



Enhanced summertime ozone and SOA from biogenic volatile organic compound (BVOC) emissions due to vegetation biomass variability during 1981–2018 in China

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Abstract. Coordinated control of fine particulate matter (PM_{2.5}) and ozone (O₃) has become a new and urgent issue for China's air pollution control. Biogenic volatile organic compounds (BVOCs) are important precursors of O₃ and secondary organic aerosol (SOA) formation. China experienced a rapid increase in BVOC emissions as a result of increased vegetation biomass. We applied WRF-Chem3.8 coupling with MEGAN2.1 to conduct long-term simulations for impacts of BVOC emissions on O₃ and SOA during 1981–2018, using the emission factors extrapolated by localized emission rates and annual vegetation biomass. In summer 2018, BVOC emissions were 9.91 Tg (in June), which led to an average increase of 8.6 ppb (16.75 % of the total) in daily maximum 8 h (MDA8) O₃ concentration and 0.84 µg m⁻³ (73.15 % of the total) in SOA over China. The highest contribution to O₃ is concentrated in the Great Khingan Mountains, Qinling Mountains, and most southern regions while in southern areas for SOA. Isoprene has the greatest contribution to O₃, while monoterpene has the largest SOA production. BVOC emissions have distinguished impacts in different regions. The Chengdu–Chongqing (CC) region has the highest O₃ and SOA generated by BVOCs, while the Beijing–Tianjin–Hebei (BTH) region has the lowest. From 1981 to 2018, the interannual variation of BVOC emissions caused by increasing leaf biomass resulted in O₃ concentration increasing by 7.38 % at an average rate of 0.11 ppb yr⁻¹ and SOA increasing by 39.30 % at an average rate of 0.008 µg m⁻³ yr⁻¹. Due to the different changing trends of leaf biomass by region and vegetation type, O₃ and SOA show different interannual variations. The Fenwei Plain (FWP), Yangtze River Delta (YRD), and Pearl River Delta (PRD) regions have the most rapid O₃ increment, while the increasing rate of SOA in CC is the highest. BTH has the smallest enhancement in O₃ and SOA concentration. This study will help to recognize the impact of historical BVOC emissions on O₃ and SOA and further provide a reliable scientific basis for the precise prevention and control of air pollution in China.

1 Introduction

In recent years, China has suffered from more and more severe O₃ pollution, with continuously increasing O₃ concentration in most regions (H. Liu et al., 2018; Liu and Wang, 2020; Wang et al., 2017). On the other hand, although the ambient concentrations of PM_{2.5} have decreased, its pollution is still severe (Fan et al., 2020; Li et al., 2019b; Silver et al., 2018; Zhai et al., 2019). PM_{2.5} and O₃ are currently important atmospheric pollutants affecting air quality in urban and regional areas of China. Their coordinated prevention and control has become a new and urgent issue for China's air pollution control. Volatile organic compounds (VOCs) are precursors of O₃ and secondary organic aerosol (SOA) formation (Claeys et al., 2004; Hallquist et al., 2009; Kota et al., 2015), which can be emitted by biogenic and anthropogenic sources. On the global scale, up to 90 % of total VOC emissions come from biogenic sources, including 99 % from vegetation (Guenther et al., 1995). Therefore, the biogenic VOCs (BVOCs) in this study refer to the VOCs emitted by vegetation. BVOCs play a key role in the formation of secondary pollution due to their large emission and high activity, including isoprene, monoterpene, and sesquiterpene (Carslaw et al., 2010; Emanuelsson et al., 2013; Ng et al., 2008; Tasoglou and Pandis, 2015). Globally, BVOC emissions contribute about 20 % and 76 % to the O₃ and SOA formation, respectively (Hallquist et al., 2009; Wang et al., 2019). In China, BVOC emissions are estimated to be approximately 1.8 times those of anthropogenic VOCs (Li et al., 2016; Yang et al., 2021). The high BVOC emissions can lead to great increases by 29 %–49 % in surface O₃ concentration in most southern urban areas (Li et al., 2018; Y. Liu et al., 2018; Situ et al., 2013; Wu et al., 2020) and contribute 70 %–75 % to China's total SOA formation in summer (Hu et al., 2017; Wu et al., 2020). Therefore, it is essential to understand the emission characteristics of BVOCs and their impacts on the formation of O₃ and SOA for making effective policies of secondary air pollution control in China.

In the past 4 decades, China's BVOC emissions have increased by 55.38 % due to biomass growth and climate change, of which biomass variability dominates the interannual variations in emission, which were reported in our previous studies (Li and Xie, 2014; Li et al., 2021). During 1981–2018, the leaf biomass of forests and crops increased by 118.63 % and 316.83 %, respectively, resulting in a continuous growth trend in BVOC emissions by 0.52 Tg yr^{−1} averagely. The increasing BVOC emissions will affect their contributions to O₃ and SOA formation. Some studies investigated the impact of historical and future BVOC emissions on O₃ and SOA. Fu and Liao (2012) applied GEOS-Chem coupling with the Model of Emissions of Gases and Aerosols from Nature (MEGAN) using the default constant BVOC emission factors and annual leaf area index (LAI) data and found that the interannual variations in BVOCs led to 2 %–5 % differences in simulated O₃ and SOA in summer over the

years 2001–2006. Li et al. (2015) concluded that the BVOC emissions (including isoprene, α -pinene, and β -pinene) from the early 1990s to 2006 caused a 0.9–4.6 ppb increment in O₃ concentrations over the downwind areas in the PRD region, based on annual land cover data. Sun et al. (2019) used MEGAN2.1 to calculate biogenic emissions which only include isoprene and monoterpenes and showed that their interannual changes caused by meteorology led to an increment of 0.3 ppb O₃ over central eastern China from 2003 to 2015. Liu et al. (2019) concluded that climate-driven BVOC emission changes would enhance O₃ and SOA concentrations by 0.90 % and 7.33 % in eastern China from 2015s to 2050s under future climate scenario RCP8.5 and reduce them by 0.80 % and 6.50 % respectively under scenario RCP4.5, simulated by the same emission factors and land cover data. However, the BVOC emission inventories applied in these studies may have high uncertainties because of the lack of localized emission factors and high-resolution land cover data, which is not conducive to the accurate evaluation for the contribution of BVOC emissions on O₃ and SOA formation. In addition, China has experienced continuously increasing forest volume, while few studies focus on the impact of changes in BVOC emissions caused by vegetation biomass variability over a longer time span. This makes it difficult to accurately evaluate the impact of BVOCs on O₃ and SOA and guide the precise control of O₃ and PM_{2.5}. How to obtain the accurate historical contribution of BVOCs to O₃ and SOA based on localized accurate BVOC emissions is both an important issue and a great challenge.

