https://doi.org/10.20965/jdr.2007.p0163

Review:

Recent Trends and Future Projections in Asian Air Pollution

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[Received March 14, 2007; accepted April 12, 2007]

We studied trends in Asian air pollution in recent decades using air-quality monitoring station data, satellite retrieval data (GOME NO_2), and regional-scale chemical transport model (CTM) simulation. A newly developed annual Asian-scale emission inventory (REAS) from 1980-2003 was used in observation data analysis and CTM. Analyses of recent trends in annual emissions in China by REAS and satellite GOME NO_2 show an 8-10% increase after 2000 suggesting the impact of long-range transport of secondary air pollutants in regions and countries downwind. Detailed analyses of O_3 observation data in Japan suggest an annual averaged O_3 concentration increase of 2% yr^{-1} due to this long-range transport. We extended our regional air quality study targeting 2020. REAS provides three emission scenarios for China: the reference case (REF), the policy success case (PSC), and the policy failure case (PFC). Projected REF emissions for 2020 show O_3 concentrations rising to 75 to 90 ppbv in June and 75 to 85 ppbv in August over the North China Plain. Projected PFC emissions bring an increase of monthly averaged O_3 with greater than 20 ppbv (1 ppbv yr^{-1} growth) in the North China Plain. Surface O_3 under the PFC scenario is enhanced by 6 to 8 ppbv over the Korean Peninsula and by 2 to 6 ppbv in Japan from 2000 to 2020 despite the reduction of NO_X in Japan. This may become a critical level in air quality in Asia.

Keywords: ozone, pollution, Asia, environmental standard, chemical transport model

1. Introduction

In Asian developing countries, including China, air pollution emissions caused by anthropogenic activities have increased rapidly. For example, in China, NO_X emissions increased by 2.5 times during 1980-2003; this upward tendency has been exacerbated since 2000. In 2020, emissions of SO_2 and NO_X are predicted to increase by 1.4 and 1.7 times, respectively. Accordingly, the problem of transboundary air pollution by NO_X emissions will be important. Such emission increases trigger regional air pollution (transboundary air pollution) including tropospheric O_3 and particulate material and undesired influences on health, food production, and ecosystems. Moreover, regional air pollution influences climate change, presenting the calamitous possibility of considerable damage to both humans and ecosystems combined with air pollution. In addition, air pollution in east Asia influences global-scale air quality and is recognized as a worldwide environmental issue.

The United Nations Economic Commission for Europe (UNECE) enacted the Convention on Long-Range Transboundary Air Pollution (CLRTAP) in 1979 and has implemented various policies concerning wide-area pollution management under the Convention since then. It was later clarified that areas outside Europe have also exerted strong influences on regional air pollution in Europe. Therefore, UNECE/CLRTAP established the "Task Force on Hemispheric Transport of Air Pollution" (TFHTAP) in December, 2004; the task force is to produce an interim report in 2007 and a final report in 2009. In addition, UNECE/CLRTAP has given attention to other regions including Asia. An international workshop devoted to emission inventory in east Asia as a main subject, sponsored by TFHTAP, was held in Beijing in October 2006.

Generally NO_X has a life of one day or less in the atmosphere, but it generates photochemical O_3 through complicated photochemical reactions and is converted into pollutants such as nitrate particles, which have long lives. The pollutants are transported long distances and trigger transboundary pollution in regions leeward of the Asian Continent, such as Korea and Japan. In this context, it can be said to be a kind of man-made disaster in a broad sense.

In addition to the fact that O_3 itself is a pollution gas for which air quality environmental standards are specified, the increase of photochemical O_3 caused by the increase of NO_X emissions promotes generation of nitric acid HNO_3, nitrate particles, sulfate particles, etc., engendering acidification of the atmospheric environment. Among the acidification substances generated through photochemical reactions, those aside from NO_2 are called photochemical oxidants (or simply, oxidants) and are used as an index of photochemical smog. As far as the oxidant concentration in the atmosphere is concerned, environmental standards in 1-h values (60 ppbv and less) are set in Japan according to notification of the Environmental Agency based on the Basic Law on the Environment. Oxidant warnings, photochemical smog warnings, are issued if it is judged that oxidant concentration exceeds 120 ppbv consecutively. If levels are higher than 240 ppbv, an alert is issued. For the Tokyo Metropolitan area, oxidant warnings are issued in many cases in summer because of urban air pollution. On the other hand, in Nagasaki Prefecture, where no large industrial area exists, the first oxidant warning since the start of the observations was issued in May 2006. This might have been triggered by transboundary air pollution.

As described above, in the Asian region, a huge amount of air pollutants and greenhouse gases are emitted into the atmosphere and have strongly influenced the atmospheric environment. To evaluate and forecast such environmental influences and take appropriate measures against them, it is necessary to evaluate discharge (emission) into the air as precisely as possible and to produce an emissions inventory. In Asia, various emission inventories have been made so far. Such inventories are roughly classifiable as follows: (a) inventories based on policies and studies targeting for own countries; (b) inventories based on studies on the Asian-scale, such as those by Kato and Akimoto [13], RAINS-ASIA [12], and TRACE-P [21]. For example, TRACE-P inventory, which was developed for the observation campaign in spring of 2001, estimated emissions of many kinds of substances attributable to anthropogenic activities (including biomass burning) in 2000 at an horizontal resolution of one degree of longitude and latitude, and has often been used in Asian atmospheric environmental studies. On the other hand, if the atmospheric environmental influences were evaluated, it would be necessary to grasp the change of emission during this period. However, emissions inventories from the past to the present (historical inventories) are extremely few: only one such study gives emissions of SO_2 and NO_X estimated for 1985-1997, based on RAINS-ASIA (Streets et al., [21, 22]).

