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Variability in BVOC emissions and air quality impacts among urban trees in Montreal and Helsinki

Kaisa Rissanen[1*], Juho Aalto[2], Jaana Bäck[2], Heidi Hellén[3], Toni Tykkä[3], Alain Paquette[1]

[1]Département des Sciences Biologiques / Centre for Forest Research, Université du Québec à Montréal, Montreal, QC H3C 3P8, Canada
[2]Institute for Atmospheric and Earth System Research / Department of Forest Sciences, University of Helsinki, Helsinki, 00014, Finland
[3]Atmospheric Composition Research Unit, Finnish Meteorological Institute, Helsinki, 00101, Finland
[*]current affiliation: Institute for Atmospheric and Earth System Research / Department of Forest Sciences, University of Helsinki, Helsinki, 00014, Finland

Correspondence to: Kaisa Rissanen (kaisa.rissanen@helsinki.fi)

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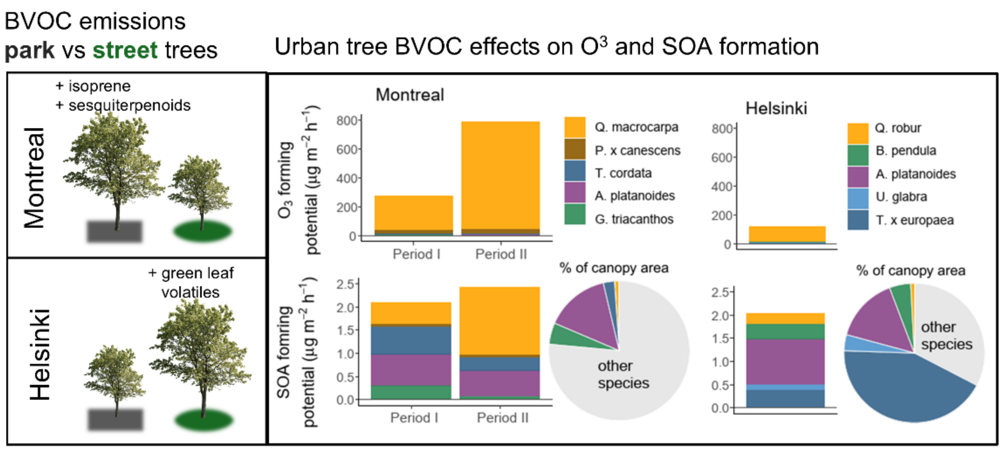
Abstract. Many cities attempt to mitigate poor air quality by increasing tree canopy cover. Trees can indeed capture pollutants and reduce their dispersion, but they can also negatively impact urban air quality. For example, trees emit biogenic volatile organic compounds (BVOCs) that participate in both ozone (O_3) and secondary organic aerosol (SOA) formation, yet these emissions have been little studied in urban contexts.

We sampled BVOCs from the leaves of mature urban trees using lightweight enclosures and adsorbent tubes in two cities: Montreal, Canada and Helsinki, Finland. In both cities, we targeted five common broadleaved species, comparing their standardised BVOC emission potentials 1) between parks and streets and 2) to nonurban BVOC emission potential estimates from emission databases. Finally, we calculated the potential O_3 and SOA formation by urban trees at the leaf scale and upscaled to the neighbourhood.

We found that the BVOC emission potentials differed slightly between park and street trees. Compared to park trees, street tree emissions were higher in Montreal (specifically isoprene and sesquiterpenoids) and lower in Helsinki (specifically green leaf volatiles). However, the measured BVOC emission potentials generally deviated little from the emission database estimates, supporting the use of database estimates for urban trees. In addition, we found that O_3 formation from urban tree BVOC emissions was dominated by isoprene, while SOA formation was also affected by lower monoterpenoid and sesquiterpenoid emissions. These findings highlight the importance of species selection and management strategies that protect trees from BVOC-inducing stresses.



1 Introduction

Poor air quality is a major health risk impacting nine out of ten people globally (WHO, 2016). Local air pollution, including both particulate matter and gaseous pollutants, is estimated to cost 8.8 million lives annually (Lelieveld et al., 2020). Particularly vulnerable to air pollution, densely populated urban areas are increasingly investing in urban greenery to improve

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the local air quality and living environment (see, e.g., the Million Tree Initiatives in Los Angeles, Pincetl et al., (2013)). Urban trees are expected to remove particulate matter (PM_{10} and $PM_{2.5}$, i.e., particulate matter smaller than 10 and 2.5 μm, respectively) and gaseous pollutants such as ozone (O_3) through dry and wet deposition or stomatal uptake (Calfapietra et al., 2016; Nowak et al., 2014). In addition, urban trees can, for example, cool air through shading and transpiration (Pataki et al., 2021; Winbourne et al., 2020), benefit the physical and mental health of urban residents (Pataki et al., 2021; Wolf et al., 2020), and provide other ecosystem services like biodiversity conservation or flood protection (O'Brien et al., 2022).

Yet, urban trees may also negatively affect local air quality (Calfapietra et al., 2013; Churkina et al., 2017; Fitzky et al., 2019). Trees emit a variety of biogenic volatile organic compounds (BVOCs), some of which carry positive health impacts (Cho et al., 2017). Many, however, are highly reactive in the atmosphere, whereby BVOCs such as isoprene, monoterpenes, and sesquiterpenes are oxidised once they react with hydroxide (OH) and nitrate (NO_3) radicals, O_3, or chlorine (Cl) (Ziemann and Atkinson, 2012). These reactions form less volatile products, which, through multiphase reactions participate in the production of secondary organic aerosols (SOAs) (Ziemann and Atkinson, 2012), constituting an important fraction of the atmospheric particulate matter (PM) (Huang et al., 2014). Moreover, although BVOCs can remove O_3 through oxidation, BVOC oxidation products also produce O_3 through reactions with nitrogen oxides (NO_x) (Atkinson, 2000). Due to the abundance of NO_x in urban atmospheres (Delmas et al., 1997) and the decrease in the anthropogenic VOC concentrations (EMEP database), urban O_3 concentrations can become increasingly sensitive to changes in BVOC concentrations (Bell and Ellis, 2004). The potential impact of BVOC emissions on both SOA and O_3 concentrations has been estimated in large cities (Bao et al., 2024; Ghirardo et al., 2016; Ren et al., 2017), illustrating the important contributions of BVOCs on local O_3 concentrations (up to 30% in Beijing, Ren et al., 2017), albeit impacting $PM_{2.5}$ concentrations to a lesser degree (1.3% in Beijing, Ren et al., 2017). However, in Helsinki, monoterpenes and sesquiterpenes (either biogenic and anthropogenic VOCs) significantly contributed to SOA formation, even in urban traffic environments (Hellén et al., 2012; Saarikoski et al., 2023).

In addition to the effect of BVOCs on air quality, street tree canopies may also decelerate air flow and ventilation in street canyons, thereby delaying pollutant removal at the pedestrian level (Abhijith et al., 2017; Karttunen et al., 2020). Overall, the net impact of urban trees on air quality remains unclear and scale-dependent (Maison et al., 2024; Venter et al., 2024). To estimate the net effect of urban trees on local air quality, the positive (capture of PM and gaseous pollutants) and negative effects (BVOC effects on PM and O_3 production and reduced ventilation) of urban trees must be considered together. Therefore, accurate estimates of BVOC emissions of urban trees are crucial. In fact, estimates of the species-specific BVOC emission potentials (BVOC emission rate normalised to a certain temperature and light intensity), measured in natural environments or laboratory conditions, have been collected in emission databases (e.g., Kesselmeier and Staudt, 1999; Karl et al., 2009; Oderbolz et al., 2013). These databases have been used in studies estimating urban BVOC emission budgets and air quality impacts (Benjamin and Winer, 1998; Donovan et al., 2005; Owen et al., 2003; Ren et al., 2017), in tools for species selection (Churkina et al., 2015; Donovan et al., 2005; Simpson and McPherson, 2011) or for quantifying urban tree services and disservices (e.g. i-Tree tools, itreetools.org). However, because these databases contain BVOC emission potentials mainly from nonurban trees, they also carry potential sources of errors.