In this study, firstly we updated the localized emission factor and detailed vegetation data to obtain more accurate BVOC emissions. The emission factors were extrapolated by localized emission rates and annual vegetation biomass at the provincial level. Then, based on the estimated historical emissions, their long-term impacts on O₃ and SOA during 1981–2018 were simulated through coupling MEGAN2.1 in WRF-Chem3.8. We aimed to investigate the impact of interannual BVOC emission variations on O₃ and SOA formation caused by vegetation biomass variability. The spatial variations in BVOC effects were discussed. Also, their contribution by different components was studied. It is expected that an accurate evaluation of BVOC contribution to O₃ and SOA can be obtained through improving the accuracy of emission inventory. Our study can help to better understand the role of vegetation biomass variability in the BVOC impact on O₃ and SOA in China in the past 4 decades.

2 Model and simulation

2.1 MEGAN

The MEGAN2.1 was utilized to estimate BVOC emissions in China at a spatial resolution of 36 km × 36 km (Guenther et al., 2012). It can calculate hourly emissions of 147 BVOC species, including isoprene, monoterpene, sesquiter-

pene, carbonyls, and other VOCs. The BVOC emissions simulated by MEGAN2.1 were the online inputs of the WRF-Chem simulation.

Meteorological and vegetation data are inputs to drive MEGAN. MEGAN2.1 requires hourly weather variables to drive the calculation of hourly BVOC emissions. The hourly meteorological fields including temperature, downward shortwave radiation, wind speed, water vapor mixing ratio, pressure, and precipitation were simulated by the WRF model in this study. Temperature and radiation play key roles in BVOC emissions. We used the observed daily average temperature at 2411 sites in 2008 and 684 sites in 2018 in China to evaluate the reliability of the 2 m temperature (T2) simulated by WRF in this study. The observations were from the National Meteorological Data Center in China (<http://data.cma.cn/>, last access: 16 February 2022). The simulated radiation was not evaluated because of a lack of available site observations. We conducted the statistical verification of meteorological variables, as shown in Table S1, including the average mean bias (MB), mean absolute error (MAE), and root-mean-square error (RMSE). The results show that the WRF simulation is considered reasonable for driving MEGAN. The vegetation data include gridded fraction of plant functional types (PFTs), leaf area index (LAI), and PFT-specific emission factors. The new detailed PFT classification and distribution were developed from the Vegetation Atlas of China (1 : 1 000 000) at a high resolution of ~ 250 m, including 82 PFTs of forests, crops, grasses, and shrubs. The atlas was compiled based on multi-year vegetation survey data and related research studies over 30–40 years and provides a more detailed distribution of vegetation in China. Previous studies typically included a coarse vegetation classification that is based on a less detailed vegetation distribution (Gao et al., 2019; Klinger et al., 2002; Wang et al., 2007). And the MEGAN2.1 defined 15 vegetation types by default (Guenther et al., 2012). For LAI, the MODIS LAI data were used.

Emission factors were extrapolated by the leaf-level emission rates and leaf biomass using the canopy environment model described in MEGAN2.1. In our previous studies on BVOC emission inventories, a large number of observations from China and other countries were summarized to obtain more accurate basal emission rates using a theoretically effective statistical approach (Li et al., 2020). The dataset contained isoprene and monoterpene emission rates of 192 plant species/genera including the dominant forest tree and crop species, and grass and shrub genera, which were expected to be more accurate and localized. In previous studies, traditional emission categories were used to determine emission rates (Guenther et al., 1994; Klinger et al., 2002; Simpson et al., 1999; Wang et al., 2007), which usually utilized coarse categories and resulted in high uncertainty. The vegetation species/type-specific leaf biomass at the province level originated from statistics of vegetation volume and production using biomass apportion models, which revealed differences

among plant species and regions compared with previous studies (Li et al., 2013, 2020; Li and Xie, 2014). Previous studies usually applied an average value for each vegetation class, such as broadleaf trees, needleleaf trees, crops, and grasses, without revealing their differences among regions and plant species (Klinger et al., 2002; Wang et al., 2007). The statistics on forest volumes were from the National Forest Inventory of China. Eight forest inventories for the periods of 1977–1981, 1984–1988, 1989–1993, 1994–1998, 1999–2003, 2004–2008, 2009–2013, and 2014–2018 were used. The last year of each period was selected to simulate the historical emissions and concentration of O₃ and SOA during 1981–2018. Crop productions were from the China Statistical Yearbook for the year 1981, 1988, 1993, 1998, 2003, 2008, 2013, and 2018. Grass production was from the Grassland Resource Data of China. Consequently, the extrapolated emission factors vary over the years. After the emission factor calculation of total monoterpene, the emission factors for each monoterpene and sesquiterpene species were allocated from total monoterpene based on the global average emission factor proportions described in MEGAN2.1. For the emission factors of other VOCs, the default MEGAN emission factors were used.

2.2 WRF-Chem

The chemistry version of the WRF model (WRF-Chem3.8) was used in this study to simulate the concentration of surface O₃ and SOA (Grell et al., 2005). WRF-Chem realizes the online coupling of meteorological models and chemical models, considering the emission, transport, diffusion, dry and wet deposition, photolysis, meteorological chemistry, and aerosol chemical processes of pollutants, which have been widely used to conduct the online calculations of meteorology and chemistry. A large number of global and regional air pollution studies widely apply it to simulate secondary pollutants, and the verification results show that it can reproduce the observed pollutant concentrations well (Gupta and Mohan, 2015; Hoshyaripour et al., 2016; Li et al., 2018; Situ et al., 2013; Wu et al., 2018).

One nested domain centered at 34.53° N and 108.92° E was adopted, with a 36 km horizontal resolution covering the whole area of the China region. Main parameterization options for physical and chemical schemes of the model setup are listed in Table S2 in the Supplement. The United States National Centers for Environmental Prediction (NCEP)/Department of Energy (DOE) Reanalysis II data at 1° × 1° were used as the initialization field and boundary conditions for WRF, which are updated every 6 h (00:00, 06:00, 12:00, and 18:00 UTC). The NOAA/ESRL RACM (Stockwell et al., 1997) gas-phase chemistry scheme and the volatility basis set (VBS) (Donahue et al., 2006) aerosol chemistry module were selected in this study. The RACM consists of 77 chemical species and considers 237 reactions, including relatively detailed organic chemistry which con-

siders the oxidation mechanism for BVOCs (e.g., isoprene, α -pinene, and β -limonene). In the VBS, a unified set of saturated vapor pressure is used, and the coupling matrix of the gas phase and condensed phase is established to describe the photochemical multi-generation oxidation and gas–particle partition process. This approach can better represent multi-generation oxidation of BVOCs in the gas phase and their aging processes in the aerosol phase (Hu et al., 2017). The photolysis scheme of Fast-J (Wild et al., 2000) was selected because it can better compute photolysis rates from the predicted O_3 , aerosol, and cloud profiles.

