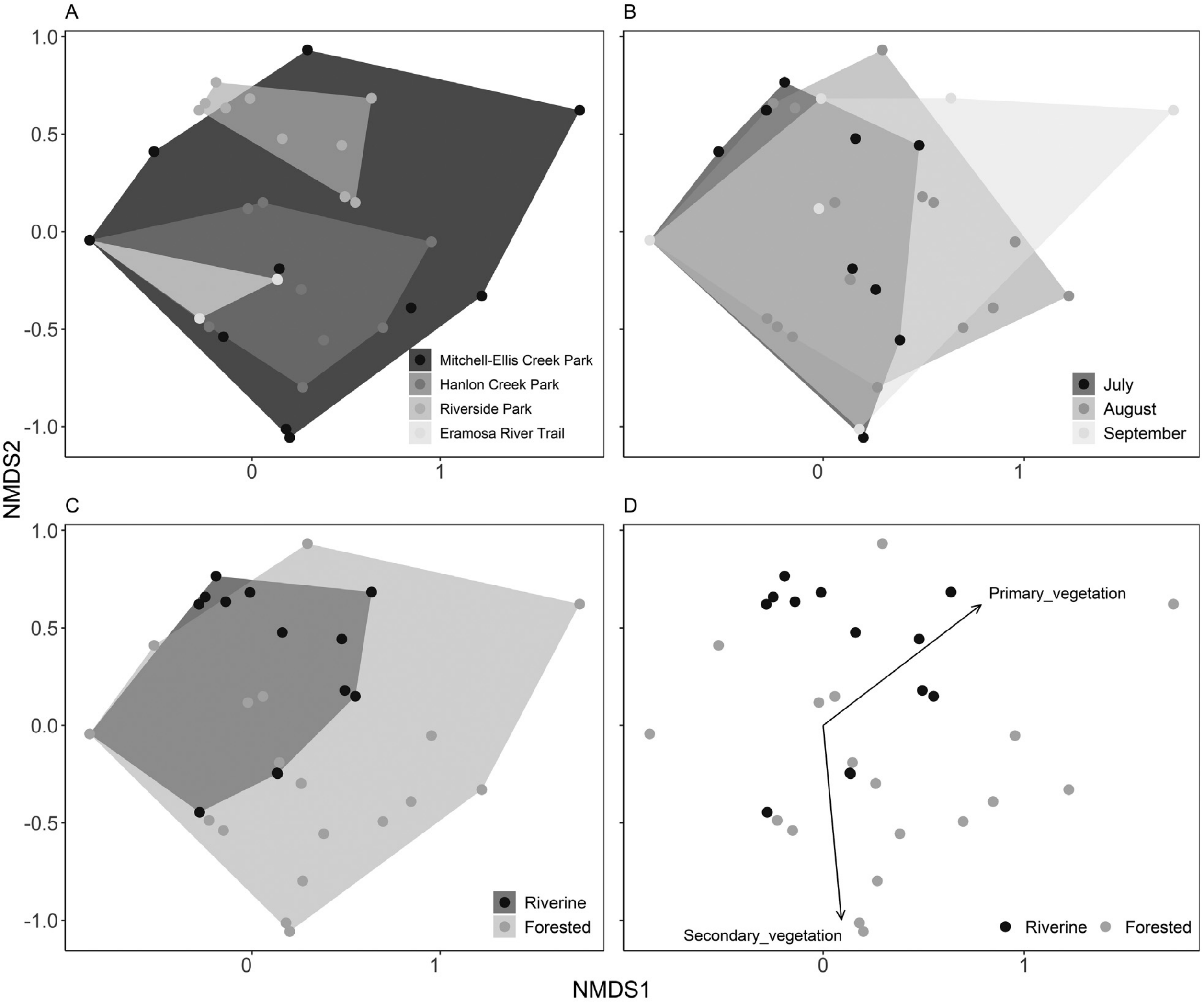

FIG. 3. Non-metric multidimensional scaling (NMDS) plots based on Bray-Curtis dissimilarities for the bat species detected across the four sites. All plots display overlap of dissimilarities for A — site; B — month surveyed (July, August, September); C — habitat type (forest or waterway); D — habitat type (forest or waterway) with significant ($P < 0.05$) environmental variables plotted