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Firstly, the emission rates and composition of BVOCs are diverse and vary between tree genera and species, yet many tree species commonly used in urban environments lack estimates in the emission databases. While the species-level emission potentials can be approximated based on genus-level estimates (Benjamin et al., 1996), the approximations sometimes introduce large errors (Dunn-Johnston et al., 2016; Noe et al., 2008). For example, in the *Quercus* genus, there are species with mainly isoprene emissions, mainly monoterpene emissions, and species that emit a mixture of both types of isoprenoids (Loreto, 2002; Steinbrecher et al., 1997). Furthermore, species-level BVOC emission potentials and blends may differ between populations (Bäck et al., 2012; Loreto, 2002; Staudt and Visnadi, 2023), which can explain some observed differences in species-level BVOC emissions between cities (Bao et al., 2023; Cui et al., 2024; Yuan et al., 2023). More species-level data are thus needed to support species selection and to model BVOC emissions and air quality (Bao et al., 2023).

Secondly, urban environmental conditions differ from natural or laboratory conditions in ways that may considerably modify tree BVOC emissions. Many urban growth environments are stressful, exposing trees to the combined effects of drought, heat, high O_3 concentrations, and mechanical damage (Fitzky et al., 2019; Lüttge and Buckeridge, 2020). Because protection against and mitigation of biotic and abiotic stresses are among the most important known roles of many BVOCs (Harrison et al., 2013; Loreto and Schnitzler, 2010), these urban stress factors could induce or increase the synthesis and emissions of BVOC even in species with normally no or low emissions (Fitzky et al., 2019; Ghirardo et al., 2016; Holopainen and Gershenzon, 2010). Other urban environment characteristics, such as a high CO_2 concentration and light availability in open spaces, can also impact BVOC emissions (Bao et al., 2023; Guenther, 1997). Thus, it is uncertain how well the BVOC database values originating from nonurban environments can be applied to urban trees.

Moreover, direct measurements of urban tree BVOC emissions remain rare, primarily concentrating on trees in urban green spaces such as botanical gardens or university campuses (Ghirardo et al., 2016; Khedive et al., 2017; Noe et al., 2008; Préndez et al., 2013; Wu et al., 2021; Yuan et al., 2023; Zhang et al., 2024). A few studies have also explored BVOC emissions in varying urban green spaces or over rural-urban gradients (Duan et al., 2023; Lahr et al., 2015; Papiez et al., 2009). However, trees within the built environment have received little attention (see, however, Dunn-Johnston et al., 2016), although they may be the most exposed to the abovementioned stress factors and thus prone to modified BVOC emissions. To quantify how well the urban tree BVOC emission potentials conform to the estimates in emission databases (species- or genus-level), more direct measurements of urban tree BVOC emissions in various urban environments are necessary. Such an understanding will allow us to estimate the net effects of urban trees on air quality more accurately and select suitable species, species compositions, and tree management strategies for urban spaces.

Here, we measured the BVOC emission rates and composition of five common urban tree species in Montreal, Canada and Helsinki, Finland, comparing the measured emission rates to earlier estimates collected from emission databases and calculating their potentials for O_3 and SOA formation. To quantify the variability between trees of the same species and between differing urban environments, we measured trees in two typical urban growth environments—streets and parks—in both cities. Being more exposed to stress factors, we expected urban trees, particularly on streets, to emit more BVOCs and contribute more to O_3 and SOA formation.

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Our specific aims were as follows:

1) To provide BVOC emission potentials for common urban tree species, measured directly from the shoots of mature urban trees.
2) To explore the tree-to-tree variability of BVOC emissions by comparing two contrasting urban environments (parks and streets).
3) To compare urban tree BVOC emission potentials to estimates from BVOC emission databases.
4) To provide O_3 and SOA formation potentials for the measured common urban tree species, at both the leaf and neighbourhood levels.

2 Materials and methods

2.1 Study sites and tree selection

The study took place in two cities: in Montreal, Canada (45.508888, -73.561668) with its 1.76 million inhabitants and in Helsinki, Finland (60.192059, 24.945831) with its 0.65 million inhabitants. Both cities have a warm summer with a humid continental climate (Dfb) based on the Köppen classification (Beck et al., 2018). Montreal's monthly mean temperatures range from 21.1°C (July) to -9.7°C (January). With an annual precipitation of 1000 mm, approximately 270 mm falls between June and August (Environment and Climate Change Canada, 1981–2010). In Helsinki, the mean temperature ranges from 18.1°C (July) to -3.1°C (January). With its annual precipitation of 653 mm, approximately 200 mm falls between June and August (Jokinen et al., 2021, 1991–2020). In both cities, we limited the study area (~2.5 km^2 in Montreal and ~1 km^2 in Helsinki) to central residential areas, with population densities exceeding 2500 inhabitants km^{-2}.

In addition, in both cities, we selected five common species present in both parks and streets (sidewalk pits or planting strips). Based on the literature, two species per city had moderate to high BVOC emission rates, while three had low rates. In Montreal, the high-emitting species consisted of the bur oak (*Quercus macrocarpa,* Michx., QM) and the grey poplar (*Populus x canescens,* Aiton., PC), while the low-emitting species included the Norway maple (*Acer platanoides,* L., AP), the little-leaved linden (*Tilia cordata*, Mill., TC), and the honey locust (*Gleditsia triacanthos*, L., GT). These species correspond to 0.7%, 0.3%, 15.1%, 5.4% and 9.1%, respectively, of all trees in the public tree inventory (City of Montreal open data a). In Helsinki, the high-emitting species consisted of the pedunculate oak (*Q. robur*, L., QR) and the silver birch (*Betula pendula,* Roth., BP), with the low-emitting species including the Norway maple (AP), the European linden (*T. x europea* Hayne, TE), and the Scots elm (*Ulmus glabra*, Huds., UG). These correspond to 3.5%, 8.5%, 11.0%, 22.8% and 5.7%, respectively, of trees in the public tree inventory (City of Helsinki open data).

For each species in each city, we selected three trees from three different parks and streets within the study areas (Table 1, Supplement Fig. S1). As an exception, all available PC trees were located inside one park and on one street. All selected trees had a minimum diameter at breast height (DBH) of 9 cm (Table 1) and were visibly healthy with no dead branches or large wounds.

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Table 1. The study species in each city, the number of trees successfully sampled on streets and in parks and the range of diameter at breast height (DBH) of the study trees. An n of 2 to 3 in Montreal means that, during measurement period I or II, one sampling was unsuccessful (see Sect. 2.4).

Montreal					Helsinki				
	Street		Park			Street		Park	
Species	n	DBH	n	DBH	Species	n	DBH	n	DBH
Quercus macrocarpa (QM)	3	16–18	3	12–44	*Quercus robur* (QR)	3	9–13	3	16–48
Populus x canescens (PC)	3	16–18	3	15–17	*Betula pendula* (BP)	3	19–26	3	21–54
Acer platanoides (AP)	2 to 3	14–28	3	19–44	*Acer platanoides* (AP)	3	13–29	3	24–53
Tilia cordata (TC)	3	32–42	3	20–42	*Tilia x europaea* (TE)	3	11–25	3	21–48
Gleditsia triacanthos (GT)	3	11–13	2-3	31–34	*Ulmus glabra* (UG)	3	24–43	3	14–74

2.2 BVOC sampling and auxiliary measurements

2.2.1 BVOC sampling

We sampled BVOC emissions from the Montreal trees twice—on 2 to 15 June (period I) and 11 to 25 August (period II) 2022—and from Helsinki trees once on 6 to 25 July 2022 (for exact sampling dates, see Table S1). The sampling days were preferably sunny—without rain or heavy cloud cover—and sampling always took place between 11.00 and 15.00. Ambient temperatures during sampling were 19 to 33 °C in period I and 21 to 37 °C in period II in Montreal, and 22 to 33°C in Helsinki (Supplement Table S1).