The observed daily maximum 8 h (MDA8) O_3 and daily average $PM_{2.5}$ concentrations at 1588 sites in June 2018 in China were applied to evaluate the WRF-Chem simulations in the control run (as listed in Table 1) in this study. The observations were from the daily updated national air quality released by the China National Environmental Monitoring Centre (<http://www.cnemc.cn/>, last access: 16 February 2022). The verification statistics are shown in Table S1. Notably, $PM_{2.5}$ had no systematic bias between the observation and simulation, while the model-predicted O_3 concentrations were lower than measurements. The errors could be mainly attributed to the anthropogenic emissions data used in this study, as described in Sect. 2.3.

2.3 Simulation setups

In this study, June was selected as the simulated period to investigate the effects of summertime BVOC emissions on O_3 and SOA generation. Seven simulation scenarios were set up to investigate the effects of BVOC emissions by each compound category and historical emission variation. The details of simulation setup are listed in Table 1. We focused on the contribution of historical BVOC emissions caused by biomass variability, so the anthropogenic emissions in all scenarios were fixed using the MIX Asian anthropogenic source emission inventory (Li et al., 2017).

In the BASE scenario, a 1-month-long simulation using anthropogenic emissions and total BVOC emissions in June of 2018 was conducted as the control run for this study. Excluding all the BVOC emissions, the BIO scenario was set to simulate the O_3 and SOA concentration to quantify the impacts of total BVOC emissions compared with the results of BASE scenario (BASE-BIO). Different compound categories have different contributions to the formation of secondary pollutants. Thus, scenarios “ISOP”, “MTP”, “SQT”, and “ISOPRENOID” were designed to simulate the impacts of isoprene, monoterpene, sesquiterpene, and isoprenoid (total of the above three categories) on O_3 and SOA, by excluding their emissions. For these simulations, the meteorology of 2018 was used to drive MEGAN to estimate biogenic emissions in June 2018. The “HISTORY” scenario was an interannual comparison simulation to estimate the impacts of historical BVOC emissions caused by the change of vegetation leaf biomass. It was run using annual emis-

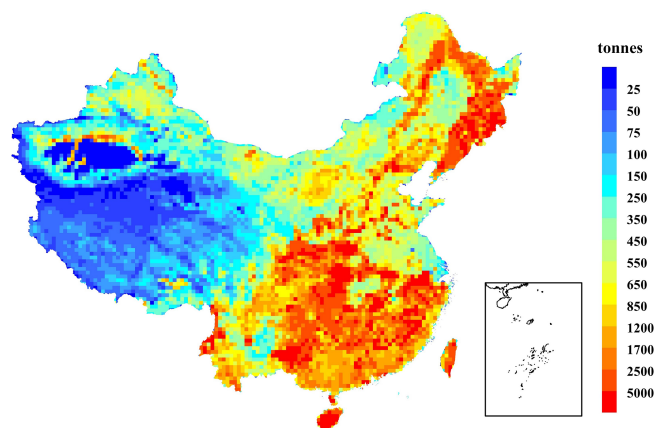


Figure 1. Spatial variations in BVOC emissions in June 2018.

sion factors extrapolated from emission rates and annual leaf biomass during 1981–2018. For the meteorology, the fixing set of 2008, a mid-range year from 1981–2018, was used for all the HISTORY simulations. To explore the impacts of interannual BVOC emission variations caused by vegetation biomass variability, influences of annual meteorology on BVOC emissions and formation of secondary air pollutants were not considered.

3 Results and discussion

3.1 BVOC emission

3.1.1 Spatial distributions of BVOC emissions

The total BVOC emissions in China estimated by MEGAN2.1 are 9.91 Tg in June 2018, of which isoprene, monoterpene, sesquiterpene, and other BVOCs account for 64.21 %, 10.58 %, 2.12 %, and 23.09 %, respectively. As shown in Fig. 1, BVOC emissions show significant spatial variations with the highest emissions in the Changbai Mountains, Greater Khingan Mountains, Qinling Mountains, the southeast and southwest China forest regions, and Hainan and Taiwan provinces and the lowest in the Qinghai–Tibet Plateau and southern Xinjiang province.

The emission simulations were validated using the formaldehyde (HCHO) concentration and measurements of BVOC emission flux. Isoprene is the main compound in BVOC species, accounting for nearly half of total BVOC emissions in China. It undergoes chemical and photochemical reactions in the atmosphere, and the oxidation product is mainly HCHO (Bai and Hao, 2018; Orlando et al., 2000). In forest areas and in summer, biogenic isoprene is the dominant source of HCHO, so satellite HCHO column concentration is widely used to constrain isoprene emissions (Opacka et al., 2021; Palmer et al., 2003; Stavrou et al., 2018; Wang et al., 2021; Zhang et al., 2021). In this study, we used the HCHO vertical column detected by the Ozone Monitoring

Table 1. Description of different model simulations in this study.

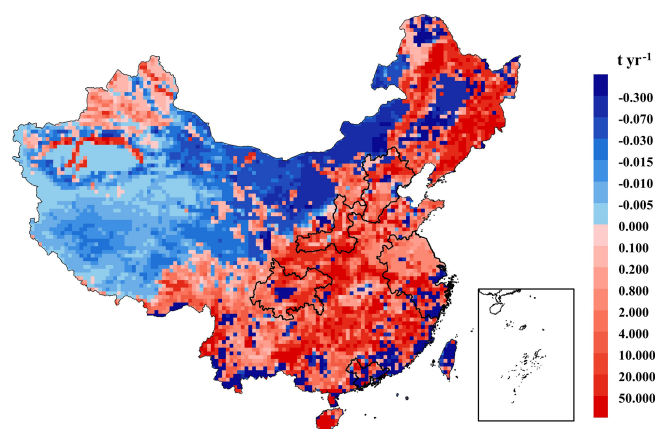
Simulation	Anthropogenic emissions	BVOC emissions	BVOC emission factor	Meteorology
BASE	All	All BVOCs	Year 2018	Year 2018
BIO	All	No BVOCs	–	Year 2018
ISOP	All	No isoprene	Year 2018	Year 2018
MTP	All	No monoterpene	Year 2018	Year 2018
SQT	All	No sesquiterpene	Year 2018	Year 2018
ISOPRENOID	All	No isoprenoid	Year 2018	Year 2018
HISTORY	All	All BVOCs estimated with annual emission factors	Years 1981–2018	Year 2008