Comparisons of NO_X emission inventory and NO_2 tropospheric column concentration from environmental measurement satellite have been reported by Richter et al. [19]. According to these results, in the cases of Europe and North America, both indicate good correspondence, but in the case of China, large discrepancy with the estimation of NO_X emission might be recognized so that inaccuracy of basic statistical information (energy data, etc.) that are used in emission calculation in China are pointed out (e.g. [1]).

This paper reports the recent emission trends, observed/modeled NO_2 and O_3 trend analysis, and the results of future air pollution level for the year 2020 based on three-emission scenario from the standpoint of environmental standard level and possible damage to vegetations.

2. Recent Trend of NO_X Emission in Asia and its Future Projection

Ohara et al. [16] reported a regional emission inventory for Asia [Regional Emission inventory in ASia (REAS) ver. 1.1] during 1980-2003 as well as the years 2010 and 2020. As the first inventory in the Asian region, REAS estimates emissions from the past to the present using uniform methods. The species covered with REAS are nitrogen dioxides (NO_X), sulfur dioxide (SO_2), carbon monoxide (CO), black carbon (BC) particulates, organic carbon (OC) particulates, non-methane volatile organic compounds (NMVOC) emitted through anthropogenic activities (combustion of fossil fuels and biomass fuel, industrial processes and agricultural activities). The area covered with REAS is the Asian region east of Afghanistan. In this section, recent trends and future projections of NO_X emissions are introduced.

The NO_X emissions throughout Asia were estimated as approximately 27.3 Mt in 2000. Emissions in China (11.2 Mt; 41%) and India (4.7 Mt; 17%) were very high, but those of Japan (7%), Korea (6%), and Indonesia (6%) were also high. Classified by sectors, the percentages of oil burning at the transport sector and coal combustion of the power plants and industries were high, accounting respectively for 34%, 22%, and 14%. Classified by fuels, oil (41%) is slightly more than coal (37%). In China, the country with the largest emissions, the percentages of coal power plants, coal combustion in the industrial sector, and oil combustion in the transport sector are high, accounting respectively for 34%, 25%, and 25% of the total.

Figure 1 shows the trends of emissions by regions and sectors for 1980-2003; **Fig. 2** shows spatial distributions (0.5 degree grid of longitude and latitude) for 1980 and 2000. The total NO_X emission in Asia has increased during 1980-2003: 2.8 times in 24 years. During that period, the rate of increase during 1996-1999 was low, but those since 2000 are notably higher. Among others, the rate of increase in China's emissions is very large: 3.8 times (annually averaged rate of increase: 6%). Particularly, the increase since 2000 shows a rate of increase that has been never experienced before (1.3 times in only 3 years). This change since 1996 has been verified by satellite observation data [1].

In terms of sectoral contribution of Asian NO_X emissions, power plants, industry, transport and domestic consumption accounts for 17%, 29%, 37%, 17%, respectively, to the total emissions in 1980 and 33%, 22%, 35%, 10%, respectively, in 2003. The increase for the contribution of power plants is very high. Devoting attention to spatial distributions, emissions have increased rapidly for 20 years in all regions. In particular, increases in Chinese coastal regions and in Indian urban regions (Hindustan Plain and the western coastal region) are quite high. In contrast, the changes in Japan and the Korean Peninsula are small.

In addition to the estimation over 24 years described above, future projections of Asian emissions were performed based on emission scenarios and emissions for

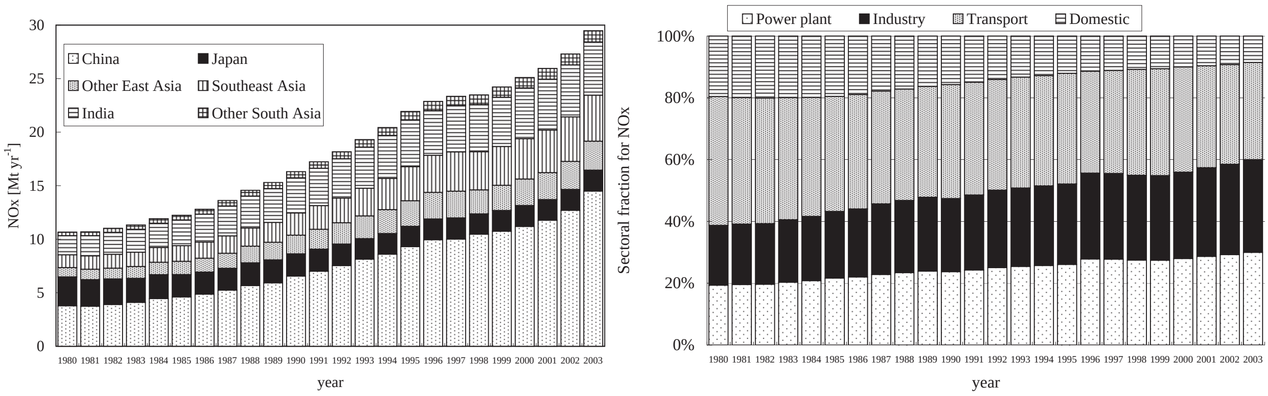

Fig. 1. Changes in NO$_X$ emissions by regions (left) and their combustion by sector (right) from 1980 to 2003.