From each tree, we sampled BVOC emissions from one undamaged shoot with mature leaves in a sunny position in the lower or mid-canopy area. We carefully enclosed the shoot in a polyethylene terephthalate (PET) bag (LOOK oven bag 35 x 43 cm, heated at 120°C for at least 1 h, (Vedel-Petersen et al., 2015); for recovery tests, see Supplement methods S1 and Table S2), with a 6.35 mm diameter fluorinated ethylene propylene (FEP) tubing for replacement and sample air (Fig. S2). We first flushed the bag at a flow rate of 2 L min^{-1} for 15 minutes and then sampled portions of the replacement and sample air into adsorbent tubes (Tenax TA and Carbopack B) with a flowrate of 0.08 L min^{-1} for 30 minutes. The sampling time was reduced to 15 minutes for QM and PC in Montreal in August to avoid isoprene saturation. For quality control, we also sampled an empty bag at the beginning of each sampling day using the same setup. To pump the replacement air, controlling and logging the air flowrates, and collecting the air samples into adsorbent tubes, we used a custom-made gas sampler. The replacement air relied on non-filtered ambient air to maintain close-to-ambient conditions within the sampling bag and to allow for potential ozone effects on leaf BVOC emissions. To ensure BVOC stability in the adsorbent tubes during sampling and storage, we added sodium thiosulfate–impregnated filters to remove any ozone before reaching the adsorbent tube (Hellén et al., 2024).

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Between 7 June and 25 July, we could only use the filters for outgoing sample air adsorbent tubes because of supplier delays. We corrected this imbalance in the BVOC emission calculations (see methods Sect. 2.3).
After sampling, we stored the adsorbent tubes and the sampled branch in a box cooler until the end of the day, after which the tubes were stored in a refrigerator (~4 to 8°C). We photographed the sampled shoot leaves for leaf area calculation and dried them at 60°C for at least 48 h before weighing their dry mass.

2.2.2 Auxiliary measurements

During sampling, we tracked the temperature, humidity, and light conditions (photosynthetically active radiation, PAR) within and outside the sampling bag by placing a humidity/temperature sensor (Rotronic HC2-S3C03, Bassersdorf, Switzerland) within the sampling bag and a quantum sensor (LI-190R-SMV-5, LI-COR Biosciences, Lincoln, NE, USA) beside the bag. In addition, we logged the ambient air temperature and humidity outside the bag using a sensor (RuuviTag pro, Riihimäki, Finland) shaded from direct solar radiation. We measured the ambient air O_3 concentration during BVOC sampling using a multi-gas monitor (Gasmaster Gas Monitor 2710 with O_3 head from 0–0.15 ppm, Kanomax, NJ, USA). Following BVOC sampling, we measured the leaf water potential of three leaves near the sampled branch using a portable pressure chamber (Pump-Up Chamber, PMS Instrument, OR, USA). For information on the meteorological conditions before and during the sampling periods, we accessed the temperature and precipitation records of the Université du Québec à Montréal weather station (Université du Québec à Montréal / ESCER) and the Finnish Meteorological Institute Kumpula weather station (Finnish Meteorological Institute).

2.3 BVOC analysis and calculations

We analysed the adsorbent tubes by using a gas-chromatograph mass-spectrometer (GC-MS; Clarus 680 and Clarus SQ T or Clarus SQ 8 C, PerkinElmer, Waltham, MA, USA) with thermal desorption (TurboMatrix 350, PerkinElmer, Waltham, MA, USA) at the Finnish Meteorological Institute within 30 days of sample collection. Relying on the analytical method of Helin et al. (2020), we used six calibration standards of 25 compounds in each GC-MS run to quantify the compound concentrations in the sample tubes (see the list of detected and calibrated compounds in Table S3) and identified the compounds by comparing their retention times and mass spectra to the calibrated standard. For isoprene, we used one gaseous calibration standard (National Physical Laboratory, Teddington, UK). Compounds which were not in the calibration standards were tentatively identified by comparing their retention times and mass spectra with the National Institute of Standards and Technology (NIST, Gaithersburg, Maryland, USA) mass spectral library, and calculated as the calibrated compound closest in composition and retention time (Table S3). When the tentative identification did not provide a confident match, we grouped the compound into a subgroup following Guenther et al. (2012) (hemiterpene, monoterpenoid, sesquiterpenoid, GLV).
Two corrections to the measured BVOC concentrations were necessary. First, as mentioned in Sect. 2.2.1, samples collected between 7 June and 25 July did not include ozone removal for the incoming replacement air samples. For these samples, we

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corrected the measured BVOC concentrations using terpenoid losses due to ozone reactivity in adsorbent tubes quantified by Helin et al. (2020) and the measured ambient ozone concentrations to correct the concentration values (for details, see Supplement methods S2). The potential error and the correction likely had a minor impact on the calculated emission rates (Fig. S3). Second, in 13 samples from QM, QR, and PC, the isoprene concentrations in the outgoing samples were higher than the isoprene standard. In nine of these samples, the concentration saturated the selected ion monitoring (mass 67), for which we thus quantified and calibrated the peak using the ion scan mode (mass 65). In these samples, the isoprene concentrations are less certain than in other samples and compounds, although they serve as the conservative minimum estimates of the isoprene concentration. In addition, we did not use the samples from two extreme cases (one PC on the street and one QR in the park) in further analyses.

In Eq. (1), we calculated the shoot emission rates (E_{shoot}) or empty bag emission rates (E_{bag}) for each compound based on the mass of the incoming replacement air and outgoing sample (C_{in} and C_{out} ng), the sampling air flowrate (F_S L min^{-1}), the sampling time (t, min), and the replacement air flowrate (F_R, L min^{-1}) as follows:.

$$E_{\text{shoot or bag}} = \left(\frac{C_{out}}{F_S{*}t} - \frac{C_{in}}{F_S{*}t}\right){*}F_R \qquad (1).$$

To account for impurities or the retention of compounds within the sampling system, we subtracted the empty bag emission rates from the shoot emission rates measured on the same day ($E_{\text{shoot corrected}} = E_{shoot} - E_{bag}$). Finally, we divided the corrected shoot emission rates by the dry leaf mass and multiplied it by 60 to obtain the emissions per dry mass in an hour (ng g^{-1} DW h^{-1}).

We calculated the detection limits of the sampling system as 3x the standard deviation (SD) of all incoming sample concentrations (Table S3). When the concentration difference between the incoming and outgoing samples (C_{out} - C_{in}) did not exceed the detection limit, we flagged the compound emissions as uncertain. We included the flagged compounds to calculate the total emission rates over the compound groups to avoid a consistent negative bias, but did not use them in compound-specific analyses. We also calculated the analytical detection limits per measurement period across all the TD-GC-MS runs during the period as the mean peak concentration of empty tube measurements + 3x SD for analytical quality control (Table S3).

2.4 Data analysis

To prepare the shoot emission data for further analysis, we first removed any data with potential signs of rough handling in the BVOC emissions because of their sensitivity to mechanical stress. To do so, we plotted the emission rates (isoprene, monoterpenoid total, sesquiterpenoid total and green leaf volatile (GLV) total) of all trees per species against the temperature and PAR while removing trees which had abnormally high emissions (3- to 100-fold emission rates) for the given temperature or PAR. This removed one GLTR in a park in Montreal sampled during period I and one ACPL measured on a street in

Montreal during period II. Next, we calculated the emission potentials, that is, the emission rates normalised to temperature 30°C and PAR 1000 µmol m^{-2} s^{-1} based on G97 functions (Guenther, 1997). The occurrence of low light conditions (PAR< 50 µmol m^{-2} s^{-1}) during sampling can inflate uncertainty in the normalisation. Thus, we removed one ACPL and one TICO measured on streets in Montreal during period I and one ACPL measured on streets in Montreal during period II from the analysis that used the emission potentials.

