# Changing Trends in Sulfur Emissions in Asia: Implications for Acid Deposition, Air Pollution, and Climate

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**In the early 1990s, it was projected that annual $SO_2$ emissions in Asia might grow to 80-110 Tg $yr^{-1}$ by 2020. Based on new high-resolution estimates from 1975-2000, we calculate that $SO_2$ emissions in Asia might grow only to 40-45 Tg $yr^{-1}$ by 2020. The main reason for this lower estimate is a decline of $SO_2$ emissions from 1995 to 2000 in China, which emits about two-thirds of Asian $SO_2$. The decline was due to a reduction in industrial coal use, a slow-down of the Chinese economy, and the closure of small and inefficient plants, among other reasons. One effect of the reduction in $SO_2$ emissions in China has been a reduction in acid deposition, not only in China but also in Japan. Reductions should also improve visibility and reduce health problems. $SO_2$ emission reductions may increase global warming, but this warming effect could be partially offset by reductions in the emissions of black carbon. How $SO_2$ emissions in the region change in the coming decades will depend on many competing factors (economic growth, pollution control laws, etc.) However a continuation of current trends would result in sulfur emissions lower than any IPCC forecasts.**

Estimating past, present and future levels of sulfur deposition, and ambient levels of $SO_2$ and sulfate aerosol is central to the evaluation of risks to ecosystems and human health, and net changes in radiative forcing and resulting changes in climate. This is particularly true in Asia where the pressing environmental problems of urban pollution, acid deposition and climate change are intimately linked to sulfur (*1*). Over the last 25 years the primary energy demand in Asia has grown at a pace twice as fast as the world average (*2*). Presently about 80% of the energy demand in Asia is satisfied by fossil fuels, with coal being the primary energy source. Energy scenarios out to 2020 are characterized by a further increase in energy use, with fossil fuels remaining the dominant source (*3*). The demand for coal and oil is expected to double or triple in the next thirty years. As a consequence, the conventional wisdom is that current sulfur emissions throughout the continent, already nearly equal to those from Europe and North America combined (*4*), will continue to increase in the coming decades (*2*). This projected growth in emissions has led to increased scientific interest and political concern regarding the fate of Asian emissions, because countries are receiving increasing amounts of pollutants from neighboring and even distant countries (*5*). However, patterns of energy use in Asia are changing quickly, and the forecasts of continued growth in S-emissions hold true in some countries, but are outdated in others.

We estimate (*6*) (as shown in Fig. 1) that $SO_2$ emissions in Asia grew from 17.1 Tg in 1975 to 38.5 Tg in 1995, an annual-average growth rate of 4.1%. This increase was driven by expansion of the economies of Asia, particularly the coal-based industrial economy of China, which contributes about two-thirds of Asian $SO_2$ emissions. During this 20-year period, only a few countries—principally, Japan, Taiwan, and the Republic of Korea—took effective steps to abate emissions of $SO_2$ by installing flue-gas desulfurization technology or adopting similar measures.

In the early 1990s, several countries began to appreciate the dangers of uncontrolled sulfur emissions and instituted modest abatement measures to limit the sulfur content of petroleum products. In addition, China began to restrict its use of high-sulfur coal, with a resulting reduction in the average sulfur content of coal from about 1.31% in 1990 to 1.12% in 1995. These factors led to a deceleration in the growth of Asian $SO_2$ emissions between 1990 and 1995 (*4*).

Since 1995, we estimate that Asian $SO_2$ emissions have actually declined, from 38.5 Tg in 1995 to 34.4 Tg in 2000, a decrease of 2.3% per year. This remarkable change is almost entirely due to a reduction in $SO_2$ emissions in China brought about by several factors: a marked reduction in industrial coal use from the closure of small and inefficient plants, a slow-down in the Chinese economy, the improved efficiency of energy use, the closure of some high-sulfur coal mines, a general reform in industry and power generation, and a rising awareness of the dangers of air pollution (*7*). Also, in January 1998, China instituted an $SO_2$ reduction program—known as the "two-control-zone" policy—to further abate emissions in industrial and environmentally sensitive regions (*8*) and protect human health in Chinese cities (*9*). We estimate that China's $SO_2$ emissions dropped from 25.7 Tg in 1995 to 20.8 Tg in 2000 (*10*). The economic recession in Asia in 1997-1998 also contributed to lower emissions in many other East Asian and Southeast Asian countries.

In contrast, emissions on the Indian subcontinent and in Southeast Asia continue to increase. To meet increasing electricity demand in India in the coming decade, several hundred new power stations need to be constructed (*11*). With a continued growth of 6-7 percent annually, coal is considered to remain the primary fuel for power generation. In particular, we estimate that emissions in India reached 5.5 Tg in 2000, with no signs of abatement. Because Indian soils are not particularly susceptible to acid deposition, the acid-rain issue has not been embraced in India; only recently have steps been taken to reduce the sulfur content of petroleum products in Delhi and the Taj Trapezium (to protect the Taj Mahal). In Southeast Asia, the developing economies still rely heavily on coal and fuel oil to boost industrial production. Only in the more developed countries (e.g., Singapore and most recently Thailand) have sulfur emission controls been implemented.