Instrument (OMI) to validate the spatial variability of isoprene estimates. The monthly OMI HCHO data from the EU FP7 project QA4ECV product (Quality Assurance for Essential Climate Variables; <http://www.qa4ecv.eu>, last access: 16 February 2022) were used in this study. The result of the statistical analysis with a confidence interval of 99 % indicates that the monthly averaged OMI HCHO vertical column in June 2018 is significantly correlated to the model-estimated isoprene emissions. The flux measurements of BVOCs conducted in China were collected (Bai et al., 2015, 2016, 2017). The details of these measurements are provided in Table S3, including the location of sampling sites and observed BVOC emission fluxes. The gridded BVOC emissions estimated by MEGAN were extracted where the flux measurement sites were located to do the comparison (Fig. S1 in the Supplement). The modeled fluxes of BVOCs in this study capture the spatial variability of observations better, with a correlation coefficient of 0.84. But the estimation is higher than the measurement, with an average mean bias of $1.11 \text{ mg m}^{-2} \text{ h}^{-1}$ in these sampling sites, mainly because of the differences in time between them.

3.1.2 Influence of leaf biomass variability

The leaf biomasses increased from $378.35 \times 10^{12} \text{ g}$ in 1981 to $1107.16 \times 10^{12} \text{ g}$ in 2018 at an average rate of $17.97 \times 10^{12} \text{ g yr}^{-1}$. Among them, the forest and crop leaf biomass increased from 237.10×10^{12} to $518.38 \times 10^{12} \text{ g}$ and from 141.25×10^{12} to $588.79 \times 10^{12} \text{ g}$, respectively, totally increasing by 192.63 %. The spatial distribution of interannual variations in leaf biomass is presented in Fig. S2. The increase of leaf biomass is most significant in the Great Khingan, Changbai Mountains, the North China Plain, and south and southwest China. This is mainly due to the increased stock of broadleaf and coniferous forests as a result of afforestation. The northern Qinghai–Tibet area and northwest China have a relatively high grass cover rate but insignificant increase in leaf biomass of vegetation. This is because the grass biomasses were the same over the historical simulations due to a lack of data.

Due to the increased volume and production of vegetation, the total BVOC emissions increased by 58.66 % at av-

**Figure 2.** Spatial distribution of interannual variations in BVOC emissions caused by leaf biomass changes.

erage rates of 96.64 Gg yr^{-1} , of which isoprene, monoterpene, and sesquiterpene increased by 108.57 %, 38.17 %, and 33.35 % at average rates of 11.10, 0.99, and 0.17 Gg yr^{-1} , respectively. Isoprene emissions increased more rapidly over the past 40 years, which is primarily due to the greater increase in the biomass of broadleaf trees, which have the highest isoprene emission rates. Monoterpene and sesquiterpene increased at a lower rate because the increase of leaf biomass of conifers is relatively small. Figure 2 shows the spatial distribution of interannual variations in BVOC emissions caused by the changing leaf biomass. Since the needle-leaf and broadleaf trees tend to have a higher emission potential than grass or crop (Guenther et al., 2012), their wide distribution and the substantial increase in biomass result in the largest interannual variability of BVOC emissions in the Great Khingan, Changbai Mountains, North China Plain, Central and Southern China, and Hainan province. However, the emission of BVOCs in the northwest and southern coastal areas has decreased.

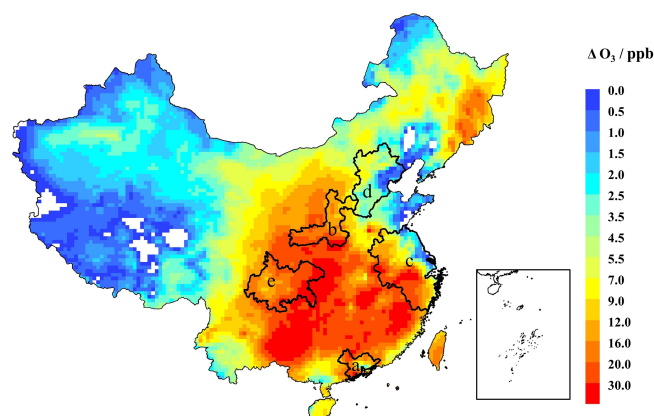


Figure 3. Spatial variations in impact of BVOC emission on MDA8 O_3 concentration. The key regions include the (a) Pearl River Delta (PRD), (b) Fenwei Plain (FWP), (c) Yangtze River Delta (YRD), (d) Beijing–Tianjin–Hebei (BTH), and (e) Chengdu–Chongqing (CC).

3.2 Impacts of BVOC emissions on O_3 and SOA

3.2.1 O_3

BVOC emissions generate additional O_3 and contribute 8.61 ppb MDA8 O_3 over China on average in summer 2018. Their nationally averaged contribution is 16.75 %. There are distinct differences spatially in the impact of BVOC emissions. The spatial distribution of the difference in MDA8 O_3 concentrations between simulations with and without BVOC emissions in summer 2018 is shown in Fig. 3. BVOC emissions lead to the increase of O_3 concentration in most regions, with the highest contribution of > 30 ppb. The highest contribution is concentrated in southern China, which is mainly caused by their higher BVOC emissions due to higher temperature and vegetation biomass (Li et al., 2020). Their impact ranges from 10.50 to 77.17 ppb, contributing 36.20 %–70.83 % to the total O_3 formation, respectively. Because most land cover in western China, including Qinghai–Tibet Plateau and Xinjiang province, is sparse or herbaceous vegetation with low emission potential, O_3 is less affected by BVOC emission. In addition, it is interesting to note that the extremely low contribution in some areas of the Bohai Rim in eastern region is inconsistent with the high BVOC emissions. This is likely because O_3 is more sensitive to anthropogenic emissions.

The spatial pattern of estimated MDA8 O_3 impacted by BVOC emissions differs from the spatial distribution of BVOC emissions, mainly because of the variability of the nonlinear response relationship between O_3 formation and precursors. As important precursors, VOCs and NO_x react in the presence of hydroxyl (OH) and hydroperoxyl (HO_2) radicals to create O_3 . The O_3 formation is expected to be affected by the different levels of O_3 precursors in different land use functional areas. According to different VOC/ NO_x

ratio, O_3 formation regimes can be classified into VOC-limited (VOC-sensitive), transition, and NO_x -limited (NO_x -sensitive) regimes (Lu et al., 2019; Wang et al., 2008). From the spatial distribution of the BVOC effect (Fig. 3), the surface O_3 is sensitive to BVOC emissions in most regions in China, which can further indicate they are usually VOC-limited. It confirms the conclusion made by Lu et al. (2019), Lyu et al. (2016), and Tan et al. (2018) that the VOC-limited regime is dominant in southern China. Compared with the spatial distribution of BVOC emissions (Fig. 1), the areas with high BVOC emissions usually have a higher contribution to O_3 . Because the dense population leads to a large number of NO_x emitted by human activities, NO_x is saturated with the formation of O_3 , which is more sensitive to VOC emissions. Therefore, the higher BVOC emissions usually cause a greater contribution to O_3 in these areas. In the VOC-limited regime, the reduction of VOC emissions reduces the chemical production of organic radicals (RO_2), which in turn leads to decreased cycling with NO_x and consequently lower concentration of O_3 (Jin and Holloway, 2015; Milford et al., 1989). Decreasing BVOC emissions by planting plants with low emission potential may contribute to O_3 pollution control. Notably, both northeastern and southern regions have the highest BVOC emissions, but their contributed O_3 levels differ much, which indicates that O_3 formation in the south is more sensitive to VOCs than in the northeast. Hainan province also has higher BVOC emissions but a relatively low contribution to O_3 .

Five key regions with stronger economy, larger population, and higher O_3 pollution are selected to further investigate the regional variability in the impact of BVOC emissions on O_3 , which are Beijing–Tianjin–Hebei (BTH), Fenwei Plain (FWP), PRD (Pearl River Delta), Yangtze River Delta (YRD), and Chengdu–Chongqing (CC) (Table 2). Their BVOC effects were more significant than the national average, except for the BTH region. These four key areas are populous and economically developed, and power grids are dense. The high concentration of NO_x emissions may lead to saturation of NO_x and excessive consumption of OH, so ozone generation is more sensitive to VOC emissions (Jin and Holloway, 2015). The contribution is the largest in the CC region, with a 23.29 ppb O_3 increment, which accounts for 44.98 % of the total O_3 concentration, although its BVOC emissions are relatively low among the five regions. This is mainly due to its low concentration of other precursors including NO_2 and CO in summer and its low topography that is favorable for the accumulation of O_3 (Cao et al., 2018). The difference in MDA8 O_3 with and without BVOC emissions shows a small gap between FWP and PRD regions, both of which are ~ 19 ppb, but the contribution to total O_3 in PRD (41.83 %) is 1.17 times that in FWP (35.81 %), which could indicate a higher contribution by the anthropogenic VOC emissions in FWP. In addition, FWP has a higher O_3 concentration in summer than PRD (Li et al., 2019a; Wang et al., 2019; Xie et al., 2021). The BVOC emissions in BTH are

the lowest among the five regions, so the impact on MDA8 O_3 is the least significant, which is 4.10 ppb and 15.84 % of the total O_3 . This can be explained by the greater contribution of anthropogenic emissions to surface O_3 caused by manufacturing, electricity production, and transportation.

3.2.2 SOA

BVOCs undergo a series of atmospheric degradation processes to produce oxidation products, which may contribute to the formation and growth of SOA. This process is affected by the concentration of precursors and meteorological conditions. In summer, high biogenic emissions combined with enhanced photochemical levels, caused by high temperature and strong solar radiation, lead to the peak of SOA production in this season (Kelly et al., 2018). From our simulations, SOA produced by BVOCs (BSOA) plays an important role in SOA production, accounting for 73.15 % of the total concentration, which is in good agreement with previous studies that considered BSOA a major contributor (Hu et al., 2017; Wu et al., 2020). The BVOC emissions generate an additional $\sim 0.8 \mu\text{g m}^{-3}$ SOA on average, with a gridded maximum of $4.12 \mu\text{g m}^{-3}$. Figure 4 shows the spatial distribution of BSOA contribution in June 2018. The whole country has increases in SOA concentration after considering BVOC emissions compared with simulations without them. Generally, spatial variability in BVOC contribution to SOA corresponds with that in BVOC emissions. Southern China has a higher contribution than northeastern areas, although they both have the highest BVOC emissions in summer. This is mainly because the southern area has higher emissions of monoterpenes, which contribute most to SOA generation. Notably, the hotspots of BVOC contribution to SOA with $> 2 \mu\text{g m}^{-3}$ (accounting for $> 70\%$ of the total SOA) are mainly distributed in the Sichuan Basin, northeast Yunnan, and northwest Guizhou, which may be because the high level of BVOCs in southern China can be transported to these regions due to the influence of the southeast monsoon in the summertime and accumulate under the barrier of the western plateau (Wang et al., 2018). The sensitivity of SOA to BVOC emission is the lowest in the western Qinghai–Tibet Plateau and southwestern Xinjiang, where the contribution of BVOCs is only about $0.15 \mu\text{g m}^{-3}$, lower than 60 % of the total produced SOA. This is mainly because of their lower biogenic emissions contribution.

The impacts of BVOC emissions on SOA differ much among the five key regions (Table 2). The results indicate that SOA production is most sensitive to BVOC emissions in CC, followed by PRD, FWP, and YRD, and the least in BTH. Among the five regions, CC has the lowest BVOC emissions but the strongest impacts on SOA, with an additional $2.51 \mu\text{g m}^{-3}$ SOA and contributing 84.27 % to the total, which is a characteristic that coincides with impacts on O_3 . This can be explained by the widely distributed forest in the southwest forest region and the topography that is bene-

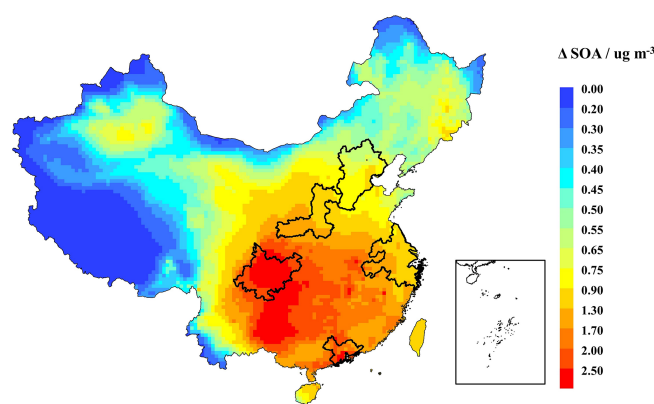


Figure 4. Spatial variations in impact of BVOC emission on SOA concentration.

ficial to the accumulation of BVOCs and SOA. The BVOC emissions of 462.92 Gg in PRD generate $1.96 \mu\text{g m}^{-3}$ SOA, accounting for 74.63 % of the total SOA. YRD and FWP regions have similar emission budgets and additional SOA generation by BVOCs but different proportions of total BSOA. The contribution of BSOA in FWP (77.75 %) is greater than that in YRD (69.12 %), which indicates a higher contribution of anthropogenic VOCs in YRD. BTH has the lowest production of BSOA with $0.74 \mu\text{g m}^{-3}$, which is lower than the national average, mainly due to its lowest biogenic emissions among the five regions.

3.2.3 BVOC component contribution

Table 2 shows the contributions of different BVOC categories to MDA8 O_3 and SOA. Isoprene, monoterpene, and sesquiterpene contribute 7.01, 1.17, and $0.16 \text{ ppb } O_3$ and 0.25, 0.52, and $0.22 \mu\text{g m}^{-3}$ SOA over China, respectively. Among the emission-dominated categories, including isoprene, monoterpene, and sesquiterpene, isoprene has the largest contribution to O_3 production. Owing to its highest emissions and reactivity with various oxidants, isoprene plays an important role in the formation of O_3 (Wennberg et al., 2018). Sesquiterpene emission is the lowest, resulting in the smallest contribution to O_3 . As key precursors of SOA formation, monoterpene emissions are only one-sixth of isoprene, but their contribution to SOA is the highest, about 2 times the contribution by isoprene. The reaction of monoterpenes with atmospheric oxidants such as O_3 , OH, and NO_3 is considered to be an important reason for the resultant SOA formation (Hoffmann et al., 1997; Mutzel et al., 2016; Watne et al., 2017). The contribution of isoprene to SOA is relatively low because isoprene mainly reacts with OH radicals to form SOA, while the reactions with other oxidants in the atmosphere (including O_3 and NO_3) contribute little to SOA production (Claeys et al., 2004; Henze and Seinfeld, 2006; Ruppert and Becker, 2000). Sesquiterpene has much lower emissions than isoprene, but their generated SOAs are

equivalent, indicating the higher SOA yield of sesquiterpene (Hoffmann et al., 1997). From the simulation of the scenario “ISOPRENOID”, which excludes the emissions of isoprene, monoterpene, and sesquiterpene together, the additional generation of O_3 and SOA by isoprenoid is not a linear combination of those by each category. The contributions by isoprenoid are usually larger than the sum of those by each category. It can be concluded that there is a complex nonlinear relationship between BVOCs and the formation of O_3 and SOA.

For all the five regions, isoprene has both the largest emissions and the highest contribution of O_3 generation, followed by monoterpene and sesquiterpene. O_3 in CC, FWP, and PRD is greatly impacted by isoprene, where the impacts are on average 20.49, 18.01, and 16.81 ppb, respectively. Among them, CC has the lowest isoprene emissions, but the concentration of O_3 generated by isoprene is the highest, which is mainly because of the low concentration of other precursors. Isoprene emissions are the greatest in PRD. For the impact of monoterpene on O_3 , CC contributes the most because it has higher monoterpene emissions. Monoterpenes are the largest contributor to SOA production among the key regions except YRD. In YRD, the generated SOAs by isoprene and monoterpene are equivalent, although their emissions can be distinguished. These also indicate the different SOA yields of isoprene and monoterpene. SOA contributed by isoprene is also higher in CC and PRD regions due to their absolute advantage in topography and emission, respectively. BTH and FWP have similar monoterpene emissions, but their generated SOAs differ much due to the different ratios of the different BVOC component emissions. Compared with isoprene and monoterpenes, the contribution of sesquiterpene to SOA is small.

3.3 Interannual variability

3.3.1 O_3

The concentration of O_3 enhances from 48.82 to 52.43 ppb at an average growth rate of 0.11 ppb yr^{-1} during 1981–2018. At the 95 % confidence level, the increasing trend for O_3 is significant. But the interannual variability in O_3 can be distinguished regionally. Figure 5 shows the spatial distribution for changes of MDA8 O_3 concentration due to the changed BVOC emissions caused by vegetation leaf biomass variability in the past 40 years. The strongest enhancement of O_3 from interannual variation of BVOC emissions is mainly distributed in most of the southern areas, which have an average annual rate of $> 0.2 \text{ ppb}$ and some $> 0.4 \text{ ppb}$. In these areas, the increase in biomass of broadleaf trees and needle-leaf trees with high emission potential has contributed to the increased BVOC emissions and O_3 concentration as a result. The enhancement of O_3 in the area of the Changbai Mountains is also large mainly due to the increased broadleaf tree biomass. The weakest increments are found

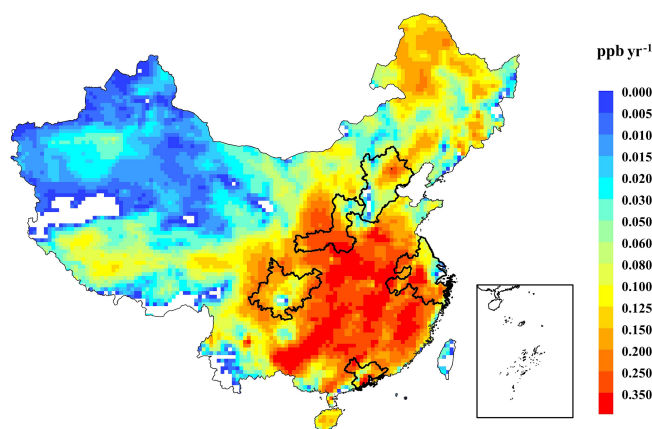


Figure 5. Spatial distribution of interannual variations in O_3 simulated using annual BVOC emission factors.

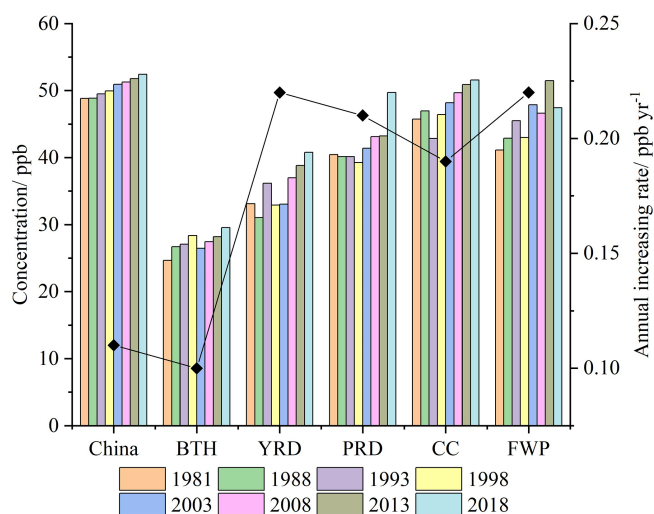
mainly in Xinjiang, southwestern Yunnan province, and Taiwan. In Xinjiang province, there is a greater distribution of vegetation with lower emission potentials, including crops and shrubs, which has a small change in leaf biomass. In the past 4 decades, the leaf biomass in Xinjiang increased by $17.55 \times 10^{12} \text{ g}$, accounting for only 2.41 % of the total growth in China. The tropical forest in the southwest of Yunnan province has declined in volume because of commercial deforestation and expanded plantations of economic trees.

Although there is evidence of the overall positive influence of historical BVOC emissions on MDA8 O_3 , different trends are shown during different periods within 1981–2018. In the 1980s, due to the demand for social development, urban land continued to expand, resulting in massive deforestation. The decrease of forest biomass led to a decrease in BVOC emissions, resulting in little change in O_3 concentration (only 0.03 ppb) from 1981 to 1988. In order to offset the deterioration of the ecological environment caused by a large amount of deforestation, large-scale afforestation activities have been carried out all over the country, and forest biomass has been increasing. Most of the planted species selected for this ecological project were broadleaf trees with high emission potential (Klinger et al., 2002). During 1988 and 2018, the forest volume and crop production increased by $6.60 \times 10^9 \text{ m}^2$ and $1487.25 \times 10^6 \text{ t}$, respectively. As a result, the vegetation leaf biomass continued to grow by 95.89 %, which led to 3.58 ppb O_3 enhancement by 6.88 %. Among the annual growth rates, the growth was the biggest in 2003, with an increase of 1.99 % compared with 1998.

The interannual variations in O_3 have significant differences among the five key regions (Fig. 6). But the overall growth trend can be found. From 1981 to 2018, MDA8 O_3 concentration in BTH, YRD, PRD, CC, and FWP increased by 19.87 %, 23.24 %, 22.91 %, 12.71 %, and 15.44 % at average rates of 0.10, 0.22, 0.21, 0.19, and 0.22 ppb yr^{-1} , respectively. This reflects O_3 increments of 4.90, 7.70, 9.26, 5.81, and 6.35 ppb, respectively. As the surface O_3 in urban areas

Table 2. Emissions of each BVOC category and their corresponding contribution to MDA8 O₃ and SOA concentration in the five key regions of China in June 2018.

	BVOC category	China	BTH	FWP	PRD	YRD	CC
Emission (10 ⁴ t)	Isoprene	636.28	15.66	30.29	34.74	53.43	24.50
	Monoterpene	104.81	2.77	2.16	5.09	4.38	11.59
	Sesquiterpene	20.98	0.64	0.32	1.27	1.42	1.65
	Total BVOCs	990.91	29.23	40.67	46.29	74.18	51.32
Contribution to MDA8 O ₃ (ppb)	Isoprene	7.01	3.42	18.01	16.81	12.55	20.49
	Monoterpene	1.17	1.74	0.32	4.07	2.92	6.89
	Sesquiterpene	0.16	0.93	−1.36	1.66	1.06	2.88
	Isoprenoid	7.77	2.94	17.43	16.34	14.23	24.55
	Total BVOCs	8.61	4.10	18.94	18.74	13.40	23.29
Contribution to SOA (μg m ^{−3})	Isoprene	0.25	0.20	0.53	0.95	0.63	0.91
	Monoterpene	0.52	0.45	0.72	1.21	0.62	1.59
	Sesquiterpene	0.22	0.21	0.26	0.49	0.31	0.75
	Isoprenoid	0.84	0.78	1.30	1.96	1.29	2.52
	Total BVOCs	0.84	0.74	1.29	1.96	1.27	2.51

**Figure 6.** The interannual changes of O₃ in China and the key regions.

is more sensitive to VOCs, the increment of O₃ concentration caused by BVOC emissions in all the regions is greater than the national average. FWP, YRD, and PRD regions have the highest increasing rates annually, which may be because of the rapid increase in leaf biomass of broadleaf trees with high isoprene emission potential. At the end of the last century, the implementation of the reform and opening-up policy led to the continuous acceleration of the level of urbanization and the reduction of forest and arable land biomass, eventually resulting in the fluctuating decline of MDA8 O₃ in YRD and PRD during 1981–2003. The rapid growth of MDA8 O₃ in PRD led to a greater average annual growth rate of MDA8

O₃ in PRD than that in YRD from 1981 to 2018. The lowest growth rate occurs in BTH. Due to the Three-North Shelterbelt System Project, the area coverage of natural forest in the north of BTH has more than tripled compared to that in 2003, resulting in an increase in leaf biomass and BVOC emissions (Ma et al., 2019). However, because of the rapid urbanization in Beijing, a large amount of forest has been converted to urban construction land, and the reduction of BVOC emission related to losses of trees may offset part of their increase associated with rising coverage and volume in the surrounding areas.

3.3.2 SOA

The historically varied BVOC emissions caused by changes in leaf biomass had a significant influence on SOA formation in China from 1981–2018 according to our simulation. The national SOA was enhanced at an annual rate of 0.01 μg m^{−3}, showing a significantly increasing trend ($p < 0.05$). As shown in Fig. 7, which depicts the spatial distributions of interannual changes in SOA, most of southern China shows a significantly increasing trend, with an average growth rate higher than 0.02 μg m^{−3}, even $> 0.03 \mu\text{g m}^{-3}$ in some areas. It is mainly because of the continuous expansion of vegetation coverage and increase of biomass, which results in enhanced BVOC emissions. Monoterpene is the biggest contributor to BSOA, as described in Sect. 3.2.2. The spatial distribution of monoterpene emissions is consistent with that of needleleaf trees. The leaf biomass of needleleaf trees increased from 118.68×10^{12} to 212.04×10^{12} g during 1981–2018. Needleleaf trees are densely distributed in southern China, including Masson pine, spruce, and hemlock, with high monoterpene emission rates (Li et al., 2020).

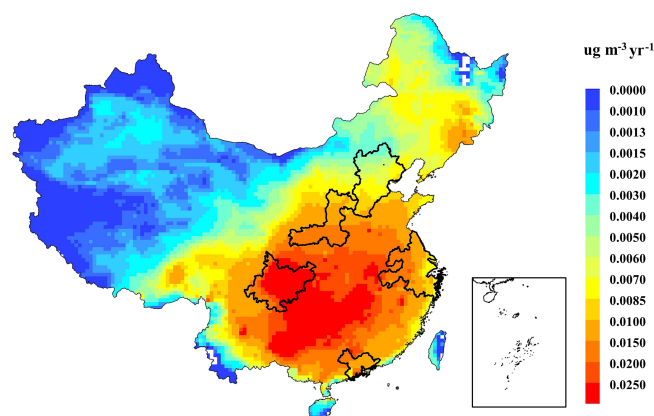


Figure 7. Spatial distribution of interannual variations in SOA simulated using annual BVOC emission factors.