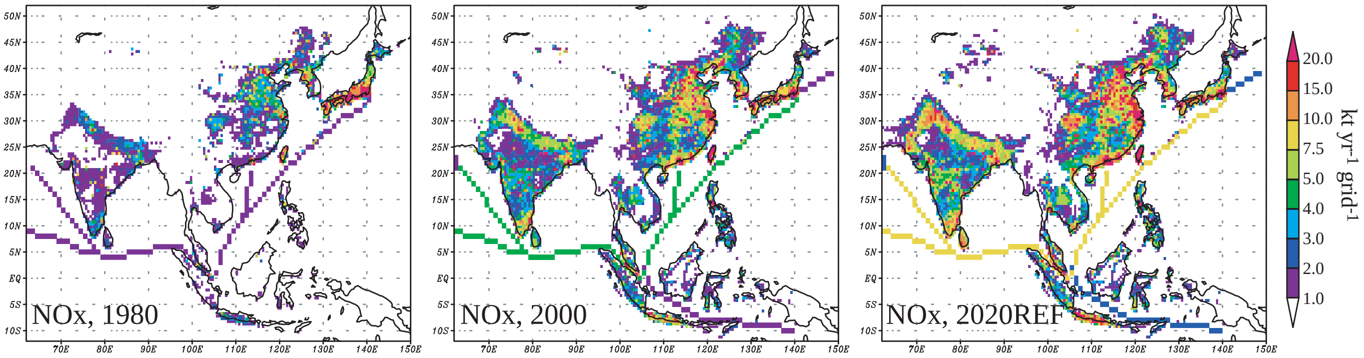

Fig. 2. Spatial distributions of NO$_X$ emissions in 1980, 2000, and 2020 (REF scenario).

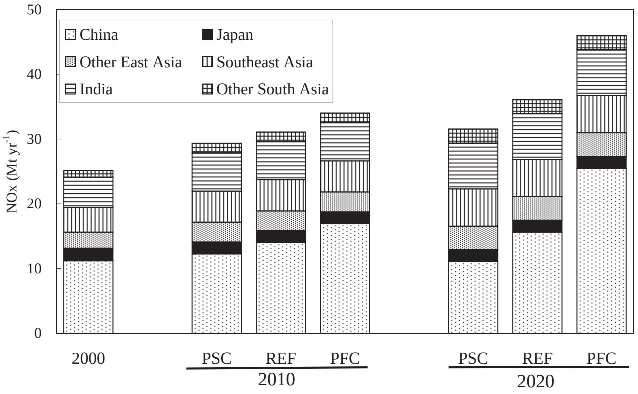

Fig. 3. NO$_X$ emissions in 2000, 2010, and 2020 (REF scenario).

2000. Three emissions scenarios for China were developed for 2010 and 2020 by Zhao et al. [29]. The socio-economical indices, such as population, urbanization, and GDP, are almost the same in these scenarios. The population in 2020, 1.45-1.49 billion, is similar to that of the IPCC B2 scenario (1.45 billion) [15]. The GDP growth rate during 2000-2020 (7.3% yr^{-1} before 2010 and 6.7% yr^{-1} after 2010) is close to that of the A2 scenario of IPCC (7% yr^{-1}) [15].

The first scenario is termed the Policy Failed Case scenario (PFC), which means “pessimistic scenario”, having high emissions caused by the continuing of current energy structure, increased energy consumption, and the slow deployment of new energy technologies and of new emission-control technologies. The second scenario is termed the Reference scenario, or REF. This means “sustainable scenario”, having a moderate emission caused by suppression of energy consumption through energy conservation, a change to clean energy, and moderate deployment of new energy technologies and of new emission control technologies. We think that this presents our “best guess”, as to what emissions in Asia will be in the years 2010 and 2020. The third scenario is termed the Policy Success Case scenario, or PSC. An “optimistic case”, it assumes low emissions achieved through implementation of strong energy and environmental policies and the rapid deployment of new energy technologies and of new emission-control technologies. Concepts of PFC, REF, and PSC respectively resemble those of the A2 scenario, the B2 scenario, and the B1 scenario of IPCC [15].

In the research project of the China Energy Research Institute and National Lawrence Berkeley Laboratory [29], the energy consumption under the REF, PSC, and PFC scenarios were provided from forecasts by a simulation model, the Long-range Energy Alternatives Planning system (LEAP; available at http://forum.seib.org/leap) developed by the Stockholm Environment Institute. Total energy consumption in China in 2010 is 1.22 (PSC), 1.30 (REF), and 1.35 (PFC) times that in 2000. There is expected to be a further marked increase by 2020. There should be increases in energy consumption of 39% (PSC), 63% (REF), and 77%

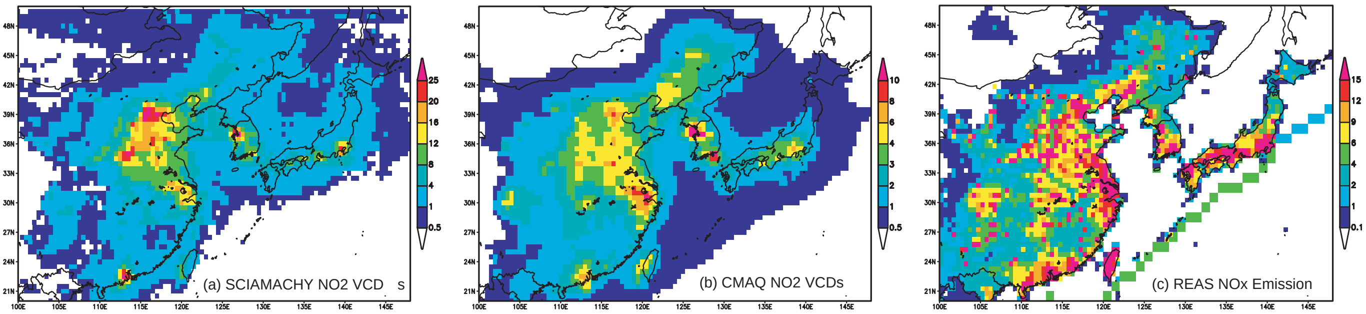

Fig. 4. (a) Annually averages in 2003 of tropospheric NO_2 vertical column density by SCIAMACHY ($\times 10^{15}$ molecules/cm^2), (b) tropospheric NO_2 vertical column density by CMAQ ($\times 10^{15}$ molecules/cm^2), (c) spatial distributions of NO_X emissions by REAS 1.1 for 2003 ($\times 10^3$ Tg-NO_2/year/grid).

(PFC), during 2000-2020. The differences in coal consumption by power plants between the three scenarios in 2020 are important to note. For other countries, the fuel consumption values in the years 2010 and 2020 were calculated on the basis of References Scenario Projections in the "World Energy Outlook" [10].

Based on the results of that estimation of energy consumption, emissions of each pollutant substance were estimated. Emission factors follow a value in 2000. However, for automobiles and power plants, the emission factors were changed, taking account of the tendency of emissions regulation based on the concept of emissions scenarios. **Fig. 3** shows the change of NO_X emissions in the years of 2010 and 2020. The NO_X emissions in China in 2020 are estimated as -1% (PSC), $+40\%$ (REF), $+128\%$ (PFC) compared to that of 2000. According to the PFC scenario, emissions will increase by 2.3 times compared to those for 2000; also according to the PSC scenario, it will decrease, but the change is not great. It is indicated that emissions will change considerably depending on the emissions scenario. In other words, implementation of suitable environmental measures would make the conservation of the atmospheric environment possible. However, overall emissions for Asia in 2020 are estimated to increase compared to those of 2000 under all scenarios: $+26\%$ (PSC), $+44\%$ (REF), $+83\%$ (PFC). Worsening of the atmospheric environment appears likely. Future projections of O_3 on the basis of emissions under those scenarios will be described later.

3. Recent Increasing Trend of NO_2 over Asia

An important air pollutant among NO_X, NO_2 was observed on a global scale using data from the Global Ozone Monitoring Experiment (GOME) sensor loaded aboard the Europe remote sensing satellite ERS-2 for environmental observation. It was launched in 1995. Later, the SCIAMACHY sensor loaded on the ENVISAT observation satellite, which is the successor to ERS-2 and Aura/Ozone Monitoring Instrument (OMI) sensor that the United States NASA launched in June 2004, began measurements from space using high-definition 13 × 24 km^2 resolution. In this section, results of comparative analyses

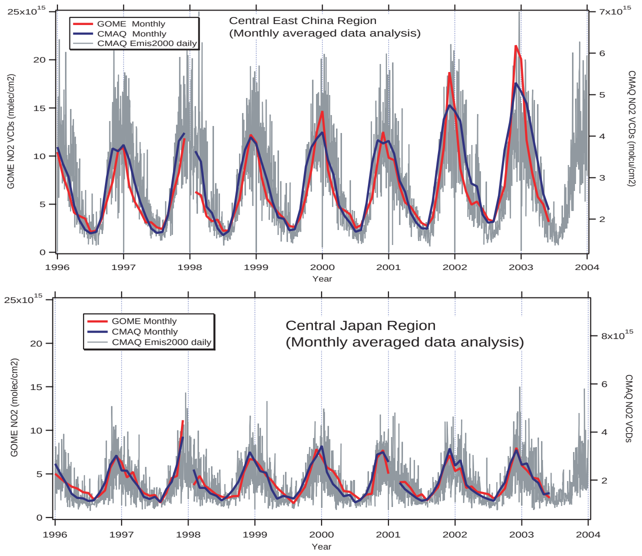

Fig. 5. Changes in GOME NO_2 and CMAQ NO_2 for 1996 to 2003 in central east China (above) and in Japan (below).

are described, including the modeling calculation through the RAMS/CMAQ cooperation system using REAS emission source inventory [8, 24, 25] and the year-by-year change of NO_2 troposphere column concentration data measured from the environmental observation satellite.

First, observation data that are used must be explained. For the satellite observation data during 1996-2002, the NO_2 troposphere column concentration obtained from GOME was used. The sensor had spatial resolution capability of 40 km (latitude) × 320 km (longitude), covered the globe in 3 days, and carried out measurements at about 1030 a.m. local time. To obtain NO_2 troposphere column concentration, the NO_2 air column in the stratosphere is estimated using the three-dimensional chemical transport modeling SLIMCAT; this is deduced from all quantities of the air column [19]. The results are published on GOME HP by Bremen University (http://www.iup.uni-bremen.de/doas/gome_no2_data.htm). Data from January 1996 to April 2003 are available. For data during and after 2003, the measurement results from SCIAMACHY were used. The sensor had spatial resolution ability of 30 km (latitude) × 60 km (longitude) and covered almost all of the globe in 12 days. Measurement results until 2005 were used.

Details of analysis results are described in Uno et al. [25] and He et al. [8]. Here, using these two results, the comparison period is extended to 1995-2006 and shown. Because REAS emission is available only until 2003, values in and after 2004 are replaced with those in 2003.

Figure 4 shows, in terms of annually averaged values in 2003, (a) the NO_2 column from SCIAMACHY, (b) the NO_2 troposphere column concentration from CMAQ, and (c) spatial distributions of NO_X emissions from REAS 1.1 in 2003. Although these three values (CMAQ NO_2, SCIAMACHY NO_2, REAS NO_X) have good correspondence. The CMAQ is too small, especially so near strong emission sources. Among others, in the Beijing area, CMAQ has factor 1-4 times smaller than that from SCIAMACHY.