2.4.1 Differences between park and street tree BVOC emission potentials and blends

To explore the effect of the site type (park or street) on the emission potentials of isoprene, and the monoterpenoid, sesquiterpenoid, and GLV totals, we used the analysis of variance (ANOVA) F-test (Anova of R package car, version 3.1.1). We applied the test first separately by city and in Montreal by measurement period (I or II), using site type, species, and the interaction between species and site type as explanatory variables (n_{obs}= 26 to 30). We applied log + i-transformation (where i was a small number, 0.5x the smallest nonzero value) on the emission potential data for a normal distribution of the residuals (calculated using Shapiro–Wilk test, Shapiro.test of R package stats, version 4.2.2).

2.4.2 Differences between measured urban tree BVOC emission potentials and estimates from databases

We collected the estimates of the species- or genus-level emission potentials for isoprene, and the monoterpenoid and sesquiterpenoid totals from several BVOC emission databases (Table S4). We then calculated the mean for all unique database values per species and compound group for the "reference emission potential" (Tables S5 and S6), which we compared with our measured urban tree BVOC emission potentials. In this comparison, we used one-sample Wilcoxon tests (wilcox.test function of R-package stats, version 4.2.2), separately per city, species and compound group, and the sampling period in Montreal (n_{obs} = 4 to 6 per species). Although the Wilcoxon test accommodates small sample sizes and nonnormal distributions of the emission data, the results here should be interpreted with care.

2.4.3 Potential air quality effects of the study species

To estimate the O_3-formation potential per tree and compound, we used OFP = E x MIR, where OFP is the O_3-formation potential (µg g^{-1} h^{-1}), E is the emission potential of the BVOC (µg g^{-1} h^{-1}), and MIR is the maximum incremental reactivity (g g^{-1}), which is based on previous work by Carter, (1994, 2010). OFP estimates the maximum potential of BVOC to contribute to the production of photochemical O_3 (Carter, 1994, 2010). When a compound-wise MIR was not available for a monoterpenoid in our study, we used the average value of 4.04 g g^{-1} (Carter, 1994, 2010). Following Su et al. (2016), Wang et al. (2019) and Yang et al. (2023), for sesquiterpenoids, we used the value of 1.71 g g^{-1} for C15 alkenes. To estimate the SOA-formation potential (SOAFP) per tree and compound, we used SOAFP = E x FAC, where SOAFP is the SOA-formation potential, E is the emission potential of BVOC (µg g^{-1} h^{-1}), and FAC is the fractional aerosol coefficient (%, the fraction of BVOC which converts into aerosol, according to Grosjean (1992) and Grosjean and Seinfeld (1989)). We collected the

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compound-wise FAC values from Grosjean (1992), Hoffmann et al. (1997), Griffin et al. (1999), and Carlton et al. (2009). When a compound-wise FAC was unavailable, we used the mean over the compound group (monoterpenoids or sesquiterpenoids). When we found multiple FAC estimates per compound, we used the mean across estimates. The MIR and FAC values we used per compound are listed in Table S7.

We summed the compound-wise OFP and SOAPF per tree to present the total OFP and SOAFP. We then applied the ANOVA F-test to explore the differences in OFP and SOAFP between species and site types (n_{obs}= 26 to 29), similar to what we did for the BVOC emission potentials (see Sect. 2.4.1).

2.4.4 Upscaling BVOC emission potentials, OFP and SOAFP

Finally, we calculated rough estimates for the total BVOC emission, OFP, and SOAFP per land area for the study species in Montreal and Helsinki. We first selected smaller, representative areas of interest ("upscaling test areas") within the study areas: 1.75 km^2 in Montreal and 0.30 km^2 in Helsinki (Fig. S1). We selected the upscaling test areas to consist of street or park land uses (excluding green areas with trees not listed in the public tree databases) and to contain individuals from our study species. We then approximated the canopy area of each tree within the area of interest by cutting the canopy cover layer (estimated based on an aerial image and aerial LiDAR data, in Montreal from 2019 (City of Montreal open data b), and in Helsinki combining estimates from 2020 and 2022 (Helsinki Region Environmental Services)) using Voronoi polygons defined by tree location layer from public tree databases in QGIS (QGIS Development Team 2025, version 3.40.5). To minimise the inclusion of tree canopies not listed in the public tree databases, we limited the maximum canopy radius to 9 m. Using a leaf area index of 5 m^2 m^{-2} (as used in the MEGAN biogenic emission model, Guenther et al. (2012)), we then converted the tree-wise canopy areas to tree-wise leaf areas. We used the tree-wise leaf area (m^2), species-wise leaf mass per area (calculated from a measured leaf area and mass, g m^{-2}), and the measured species-wise BVOC emission potential, OFP, or SOAFP (µg g^{-1} h^{-1}) to calculate the total tree-wise emission potentials of isoprene, monoterpenoids, and sesquiterpenoids along with OFP and SOAFP for each tree in our study species.

We corrected the tree-wise emission potentials by a factor of 0.75 assuming that half of the tree leaf area is shaded at any given time of day and thus emits at a rate that is half the measured emission potential of the sun leaves [1 - (0.5 * 0.5) = 0.75]. In addition, we calculated the tree-wise emission potentials without correction (assuming all of a tree leaf area emits at the rate of the measured emission potential of the sun leaves) and at a higher factor of 0.5 (assuming half of the tree canopy is shaded and does not emit BVOC). Finally, we summed the corrected emission potentials of all trees from our study species and divided the sum by the area of the upscaling test area to determine the BVOC emission, and the OFP and SOAFP intensities of the study species as µg m^{-2} (land area) h^{-1}.

https://doi.org/10.5194/egusphere-2025-2500
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3 Results

3.1 Environmental conditions before and during BVOC sampling

Monthly mean temperatures in June (19.2°C) and August (21.8°C) 2022 in Montreal were slightly higher than the long-term mean monthly temperatures (18.6°C and 20.1°C, respectively, Figs. S4a–b). The ambient and chamber temperatures tended to be higher during the street tree than park tree sampling (Fig. S5a). In Helsinki, the July 2022 mean temperature (18.5°C) approximated the long-term mean (18.1°C), and the ambient or chamber temperatures did not differ between the park and street trees (Figs. S4c and S5a).

The precipitation totals in June (165 mm) and August (104 mm) in Montreal slightly exceeded the long-term monthly totals (87 mm and 94 mm, respectively, Figs. S4a–b). In June, however, most precipitation fell after the sampling period. In Helsinki, the July precipitation total (49 mm) was slightly lower than the long-term average (57 mm, Fig. S4c). With close-to-average precipitation, trees in both cities were unlikely to experience drought. Correspondingly, mid-day leaf water potentials remained mostly above -2 MPa (Fig. S5b). The lowest water potentials (-2.1 MPa) occurred in Montreal in August (Fig. S5b).

Ambient O_3 concentrations during sampling varied between 0.010 and 0.042 ppm in Montreal and between 0.019 and 0.039 ppm in Helsinki (Fig. S5a), thus remaining mostly below harmful levels compared with the European Environment Agency AOT40 standard threshold of 0.04 ppm and Canadian Ambient Air Quality Standards threshold of 0.06 ppm. For comparison, daytime ambient O_3 concentrations measured from rural background air near Helsinki (Luukki, Espoo) were on average 0.034 ppm in summer 2022 (https://en.ilmatieteenlaitos.fi/download-observations). In Montreal, ambient O_3 concentrations were generally higher during street tree than park tree sampling, while in Helsinki we observed no difference (Fig. S5a).

3.2 BVOC emission rates in urban trees

3.2.1 Measured BVOC emission potentials and variation between park and street trees

The total BVOC and isoprene emission potentials were high (>20 ng g^{-1} h^{-1}) for PC and QM in Montreal (Figs. 1a–d) and for QR in Helsinki (Figs. 2a–b), but low for other species. The isoprene emission potentials were negligible (<0.1 ng g^{-1} h^{-1}) for AP, GT, TC, TE, and UG (Figs. 1c–d, and Fig. 2b). The monoterpenoid emission potentials were highest for BP in Helsinki (Fig. 2c) and low (<1 ng g^{-1} h^{-1}) for the other species (Figs. 1e–f and Fig. 2c). The sesquiterpenoid emission potentials were generally low (<0.5 ng g^{-1} h^{-1}), except for TC in Montreal and AP in Helsinki (Figs.1g–h and Fig. 2d), which were of a similar magnitude or even higher than the emissions of monoterpenoids. The GLV emission potentials were highest among AP in both cities, but with large tree-to-tree and temporal variation (Fig. 1i–j and 2e).

