# Air quality and health effects of biogenic volatile organic compounds emissions from urban green spaces and the mitigation strategies☆

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## ABSTRACT

Biogenic volatile organic compounds (BVOCs) emissions lead to fine particulate matter (PM$_{2.5}$) and ground-level ozone pollution, and are harmful to human health, especially in urban areas. However, most BVOCs estimations ignored the emissions from urban green spaces, causing inaccuracies in the understanding of regional BVOCs emissions and their environmental and health effects. In this study, we used the latest local vegetation datasets from our field survey and applied an estimation model to analyze the spatial-temporal patterns, air quality impacts, health damage and mitigating strategies of BVOCs emissions in the Greater Beijing Area. Results showed that: (1) the urban core was the hotspot of regional BVOCs emissions for the highest region-based emission intensity (3.0 g C m$^{-2}$ yr$^{-1}$) among the 11 sub-regions; (2) urban green spaces played much more important roles (account for 62% of total health damage) than rural forests in threating human health; (3) BVOCs emissions from green spaces will more than triple by 2050 due to urban area expansion, tree growth and environmental changes; and (4) adopting proactive management (e.g. adjusting tree species composition) can reduce 61% of the BVOCs emissions and 50% of the health damage related to BVOCs emissions by 2050.

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## 1. Introduction

Volatile organic compounds (VOCs), including both biogenic sources (BVOCs) and anthropogenic sources (AVOCs), contribute to the formation of ground-level ozone and secondary organic aerosols (SOA), with great impacts on outdoor air pollution (Laothawornkitkul et al., 2009; Simpson and McPherson, 2011; Mochizuki et al., 2015). In particular, BVOCs originating from vegetation are believed to play more important roles owing to their higher source strengths and chemical reactivity compared with AVOCs (Guenther et al., 1995; Atkinson, 2000; Harrison et al., 2013). BVOCs induced increments in the atmospheric ozone and fine particulate matter (PM$_{2.5}$) concentrations will further increase

human's respiratory and cardiovascular mortality risks (Heal et al., 2013; Lelieveld et al., 2015; Madaniyazi et al., 2016). Nevertheless, only a few quantitative studies have been carried out to investigate the environmental and health outcomes of BVOCs (Benjamin and Winer, 1998; Ghirardo et al., 2016).

Urban areas are hotspots of air pollution and health damages. At present, more than half of the world's population lives in urban areas, and this ratio is expected to reach 70% by 2050 (Ramalho and Hobbs, 2012). Following the continuous urban expansion worldwide and the urban population's growing demands for ecosystem services, urban green spaces have expanded even faster than the increase in urban population (Fuller and Gaston, 2009; Seto et al., 2012). However, in addition to providing various regulating services and cultural services (Escobedo et al., 2011; Pataki et al., 2011), green spaces also become an important contributor to regional BVOCs emissions (Chang et al., 2012; Calfapietra et al., 2013). As the atmosphere in urban area is often dominated by VOC-limited regimes due to high anthropogenic NOx emissions, green spaces may contribute significantly to ozone production (Carter, 2007;

Calfapietra et al., 2013). Moreover, when BVOCs from green spaces occur in urban areas with high human population densities, they can have much greater health damages than those from natural forests (Ren et al., 2014). The neglect of urban BVOCs emissions may result in significant underestimations of regional and global emissions. More importantly, the health losses associated with BVOCs emissions in urban areas will be greatly underestimated.

Numerous efforts have been made worldwide to quantify BVOCs emissions across different spatial and temporal scales (Klinger et al., 2002; Steinbrecher et al., 2009; Guenther et al., 2012; Oderbolz et al., 2013). Nevertheless, only a few case studies focused on estimating BVOCs emissions from urban areas (Chang et al., 2012; Dunn-Johnston et al., 2016) due to the lack of suitable methods for studying urban vegetation (Kaye et al., 2006; Guenther et al., 2012). As the result of a warmer climate, together with improvements in gardening practices, urban greenings have brought together a rich mixture of species to improve the biodiversity and aesthetic values of cities, forming species compositions that are distinct from surrounding rural forests (Niinemets and Peñuelas, 2008). Since BVOCs emissions are species specific (Benjamin et al., 1996; Tani and Kawawata, 2008), the different species compositions between urban and rural areas may lead to significant disparities in BVOCs emission patterns (Chang et al., 2012). The urban heat island effect and different light conditions through the canopy can further enlarge the regional disparity (Ren et al., 2014). BVOCs estimations that fully consider the human management factors and the specific environmental factors in urban areas are thus urgently required.

In this study, we chose a megacity, the Greater Beijing Area, as a case study. As the capital city of China, Beijing is one of the largest and most developed cities in China. In recent years, ozone pollution in Beijing has been aggravated despite a significant decline in anthropogenic emissions by the implementation of various control measures (Wang et al., 2015a; Li et al., 2016a). The contrasting trends may be explained by the rapid increase of BVOCs emissions. The aim of this study is to (1) investigate the BVOCs emissions from urban green spaces; (2) quantify the air quality impacts and health damage associated with BVOCs emissions; and (3) discuss the effectiveness of green space management strategies on mitigating future BVOCs emissions to aid decision making.