Figure 5 shows changes of GOME NO_2 and CMAQ NO_2 during 1996-2003 in central east China (CEC; 30°N, 110°E to 40°N, 123°E) and Japan.

The NO_2 column in the area of Japan has good correspondence with NO_2 of GOME/SCIAMACHY in terms of both the year-by-year change and the seasonal variation. No strong tendency toward year-by-year increase in concentration levels is recognizable, which reflects the fact that the change of emissions is quite small. However, during 2004-2005, a decreasing tendency is apparent. The regression equation for CMAQ NO_2 and GOME NO_2 is

$$\mathrm{GOME\_NO_2} = -7.89\mathrm{E}14 + 2.67 \times \mathrm{CMAQ\_NO_2}$$
$$(\mathrm{molecule} \cdot \mathrm{cm}^{-2}) \quad (R = 0.833).$$

The value of CMAQ is about 37% that of GOME.

For China, the tendency for change, including seasonal variation, as observed by satellite, are reproduced by the result of CMAQ NO_2 well. The seasonal variation has an asymmetric pattern: its minimum concentration is in July and August in summer; the maximum is in December and January in winter. In addition, in recent observations by satellite, large peaks of NO_2 in winter are quite marked. During and after 2002, the rate of increase of NO_2 in winter, as calculated from satellite measurements, might reach 20% yr^{-1}. This rate of increase is nearly double that of GDP, as announced by the Chinese government [8]. Regarding satellite measurements, the growth rate in the Beijing and Shanghai areas are different [8].

Although more detailed analysis is presented in Uno et al. [25], it is clarified through systematic comparison of NO_2 column concentrations from environmental observation satellites that the long-term simulation by CMAQ reproduces the year-by-year change and the seasonal variation of NO_2 that strongly influences the concentration of O_3 in the troposphere. However, NO_2 column concentrations by GOME/SCIAMACHY are higher than CMAQ in the area of high emission sources in China. Therefore, it is necessary to review the emission inventory based on more reliable energy statistical data.

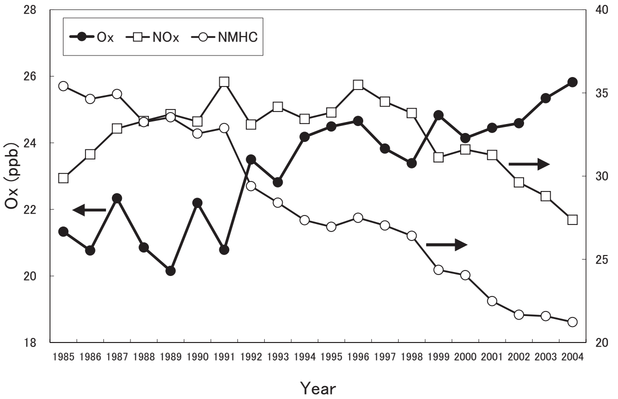

Fig. 6. Changes in annually averaged concentrations of O_X, NO_X, and NMHC at air quality monitoring stations (general stations for continuous measurement).

4. Year-by-Year Trend of O_3 Concentration in Japan

Tropospheric O_3 concentrations tend to increase globally; it is a concern that O_3 concentrations will continue to increase also in the future. Here the upward tendency is introduced based on results of O_3 observations in Japan over the last 20 years.

Figure 6 shows the change of annually averaged concentrations of photochemical oxidants (O_X) measured at general air quality monitoring stations (597 stations) over Japan. Most of these stations are located in urban areas. A large part of O_X is considered to be ozone. Average O_X concentrations measured every 5 years during the 20 years of 1985-2004 are 21.1 ppbv, 22.7 ppbv, 24.2 ppbv, and 24.9 ppbv, increasing at an increment of about 0.25 ppbv yr^{-1} (1% yr^{-1}). Moreover, the ozone concentrations measured at remote site in Japan also indicate an upward tendency after 1990. The average O_3 concentrations during the 10 years of 1992-2002 at Happo (mountainous area) and Ryori (rural coastal area) reveals increments of 9.0 ppbv (2% yr^{-1}) at Happo and of 7.3 ppbv (2% yr^{-1}) at Ryori.

On the other hand, the surface concentrations of NO_X and NMHC, the precursors of O_3, have decreased over Japan as shown in **Fig. 6**. Judging from this fact, transboundary pollution from the Asian continent such as China, where the increase of emissions is considerable, is likely to have contributed to the increase in tropospheric O_3 in Japan.

5. Numerical Analysis of Recent Trend of O_3 Concentration in Asia and its Future Projection

Present and the future atmospheric compositions over east Asia were simulated using the Models-3 Community Multi-scale Air Quality Modeling System (CMAQ) [5] coupled with the Regional Emission inventory in Asia