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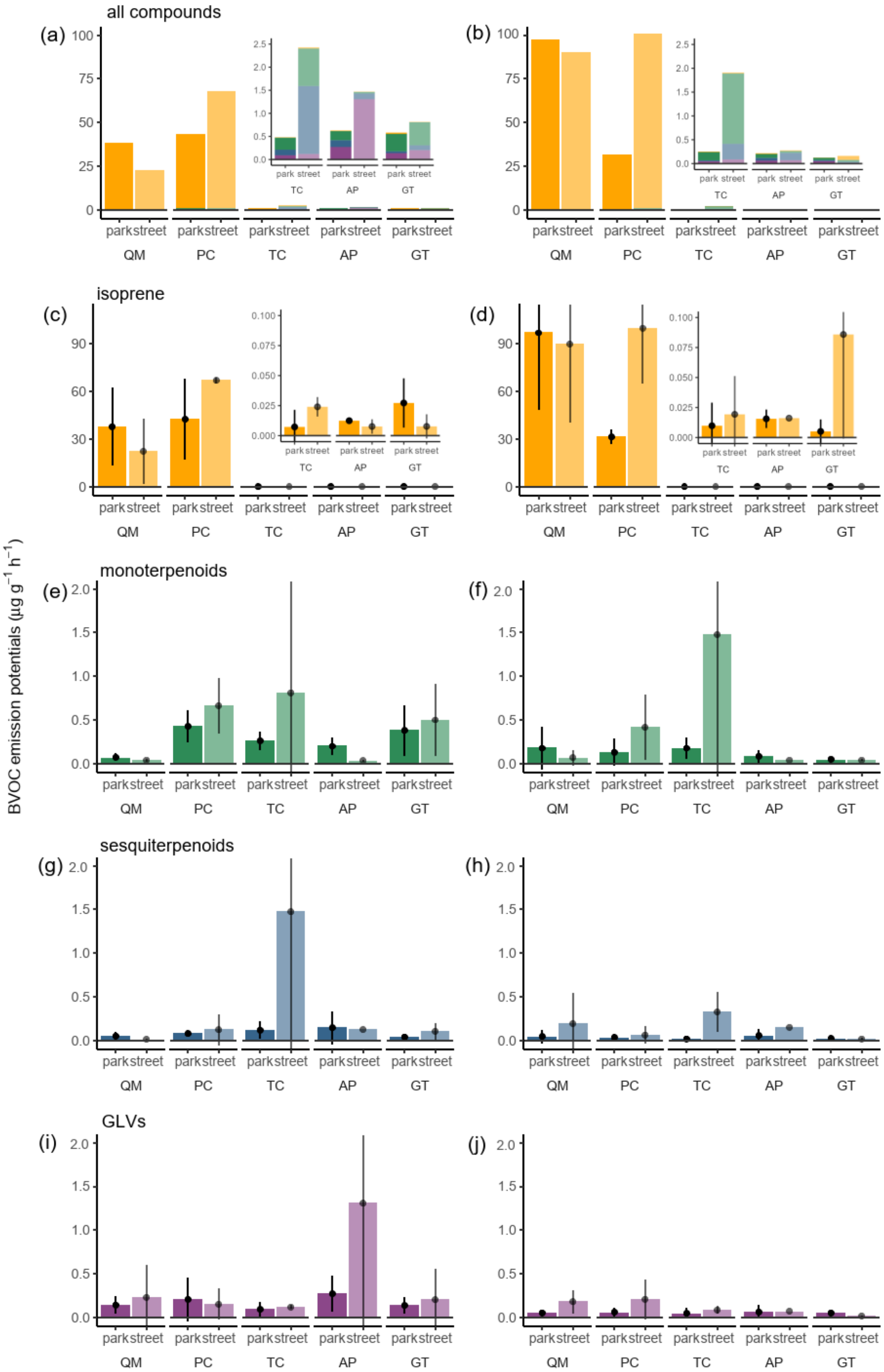

315 **Figure 1. BVOC emission potentials across all compound groups (a–b) and separately for isoprene (c–d), monoterpenoids (e–f), sesquiterpenoids (g–h) and green leaf volatiles (GLVs, i–j), for study species (QM, *Quercus marcrocarpa;* PC, *Populus x canescens*; TC, *Tilia cordata;* AP, *Acer platanoides*; GT, *Gleditsia triacanthos*) in parks and streets in Montreal, measured in June (period I: a, c, e, g, and i) and August (period II: b, d, f, h, and j) 2022. The error bars show the 95% confidence intervals for the mean emission potentials. For each bar, $n_{obs} = 2–3$.**

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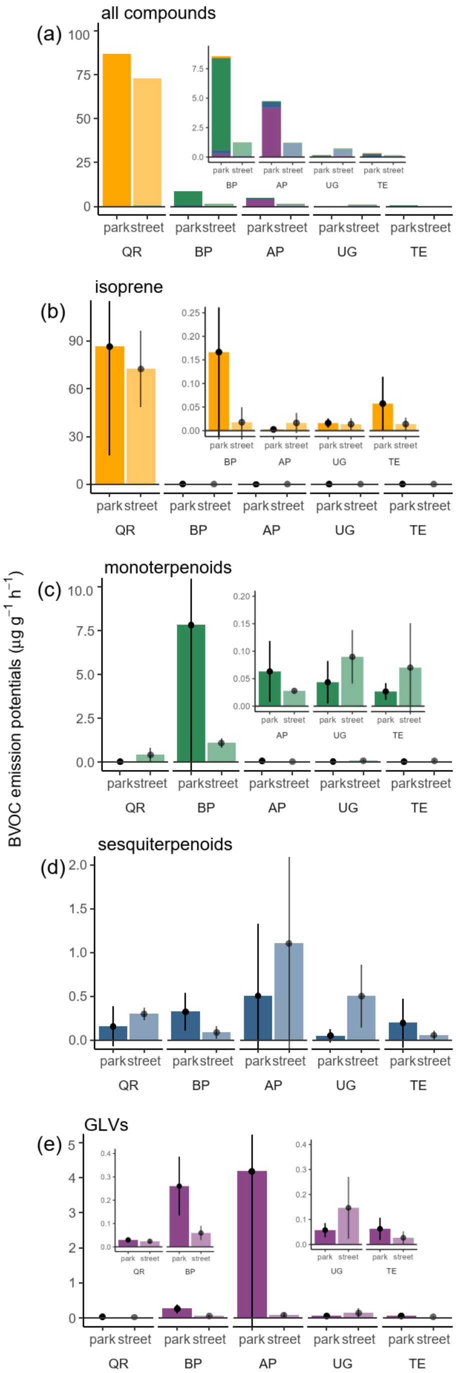

Figure 2. BVOC emission potentials across all compound groups (a) and separately for isoprene (b), monoterpenoids (c), sesquiterpenoids (d), and green leaf volatiles (GLVs, e) for the study species (QR, *Quercus robur*; BP, *Betula pendula*; AP, *Acer platanoides;* UG,*Ulmus glabra*; TE, *Tilia x europea*) in parks and streets in Helsinki, in July 2022. The error bars show the 95% confidence intervals for the mean emission potentials. For each bar, $n_{obs} = 3$.

The total BVOC emission potentials appeared generally higher for street than park trees in Montreal, except for QM (Fig. 1), but higher for park than street trees in Helsinki, except for UG (Fig. 2). Across all tree species, the isoprene and sesquiterpenoid emission potentials were higher for street than park trees in Montreal during period II ($F_{1,22} = 5.91$, $p = 0.024$ for isoprene; $F_{1,23} = 5.00$, $p = 0.042$ for sesquiterpenoids, Fig. 1h, Table S8), specifically for PC (isoprene, Fig. 1d) and TC (sesquiterpenoids, Fig. 1h). In Helsinki, the GLV emission potentials were higher for park than street trees ($F_{1,24} = 6.72$, $p = 0.016$, Table S8), specifically for AP and BP (Fig. 2e). The monoterpenoid emission potential differences between park and street trees were varied and depended upon the species, particularly in Helsinki (for site type and species interaction, $F_{4,20} = 6.72$, $p = 0.016$, Table S8, Fig. 2c).