## 2. Materials and methods

### 2.1. Study area

The Greater Beijing Area ($39°28'–41°25'$ N, $115°25'–117°130'$ E) is located in the northeast of the North China plain, containing an urban core and periphery, covering a total land area of 16, 410 km$^2$. This region is characterized by a temperate, humid, continental monsoon climate, with annual average temperature 11–12 °C and annual precipitation around 500 mm. Beijing has been undergoing rapid urbanization and its built-up area increased from 397 km$^2$ in 1990 to 1386 km$^2$ in 2015 (NBSC, 1991–2015). The city has also been plagued by serious air pollution for years (Cumming et al., 2014). In 2015, its annual average PM$_{2.5}$ concentration and O$_{3-8h}$ 90 per concentration (90th percentile of the daily maximum 8-h average ozone concentrations) were 80.6 μg m$^{-3}$ and 202.6 μg m$^{-3}$, which exceeded the National Ambient Air Quality Standards (GB 3095–2012) Grade II by 1.3 and 0.3 times, respectively (BMEPB , 2016).

### 2.2. Field measurement and data collection

In this study, we divided the Greater Beijing Area into 11 subregions: an urban core, 5 semi-urbanized sub-regions, and 5 less urbanized rural sub-regions (Fig. 1A). We conducted an urban vegetation survey in the years 2012, 2013 and 2016. Urban green spaces were divided into five types: street green space, park green space, affiliated green space (in schools and institutional units), protection green space (e.g. greenbelts, windbreak green space) and residential green space. A stratified random selection method was applied and a total of 282 plots (400 m$^2$) were investigated. In each plot, tree attributes for species (Table S1), stem diameter at breast height (DBH), and tree height of all trees (DBH>2.5 cm) were recorded. Historical data on the number of trees, species composition and areal extent of green spaces were collected from published literature (BGGB, 1995–2005; Ghirardo et al., 2016).

Natural forests were divided into 22 forest types based on their dominant tree species (Table S2). The data of forest area, tree density, spatial distribution, and DBH of each forest type were from the latest (8th) national forest resource inventory (2009–2013) and published literature (Li et al., 2011; Luo et al., 2013). Historical forest areas by age classes as well as by forest types were derived from the 2nd–7th national forest resource inventory (SFAC, 1982–2010). Specific leaf area (SLA) and annual increment of DBH (Tables S3 and S4) of different tree species were also collected to simulate the changes in leaf area index (LAI) and foliar mass during tree growth.

BVOCs were grouped into four categories: isoprene, monoterpenes, sesquiterpenes, and other VOCs (OVOCs). The raw emission rates (Table S5-S7) measured in our previous studies (Chang et al., 2012; Ren et al., 2014) and other studies (e.g. Klinger et al., 2002; Wang et al., 2003) were assigned to primary tree species using a taxonomic approach (Benjamin et al., 1996). All these emission rates have been standardized as basal emission rates under standard environmental conditions. The light dependent fraction (LDF) of moneterpenes was obtained from published literature (Table S8). In this study, we defined a tree species as high-emitting species if the sum of its isoprene and monoterpenes basal emission rates exceeded 10 μg of C g$^{-1}$ h$^{-1}$; if the sum is less than 10 μg of C g$^{-1}$ h$^{-1}$ and higher than 1 μg of C g$^{-1}$ h$^{-1}$, the species was defined as middle-emitting species; otherwise it is a low-emitting species (Ren et al., 2014).

To calculate the environmental corrections on BVOCs emissions, hourly air temperature and photosynthetically active radiation (PAR) data obtained from the meteorological data center of the China Meteorological Administration and monthly CO$_2$ concentration data from the Shangdianzi Baseline Observatory were used.

Base incremental reactivity (Carter, 2007) and fractional aerosol coefficient (e.g. Griffin et al., 1999; Karl et al., 2009; Kiendler-Scharr et al., 2012) of different BVOCs species were collected (Table S9) to estimate the effects of BVOCs on air quality. Besides BVOCs, other precursors exist for both ozone and SOA. To assess the relative contributions of BVOCs and other precursors, we also collected the summertime ambient concentrations of AVOCs, carbon monoxide and methane, and the chemical speciation of AVOCs from published literature (Wang et al., 2015b; Li et al., 2015b; Li et al., 2016a; CNEMC , 2016).

### 2.3. Calculation of BVOCs emissions

In this study, the basic BVOCs algorithms are derived from the widely used equations by Guenther (Guenther et al., 1995; Guenther et al., 1999) and are revised based on other studies (Staudt et al., 2000; Heald et al., 2009). Isoprene emission was treated as synthesis emission (light-dependent); sesquiterpenes and OVOCs were treated as pool emissions (light-independent); while monoterpenes were assumed to have both synthesis emissions and pool emissions (Steinbrecher et al., 2009; Oderbolz et al., 2013). The synthesis isoprene ($E_{ISP}$) and monoterpenes ($E_{MNS}$) emissions were quantified as: