



Significant increase of summertime ozone at Mount Tai in Central Eastern China

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Abstract. Tropospheric ozone (O_3) is a trace gas playing important roles in atmospheric chemistry, air quality and climate change. In contrast to North America and Europe, long-term measurements of surface O_3 are very limited in China. We compile available O_3 observations at Mt. Tai – the highest mountain over the North China Plain – during 2003–2015 and analyze the decadal change of O_3 and its sources. A linear regression analysis shows that summertime O_3 measured at Mt. Tai has increased significantly by 1.7 ppbv yr^{-1} for June and 2.1 ppbv yr^{-1} for the July–August average. The observed increase is supported by a global chemistry-climate model hindcast (GFDL-AM3) with O_3 precursor emissions varying from year to year over 1980–2014. Analysis of satellite data indicates that the O_3 increase was mainly due to the increased emissions of O_3 precursors, in particular volatile organic compounds (VOCs). An important finding is that the emissions of nitrogen oxides (NO_x) have diminished since 2011, but the increase of VOCs appears to have enhanced the ozone production efficiency and contributed to the observed O_3 increase in central eastern China. We present evidence that controlling NO_x alone, in the absence of VOC controls,

is not sufficient to reduce regional O_3 levels in North China in a short period.

1 Introduction

Ozone (O_3) in the troposphere is a trace gas of great importance for climate and air quality. It is the principal precursor of the hydroxyl radical (OH) which plays a central role in atmospheric chemistry (Seinfeld and Pandis, 2006), and the third most important greenhouse gas contributing to the warming of the Earth (IPCC, 2013). At ground level, high levels of O_3 have adverse effects on human health and ecosystem productivity (National Research Council, 1991; Monks et al., 2015). In the troposphere, the ambient O_3 burden is the product of the flux from the stratosphere (e.g., Stohl et al., 2003; Lin et al., 2015a), dry deposition, and net photochemical production involving the reactions of nitrogen oxides (NO_x) with carbon monoxide (CO) and volatile organic compounds (VOCs) in the presence of sunlight (Crutzen, 1973; Ma et al., 2002). Increases in anthro-

pogenic emissions of ozone precursors have contributed to changes in the tropospheric O₃ abundances, both globally and regionally (The Royal Society, 2008; Cooper et al., 2014; Monks et al., 2015; Lin et al., 2015b). Decadal shifts in climate and circulation regimes can also contribute to changes in tropospheric O₃, as found in the 40-year record at Mauna Loa Observatory in Hawaii (Lin et al., 2014). Conversely, the changes in tropospheric O₃ may also pose significant feedbacks to the environment and climate (e.g., Shindell et al., 2012; Stevenson et al., 2013). Therefore, the long-term changes (or trends) of tropospheric O₃ has long been a topic of great interest in the atmospheric sciences.

Since the 1970s, long-term measurements of surface O₃ (and O₃ precursors) have been increasingly carried out worldwide, mostly in North America and Europe (e.g., Logan et al., 2012; Oltmans et al., 2013; Cooper et al., 2014). The existing knowledge of tropospheric O₃ trends has been recently reviewed (UNEP and WMO, 2011; Cooper et al., 2014; Monks et al., 2015). Overall, upward trends have been recorded around the world since the 1970s, but trends over the past two decades have varied regionally. In Europe, the surface O₃ in rural or remote areas, usually regarded as the regional baseline O₃, rose until the year 2000 but has since leveled off or decreased (Logan et al., 2012; Oltmans et al., 2013; Parrish et al., 2012). In the eastern US, summertime O₃ at most rural and urban stations has decreased over 1990–2010 (Lefohn et al., 2010; Cooper et al., 2012). In the western US, extreme ozone events have decreased in urban areas, particularly in southern California (Warneke et al., 2012), but springtime O₃ at remote mountain sites has shown large interannual variability due to stratospheric influence (Lin et al., 2015a), with little overall trends or small increases (Cooper et al., 2012; Fine et al., 2015). Analysis of available observations and chemistry-climate model hindcast indicates that springtime O₃ in the free troposphere over western North America has increased significantly by 0.4 ppbv yr⁻¹ over 1995–2014 (Lin et al., 2015b).

In comparison with North America and Europe, investigations of long-term O₃ trends are scarce in China, where rapid urbanization and industrialization has occurred over the past 3 decades. Significant increasing trends of surface O₃ have been derived from a handful of long-term monitoring stations over China, including Beijing (Tang et al., 2009; Zhang et al., 2014), Hong Kong (Xue et al., 2014), Taiwan (Lin et al., 2010) and Mount Waligian (Xu et al., 2016). Based on the MOZAIC commercial aircraft measurements, Ding et al. (2008) derived an O₃ increase of ~2% per year from 1995 to 2005 for the lower troposphere above Beijing by analyzing the difference between observations from 1995 to 2000 and 2000 to 2005. Wang et al. (2009) reported the first long-term continuous observations of Chinese surface O₃ at a regional background site in southern China (Hok Tsui), and indicated an average increase of 0.58 ppbv yr⁻¹ during 1994–2007. Xu et al. (2016) recently reported another continuous O₃ record (1994–2013) at Mt. Waliguan, a global at-

mospheric watch station in western China, and found significant positive trends with 0.15–0.27 ppbv yr⁻¹ for daytime O₃ and 0.13–0.29 ppbv yr⁻¹ for nighttime O₃. Despite the valuable information obtained from the abovementioned efforts, additional studies are required to improve our understanding of tropospheric O₃ trends across the rapidly developing China. In particular, long-term observations covering more than 10 years remain very limited over the highly polluted regions of central eastern China.

In recent years, China has implemented a series of stringent air quality control measures. Following the successful reductions in sulfur dioxide (SO₂) emissions since 2006 (Lu et al., 2010), China has recently launched a national programme to reduce NO_x emissions during its “Twelfth Five-Year Plan” (2011–2015) (China State Council, 2011). However, to our knowledge, fewer controls have been placed on VOC emissions in China. Rather, anthropogenic VOC emissions have continued to increase (Bo et al., 2008; Wang et al., 2014). Therefore, it is of great interest and critical importance to evaluate the effect of the current control policy (i.e., controlling NO_x with little action for VOCs) on regional O₃ and other secondary air pollution problems in China.

Mt. Tai (36.25° N, 117.10° E; 1534 m altitude) is the highest mountain in the center of the North China Plain (NCP; see Fig. 1) – a fast developing region facing severe air pollution. During daytime, the summit is well within the planetary boundary layer (PBL), and the site is therefore regionally representative of the region (Kanaya et al., 2013). Since 2003, several field measurement campaigns have been conducted at this site, with surface O₃ of major interest. In this paper, we analyze all available O₃ and its precursor observations at Mt. Tai to understand the summertime O₃ characteristics, including trends. To place the short observational record into a long-term context, we analyze multi-decadal hindcast simulations (1980–2014) conducted with the GFDL-AM3 chemistry-climate model (Lin et al., 2014, 2015a, b). In the following sections, we first present the seasonal and diurnal ozone variations followed by the climatological air mass transport pattern in summer. We then derive the O₃ trend (or systematic change) during 2003–2015 using linear regression. We finally elucidate the key factors affecting the O₃ trends by examining satellite and in situ observed trace gas data. Our analysis demonstrates a significant increase of summertime surface O₃ at this important regional site in northern China, and indicates the urgent need of VOC control in China to reduce regional O₃ pollution.

2 Observational data set

The observations analyzed in the present study are summarized in Table 1. It comprises several sets of field observations from different periods. The earliest O₃ measurements at Mt. Tai were made from July to November 2003 (Gao et al., 2005), and the longest observations lasted for 3 years

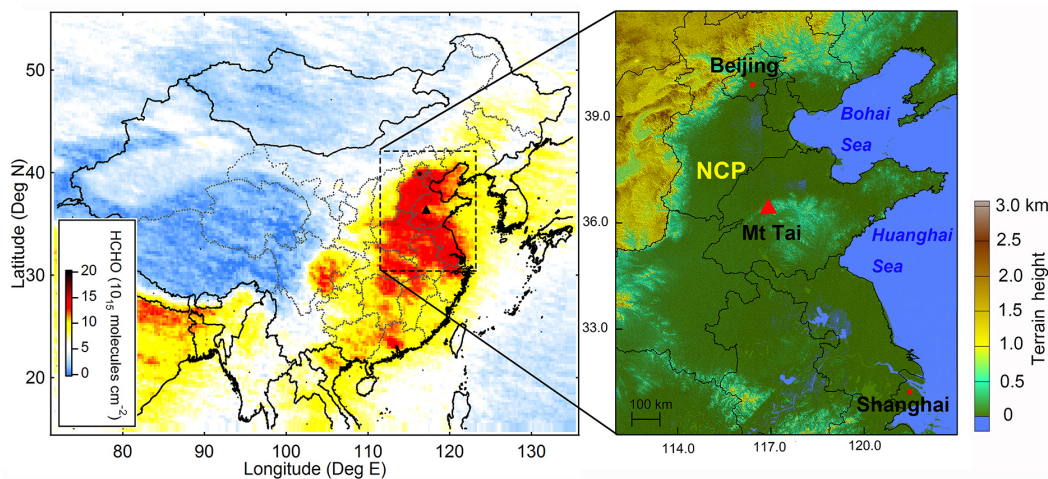


Figure 1. Geographical map showing the North China Plain and the location of Mt. Tai. The left map is color-coded by the HCHO column density in the summer period (JJA; 2003–2014) retrieved from SCIAMACHY and GOME-2(B).

Table 1. The observations at Mt. Tai analyzed in the present study.

Year	Month	Observed species	Data source
2003	July–November	O ₃ , CO	Gao et al. (2005) and our study
2004	June–August	O ₃	Kanaya et al. (2013) ^a
2005	June–August	O ₃	Kanaya et al. (2013) ^a
2006	June–December	O ₃ , NO ₂ [*] , NO	Our study
2007	March–December	O ₃ , NO, NO ₂ [*] , CO	Our study ^b
2008	January–December	O ₃ , NO, NO ₂ [*] , CO	Our study
2009	January–June	O ₃ , NO, NO ₂ [*] , CO	Our study
2014	June–August	O ₃ , NO _x [*] , CO	Our study ^c
2015	June–August	O ₃ , NO, NO ₂ [*] , CO	Our study ^c

^a Monthly average data were taken from Kanaya et al. (2013). ^b Valid NO and NO₂^{*} data were only available in March–April and July–December, and CO was available in March–April in 2007. ^c Note that the intensive measurement periods were 6 June–3 July and 24 July–26 August in 2014 and 14 June–8 August in 2015.

from June 2006 to June 2009 (despite a data gap in January–February 2007 owing to instrument maintenance). In recent years, we conducted intensive measurements at Mt. Tai during June–August of 2014 and 2015. In addition, another set of 3-year measurements was carried out from March 2004 to May 2007, as an international joint effort between Japanese and Chinese scientists (Kanaya et al., 2013); monthly average O₃ data are taken from this work. As most of the measurements are available for the period of June–August each year, we focus on the summertime O₃ in this study.

Two study sites have been used for field observations at Mt. Tai. One was the Mt. Tai Meteorological Observatory (Site 1) at the summit with an altitude of 1534 m a.s.l., and the other was in a hotel (Site 2) that is ~ 1 km to the northwest of Site 1 and slightly lower (1465 m a.s.l.; see Fig. S1 in the Supplement for the site locations). Such elevations position these sites either within the PBL in the afternoon in summer, or in the free troposphere during the night. Although Mt. Tai is a famous tourism spot, both sites are located in

the less frequently visited zones. Hence the impact of local anthropogenic emission should be small, and the data collected are believed to be regionally representative. Details of these sites have been described elsewhere (Site 1: Gao et al., 2005; Kanaya et al., 2013; Site 2: Guo et al., 2012; Shen et al., 2012). For our data set, most of the measurements were taken at Site 1, and only the intensive campaign in 2014 took place at Site 2.

All measurements were implemented using standard techniques, which were detailed in the previous publications (e.g., Gao et al., 2005; Xue et al., 2011). Briefly, O₃ was measured using a commercial ultraviolet photometric instrument (*Thermo Environment Instruments (TEI), Model 49C*) with a detection limit of 2 ppbv and a precision of 2 ppbv. CO was monitored with a gas filter correlation, non-dispersive infrared analyzer (*Teledyne Advanced Pollution Instrumentation, Model 300E*), with automatic zeroing every 2 hours. This technique has a detection limit of 30 ppbv and a precision of 1% for a level of 500 ppbv. NO and NO₂^{*} were

measured by a chemiluminescence analyzer equipped with an internal MoO catalytic converter (*TEI, Model 42C*), with a detection limit of 0.4 ppbv and precision of 0.4 ppbv. Inter-comparison with a highly selective photolytic NO₂ detection approach indicated that the NO₂* measured with MoO conversion significantly overestimated NO₂ (i.e., up to 130 % in afternoon hours), and NO₂* actually represented a major fraction (60–80 %) of NO_y at Mt. Tai (Xu et al., 2013). During the measurements, the O₃ analyzer was calibrated routinely (i.e., quarterly for the 3-year observations in 2006–2009, and before and after the other campaigns) by an ozone primary standard (*TEI, Model 49PS*). For CO and NO_x*, zero and span calibrations were performed weekly during 2006–2009 and every 3 days during the intensive campaigns. Meteorological data including temperature, relative humidity (RH), and wind vectors were obtained from the Mt. Tai Meteorological Observatory, where Site 1 is located.

It is noteworthy that some portion of this long-term data set has been reported previously. Gao et al. (2005) analyzed the measurements of O₃ and CO from July to November 2003 and examined their diurnal variations and relations to backward trajectories. Kanaya et al. (2013) reported the observations in 2004–2007 and examined the processes influencing the seasonal variations and regional pollution episode. The major objective of the present study is to compile all of the available O₃-related observations at Mt. Tai and to establish the trend (or systematic change), if any, of ambient O₃ levels in the past decade at this unique site, regionally representative of the North China Plain.