3.2.2 Comparison to emission databases

The isoprene, monoterpenoid, and sesquiterpenoid emission potentials of urban trees lay primarily near or below the reference emission potentials (Fig. 3, Tables S5 and S6). More specifically, the isoprene emission potentials of QM and PC in Montreal during period I (Fig. 3a), the monoterpenoid emission potential of AP in both cities (Figs. 3d–f) and the sesquiterpenoid emission potential of BP in Helsinki (Fig. 3i) were lower than expected based on the database values. The largest positive deviations from the reference emission potentials were for the QM isoprene emission potentials in Montreal during period II (Fig. 3b) and the QR isoprene emission potentials and the BP monoterpenoid emission potentials in Helsinki (Figs. 3c and 3f). In addition, the sesquiterpenoid emission potentials were slightly higher than the reference emission potentials for TC in Montreal during period I and for AP in Helsinki (Figs. 3g and 3i). Given the large tree-to-tree variation, these differences were, however, not significant (Tables S5 and S6). Significant but small positive deviations from the reference emission potentials were found for the PC and TC monoterpenoid emission potentials in Montreal (Figs. 3d–e, Table S5).

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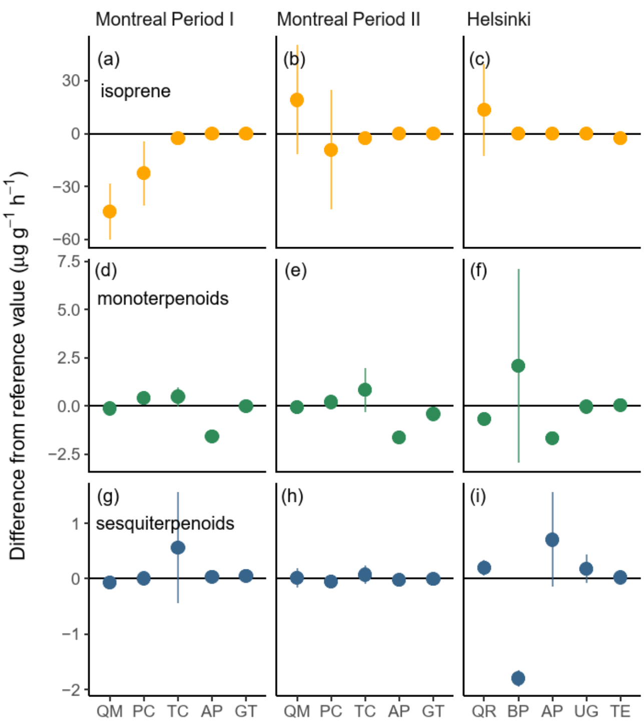

Figure 3. The difference between the isoprene (a–c), monoterpenoid (d–f), and sesquiterpenoid (g–i) emission potentials measured from urban trees and the reference values (mean value of the emission potentials collected from the emissions databases), separately for species in Montreal in June (period I: a, d and g) and August (period II: b, e, and h) and in Helsinki in July (c, f, and i). The dots indicate the mean difference across the trees (n_{obs} = 4–6) and the error bars show the 95% confidence intervals (some are smaller than the dot diameter). Species: AP, *Acer platanoides;* GT, *Gleditsia triacanthos*; PC, *Populus x canescens*; QM, *Quercus marcrocarpa*; TC, *Tilia cordata*; BP, *Betula pendula*; QR, *Quercus robur*; TE, *Tilia x europea;* UG, *Ulmus glabra*.

3.3 O_3 and SOA formation potentials of urban tree BVOCs

The calculated OFP and SOAFP differed significantly between tree species, both in Montreal and Helsinki (Fig. 4, Table S8). OFP was highest in PC and QM in Montreal and QR in Helsinki because of their strong isoprene emission potentials (Figs. 4a–c). The SOAFP differences between species were also driven by the sesquiterpenoid and monoterpenoid emission potentials (Figs. 4d–f). Thus, TC in Montreal during period I and BP in Helsinki approached the PC, QM, and QR SOAFP because of their higher sesquiterpenoid or monoterpenoid emission potentials (Figs. 4d and 4f).

Across species, OFP and SOAFP were higher in street trees than in park trees in Montreal during period II ($F_{1,22}$ = 7.74, p = 0.011 for OFP; and $F_{1,22}$ = 8.41, p = 0.008 for SOAFP; Table S8). In Helsinki, we observed no similar differences (Figs. 4c and 4f, Table S8).

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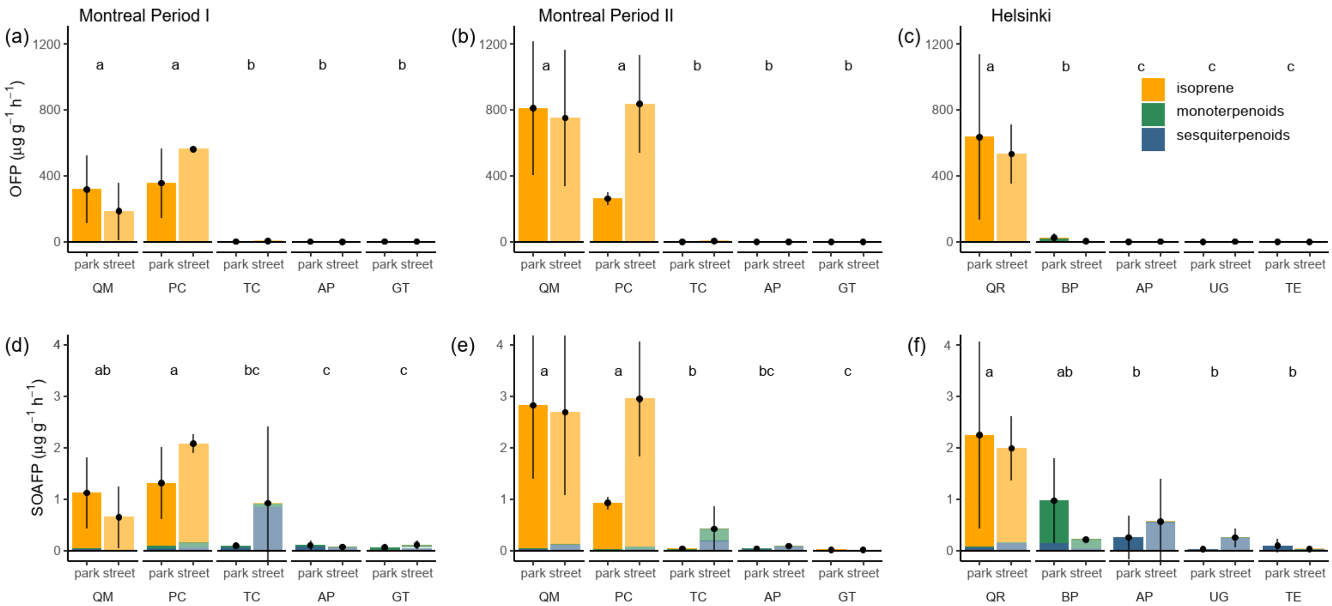

Figure 4. O_3 formation potential (OFP, a–c) and SOA formation potential (SOAFP d–e) of the isoprene, monoterpenoid, and sesquiterpenoid emissions by urban tree species in Montreal in June (period I: a and d) and August (period II: b and e) and in Helsinki in July (c and f). The dots with error bars indicate the mean and the 95% confidence intervals for the OFP or SOAFP per species (n_{obs} = 1–3) and the colours indicate the contributions of each compound group. OFP or SOAFP were calculated from the BVOC emission potentials normalised using the median temperature for the sampling period (28°C in Montreal and 27°C in Helsinki). For the OFP or SOAFP calculated from the BVOC emission potentials normalised for 30°C, see Fig. S8. The different lowercase letters indicate a significant difference between species within the city and measurement period in Montreal. Species: AP, *Acer platanoides*; GT, *Gleditsia triacanthos*; PC, *Populus x canescens;* QM, *Quercus marcrocarpa;* TC, *Tilia cordata*; BP, *Betula pendula*; QR, *Quercus robur*; TE, *Tilia x europaea*; UG, *Ulmus glabra*.