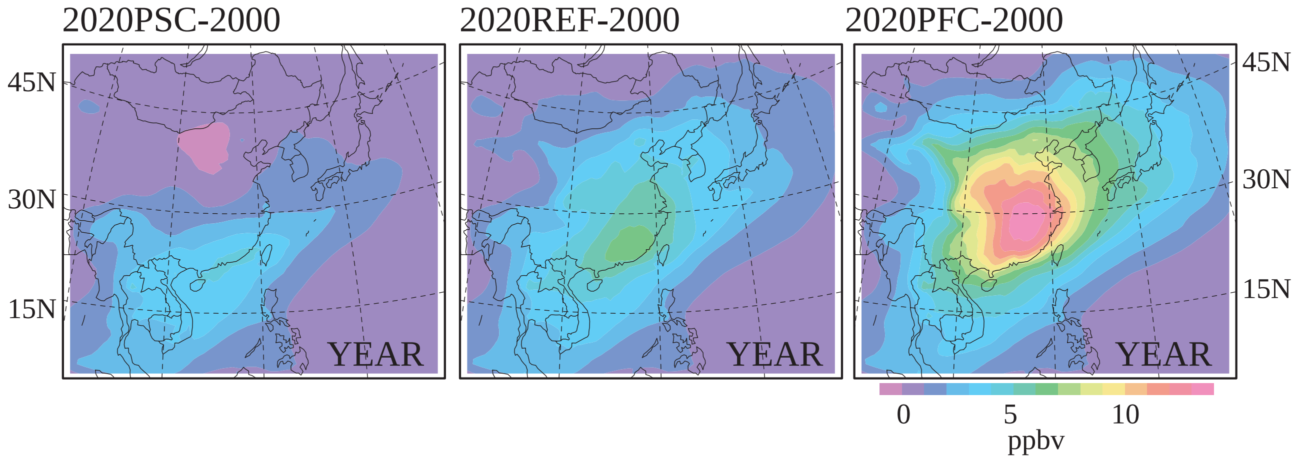

Fig. 7. Spatial distribution of O_3 increase from 2000 to 2020 under three scenarios (PSC, REF, and PFC) for annually averaged concentrations in the boundary layer (below 2 km).

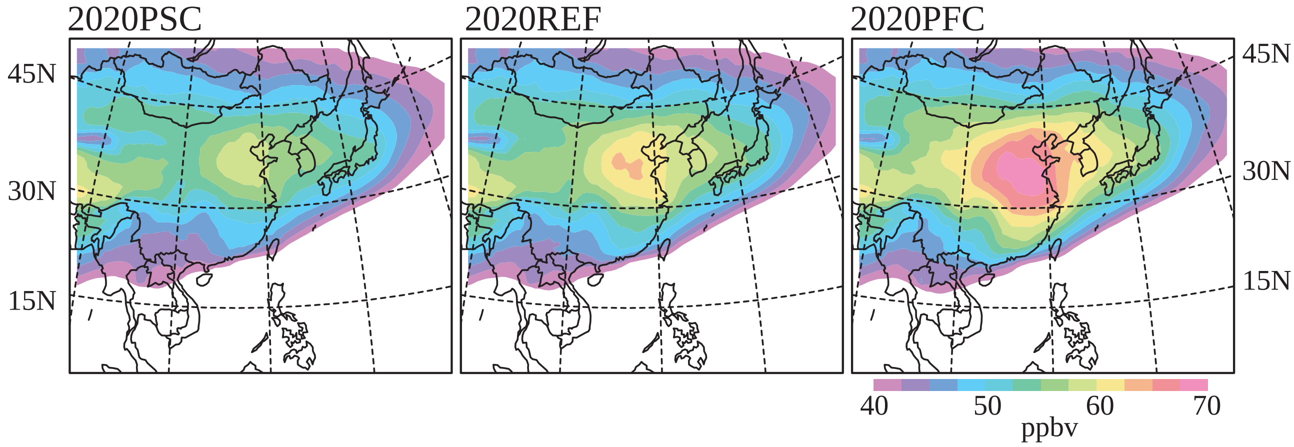

Fig. 8. Spatial distribution of annually averaged O_3 concentrations under three scenarios (PSC, REF, and PFC) from 2020 in the boundary layer (below 2 km).

(REAS) [16] to investigate air quality changes caused by future anthropogenic emissions transitions [28]. This section specifically describes surface ozone (O_3) concentrations (below 2 km) and its critical level. As for the past-present simulation, anthropogenic emissions during 1980-2000 were used under the year-by-year meteorological conditions, which were simulated using the Regional Atmospheric Modeling System (RAMS) [6, 17] with NCEP/NCAR 2.5 degree × 2.5 degree reanalysis data sets with 6-h intervals. The simulated O_3 was able to reproduce observed O_3 concentration levels and daily variations at observation sites in Japan [28]. The future predictions used three types of future emission inventories for 2020 (2020REF, 2020PSC, and 2020PFC) under meteorological conditions for 2000.

Figure 7 illustrates spatial distribution of the growth of annually averaged O_3 concentration for 2000-2020 using each scenario, the 2020PSC (left map), the 2020REF (middle map), and the 2020PFC (right map), respectively. Annually averaged O_3 concentrations under three emission scenarios for 2020 are shown in **Fig. 8**.

According to the model calculation of monthly averaged O_3 concentration in April using the estimated emissions in 2000, it exceeds the environmental standard of 60 ppbv in a wide range of areas around the Sea of Japan including Honshu. The contribution of the regional east Asian emissions was approximately 10-30% [27]. These results mean that photochemically produced O_3 by its precursors emitted from human activities in east Asia has a large impact upon the air environment in Japan.

As for the 2020PSC (left map in **Fig. 7**), O_3 concentration shows only a slight change (-1 to 2 ppbv) over north of 30°N, from central east China to Japan. Especially at a part of central China, annually averaged O_3 concentration decreases by 1 ppbv during 2000-2020, which is mainly caused by a slight NO_X decrease. Meanwhile, O_3 increases (2-5 ppbv) are confirmed over southern China and the Indochinese Peninsula. The highest O_3 area, the North China Plain, is covered with annually averaged O_3 of 57.5-60 ppbv (left map in **Fig. 8**). Most parts of 30-45°N are covered with annually averaged O_3 of 52.5-60 ppbv. Annually averaged O_3 concentration over Japan is below 55 ppbv.