The highest average annual growth rates of SOA occur in the intersecting area of southwest and southeast China, and then the growth rate gradually declines to the surrounding areas. This can be partly attributed to the spatial distribution and increasing volume of needleleaf trees in the southwest forest area. The lowest growth is found in the northwestern areas, Hainan, and southwest Yunnan. Some of these areas show negative growth. In northwestern areas, BVOCs, especially monoterpene emissions, experienced the weakest growth, and some even decrease. Although the ecological shelterbelt projects were conducted in northwest China, broadleaf tree species were mostly planted with lower monoterpene emissions. Because of the tropical climate in southwestern Yunnan and Hainan, needleleaf trees with greater monoterpene emission potential are less distributed, and the change of leaf biomass is small, which led to small or even negative changes in the growth of SOA.

The national average SOA in 2018 is $1.06 \mu\text{g m}^{-3}$, 39.30 % higher than that in 1981. In the past 40 years, China has had an overall growth for the SOA from BVOCs and a more rapid growth in 2003 and 2018. The historical SOA concentrations and their changes in BTH, YRD, PRD, CC, and FWP regions are presented in Fig. 8. They underwent SOA increments of 31.08 %, 53.21 %, 37.97 %, 54.61 %, and 54.57 % during 1981–2018, at average rates of 0.007, 0.014, 0.013, 0.027, and $0.014 \mu\text{g m}^{-3} \text{ yr}^{-1}$, respectively. CC is the region with the largest annual increasing rate since it has the more significantly enhanced leaf biomass of needleleaf trees and crops, which contribute much to monoterpene emissions. In BTH, SOA annual growth rate is the lowest and lower than the national average rate. It can be attributed to the obvious decrease of leaf biomass, with $17.57 \times 10^{12} \text{ g}$ from 1998 to 2008. With a similar overall growth rate, however, YRD and PRD have different interannual variability. YRD experienced two stages of increasing before and after 2003. In PRD, SOA showed a striking growth during the last 5 years owing to the increase of leaf biomass by up to $4.21 \times 10^{12} \text{ g}$. To sum up,

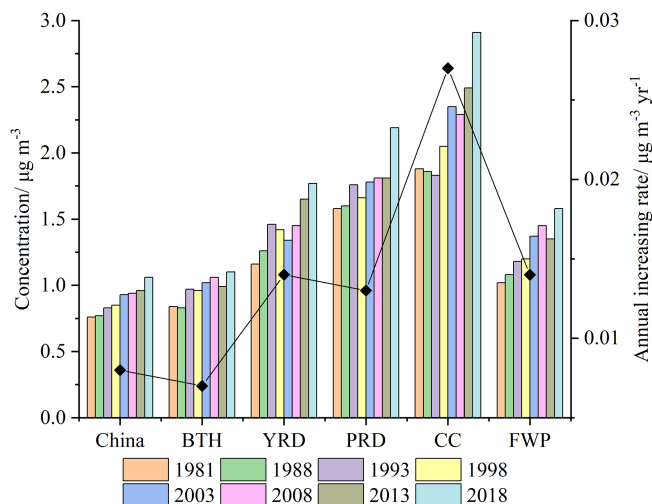


Figure 8. The interannual changes of SOA in China and the key regions.

there are significant increases in SOA in the five key regions, but they have marked differences in the regularity of growth due to the different changes of leaf biomass over time.

4 Conclusions

BVOC emissions play a key role in the formation of O_3 and SOA. In the summer of 2018, China's BVOC emissions of 9.91 Tg cause ambient O_3 and SOA concentrations to increase by 8.6 ppb (16.75 %) and $0.84 \mu\text{g m}^{-3}$ (73.15 %) on average, respectively. The impacts of BVOCs have an obvious spatial difference: O_3 and SOA in most southern regions show high sensitivities to BVOCs. Due to the different emissions and level of biogenic and anthropogenic precursors, BVOC emissions have distinguished impacts in different regions. The CC region has the highest O_3 and SOA generated by BVOCs. Some areas with equivalently high BVOC emissions, however, have different contributions to O_3 , such as northeastern and southern China. The impacts of BVOCs are affected not only by the relative abundance of biogenic and anthropogenic VOCs, but also by the VOC/ NO_x ratio. The sensitivity to BVOCs differs over regions. Decreasing BVOC emissions by planting plants with low emission potential may contribute to O_3 pollution control in most regions of southern, central, and northeastern China. For the abatement of BSOA in summer, the decreased plantation of needleleaf trees or the replacement by trees with low monoterpene emission potential is expected to be helpful.

Considering the increasing vegetation coverage and greening trend in China in recent decades (Chen et al., 2019; Piao et al., 2015), the resultant changes in leaf biomass will influence BVOC emissions, which then affect the formation of O_3 and SOA. The interannual variation of BVOC emissions caused by increasing leaf biomass results in significant

increases ($p < 0.05$) of O_3 and SOA concentrations at average rates of 0.11 ppb yr^{-1} and $0.008 \mu\text{g m}^{-3} \text{ yr}^{-1}$, respectively. It shows different interannual variations, which can be attributed to the differences in changing trends of leaf biomass. The southern region with an obvious increase of leaf biomass showed large enhancement of O_3 and SOA. In the future, in order to achieve the goal of carbon neutrality, China will not only reduce carbon emissions through energy conservation and emission reduction, but also increase carbon sinks through the development of carbon sequestration technologies and biological carbon sinks (Wang and Zhang, 2020). Increasing forest carbon sinks will inevitably lead to an increase in vegetation coverage, so that BVOC emissions will continue to increase, leading to higher contributions to O_3 and SOA production in the future. Therefore, studying the influence of land cover changes on BVOC emissions and their impact on the generation of O_3 and SOA is of great significance for future research studies on the precise prevention and control of air pollution in China in the context of fighting climate change.

In this study, we aimed to explore the impact of interannual BVOC emission variations on O_3 and SOA caused by vegetation biomass variability. The historical O_3 and SOA concentrations were simulated by fixing the anthropogenic emissions and meteorological data in 2008. The influences of annual meteorology on BVOC emissions and formation of secondary air pollutants were not considered. Although vegetation change is the main driver of interannual variations of BVOC emissions, it still brings uncertainty to the simulation in this study. Future work can update the anthropogenic emission inventory and use dynamic meteorological data to explore multivariate effects and provide more accurate data for evaluating the roles of biogenic emissions in air quality. At the same time, the impact of meteorological changes on the long-term changes of BVOC emissions and formation of secondary air pollutants should also be considered to provide a scientific basis for the precise prevention and control of air pollution in response to climate change.

Code and data availability. The code and data used in this study are available upon request from the corresponding author, Lingyu Li (lilingyu@qdu.edu.cn).

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Author contributions. LL planned and organized the project. SS provided the code, JC performed the simulation, analysis, and writing, and YH made adjustments to the paper's visualization. SX, YH, and LL revised the manuscript.

Competing interests. The contact author has declared that neither they nor their co-authors have any competing interests.

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