L. noctivagans, however *L. borealis* are known to roost-switch frequently during summer months (every 1–3 days), which could explain the lack of detection at Hanlon Creek roost location prior and after (Mager and Nelson, 2001; Monarchino *et al.*, 2020), however this also could be the result of *L. borealis* temporarily roosting at this location on a migratory stopover on way to winter grounds (McGuire *et al.*, 2012).

For the rarer forest-dwelling *P. subflavus*, detections were positively associated with primary vegetation cover similar to *M. lucifugus*, suggesting deciduous swamp is an important vegetation class for activity of this species (Layng *et al.*, 2019). We visually observed this species foraging in a canopy opening at Mitchell-Ellis Creek Park, which corresponds with their known feeding ecology(Loeb and O'Keefe, 2011). Home range for this species is mostly unknown, however some research from the US states individuals have be captured between 300–5000 m from roosting location (Schaefer, 2017). Mitchell-Ellis Creek Park was also one of two locations where *M. leibii* was detected. This rarest, least known species is largely undescribed in terms of foraging, home ranges, hibernation and maternity sites (Johnson and Gates, 2007; O'Keefe and LaVoie, 2011; Ministry of Natural Resources and Forestry, 2017). Lepidoptera however are considered to be an important prey group in US *M. leibii*, and we detected feeding buzzes on one location at Mitchell-Ellis Creek Park. In spite of suitable habitat types being present at some of the sites, we failed to confirm calls from *M. septentrionalis* (Broders *et al.*, 2006). We did detect the species at Riverside