Under the 2020REF (middle map in **Fig. 7**), annually averaged O_3 concentration increases by 5-7 ppbv in southeast China. The O_3 growth is more than 2-5 ppbv from central east China to central Japan. Annually averaged O_3 concentration for the North China Plain reaches 60-65 ppbv in 2020 (middle map in **Fig. 8**). Meanwhile, most parts of Japan excepting Hokkaido and Okinawa, are covered with annually averaged O_3 of 50-57.5 ppbv.

The PFC scenario has high O_3 growth over east Asia during 2000-2020. Considerably rapid O_3 growth with 10-20 ppbv (0.5-1 ppbv yr^{-1}) appears in southeast China (right map in **Fig. 7**). A large part of east Asia, from central east China to southern Japan, is covered with more than 5 ppbv growth. The North China Plain and the Yangtze Delta are covered with annually averaged O_3 of 65-70 ppbv (right map in **Fig. 8**). High O_3 of 60-70 ppbv is confirmed over east China and the Korean Peninsula. Meanwhile, most parts of Japan are covered with annually averaged O_3 of 52.5-60 ppbv.

During 2000-2020, the O_3 growth of 0-2 ppbv for annually averaged O_3 concentration under the PSC is enhanced up to 6-8 ppbv in the Korean Peninsula and up to 2-6 ppbv in Japan under the PFC, but no difference between scenarios is apparent over these areas. These results suggest that future anthropogenic emissions transitions of O_3 precursors over China strongly affect O_3, even over the Korean Peninsula and Japan.

The increase of O_3 concentration during 2000-2020 is estimated as 10-15 ppbv within China, slightly over 4 ppbv in areas from the East China Sea through Kyushu to western Japan, and 3 ppbv in the Kanto region. Paying attention to the oxidant environmental standard of 60 ppbv, even the annually averaged values approach the concentration level on the Sea of Japan side from Kyusyu to western Japan. As for north parts of Kyushu in 2020, the frequencies that exceed the oxidant environmental standard, 60 ppbv, increase from 20% in 2000 to approximately 30% in 2020 and are affected by O_3 growth caused by anthropogenic emissions increases during this period, though the increase of annually averaged values is small.

Figure 9 shows the year-by-year changes of simulated O_3 for a long period, 1980-2003 and future projection. The simulated O_3 concentration increases at the rate of about 0.22 ppbv yr^{-1} during 1980-2003, which corresponds to the observed O_3 growth shown in **Fig. 6**. **Fig. 9** also shows the trends of anthropogenic emissions of NO_X and NMVOC in China. The upward tendency of surface O_3 concentrations in Japan and that of emissions in China are quite similar. It is indicated that the annual averaged concentration of surface O_3 in the North China Plains increased by 1 ppbv with the annual increase of a million ton of the NO_X emissions in China. Consequently, the average concentration in summer increased by about 8 ppbv during 1996-2003. Additionally, the ozone concentration in Japan is also enhanced by the increasing of Chinese emissions and its increasing rate is 30-50% of that in China. This means that the O_3 produced by the increased emissions from the Asian Continent caused the rising of the surface O_3 concentration in Japan by the transboundary air pollution.

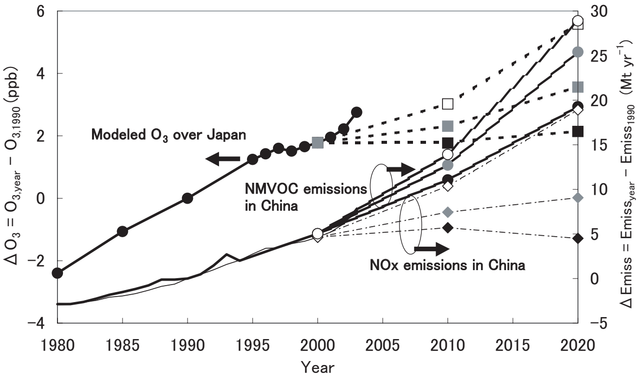

Fig. 9. Changes in surface O_3 concentrations in Japan (calculated) and emissions of NO_X and NMVOC in China.

These O_3 growths impart various influences on ecosystem. The following describes the result of evaluation using accumulated exposure over a threshold of 40 ppbv (AOT40), which sets criteria to influence plants. In this paper, AOT40 is used as in the United Nations Economic Commission for Europe (UN-ECE); AOT40 is calculated as the sum of differences between the hourly averaged O_3 concentration and the threshold value of 40 ppbv for each hour that the averaged O_3 concentration exceeds 40 ppbv [3, 7, 26]. The equation is expressed as

$$\mathrm{AOT40} = \sum_{i=1}^{n} ([\mathrm{O_3}] - 40)_i \quad \text{for } [\mathrm{O_3}] > 40 \text{ ppbv}.$$

AOT40 is currently used as a guideline for an exposure index for O_3 effects on forests and crop yields in Europe [2, 4, 26].

Figure 10 illustrates the AOT40 for forest tree during a 6-month period, from April to September, in 2000 and for 2020REF, 2020PSC, and 2020PFC over east Asia. Over most parts of east Asia, the AOT40 is greater than the critical level (10 ppmh), even in 2000. The AOT40 at northeast China and the Korean Peninsula reaches 50-70 ppmh, which is a factor 5-7 of the critical level (10 ppmh). In Japan, excluding Okinawa and Pacific Ocean areas, the AOT40 is 20-45 ppmh (a factor 2-4.5 times higher than the critical level), which is the same level as the AOT40 at Oki in 1996 and 1997 [18].

For O_3 levels under the 2020 PSC, the AOT40, which is 40-75 ppmh in northeast China and the Korean Peninsula, and 20-45 ppmh in Japan does not show critical increases over most parts of east Asia.