3 Results and discussion

3.1 Seasonal and diurnal variations from more recent data

Figure 2 depicts the seasonal variation of surface O₃ at Mt. Tai derived from the more recent year-round observations from 2006 to 2009. Overall, O₃ shows higher levels in the warm season, i.e. April–October, compared to the cold season, i.e. November–March, with two peaks in June and October. The elevated O₃ levels in April and May should be affected by the stratosphere-troposphere exchange process which usually occurs at its maximum in the spring season (Yamaji et al., 2006). In addition to the high temperatures and intense solar radiation (especially in June), biomass burning is believed to be another factor shaping the O₃ maximums in June and October, both of which are major harvest seasons of wheat and corn in northern China. The significant impacts of biomass burning on air quality over the North China Plain during June have been evaluated by a number of studies (Lin et al., 2009; Yamaji et al., 2010; Suthawaree et al., 2010). It is also noticeable that the O₃ concentrations in July and August at Mt. Tai are substantially lower than those in June. This is attributed in part to the more humid weather and greater

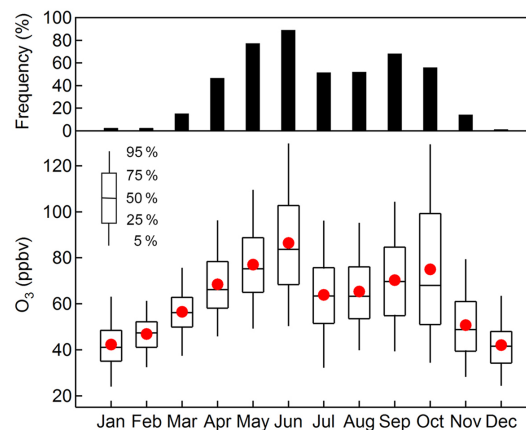


Figure 2. Seasonal variation of surface O₃ mixing ratios at Mt. Tai derived from the continuous observations from 2006 to 2009. Red dots indicate the monthly average O₃ concentrations. Shown in the upper panel is the frequency of the MDA8 O₃ exceeding the Chinese national ambient air quality standard, i.e. 75 ppbv (Class II).

precipitation in July and August in this region (see Table 2 for the RH condition). Inspection of meteorological conditions day by day also indicates a frequency of cloudy days (i.e., with RH \geq 95 %) of \sim 25 % in June and of \sim 51 % during July–August. In the following analyses, therefore, we assess the ozone characteristics separately for June and July–August.

Shown in the upper panel of Fig. 2 is the frequency of the maximum daily 8-hour average O₃ mixing ratios (MDA8 O₃) exceeding the 75-ppbv National Ambient Air Quality Standard (Class II). Although located in a relatively remote mountain-top area, the observed O₃ pollution at Mt. Tai was rather serious in the warm season, with frequencies of the O₃-exceedence days of over 45 % throughout April–October. In particular, the occurrence of O₃-exceedence days was as high as 89 % in June. These results demonstrate the severe O₃ pollution situation across the North China Plain.

Figure 3 illustrates the well-defined diurnal variations of surface O₃ with a trough in the early morning and a broad peak lasting from afternoon to early evening, which is commonly observed at polluted rural sites. As the summit of Mt. Tai is well above the PBL at nighttime, the O₃ concentrations during the latter part of the night (e.g., 02:00–05:00 LT) are usually considered to reflect the regional baseline O₃ (defined hereafter as regional O₃ without impact of local photochemical formation). Comparing the diurnal profiles in June and July–August clearly reveals the significantly higher regional baseline O₃ in June (with a mean difference of \sim 17 ppbv). On the other hand, the daytime O₃ build-up, defined as the increase in O₃ concentrations from the early-morning minimum to the late-afternoon maximum, may reflect the potential of regional O₃ formation. For the Mt. Tai case, the average daytime O₃ build-up was 22 ppbv in June and 15 ppbv in July–August, indicating the stronger photo-

Table 2. Summary of meteorological conditions recorded at Mt. Tai in June and July–August over 2003–2015*.

Year	June			July–August		
	Temperature (°C)	RH (%)	Prevailing WD	Temperature (°C)	RH (%)	Prevailing WD
2003	15.4 ± 3.1	70.9 ± 19.2	SW	17.3 ± 2.9	88.3 ± 16.3	SW
2004	15.4 ± 3.9	74.4 ± 21.2	SW	17.0 ± 2.5	86.8 ± 15.5	SSW
2005	18.1 ± 3.2	65.0 ± 22.0	SSW	17.4 ± 2.9	86.8 ± 16.3	SW
2006	17.2 ± 2.9	67.8 ± 21.4	SSW	18.3 ± 2.0	88.3 ± 16.3	SSW
2007	17.0 ± 2.8	71.4 ± 27.3	E	17.8 ± 2.3	85.2 ± 21.1	E
2008	15.0 ± 3.6	76.9 ± 20.0	S	17.0 ± 2.3	86.3 ± 14.6	S
2009	17.9 ± 3.1	57.8 ± 22.2	SSW	17.4 ± 2.9	78.7 ± 21.8	S
2010	15.9 ± 3.6	80.1 ± 18.7	SW	18.2 ± 2.7	91.0 ± 16.9	SW
2011	16.6 ± 2.7	72.6 ± 21.1	SSW	17.6 ± 2.5	87.6 ± 16.1	S
2012	16.8 ± 3.2	71.0 ± 23.8	SW	18.2 ± 2.4	89.5 ± 17.9	E
2013	16.1 ± 3.1	79.8 ± 19.6	SW	19.3 ± 2.4	88.2 ± 15.6	SW
2014	15.3 ± 2.9	79.9 ± 17.6	SW	16.5 ± 2.3	86.7 ± 14.6	SW
2015	15.8 ± 2.8	70.5 ± 21.0	SW	18.0 ± 2.1	86.1 ± 16.9	SW

* Average and standard deviations are provided.

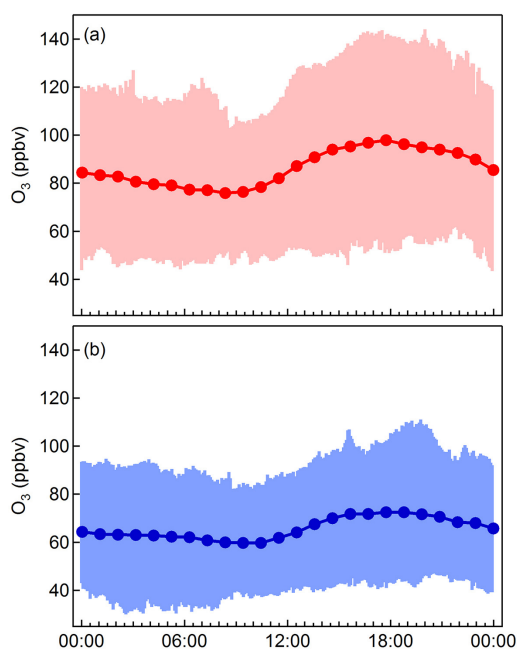


Figure 3. Average diurnal variations of surface O_3 at Mt. Tai in (a) June and (b) July–August derived from the continuous observations from 2006 to 2009. The shaded area indicates the 5th and 95th percentiles of the data.

chemical ozone production in June. Hence, the more intense photochemistry and higher regional baseline bring about the more serious O_3 pollution in June at Mt. Tai.

Another remarkable feature of surface O_3 at Mt. Tai is the relatively high nighttime levels, with average concentrations of 75–85 ppbv in June and 60–70 ppbv during July–August. This should be the composite result of the residual O_3 pro-

duced in the preceding afternoon in the boundary layer, less O_3 loss from NO titration, and long-range transport of processed regional plumes. The transport of regional plumes was evidenced by the coincident evening NO_2^* (including NO_2 and some higher oxidized nitrogen compounds) maximums and relatively low NO levels (indicative of the aged air mass), as shown in Fig. 4. Inspection of the time series day by day also reveals the frequent transport of photochemically aged air masses containing elevated concentrations of O_3 (over 100 ppbv), CO and NO_2^* to the study site during the late evening (figures not shown). Similarly, the MOZAIC aircraft measurements have also found ~ 60 ppbv on average of O_3 at around 1500 m a.s.l. over Beijing at 05:00–06:00 LT in summer (i.e., May–July; Ding et al., 2008), which is comparable to what we observed at Mt. Tai. These results imply the existence of the O_3 -laden air in the nocturnal residual layer over the North China Plain region. Moreover, Ding et al. (2008) also showed in their Fig. 11 that the O_3 enhancement extended from the surface up to about 2 km, further evidence that during the daytime Mt. Tai should be within the boundary layer and is sampling at a vertical level where ozone enhancements are expected.

3.2 Impact of long-range transport

Long-range transport associated with synoptic weather and large-scale circulations is an important factor for the variation of O_3 in rural areas (Wang et al., 2009; Ding et al., 2013; Lin et al., 2014; Zhang et al., 2016). To elucidate the history of air masses sampled at Mt. Tai, we analyzed the summertime climatological air mass transport pattern during 2003–2015 with the aid of cluster analysis of back trajectories. The NCEP reanalysis data and GDAS archive data (<http://ready.arl.noaa.gov/archives.php>) were used to compute tra-

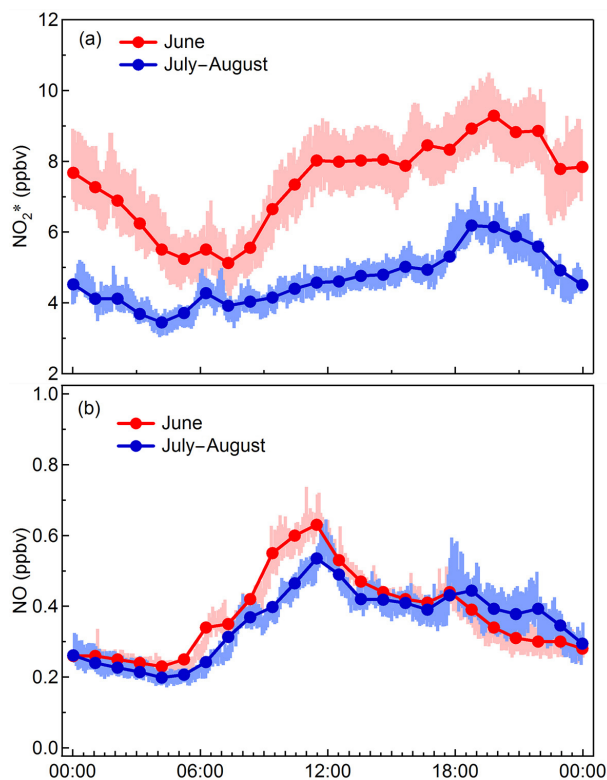


Figure 4. Average diurnal variations of (a) NO_2^* and (b) NO at Mt. Tai in June and July–August derived from the continuous observations from 2006 to 2009. The shaded area indicates the standard error of the mean.

jectories during 2003–2004 and 2005–2015. The detailed methodology has been documented by Wang et al. (2009) and Xue et al. (2011). Briefly, three-dimensional 72-hour back trajectories were computed four times a day (i.e., 02:00, 08:00, 14:00 and 20:00 LT) for June–August with the Hybrid Single-Particle Lagrangian Integrated Trajectory model (HYSPPLIT, v4.9; Draxler et al., 2009), with an endpoint of 300 m above ground level exactly over Mt. Tai. All the trajectories were then categorized into a small number of major groups with the HYSPPLIT built-in cluster analysis approach. Total spatial variance (TSV) and the variance between each trajectory component were calculated to determine the optimum number of clusters (Draxler et al., 2009).

A total of five air mass types were extracted for the summer period, with four identified for June and July–August respectively. These air mass types are named according to the regions they traversed, and are described as follows: “Marine and East China” (M&EC) – air masses from the southeast passing over the ocean and polluted central eastern China; “Northeast China” (NEC) – air masses from the north passing over Northeast China; “Central China” (CC) – air masses from the south moving slowly over central China; “Southeast China” (SEC) – air masses from the south moving fast from southeast China; “Mongolia and North China” (M&NC) –

air masses from the northwest passing over Mongolia and central northern China.

The above identified major types of air masses are presented in Fig. 5. In June, M&EC was most frequent (57 %), followed by CC (26 %), NEC (9 %) and M&NC (8 %; only identified in June). During the July–August period, the most frequent air mass type was still M&EC (36 %), then CC (29 %) and NEC (29 %), with a minor fraction of SEC (6 %; only identified in July–August). Overall, the transport patterns in June and July–August are quite similar, and it is evident that southerly and easterly air flows (e.g., M&EC and CC) dominated the air mass transport to Mt. Tai in summer. Such patterns are believed to be driven by the summer Asian monsoon (Ding et al., 2008).

The chemical signatures of the different air masses were also inspected and summarized in Table 3. The air masses of M&EC, CC and NEC, which passed over several polluted regions of eastern China, contained higher abundances of O_3 (with averages of 89–94 ppbv in June and 64–77 ppbv in July–August), CO and NO_2^* . In comparison, the more aged air masses of M&NC and SEC showed relatively lower concentrations of O_3 (78 ± 21 ppbv for M&NC in June and 58 ± 17 ppbv for SEC in July–August) and its precursors (except for NO_2^* in the SEC air mass). In view of the higher frequency and higher O_3 levels of the M&EC, CC and NEC air masses, it could be concluded that the regions with the greatest influence on O_3 at Mt. Tai in summer are primarily located in the southern and eastern parts of central eastern China.

3.3 Observed ozone trend

Figure 6 presents the monthly average hourly O_3 and MDA8 O_3 mixing ratios in June and July–August whenever available from 2003 to 2015 at Mt. Tai. The least square linear regression analysis reveals the significant increase of surface O_3 at Mt. Tai since 2003. Monthly mean O_3 values based on hourly data increased at rates of 1.7 ± 1.0 ppbv yr^{-1} ($\pm 95\%$ confidence intervals) in June and 2.1 ± 0.9 ppbv yr^{-1} in July–August, and the increases were statistically significant ($p < 0.01$). For the monthly means based on MDA8 O_3 , the fewer available data points (as we only have monthly average data during 2004–2005 from Kanaya et al., 2013) likely reduced the significance of the trend in June, with a positive but statistically insignificant increase (rate = 1.4 ± 1.9 ppbv yr^{-1} ; $p = 0.12$). However the site had a significant positive trend in July–August (rate = 2.2 ± 1.2 ppbv yr^{-1} ; $p < 0.01$). Therefore we conclude that summertime surface O_3 levels at Mt. Tai have increased over the period 2003–2015.

Given the fact that Mt. Tai is above the PBL at night when there is no photochemistry, the ambient O_3 levels before dawn (e.g., 02:00–05:00 LT) are representative of the regional baseline O_3 . The diurnal variation in Fig. 3 shows a slight but steady decrease in O_3 concentrations overnight,

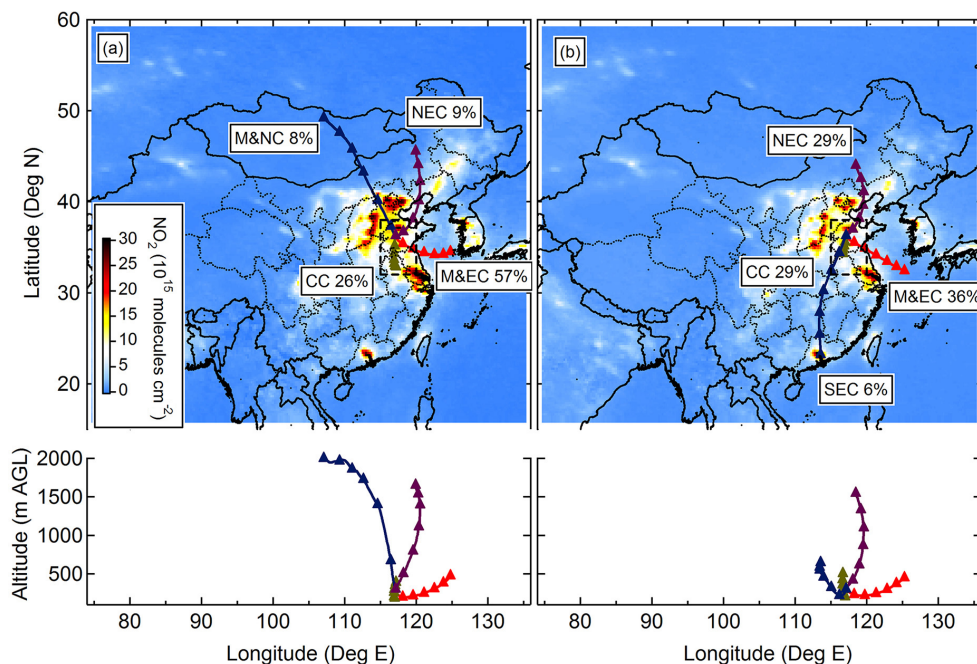


Figure 5. Climatological air mass transport pattern at Mt. Tai in (a) June and (b) July–August over 2003–2015. The maps are color-coded by the NO_2 column density retrieved from SCIAMACHY (2003–2011) and GOME-2(B) (2013–2015). The box (dashed line) refers to the domain for which the satellite retrievals were averaged. Five major air masses: (1) M&EC: Marine and East China, (2) NEC: Northeast China, (3) CC: Central China, (4) SEC: Southeast China, (5) M&NC: Mongolia and North China.

Table 3. Statistics of O_3 , NO_2^* and CO in different air mass categories.^a

Air mass ^b	June			July–August			
	O_3	NO_2^*	CO	O_3	NO_2^*	CO	
M&EC	92 ± 27	7.2 ± 5.3	500 ± 300	M&EC	64 ± 22	3.6 ± 2.9	370 ± 180
CC	89 ± 24	6.3 ± 5.0	550 ± 300	CC	77 ± 21	3.3 ± 2.5	440 ± 180
NEC	94 ± 25	7.0 ± 4.5	380 ± 180	NEC	73 ± 22	4.5 ± 4.2	380 ± 180
M&NC	78 ± 21	4.0 ± 3.3	280 ± 180	SEC	58 ± 17	4.1 ± 2.4	350 ± 80

^a The unit is ppbv; average and standard deviations are provided. ^b Refer to Fig. 5 and Sect. 3.2 for the derivation and description of the air mass types.

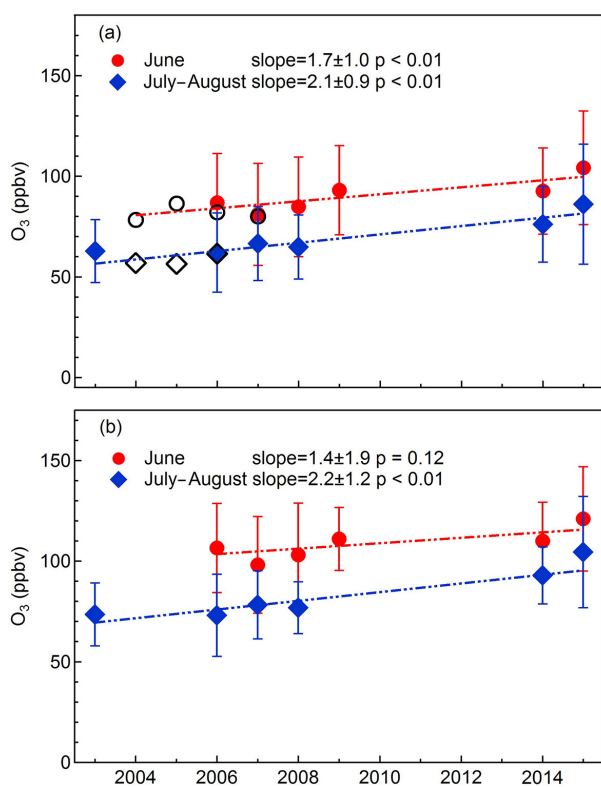
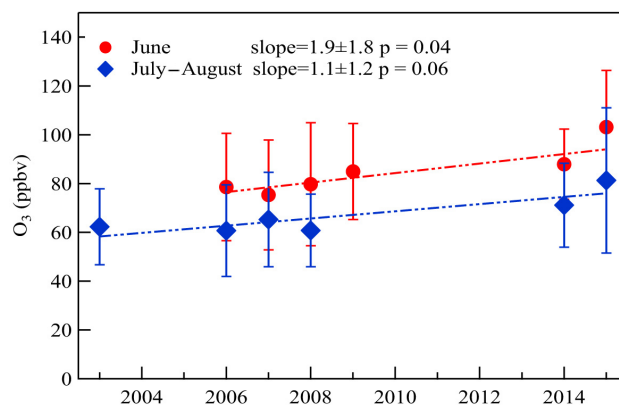
which should arise from dry deposition. It was assumed that dry deposition was essentially the same every year and did not affect the derived trends. Figure 7 shows the monthly averaged late-night O_3 mixing ratios in June and July–August available from 2003 to 2015 at Mt. Tai. Again, positive trends were found. The rate of increase was quantified at $1.9 \pm 1.8 \text{ ppbv yr}^{-1}$ ($p = 0.04$, significant) in June and $1.1 \pm 1.2 \text{ ppbv yr}^{-1}$ ($p = 0.06$, insignificant) during July–August. The increase of regional baseline likely explains the observed O_3 rise at Mt. Tai in June and also accounts for the majority of the increase during July–August. These results indicate the significant increase of surface O_3 in summer on the regional scale across northern China.

Table 4 compares the surface and lower tropospheric ozone trends available in East Asia in recent decades. Two as-

pects are particularly noteworthy from this comparison. First, most studies have deduced significant positive trends demonstrating the broad increase of tropospheric O_3 over East Asia, especially in China. This pattern is distinct from that found in Europe and the eastern U.S., where O_3 levels have begun to decrease or level off since the 1990s or 2000s (e.g., Cooper et al., 2014; Lefohn et al., 2010; Oltmans et al., 2013; Parrish et al., 2012). The O_3 increase in East Asia is expected due to the rapid economic growth and increasing anthropogenic emissions of O_3 precursors in the past 3 decades (e.g., Ohara et al., 2007). Second, the magnitude of O_3 increase is quite heterogeneous in different regions and the fastest rise was found in the North China Plain. For example, the rates of O_3 increase were in the range of $0.54\text{--}0.58 \text{ ppbv yr}^{-1}$ in Hong Kong (1994–2007, Wang et al., 2009; 2002–2013, Xue

Table 4. Summary of surface and lower tropospheric ozone trends recorded in East Asia.

Station	Site type	Period	Rate of change (ppbv yr ⁻¹)	Reference
Mt. Tai	rural	2003–2015 (summer)	1.7 ± 1.0 (June) 2.1 ± 0.9 (July–August)	This study
Beijing	rural (MOZAIC)	1995–2005	~ 1 (annual average) ~ 3 (summer afternoon)	Ding et al. (2008)
Hong Kong (Hok Tsui)	rural	1994–2007	0.58	Wang et al. (2009)
Hong Kong	urban & suburban	2002–2013 (autumn)	0.54 ± 0.49	Xue et al. (2014)
Lin'an	rural	1991–2006	2.7 % (summer daily maximum)	Xu et al. (2008)
Waliguan	remote	1994–2013 (summer)	0.15 ± 0.19	Xu et al. (2016)
Taiwan (Yangming)	rural	1994–2007	0.54 ± 0.21	Lin et al. (2010)
Mt. Happo, Japan	rural	1991–2011 (summer)	0.64 ± 0.40	Parrish et al. (2012)
Tokyo, Japan	urban & suburban	1990–2010	0.31 ± 0.02	Akimoto et al. (2015)
Nagoya, Japan	urban & suburban	1990–2010	0.22 ± 0.05	Akimoto et al. (2015)
Osaka/Kyoto, Japan	urban & suburban	1990–2010	0.37 ± 0.03	Akimoto et al. (2015)
Fukuoka, Japan	urban & suburban	1990–2010	0.37 ± 0.04	Akimoto et al. (2015)
South Korea	124 urban sites average	1999–2010	0.26	Seo et al. (2014)
South Korea	56 urban sites average	1990–2010	0.48 ± 0.07 (annual average) 0.55 ± 0.13 (summer)	Lee et al. (2013)

**Figure 6.** Monthly averaged (a) 1-hour and (b) MDA8 O₃ mixing ratios at Mt. Tai in June and July–August over 2003–2015. Error bars indicate the standard deviation of the mean. The black open circles and squares represent the data taken from Kanaya et al. (2013). The fitted lines are derived from the least square linear regression analysis with the slopes (±95 % confidence intervals) and *p* values annotated.**Figure 7.** Monthly averaged nighttime O₃ mixing ratios (inferring the regional background O₃) at Mt. Tai in June and July–August over 2003–2015. Error bars indicate the standard deviation of the mean. The fitted lines are derived from the least square linear regression analysis with the slopes (±95 % confidence intervals) and *p* values annotated.

et al., 2014), 0.54 ppbv yr⁻¹ in Taiwan (1994–2007, Lin et al., 2010), 0.26–0.55 ppbv yr⁻¹ over South Korea (1990–2010, Lee et al., 2013; 1999–2010, Seo et al., 2014), 0.22–0.37 ppbv yr⁻¹ in major Japanese metropolitan areas (1990–2010; Akimoto et al., 2015), 0.64 ppbv yr⁻¹ at Mt. Happo, Japan (1991–2011, summer scenario; Parrish et al., 2014), and 0.15 ppbv yr⁻¹ at Mt. Waliguan, a GAW station in western China (1994–2013, summer scenario; Xu et al., 2016). According to the MOZAIC aircraft observations, in comparison, Ding et al. (2008) have reported the PBL O₃ increases of ~ 1 ppbv yr⁻¹ for the annual average and ~ 3 ppbv yr⁻¹ for the summer afternoon peaks over the period of 1995–

2005. Zhang et al. (2014) analyzed their field measurements at an urban site in Beijing during 2005–2011 and quantified an increasing rate of 2.6 ppbv yr^{-1} for the daytime average O_3 in summer. Comparable rates of O_3 increase (1.7 – 2.1 ppbv yr^{-1}) were determined in the present study from the measurements of longer time coverage and at a more regionally representative mountain site, affirming the significant rise of surface O_3 levels over the North China Plain region. Furthermore, the magnitude of O_3 increase in this region is also among the highest records currently reported in the world (Cooper et al., 2012, 2014; Lin et al., 2014; Parrish et al., 2014).

3.4 Comparison with multi-decadal chemistry-climate simulations

We draw on a multi-decadal GFDL-AM3 chemical transport model simulation to provide context for the trends derived from the short observation record. In contrast to the free-running models used in Cooper et al. (2014) that generate their own metrology, the GFDL-AM3 simulations used in the present study are relaxed to the NCEP/NCAR reanalysis using a pressure-dependent nudging technique (Lin et al., 2012) and thus allow for an apples-to-apples comparison with the observational records. The simulations are described in detail in a few recent publications, which show that AM3 captures inter-annual variability and long-term trends of baseline O_3 measured at Mauna Loa Observatory (Lin et al., 2014), western U.S. free tropospheric and surface sites (Lin et al., 2015a, b). Ozone measurements at Mt. Tai provide an important test for the model to represent O_3 trends in a polluted region where emissions have changed markedly over the past few decades.

Figure 8 shows comparisons of O_3 trends at Mt Tai as observed and simulated by the GFDL-AM3 model for June and July–August, respectively. The model has a mean state ozone bias of 10–20 ppbv as in the other global models (Fiore et al., 2009). For illustrative purposes, both observations and model results in Fig. 8 are shown as anomalies. With anthropogenic emissions varying over time (based on Lamarque et al., 2010, with annual interpolation after 2000 to RCP 8.5), AM3 simulates significant MDA8 O_3 increases of $1.04 \pm 0.45 \text{ ppbv yr}^{-1}$ over 1995–2014 for June and $1.65 \pm 0.41 \text{ ppbv yr}^{-1}$ for July–August. A greater rate of O_3 increase is simulated for July–August with warmer temperatures compared to June, consistent with the trends derived from the shorter observational records. With constant emissions, AM3 gives no significant long-term ozone trends over the entire 1980–2014 period despite large inter-annual variability (see gray lines in Fig. 8). The model indicates that changes in regional emissions have raised surface ozone over the North China Plain by $\sim 30 \text{ ppbv}$ in June and $\sim 45 \text{ ppbv}$ in July–August over the past 35 years, with an accelerating trend in the most recent 20 years.

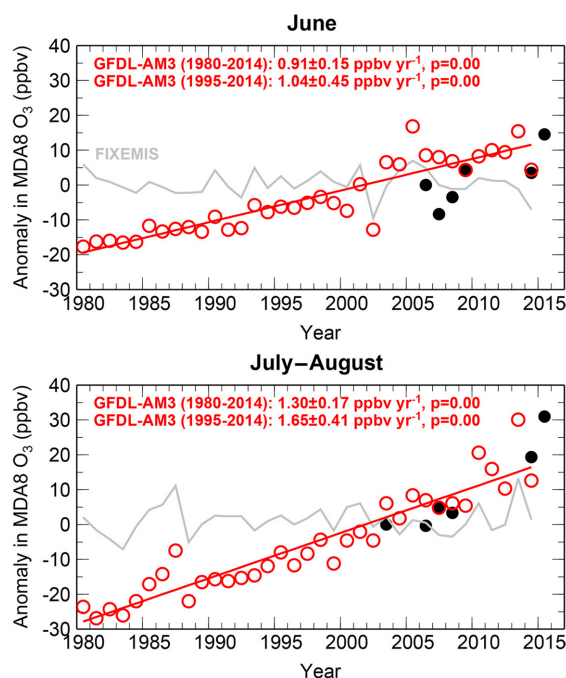


Figure 8. Comparison of ozone trends at Mt. Tai. (a) Anomalies in the June average of MDA8 O_3 from 1980 to 2015 as observed (black dots) and simulated by the GFDL-AM3 model with time-varying (red circles) and constant anthropogenic emissions (gray lines). (b) Same as panel (a) but for the July–August average. The model is sampled in the surface level. The linear trends over 1980–2014 and 1995–2014, including the 95 % confidence limits and p -values, are shown.

3.5 Roles of meteorology and anthropogenic emissions

To further elucidate the factors contributing to the observed surface O_3 change at Mt. Tai, we examined variations in both meteorological conditions and anthropogenic emissions of O_3 precursors during the past decade. Table 2 shows year-to-year variability in summertime mean meteorological conditions including average temperature, RH, and prevailing wind direction recorded at Mt. Tai over the period of 2003–2015. No systematic change was found with regard to the overall meteorological conditions although some variability is clearly shown. For instance, substantially warmer temperatures were recorded in June 2015 and July–August 2013 as a result of large-scale heat waves (Yuan et al., 2016). During these years, the GFDL-AM3 model with constant emissions simulates high- O_3 anomalies (up to $\sim 10 \text{ ppbv}$) relative to the climatological mean (Fig. 8), indicating that heat waves can enhance summertime O_3 pollution in Central Eastern China. Unfortunately, there were no observations available in 2005 and 2013. Observations show greater O_3 levels during July–August of 2014–2015 compared to 2004–2009 (Fig. 8) but no significant change in temperature were found between the two time periods (Table 2), indicating the key role of regional

emission changes in contributing to the observed ozone increase at Mt. Tai.

We also explored the air mass transport pattern deduced from cluster analysis of back trajectories (see Sect. 3.2) year by year over 2003–2015 (Table S1). Despite the large year-to-year variability, again, no systematic change in the air mass transport pattern was indicated during the target period. Although the impact of meteorology on tropospheric O_3 is very complex and might not be quantified by such a simple analysis, the significant increase of surface O_3 observed at Mt. Tai should not primarily arise from the change in meteorological conditions.

We analyzed the satellite retrievals of formaldehyde (HCHO) and NO_2 to track the variations in the abundances of O_3 precursors (i.e., NO_x and VOCs) during the study period. Considering that HCHO is a major oxidation product of a variety of VOC species and due to the availability of the satellite-retrieved products, HCHO was selected as an indicator of the VOC abundances. The satellite data were obtained from SCIAMACHY for 2003–2011 and GOME-2(B) from 2013 onwards, with the Level-2 products taken from the TEMIS archive (Tropospheric Emission Monitoring Internet Service; <http://www.temis.nl/index.php>). Considering the geographical representativeness of Mt. Tai, a larger domain ($32\text{--}38^\circ\text{N}$, $115\text{--}120^\circ\text{E}$; see Fig. 5) was selected to process the regional mean satellite data. The monthly averaged HCHO and NO_2 column densities in June and July–August from 2003–2015 are documented in Fig. 9. Significant positive trends are seen for HCHO, with rates of $2.7\pm 2.2\%$ ($p = 0.02$) for June and $2.2\pm 1.4\%$ ($p < 0.01$) for July–August, indicative of the strong increase of VOCs in this region. This result agrees very well with the emission inventory estimates which showed significant increases of anthropogenic VOC emissions in China in the past decades (Bo et al., 2008; Wang et al., 2014), and is consistent with the lack of nationwide VOC controls. All of these results evidence the increase of atmospheric VOC abundances over the North China Plain.

What is more interesting is the two-phase variation of the NO_2 column, showing a significant increase first from 2003 to 2011 (June: $4.8\pm 3.4\%$, $p = 0.01$; July–August: $7.7\pm 3.6\%$, $p < 0.01$) and a decrease afterwards (see Fig. 9b). These satellite observations agree very well with the bottom-up emission inventory estimates, which clearly showed a break point occurring in 2011 in the anthropogenic NO_x emissions of China (see Fig. S2). China has just launched a national NO_x control programme during its “Twelfth Five-Year Plan” (i.e., 2011–2015) (China State Council, 2011). The strict control measures are very efficient and have resulted in an immediate reduction of NO_x emissions, as affirmed by both emission inventories and satellite retrievals. Furthermore, the reduced levels of NO_x in the most recent 5 years were also evidenced by our limited in situ NO_x^* measurements at Mt. Tai. As shown in Fig. 10, the ambient NO_2^* levels in 2014 and 2015 were indeed substan-

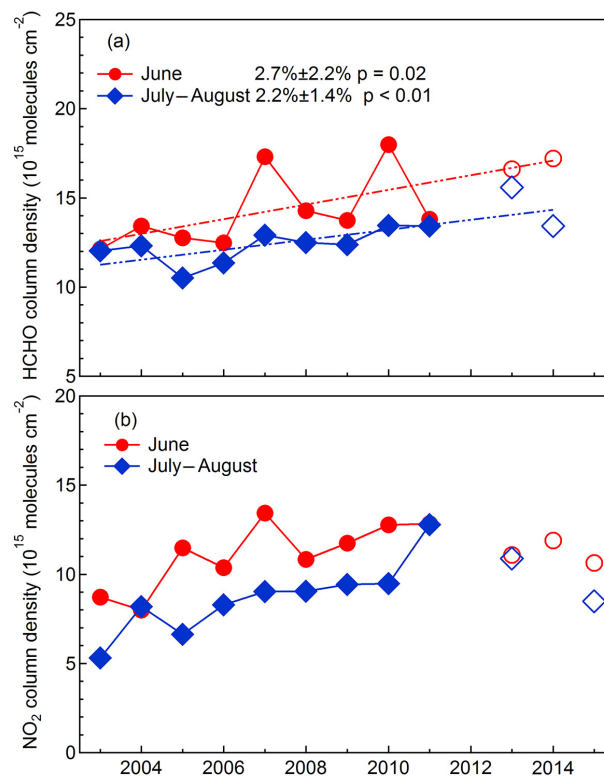


Figure 9. Monthly average column density of (a) formaldehyde and (b) NO_2 retrieved from SCIAMACHY (2003–2011; solid markers) and GOME-2(B) (2013–2015; open markers) for the target domain ($32\text{--}38^\circ\text{N}$, $115\text{--}120^\circ\text{E}$). For formaldehyde, the fitted lines are derived from the least square linear regression analysis with the slopes ($\pm 95\%$ confidence intervals) and p values also shown.

tially lower than those measured in the previous years before 2010.

From the above analyses, the O_3 increase between 2003 and 2011 is easy to understand in light of the consistent increase of both NO_x and VOCs. For the later period, i.e. after 2011, in comparison, opposite trends have taken place with NO_x decreasing but VOCs still increasing. The observed continuing O_3 rise suggests that the reduction of NO_x is not adequate to reduce the ambient O_3 levels, with a background of increasing VOCs. We then evaluated the ozone production efficiency (OPE) for the air masses sampled at Mt. Tai. OPE is usually derived from the regression slope of the scatter plots of O_3 versus NO_z (Trainer et al., 1993), and is a useful metric to infer how efficient O_3 is produced per oxidation of unit of NO_x (e.g., Wang et al., 2010; Xue et al., 2011). As NO_y (and thus NO_z) is not routinely measured in the present study, NO_2^* is used instead of NO_z to infer the OPE values. It should be reasonable considering that our measured NO_2^* significantly overestimated true NO_2 and actually contained a large fraction of NO_z , especially in the afternoon period when NO_z was at its maximum with NO_2 at the minimum (Xu et al., 2013). The scatter plots of O_3 ver-

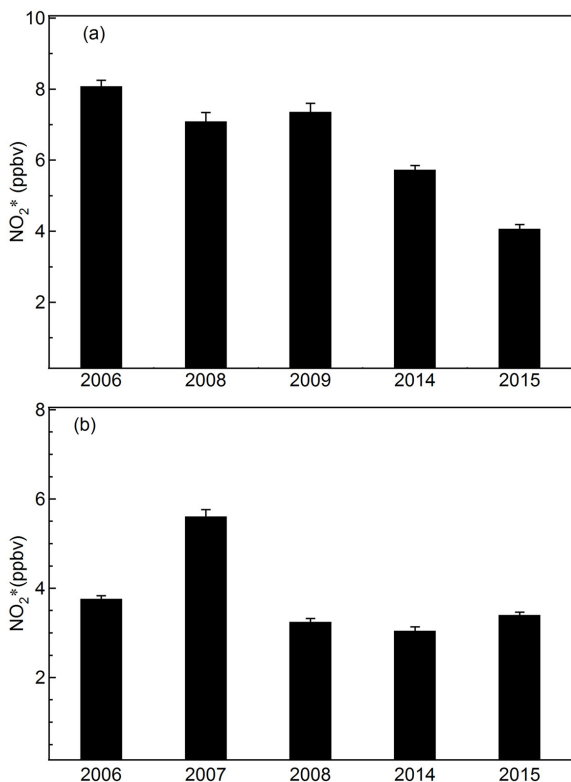


Figure 10. Monthly averaged NO_2^* concentrations measured at Mt. Tai in (a) June and (b) July–August during 2006–2015. Error bars indicate the standard error of the mean. Note that the data point in June 2014 is for NO_x^* instead of NO_2^* .

sus NO_2^* during the afternoon hours (i.e., 12:00–18:00 LT) available from 2006–2015 at Mt. Tai are presented in Fig. 11. The OPE values in 2014–2015 (i.e., 9.6–15.0) were significantly higher than those determined during 2006–2009 (i.e., 3.6–7.9). This demonstrates the greater ozone production efficiency in recent years with increasing VOCs. These results indicate that although NO_x in China has been reduced since 2011, little action on VOC control has led to increased emissions of VOCs, which could make O_3 formation more efficient per unit of NO_x . As a consequence, ambient O_3 levels have been rising in northern China. We conclude that control of VOCs is urgently needed, in addition to the ongoing strict NO_x control, to mitigate regional O_3 pollution in China.

4 Conclusions

We have compiled all field observations of O_3 and its precursors ever collected at Mt. Tai in the North China Plain and found a significant increase of summertime surface O_3 over 2003–2015 at this regionally representative site. The ozone increase derived from these valuable but sparse observations is supported by a chemistry-climate model simulation with emissions varying from 1980 to 2014. The marked increase

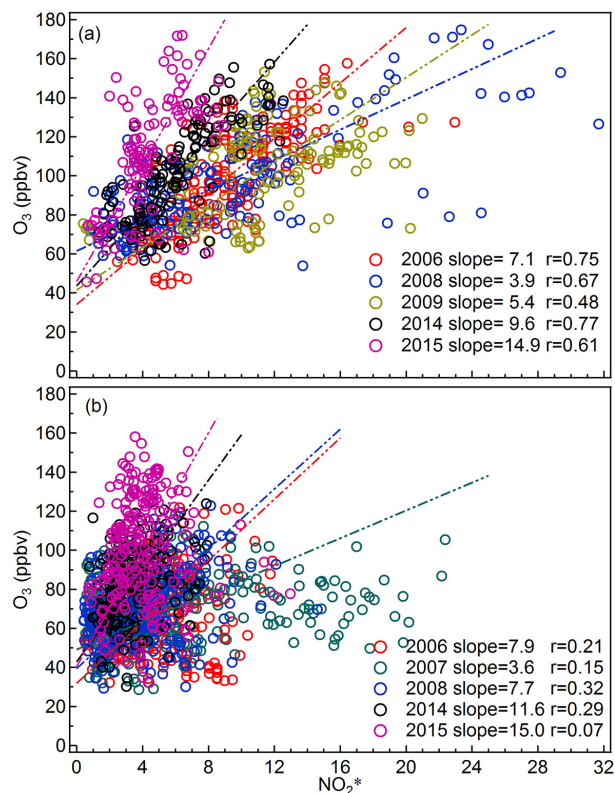


Figure 11. Scatter plots of O_3 versus NO_2^* at Mt. Tai in (a) June and (b) July–August during 2006–2015. Only the afternoon data (i.e., 12:00–18:00 local time) were used for this analysis. Note that the NO_2^* data in June 2014 stand for NO_x^* ($\text{NO}_x^* = \text{NO} + \text{NO}_2^*$), of which NO usually presents a minor fraction. The slopes are determined by the reduced major axis (RMA) method.

of surface O_3 levels over central Eastern China is primarily attributable to rising anthropogenic emissions of O_3 precursors. We provide evidence that the changes in VOCs emissions contribute to the observed ozone increase since 2011 when NO_x has reduced efficiently as a result of a strict national emission control programme. This study provides some evidence that the current Chinese control programme, focusing on NO_x alone with little action for VOCs, is not sufficient for mitigating regional O_3 pollution, and calls on the implementation of VOC controls as soon as possible. Similar to the U.S., China phased in its air pollution control measures resulting in decreasing SO_2 emission after 2006 and NO_x emission after 2011. It is foreseen that the national VOC control will be launched very soon. Thus, follow-up long-term measurements are required to evaluate the response of ambient O_3 to the upcoming VOC control and to provide observational constraints to evaluate global and regional chemical transport models.

5 Data availability

The underlying research data can be accessed upon contact with the corresponding author (Likun Xue, xuelikun@sdu.edu.cn).

The Supplement related to this article is available online at doi:10.5194/acp-16-10637-2016-supplement.

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