3.4 Neighbourhood-scale BVOC emission potentials, OFP and SOAFP of study species

The isoprene emissions of oaks (QM in Montreal and QR in Helsinki) dominated the total isoprene emission potentials in the upscaling test areas in both cities (Figs. 5a–b), despite their small proportion of the total canopy area within the upscaling test areas (Figs. 5k–l). The species most common in the canopy area of the upscaling test area in Montreal—AP, GT, and TC—were responsible for most of the total monoterpenoid and sesquiterpenoid emission potentials (Figs. 5c–e). In Helsinki, most of the monoterpenoid emissions originated from BP despite its smaller proportion of total canopy in the upscaling test area, while the sesquiterpenoid emissions primarily originated from AP and TE with large canopy areas (Figs. 5d and 5f). The total OFP closely mirrored the total isoprene emission potential per land area (Figs. 5g–h), whereas the total SOAFP in both cities was impacted more by the common species and their monoterpenoid and sesquiterpenoid emissions (Figs. 5i–j).

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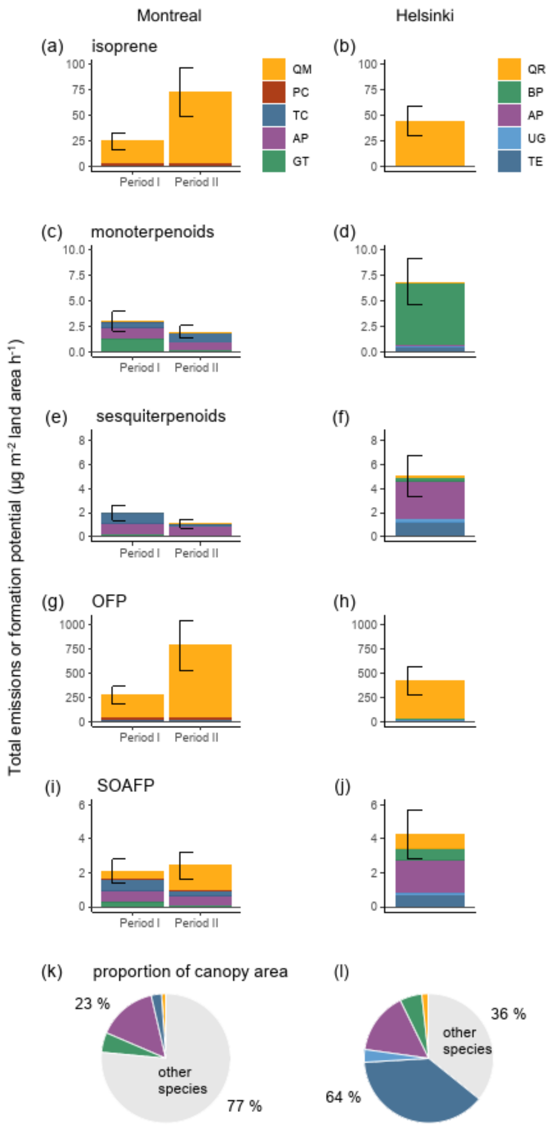

Figure 5. Upscaled total emission potential per land area for isoprene (a–b), monoterpenes (c–d), and sesquiterpenoids (e–f), as well as for their O_3 formation potential (OFP, g–h) and SOA-formation potential (SOAFP, i–j) of the study species within the upscaling test area in Montreal in June and August (periods I and II: a, c, e, g and i) and Helsinki in July 2022 (b, d, f, h and j). The BVOC emission, OFP, and SOAFP intensity estimates only include the individuals from our study species, whereby their proportions of the entire canopy area within the upscaling test areas are listed for Montreal (k) and Helsinki (l). The vertical brackets show the estimated total emissions or formation potentials without correction for shading (upper limit) and a stronger correction for shading (lower limit).

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4 Discussion

4.1 Urban tree BVOC emission rates and variation in the urban landscape

In this study, we provided the BVOC emission potentials for nine urban tree species, measured in the city from mature urban trees. We sampled BVOCs from park and street trees to account for variations in urban growth conditions, finding that the BVOC emission potentials differed between the two site types (park and street) depending upon the city. In Montreal, the BVOC emission potentials were higher among street trees than park trees, particularly for isoprene and sesquiterpenoids. By contrast, in Helsinki, the BVOC emission potentials were generally higher among park trees than street trees, particularly for green leaf volatiles (GLVs). However, the large intraspecific tree-to-tree variation we observed within sites suggests that either inherent tree characteristics or small-scale variations in growth environments also play a major role in the variation of tree BVOC emissions.

Multiple factors can affect the emission potential differences between parks and streets. On the one hand, higher temperatures in the street environment (Bowler et al., 2010) can increase the risk of heat stress and cause higher BVOC emission potentials (Bao et al., 2023; Niinemets, 2010; Pollastri et al., 2021). In addition, high concentrations of O_3 trapped in street canyons (Abhijith et al., 2017; Karttunen et al., 2020) can increase the emissions of monoterpenoids, sesquiterpenoids, and GLVs (Bao et al., 2023; Ghirardo et al., 2016; Lim et al., 2024), but inhibit those of isoprene (Bellucci et al., 2023). Moreover, mechanical damage from traffic and pedestrians may be more frequent on streets, inducing the emissions of stress-related BVOCs such as GLVs and specific terpenoids (Holopainen and Gershenzon, 2010; Panthee et al., 2022; Portillo-Estrada et al., 2015). On the other hand, higher CO_2 concentrations on streets (Gratani and Varone, 2007, 2014) may inhibit isoprene and monoterpenoid emissions (Bao et al., 2023; Bellucci et al., 2023), while a diminished availability of water (due to impermeable street surfaces) can reduce the production and emissions of BVOCs in the long term (Bao et al., 2023; Niinemets, 2010). Finally, factors such as de-icing salts (Cekstere et al., 2020; Helama et al., 2020) and variations in the light environment (Simon et al., 2019) can also impact differences in the BVOC emission levels between park and street trees.

The higher BVOC emission potentials from street than park trees in Montreal may be explained by the generally higher temperatures on streets (Fig. S5). We found that the isoprene emission potential differences between site types were largest during period II in August when the temperature difference between parks and streets was also greatest. While O_3 concentrations were also generally higher on streets than in parks in Montreal, they were generally low and thus unlikely to strongly impact the emissions. Compared with Montreal, Helsinki is a smaller city with a lower population density (3032 km^{-2} in Helsinki vs 4834 km^{-2} in Montreal) and a cooler summer climate. Consequently, we observed no differences in the park and street ambient temperatures, light intensities and O_3 concentrations during the sampling in that city (Fig. S5), suggesting small street–park differences in potential tree stress levels. To explain the generally higher BVOC emission potentials from park versus street trees in Helsinki, further research is needed which attempts to identify the emissions drivers, for example, the potential unidentified stress factors. That the GLV emission potentials were specifically higher among park trees suggests that mechanical damage or biotic effects may play a role.

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Similar BVOC emission potential comparisons have been conducted across rural–urban gradients. For instance, Lahr et al. (2015) reported higher isoprene emission rates in urban and suburban sites than in rural sites for *Quercus stellata* and *Liquidambar styraciflua* in Houston, Texas, USA. By contrast, Duan et al. (2023) reported higher isoprene and monoterpene emission rates in rural versus suburban and urban sites amongst a combination of broad-leaved species and two pine species in Xiamen, China. Correspondingly, Yu (2023) observed higher isoprene emission rates for *Pinus densiflora* and higher isoprene and monoterpene emission rates for *Acer palmatum* in rural versus urban forests in South Korea. If we consider the gradient from parks to streets as a continuation of the rural–urban gradient, our results from Montreal partially correspond to the results of Lahr et al. (2015) with the higher isoprene emissions in more urbanised environments. Our results from Helsinki, on the other hand, lie closer to those of Duan et al. (2023) and Yu (2023), finding higher BVOC emission potentials in less urbanised environments.