The AOT40 for the 2020 REF increases over central east China and the Korean Peninsula, where the value reaches 55-90 ppmh (a factor 5.5-9 of the critical level). The increase of AOT40 is also confirmed over Japan, where the value reaches 30-50 ppmh (10-25 ppmh at Okinawa and Pacific Ocean areas).

Under the 2020PFC, the AOT40 shows a significant increase over central east China and the Korean Peninsula, where the value reaches 65-115 ppmh. In particular, at

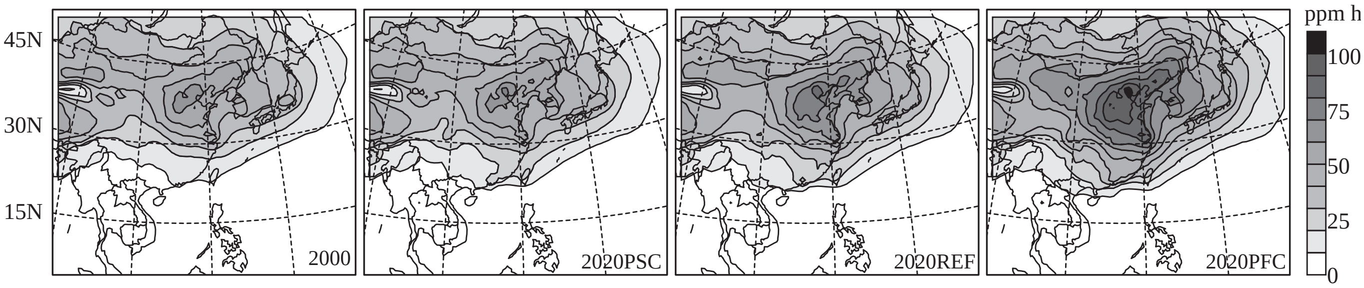

Fig. 10. Spatial distributions of 6-month AOT40 for forest trees using 2000, 2020REF, 2020PSC, and 2020PFC emissions.

the Yellow River Delta and the Yangtze River Delta, the AOT40 reach more than 90 ppmh. Meanwhile, AOT40 increases to 30-60 ppmh in Japan.

Results of studies of O_3 exposures of 16 typical Japan forest species [14] suggest that the critical level of AOT40 of Japanese forest protection imparts a 10% growth loss of forest trees might be in the range 8-21 ppmh [9]. Considering this Japanese critical level for AOT40, the AOT40 does not exceed the critical level only for those areas over a part of Hokkaido, Okinawa, and Pacific Ocean areas. However, most parts of Japan are covered with the AOT40 more than the critical level even in 2000. Under the 2020REF and the 2020PFC, the AOT40 vastly exceeds the Japanese critical level over most of Japan, except Okinawa and Pacific Ocean areas.

6. Conclusions

Trends of Asian air pollution in the last decade have been studied based on air quality monitoring station data, satellite retrieval data (GOME NO_2), and a regional scale chemical transport model (CTM) simulation. A newly developed annual Asian-scale emission inventory (REAS) for 1980-2003 was used in observation data analysis and CTM. Especially, systematic analyses of interannual trends and seasonal variations of tropospheric NO_2 vertical column densities (VCDs) based on GOME satellite data and the regional scale CTM and CMAQ were examined over east Asia during January 1996 to June 2003. Analyses of recent trends of annual emissions in China by REAS and satellite GOME NO_2 show trends of increase of 8-10% after 2000; this rapid increase suggests a strong impact of long-range transport of secondary air pollutants to downwind regions/countries. Detailed analyses of O_3 observation data in Japan suggest the increase of annual averaged O_3 concentration of 2% yr^{-1} because of this long-range transport.

The analysis by CMAQ model captured the observed O_3 concentration levels and daily variations in 2000 at observation sites in Japan. We also extended our regional air quality study for targeting 2020. Regarding future projections for 2020, REAS has provided three types of emissions data based on three emissions scenarios of REF, PSC, and PFC for China and emissions data based on REF for other countries. The projected PSC NO_X emission caused a slight decrease (-1%) during 2000-2020 in China, but the other scenarios imparted a large increase (39% for the REF and 128% for the PFC). The monthly averaged O_3 concentration in summer in 2000 was considerably high over northeast Asia: 70-80 ppbv in June and 65-75 ppbv in August. Projected 2020REF emissions enhanced the O_3 concentration there to 75-90 ppbv in June and 75-85 ppbv in August. Annually averaged O_3 concentrations increased by approximately 5 ppbv because of changes in anthropogenic emission for 2000-2020 (REF) over northeast Asia including central and east China, the Korean Peninsula, and parts of Japan. On the other hand, the projected PFC emissions imply an increase of monthly averaged O_3 of more than 20 ppbv (1 ppbv yr^{-1} growth) in the North China Plain. Surface O_3 under the PFC scenario is also enhanced by 6-8 ppbv over the Korean Peninsula and by 2-6 ppbv in Japan during 2000-2020 in spite of the reduction of NO_X emission in Japan. Air quality in Asia has possibly already be in the a critical level from the perspective of pollution standards.

Acknowledgements

The authors thank Mr. Y.-J. He of Kyushu University and Mr. M. Katayama of NIES for their help of numerical simulation and data analysis. The authors also wish to thank to Dr. N. Horii of Kyushu University and Dr. T. Hayasaka of RIHN for the development of emission inventory of this paper. This study was supported in part by the funds from a Grant in Aid for Scientific Research Grant No. 15310011 from the Ministry of Education, Culture, Sports, Science and Technology of Japan, the Global Environment Research Fund (B-051) from Ministry of Environment Japan, and the Collaborative Research Program of Research Institute for Applied Mechanics, Kyushu University.

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