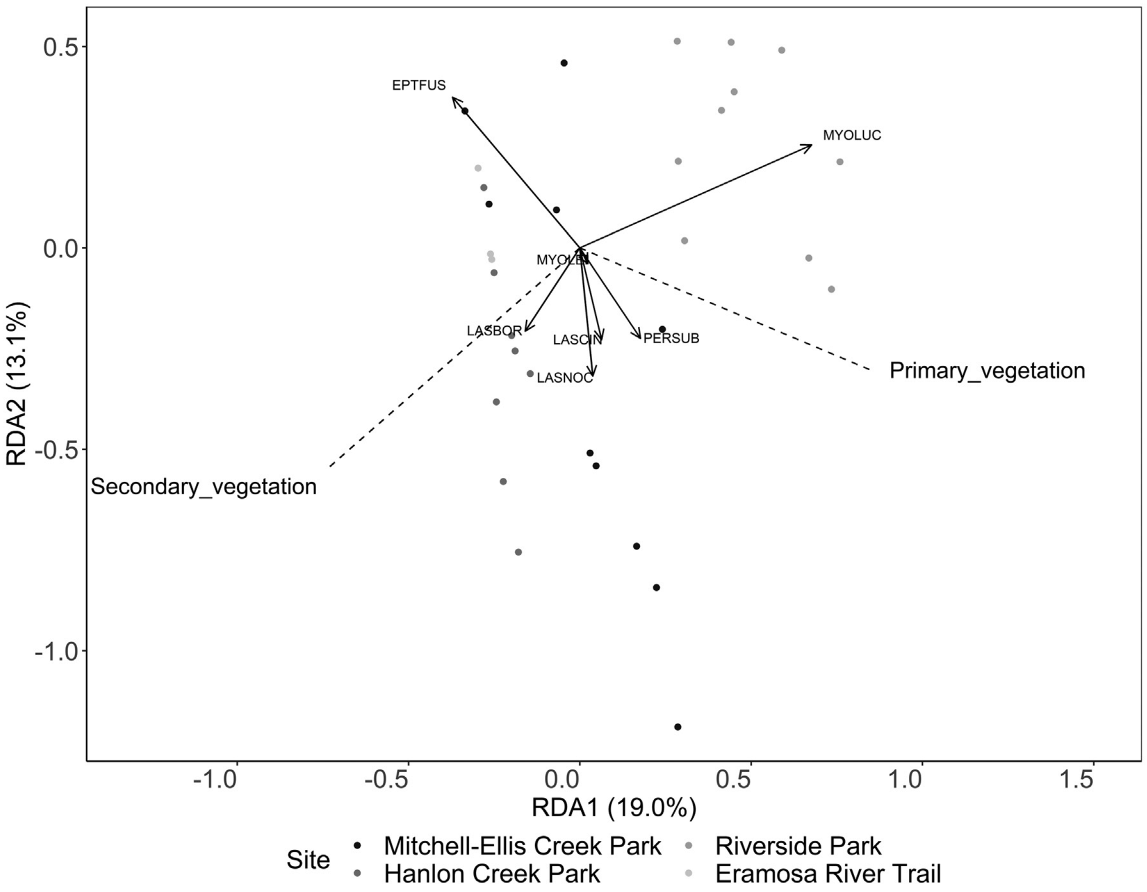

FIG. 4. Redundancy analysis (RDA) showing bat species metrics constrained by physical environmental variables across the four sites. Both axes were found to be statistically significant following a permutation ANOVA test ($P = 0.001$ and $P = 0.008$ for RDA1 and RDA2, respectively). EPTFUS = *E. fuscus*; LASBOR = *L. borealis*; LASCIN = *L. cinereus*; LASNOC = *L. noctivagans*; MYOLEI = *M. leibii*; MYOLUC = *M. lucifugus*; PERSUB = *P. subflavus*

Park; however, the call ratio was often < 0.5 and therefore calls were not processed. The likely explanation for this low call ratio is that this species was not present at all and instead the low call ratio *M. septentrionalis* calls were a mis-identification of poor-quality *M. lucifugus* calls, which are similar (Ratcliffe and Dawson, 2003).

This study demonstrates the data which has the potential to be collected by non-specialists using smartphone acoustic devices. Despite this only being an exploratory study, we have detected common and rare Ontario bat species, determined two roost locations and identified associations between particular species and existing primary and secondary vegetation classes. This information is particularly valuable for investigating activity of bat species post-WNS introduction, which is of interest both in and beyond Canada. Although we didn't specifically isolate and statistically analyse number of feeding buzzes, this is most definitely a possibility for future studies. In addition, long-term studies (i.e. year-round) of these locations would be greatly beneficial to understand seasonal differences in habitat use. We have highlighted the importance of particular vegetation classes for rare Ontario bat species, which is information crucial to successful recovery strategies for WNS-threatened species. Our study shows the potential for applying community-based monitoring to bat research, through the use of smartphone acoustic devices with auto-ID features at easy to access, city-owned sites.

SUPPLEMENTARY INFORMATION

Contents: Supplementary Tables: Table S1. Information regarding the GPS coordinates, primary and secondary vegetation coverage for the transect route using Ecological Land Classification (ELC) classes for vegetation classifications, number of days sampled and overall number of effort hours for the four sites used in this study. Table S2. Explanatory variables from the metadata with their associated variance inflation factors (VIF) and *P*-values. Variables in bold were plotted on the Redundancy Analysis (RDA) plot. Table S3. Number of recordings for bat species detected are not significantly different between habitat type (riverine versus forested) but are significantly different between sites. No significant beta dispersion was detected within groups (site, habitat, month). The significant difference detected was recordings between sites and month. Summary of PERMANOVA results based on a Bray-Curtis dissimilarity matrix of bat species recordings. Supplementary Fig. S1. Map of the four transect locations in Guelph (Ontario) included in this study. Supplementary Information is available exclusively on BioOne.

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