4.2 Urban tree BVOC emission rates in comparison to nonurban trees

We hypothesised that urban stress factors both on streets and in parks would cause the urban tree BVOC emission potentials to differ from the emission factors presented in databases often used for BVOC emission budgeting or modelling. Notably, we found generally close agreement between the measured mean BVOC emission potentials and the reference emission potentials, with a few significant deviations.

Among species with low BVOC emission potentials, such as AP (*Acer platanoides*), GT (*Gleditsia triacanthos*), TE (*Tilia x europaea*), and UG (*Ulmus glabra*), the measured isoprene or monoterpenoid emission potentials were often smaller than expected, suggesting that the emission potentials in databases likely overestimate the emission potentials of little-studied, low-emitting species. Similar but more drastic differences between the measured isoprene emission potentials and genus-level emission estimates have been reported for subtropical urban tree species in Australia (Dunn-Johnston et al., 2016). More precisely, they found that replacing the genus-level estimates with the measured species-specific isoprene emission potentials reduced the estimated total isoprene emissions of the urban study species by 97%. In contrast to isoprene and monoterpenoids, the sesquiterpenoid emission potentials from some of our study trees among the low-emitting species greatly exceeded the reference emission potentials. On the one hand, the higher-than-expected sesquiterpenoid emissions can be affected by uncertainties in the database estimates, given that the sesquiterpenoid emissions are generally low and thus less commonly measured or reported. On the other hand, given the sensitivity of the sesquiterpenoid emissions to mechanical disturbances (Duhl et al., 2008), the occasional high sesquiterpenoid emissions that also exceed the monoterpenoid emissions may have been driven by local, small-scale stress factors.

Among species with known high or moderate BVOC emission potentials, we observed larger but not significant departures from the isoprene reference emission potentials. In Montreal, PC *(Populus x canescens)* and QR (*Quercus macrocarpa)* showed, in comparison to the reference emission potentials, generally lower isoprene emission potentials during period I and higher isoprene emission potentials during period II. In Helsinki, the QR (*Q. robur*) isoprene emission potentials also exceeded

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the reference emission potentials. The highest isoprene emission potential peaks occurred on warm days with intense radiation (Figs. S6 and S7), perhaps indicating a link to the thermal- and photoprotection roles of isoprene (Peñuelas and Munné-Bosch, 2005; Pollastri et al., 2021).

Because the agreement between our results and the reference emission potentials was generally close, the BVOC emission potential estimates do not appear to serve as a large source of error in creating urban BVOC emission budgets in cities like Montreal and Helsinki (small to mid-sized cities with cool, humid climates). Greater uncertainties potentially arise from estimates of tree biomass or leaf area index (Yang et al., 2022) or from inaccuracies in tree inventories (Bao et al., 2024). However, previous research reported considerable variation in the species-wise BVOC emission potentials between cities (Bao et al., 2023; Préndez et al., 2013), highlighting the importance of further studies on BVOC emission variability among urban trees globally.

4.3 Air quality implications of urban tree BVOC emissions

In addition, we estimated the ozone-formation potentials (OFP) and SOA-formation potentials (SOAFP) for the typical urban tree species in Montreal and Helsinki, both as leaf-level potentials and tentatively upscaled to the neighbourhood level. The leaf-level OFP was highest among species with high isoprene emission rates—that is, PC and QM in Montreal and QR in Helsinki—while other BVOCs contributed minimally. At the neighbourhood-level, the isoprene emissions of oaks dominated OFP in both cities despite their small canopy coverage in the upscaling test areas, indicating that even small numbers of isoprene-emitting individuals can considerably impact the local BVOC OFP. This result agrees with earlier recommendations to avoid isoprene-emitting *Quercus* and *Populus* species in cases where local O_3 concentrations require control (Datta et al., 2021; Manzini et al., 2023). In addition, management strategies aimed at protecting already existing high-emitting trees from isoprene-inducing stress factors, such as heat or intensive light (Czaja et al., 2020; Peñuelas and Munné-Bosch, 2005; Pollastri et al., 2021), could be explored to limit their effects on O_3 concentrations.

In comparison to OFP, the leaf-level SOAFP varied less between species because of the larger contributions of monoterpenes and sesquiterpenes on SOA formation. For example, the SOAPF of TC in Montreal and BP in Helsinki were of similar magnitudes as the SOAPF of QM and QR (isoprene emitters) due to their higher sesquiterpene (TC) or monoterpene (BP) emission potentials. The same pattern also occurred at the neighbourhood-level: the small monoterpenoid and sesquiterpenoid emissions of species with large areas of the canopy (AP and TC or TE) contributed to SOAFP per land area as much (Montreal) or more (Helsinki) than the high isoprene emissions of oaks with small areas of the canopy. Thus, to control local SOA production, species with high isoprene, monoterpenoid, or sesquiterpenoid emission potentials should be avoided (Datta et al., 2021). However, even small monoterpenoid and sesquiterpenoid emission levels from species planted in large quantities accumulate considerable SOA contributions, whereby identifying and avoiding the local environmental conditions that can lead to high emission peaks from existing trees may be more important than species selection.

We note that our neighbourhood-scale upscaling aimed to explore the urban tree BVOC effects on O_3 and SOA formation when accounting for the realistic distributions of our study species. More detailed information on the impact of each BVOC

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on SOA or O_3 formation, along with the more precise quantification of urban trees, including private trees and urban forest patches, their proportion canopy coverage and leaf area index, would be necessary to comprehensively quantify the total neighbourhood or city-scale BVOC emissions and potential air quality impacts (Bao et al., 2024; Yang et al., 2022).

Conclusions

In this study, we performed direct BVOC measurements of urban trees and provided the isoprene, monoterpenoid, sesquiterpenoid, and GLV emission potentials for nine urban tree species. We observed city-dependent differences in the BVOC emission potentials between park and street trees, but no large deviations in the measured BVOC emission potentials from the nonurban reference emission potentials. While the urban tree BVOC emission potentials were not higher than expected, they may still impact air quality. Based on our findings, avoiding planting isoprene-emitting species would help control O_3 formation. To control SOA formation, however, it seems more important to protect urban trees from stress impacts, such as mechanical damage or high O_3 concentrations, which may increase their monoterpenoid and sesquiterpenoid emissions. Further studies should explore the most important stress factors impacting urban trees and potential management practices to mitigate them.

The negative air quality effects of tree BVOC emissions need to be combined with other negative (reducing airflow in street canyons) and positive (capturing pollutants and particulate matter, reducing dispersion of pollutants from the source) air quality impacts of trees to form a comprehensive view of the net effects of trees and tree species (see, e.g., Maison et al., 2024). To achieve this, accurate estimates of the urban tree BVOC emission potentials are crucial. Although our results support using nonurban database values as estimates of urban tree BVOC emissions, further direct measurements of urban tree BVOC emissions in various cities with differing sizes and climates, including during acute stress events such as high O_3 concentrations, heat waves, or prologued droughts, are needed to scope where and when the database values can be applied. Our findings regarding the slightly differing BVOC emission potentials between park and street trees also highlight the importance of including various urban site types in future sampling efforts of urban tree BVOC emissions. In addition to parks and streets, private trees in residential areas and urban forest patches would be interesting inclusions, given that they make up a large proportion of the urban forest and may be managed differently (Sousa-Silva et al., 2023).

Data availability

Data is publicly available at DOI:10.5281/zenodo.15379394

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Author contributions

KR conceptualised the study, investigated, curated the data (with TT), conducted the formal analysis, wrote the original draft, reviewed and edited the final manuscript, and acquired funding. JA and TT contributed to the methodology and reviewed and edited the written manuscript. JB, HH, and AP conceptualised the study, provided resources, and contributed to the writing through reviewing and editing the manuscript. AP supervised the project and acquired funding for this study.

Competing interests

The authors declare that they have no conflict of interest.

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