Fig. 2 presents a longer-term perspective on $SO_2$ emission trends, including a view into the future. The RAINS-Asia model (*2*) forecasts that the historic trend will continue in the absence of control measures. But with legislation put in place in recent years the growth in $SO_2$ emissions should be decreased somewhat. The maximum feasible reduction case shows just how low emissions could be reduced if aggressive environmental policies would be introduced, albeit at an estimated annual cost of about \$100 billion in 2030, with China's annual costs reaching \$38 billion (*12*). However, current estimates of ecosystem damage and human health effects related to air pollution in China exceed \$45 billion annually (*9*), and the net benefits associated with reducing emissions are being used to help shape policy. One striking conclusion from Figure 2 is that recent trends suggest that the IPCC range of possible future $SO_2$ emissions for Asia (*3, 13*), though reduced from previous forecasts, is still too high. Just exactly where sulfur emissions will go beyond 2000 depends on the outcome of the competition between renewed industrial growth and the desire for environmental protection, once the cheaper control measures have been exhausted. However, a continuation of the current trends would result in $SO_2$ emissions lower than any IPCC forecast.

Using these estimated emissions we calculated the trends in the concentrations of sulfur dioxide, sulfate aerosol, and sulfur deposition (wet, dry, and total) for the period 1975-2000 using the ATMOS model (*14*) driven by NOAA-NCEP Reanalysis meteorological data. Over the period 1975-2000 the total deposition of sulfur increased substantially throughout most of Asia (Fig. 3). The regions of high sulfur deposition (>0.5 g(S) $m^2$ $yr^{-1}$) (*15*) first appeared in Asia in the 1960s (*16*) and by 1975 had spread throughout the industrial regions of Japan, South Korea, and the Sichuan/Chongqing area of China. From 1975-2000 the regions of elevated sulfur deposition expanded dramatically throughout eastern China, and to large areas of Southeast and South Asia (now covering over 5.5 million $km^2$) (*17*). Although not shown, the spatial patterns and trends of ambient $SO_2$ and sulfate concentrations are similar to those for S-deposition.

The changes in sulfur deposition exhibit dramatic regional differences. The largest increases have occurred in Southeast Asia and the Indian subcontinent, where sulfur emissions in the 1970s and early 1980s were relatively low and due primarily to the use of biofuels associated with cooking and coal in the industrial and power sectors (*18*). Along with industrial growth in the 1980s and 1990s came increased reliance on fossil fuels (largely coal) and an attendant increase in sulfur emissions and deposition. Sub-regional features are also reflected; for example, the impact of the development of the large power generation complex in northern Thailand (Mae Moh) in the late 1980s on sulfur deposition in Southeast Asia is clearly depicted.

When viewed over the entire 25-year period (Fig. 3b), only Japan shows a net decrease in sulfur emissions and deposition. However, the efforts in the 1990s in East Asia to reduce sulfur emissions have had a marked impact on sulfur trends, as illustrated by observed ambient $SO_2$ levels in Chinese cities (Fig. 4). In Shanghai (*19*) sulfur emissions grew from 400 Gg in 1989 to a peak of over 500 Gg in 1996. Ambient $SO_2$ levels within the city began to decline in the early 1990s due to the factors discussed previously, and due to transformations of the Shanghai urban environment associated with the rapid growth in the region, to the closure of small industrial coal plants, and to the use of tall stacks. In Hong Kong, ambient $SO_2$ levels track more closely the behavior of $SO_2$ emissions. The impact of the reduction in the S-content of liquid fuels mandated in 1994 is shown in the decrease in ambient sulfur dioxide levels (*19*).

Trends in sulfur deposition within a country are not a simple function of national S-emissions as illustrated in Fig. 5, where the role of trans-boundary transport of sulfur is presented. In the case of Japan, sulfur deposition decreased from 1975-85 in response to reductions in domestic S-emissions. While Japanese emissions were level between 1985 and 1995, sulfur deposition in Japan increased due to the growth in Chinese S-emissions (the share of China's contribution to S-deposition in Japan doubled from ~20% in 1975 to nearly 40% in 1995). These results illustrate the importance to the region of reducing S-emissions in China. The decrease in China's emissions between 1995 and 2000 is shown to have had a greater impact on S-deposition in Japan than all of Japan's efforts over the last quarter of a century to reduce its domestic sulfur emissions. (For reference, China's S-emissions in 2000 are estimated to be 20 times greater than Japan's.)

The situation in Southeast Asia is different, and here S-deposition has continued to increase, as illustrated in the Malaysian sulfur deposition trends. In Malaysia sulfur deposition increased from 1975 to 1995 due to growth in domestic sources, and large increases in emissions from neighboring Singapore. Sulfur emissions decreased in Singapore after 1995, yet sulfur deposition in Malaysia continued to grow due to a progressive increase in the domestic sources and from sulfur emissions associated with the expanding ship traffic in the Strait of Malacca (*20*).

Sulfate levels in precipitation samples in Japan are beginning to show a decrease (*21*). However, the link between trends in S-emissions and deposition is confounded by the fact that interannual variability in meteorological conditions can result in annual variations in sulfur deposition that are larger than those due to annual changes in emissions, and they can either enhance or mask the trends due to emissions. This is illustrated in Fig. 4 (lower panel), where we de-trended S-deposition from emissions by repeating the calculation with emissions held constant at 1990 levels. Years with low precipitation correspond to decreases in local S-deposition, as shown in the observed sulfur deposition in Hong Kong in 1991. On a larger scale, meteorological conditions in China over the last decade (*22*) have favored a decrease in local deposition and an increase in the export of sulfur deposition away from China (and towards Japan as the result of the predominate westerlies). Thus meteorological factors in the 1990s produced trends in S-deposition over China that have both masked the impacts of increasing S-emissions (as during the period 1990-96), and during 1996-2000 exagerated the effectiveness of emission reductions. Over Japan these meteorological factors have tended to increase sulfur deposition, thus exacerbating the influence of China's emissions on S-deposition over Japan. Whether this decadal trend in climatic factors that favor an increased transport distance of sulfur in East Asia reflects a regional climate feedback involving the high aerosol loadings in eastern China remains an open but important question (*23*). These results point out the need for long-term monitoring in the region designed specifically to detect changes in Asian emissions, and the challenges of interpreting observed sulfur trends.

***Discussion***

In the early 1990s the outlook for the state of the environment in Asia, based on forecasted economic growth and the environmental protection laws and practices in place at that time, appeared bleak. It was projected that $SO_2$ emissions in Asia might grow to 80-110 Tg $yr^{-1}$ by 2020 (*24*). However, our analysis of sulfur emissions in Asia from 1975-2000 suggests that

Asian sulfur emissions will not follow these earlier projections. Already countries in East Asia and Southeast Asia have made progress in reducing the sulfur contents of their fuels and modifying their coal-use practices. For these reasons the projections of continued rapid growth of sulfur emissions in Asia hold true in some countries (e.g., those in South Asia) but are outdated in others. What we have shown (and what we expect to continue to see) for Asia is a transition of countries (or sub-regions) from conditions where sulfur emissions are largely unabated and grow proportionally to energy consumption, to situations where, through a variety of actions (such as fuel switching, reduction in sulfur contents of fuels used, energy efficiency measures, or use of pollution control technologies), sulfur emissions become decoupled from energy consumption. Today, we are optimistic that emissions will peak before 2020 in the range of 40-45 Tg $yr^{-1}$, and even lower if current trends are maintained. This is good news for the environment.

Whether this recent trend of lower $SO_2$ emissions in Asia as a whole will continue in the future will depend on many factors, but particularly on whether China can sustain a lower rate of emissions once the economy rebounds and the "easy" control measures are exhausted. Factors that will help to hold down emissions in the future include rapid growth in the use of natural gas as a substitute for coal and a spreading awareness of the dangers of sulfur pollution to the environment. Recent initiatives in China, including changes to the Air Pollution Control laws, the setting of S-emission targets, and sulfur emission trading initiatives, are encouraging signs for continued optimism for lower sulfur emissions in the region (*25*).

This more optimistic view of future Asian sulfur emissions must be tempered by the fact that emissions in Southeast and South (particularly India) Asia continue to increase rapidly, and regional problems may intensify. Furthermore, while $SO_2$ has been the main contributor to past air pollution problems in Asia, nitrogen oxides and non-methane hydrocarbon emissions are now growing much more rapidly than sulfur (*26*) (though they too are growing at slower rates than anticipated) and smog-related problems are becoming more important.

It should be noted that air pollution issues related to human health and ecosystem protection, coupled with economic issues such as a desire to improve industrial productivity and the effects of recession, have driven the trends in S-emissions in Asia. Climate change concerns have not played a major role. But clearly these changing trends in S-emissions have important climate implications. Ironically, decreases in S-emissions result in decreases in aerosol sulfate levels and hence an increase in global warming, because sulfate tends to cool the atmosphere. We have estimated the change in long-term global temperature due to the reduction in S-emissions in two ways. Method-A uses a climate model *(32)* that calculates temperature for specified constituent changes. Method-B multiplies an estimated global climate forcing for each constituent *(31)* by the fractional change due to Chinese emission changes and by an assumed global climate sensitivity of 3/4 C per $W/m^2$. These calculations yield temperature changes due to the decrease in China's S-emissions to be in the range of (+0.03 - +0.04)K (Table 1). However, emissions of carbon dioxide, methane, and black carbon (BC) have also changed in China in recent years, as a result of the same driving forces that have led to the decrease in S-emissions (*27, 28*). These are key greenhouse constituents, and their reduction may mitigate some of the warming due to S-emission reductions. We also estimated the effect of changes in the emissions of these species in China from 1995-2000. The decrease in BC emissions in China over the same period is estimated to off-set between 65-100% of the warming associated with the reduction in S-emissions. Reduction of BC emissions to counter warming due to S-emission reductions is an important element in a proposed "alternative scenario" aimed at minimizing anthropogenic climate change *(30).* The estimated *net* effect of the recent changes in China's emissions on global warming range from –0.0053K to +0.012K; underscoring the fact that due to uncertainties in these estimates we are not yet able to anticipate the sign of the net temperature change.

The future of S-emissions in Asia has a large impact on global temperatures. If China succeeds in reducing its S-emissions to its target of 10 Tg/yr by 2010, then Asian S-emissions could stabilize near a lower limit of ~20 Tg/yr. As discussed earlier, we estimate an upper limit for S-emissions to be ~ 45 Tg/yr by 2025. Using the temperature response function from Table 1, this range in future Asian S-emissions implies a global temperature change sensitivity of ~ 0.2K ($\Delta T$ = -0.16 for a 2025 S-emissions of 45 Tg/yr and $\Delta T$ = +0.04K for a 2025 emissions of 20 Tg/yr). Clearly it is important that we quantify Asian S-emissions as they evolve over the next few decades.

Finally, these results illustrate that concerns regarding air pollution, human health and industrial productivity can be powerful forces in Asia, resulting in simultaneous reductions in local air pollution and greenhouse-gas emissions. Policies that acknowledge these recent emission trends and that assist China and other Asian countries in their efforts to reduce pollution emissions, offer an opportunity to further reduce the impacts of local air pollution, regional acid deposition, and global warming.

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transboundary pollution is discussed in D. G. Streets, G. R. Carmichael, M. Amann, R. Arndt, *Ambio* **28**, 135 (1999).

6. Detailed trends for 1985-1997 were developed from the RAINS-Asia model (*2*) and the IEA Energy Statistics and Balances database (*4*). These were extended to the period 1975-2000 using the same historical database and economic growth factors for recent years. Trends are consistent with other estimates for shorter time periods, e.g., H. Akimoto, H. Narita, *Atmos. Environ*. **28**, 213 (1994) and N. Kato, *Atmos. Environ*. **30**, 757 (1996).
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9. In 1999, among the 338 cities surveyed by the State Environmental Protection Administration (SEPA) of China, only one-third of the cities met the Class II air quality standards. 60% of the cities exceeded China's Class II daily and annual TSP standards, and nearly 30% of the cities exceeded the Class II $SO_2$ standards (*Report on the State of the Environment in China 1999*, http://www.sepaeic.gov.cn/english/99gb/index.htm). The total mortality and morbidity loses are estimated to be $32 billion annually. Furthermore, SEPA estimates that economic losses due to acid rain damage to forests and farmland are $13.25 billion annually (T. M. Johnson, F. Liu, R. S. Newfarmer, *Clear Water, Blue Skies: China's Environment in the New Century*, World Bank, Washington DC, 1997).
10. China's State Environmental Protection Administration (SEPA) reports an even more striking decline in $SO_2$ emissions, from 23.7 Tg in 1995 to 18.6 Tg in 1999 (*8*). See also www.sepaeic.gov.cn/english/98gb/index.htm;www.zhb.gov.cn/bulletin/99gb.php3?category=4 (in Chinese). Our emissions are slightly higher in absolute terms because of the inclusion of some source categories not normally reported in Chinese statistics and other factors.
11. *Global Energy – India,* World Energy Council Special Report (FIRST Magazine, London, UK, 2001).
12. Costs were estimated using the RAINS-Asia Model (*2*). Emission-control and associated costs are based on international free market experience with operating pollution control equipment and extrapolating costs to the specific country context.
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Industry, Japan, 1992). Total volcanic emissions within the domain amount to 3.33 Tg $SO_2$ $yr^{-1}$, and were held constant throughout the period studied.

15. What constitutes a high sulfur deposition level from the perspective of ecosystem impact is a complicated and relative matter and involves the consideration of underlying soil chemistry and the type of vegetation, along with what other materials are deposited in addition to the sulfur (e.g., wind-blown soils, which would have a certain capacity to buffer the sulfur acidity). Highly sensitive soils/vegetation may experience adverse effects at sulfur deposition values lower than 0.5 g (S) $m^2$ $yr^{-1}$, while insensitive soils/vegetation can withstand much higher levels. The value of 0.5 has proven to be somewhat robust in generally correlating with adverse effects in areas where windblown dust is of minor importance.
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25. China's air pollution laws and enforcement activities are changing. The two major pollutants in China are sulfur dioxide and soot, both emitted during coal burning. On September 1, 2000, China again amended its Air Pollution Control law. The new law aims to reduce $SO_2$ emissions from 18.6 million tons in 1999 to 10 million tons by 2010. It also requires the phase-out of dirty coal and provides incentives for the use of low-sulfur, low-ash coal in medium and large cities. New and expanding power plants in these cities must also install sulfur- and particulate-removing equipment. Under the new law, emissions from vehicles, ships, domestic heating and cooking, and construction are controlled as well. In addition, China is considering instituting a sulfur emissions trading system (*2*).
26. Z. Klimont et al., Water Air and Soil Pollution, in presss(2001). J. A. van Aardenne *et al*., *Atmos. Environ*. **33**, 633 (1999).
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33. This research was supported in part by RAINS-Asia Phase II Project of the World Bank.

**Table 1. Estimated impact of changes in China's emissions between 1995 and 2000 on long-term global temperatures.**

| 1. Species | 2. China Emissions in 2000[1] (Tg) | 3. Change in China's emissions (2000–1995)[1] (Tg) | 4. Global anthropogenic emissions ca~1995 (Tg) | 5. Change in China's emissions as % of total anthropogenic emissions (%) | 6. Constituent Global Forcing (CGF)[2] ($W/m^2$) | 7. Constituent Temperature Response Function (CTRF)[3] (K/Tg) | 8. Estimated Change in global temperatures due to changes in China's emissions between 1995 and 2000[5] (K) **method-A[4]** | **method-B[2]** |
|---|---|---|---|---|---|---|---|---|
| $SO_2$ | 20.8 | -4.95 | 134 | -3.7% | -1.0 | $-8.2\ 10^{-3}$ | +0.040 | +0.028 |
| BC | 0.91 | -0.43 | 5.1 | -8.4% | 0.5 | $6.5\ 10^{-2}$ | -0.026 | -0.031 |
| $CO_2$ | 3,218 | -109 | 29,700 | -0.37% | 1.4 | $3.0\ 10^{-5}$ | -0.003 | -0.004 |
| $CH_4$ | 33.4 | +1.16 | 360 | +0.32% | 0.7 | $7.5\ 10^{-4}$ | +0.001 | +0.0017 |
| | | | | | | **Net change:** | **Σ = +0.012** | **-0.0053** |

[1] from Streets et al. *(28)*

[2] An alternative estimate of global temperature change due to changes in China's emissions can be obtained by multiplying the constituent global climate forcing (CGF), defined as the global climate forcing ($W/m^2$) due to total anthropogenic emissions of that constituent species, by the Chinese fraction of these emissions and climate sensitivity (C per $W/m^2$). CGFs in column 6 are from Hansen and Sato *(31)*. Climate sensitivity is assumed to be 3/4 K per $W/m^2$. Resulting equilibrium global temperature changes are given under method-B in column 8.

[3] The constituent temperature response function (CTRF) is defined as the change in temperature per unit emissions was calculated as follows *(32)*. For $CO_2$ and $CH_4$, equilibrium global simulations in which the anthropogenic ambient loading of each substance was and was not included, were run. Each resulting difference in temperature was divided by the respective difference in anthropogenic emissions. For $CO_2$, reducing emissions reduces the anthropogenic loading completely only after 100 years, so the full temperature response given is valid only after 100 years. For each constituent BC and $SO_2$, time-dependent global simulations with and without anthropogenic emissions were run. Once temperatures stabilized, the difference in temperature from the two simulations with and without emissions of the component was divided by the emission rate. Negative sign implies that temperature increases with a decrease in emissions.

[4] To estimate the change in long term global temperatures due to changes in China's emissions the temperature response function values in column 7 are multiplied by the change in China emissions (i.e., the values in column 3). Resulting equilibrium global temperature changes are given under method-A in column 8.

[5] Associated uncertainties are not shown, but are significant. The uncertainty in the climate sensitivities are of the order +/- 33%, and that associated with the CTRFs and CGFs are even larger. A further discussion of the uncertainties can be found in *(31)*.

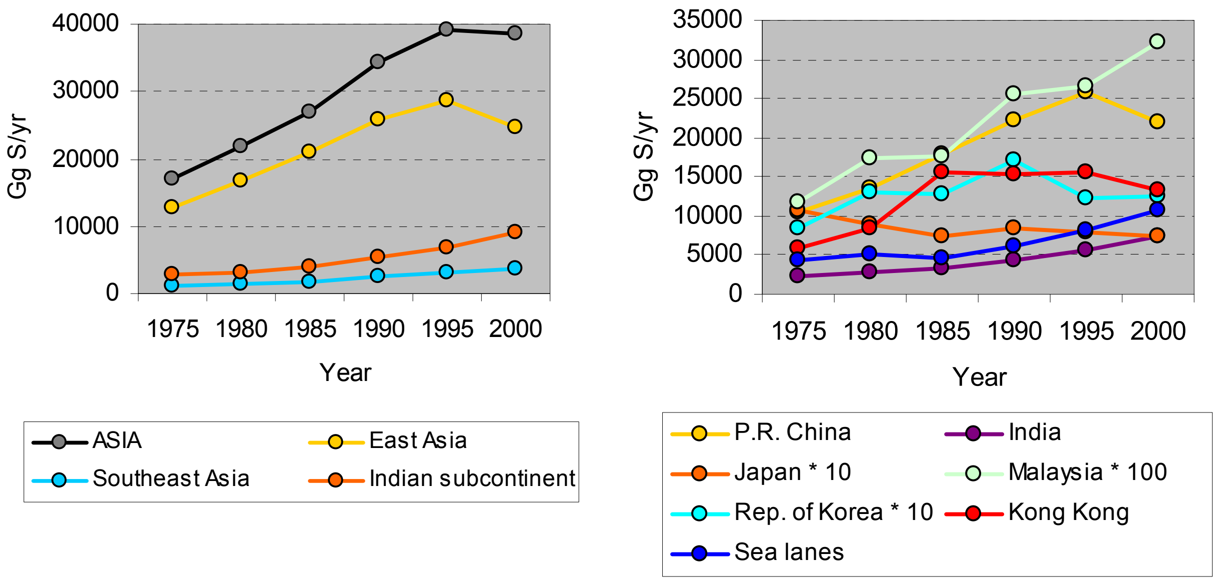



**Fig. 1**. Estimated Asia-wide annual emissions of $SO_2$ ($Gg\text{-}SO_2\ yr^{-1}$) during the period 1975-2000. Shown are the emissions for sub-regions and for selected countries. Sea lanes refer to emissions due to ships.

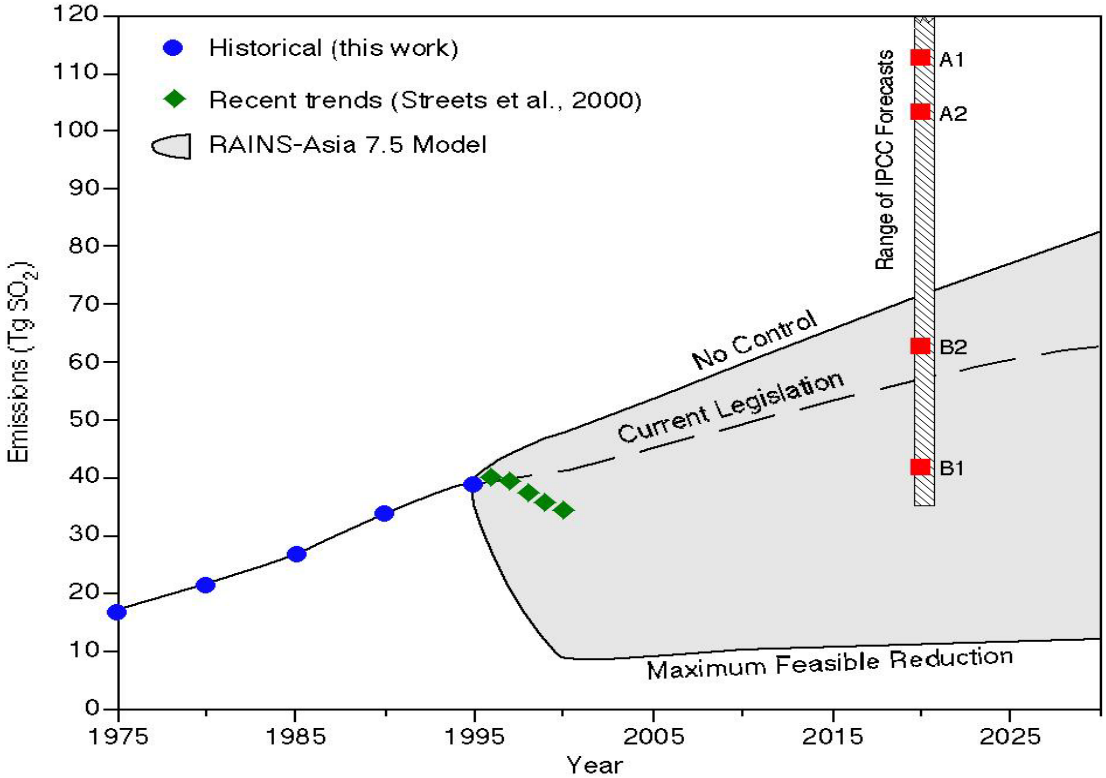
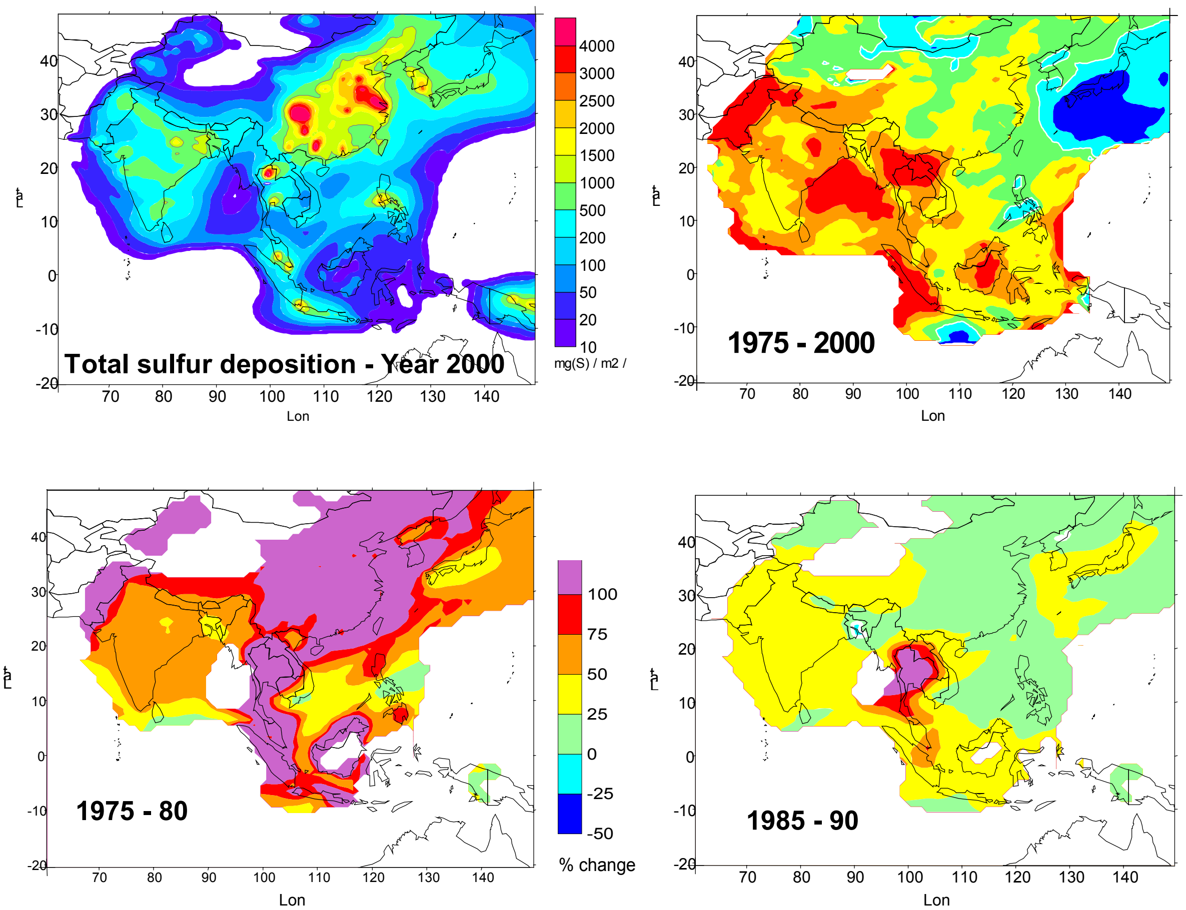



**Fig. 2**. Estimated long-term $SO_2$ Emission Trends for Asia, 1975-2030. Also shown on Fig. 2 is the range of forecasts of 2020 $SO_2$ emissions recently made by the Intergovernmental Panel on Climate Change (*3*). The range is extremely large, driven by the wide range of possible futures explored by the IPCC; specially identified on this figure are the four chief "marker" scenarios (A1, A2, B1, and B2) developed by the IPCC. The Maximum Feasible Reduction curve shows S-emissions under a case where best available technologies are applied to all sources (new and old), and with most optimistic assumptions about the degree of fuel switching and energy efficiency achievable.

a) b)




c) d)

**Fig. 3**. a) Total sulfur deposition (mg (S) $m^{-2}$ $yr^{-1}$) for 2000 computed using year-specific emissions and meteorology; b) relative change of annual total sulfur deposition in 1975-2000 period, computed with year-specific meteorology; c-d) relative change of annual total sulfur deposition in 1975-80, and 1985-1990 due to changes in S-emissions only, computed using estimated year-specific S-emissions but 1990 meteorology.

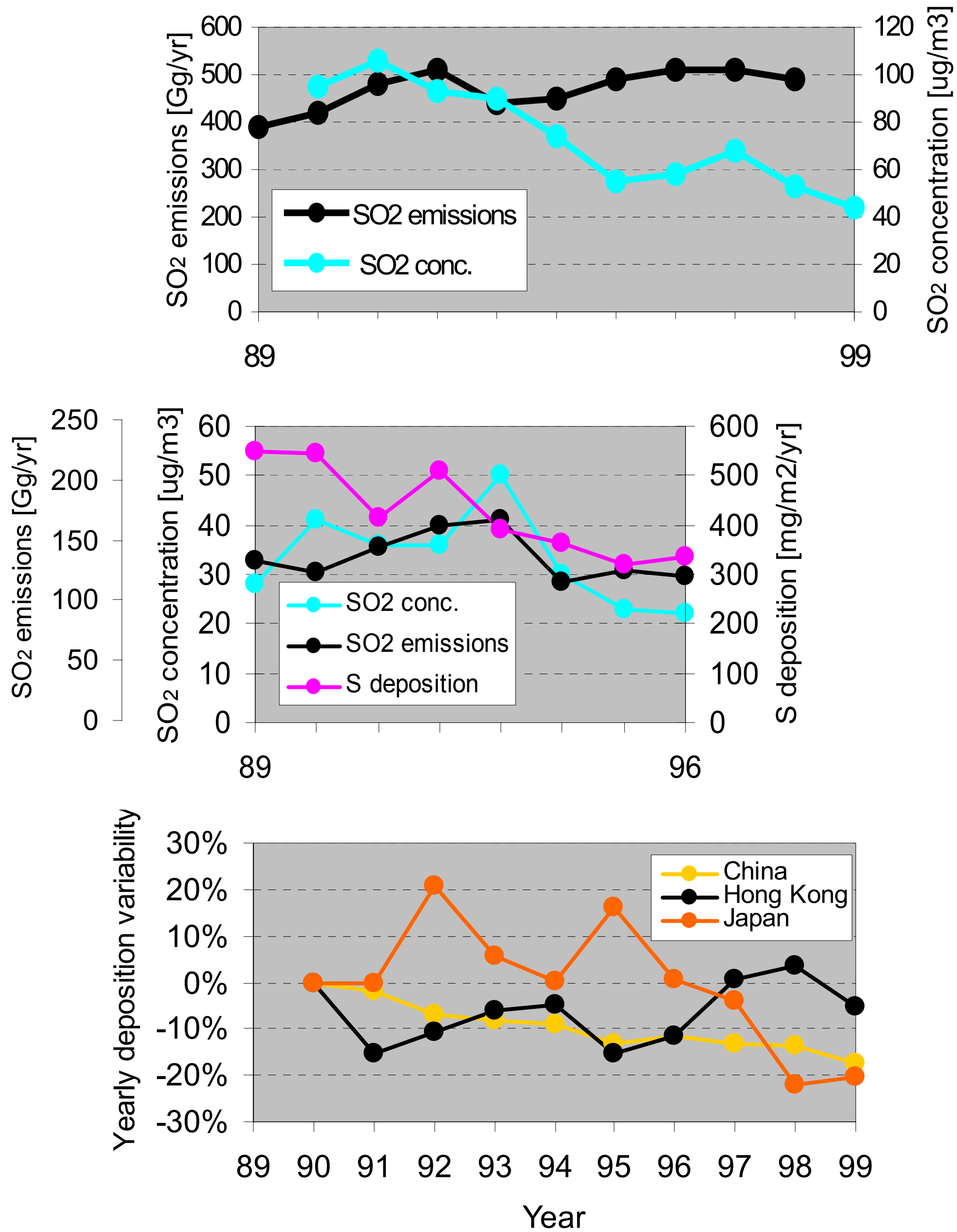



**Fig. 4**. Multi-year sulfur trends. Upper panel - reported $SO_2$ annual emissions and measured city-wide annual averaged ambient $SO_2$ concentrations in Shanghai. Middle panel - annual $SO_2$ emissions, measured $SO_2$ yearly average concentration, and observed total sulfur deposition in Hong Kong (*19*). Lower panel: modeled inter-annual variability of total sulfur deposition, relative to year 1990 calculated by fixing S-emission levels to those in 1990 for all years but using year-specific meteorological fields.

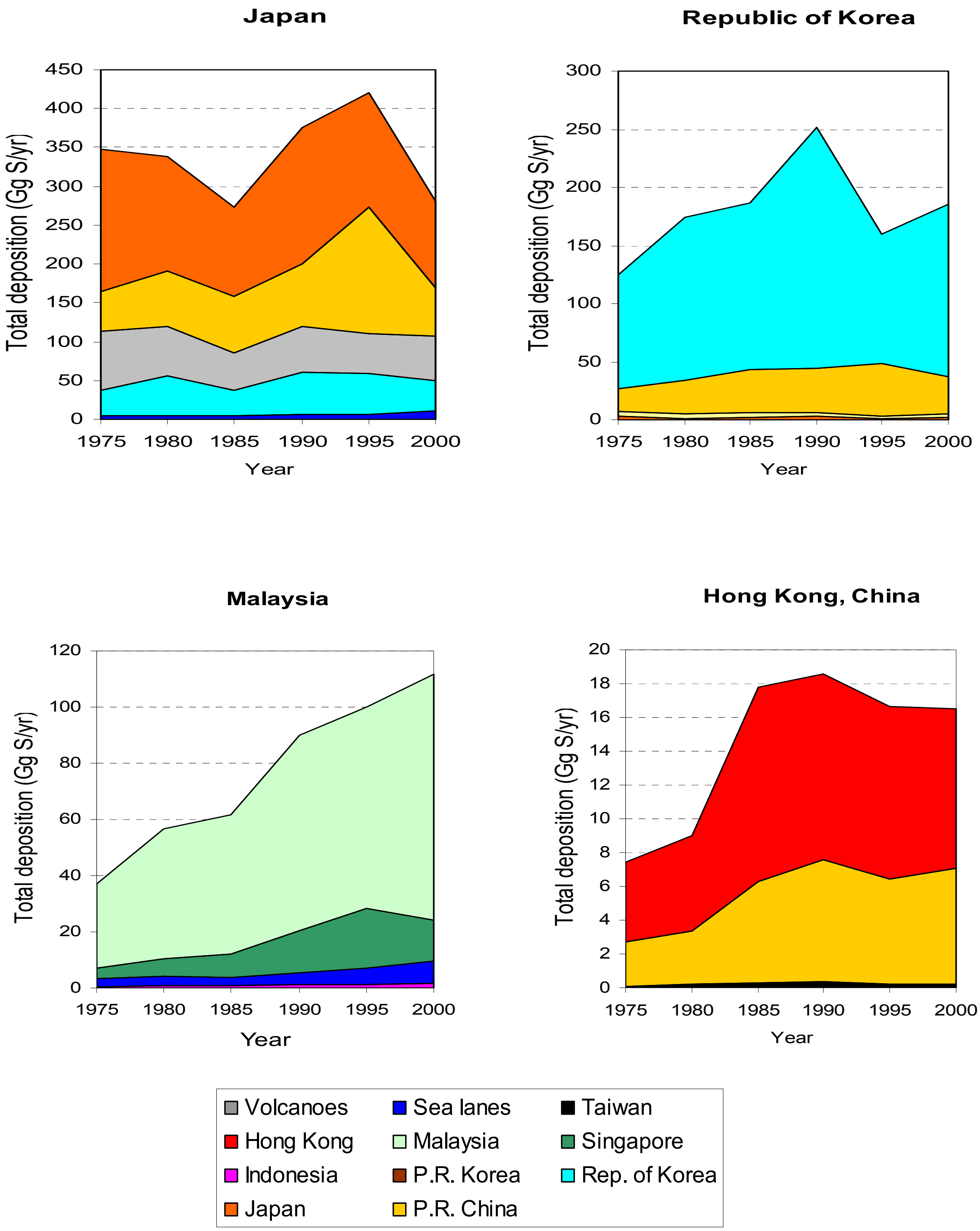



**Fig. 5.** Calculated annual total (wet and dry) sulfur deposition for selected countries from 1975 to 2000, using year-specific emissions and meteorological fields. Also shown are the *source-receptor* relationships that indicate where the sulfur deposited was originally emitted. The source contributions are coded by color (e.g., dark-yellow depicts sources from China). Emission trends are shown in **Fig. 1**.