3. Effects of Atmospheric Environment on Vegetation

The Effects of Air Pollution and Acid Precipitation on Vegetation in China —Past, Present and Future—

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Abstract

China has been experiencing a rapid economic development since the beginning of 1980s. This rapid expansion in the economy and in the population has resulted in a growing demand for energy. In China, more than 75% of its primary energy is domestic coal. This, with the rapid increasing vehicle numbers, has been resulting a large air pollutant (SO_2, NO_x and CO_2) emission. The ambient air pollution has already reached an alarming level in many cities and industrial areas. The air pollution problem is so severe that it is believed to be responsible for more than 1 million deaths per year across the country and the total economic loss resulting from acid rain and SO_2 was equivalent to 2% of the gross national product in 1995. As the economy continuously increase, the amount and structure of energy consumption are predicted to change as well. These changes are very likely going to cause changes in the ambient air pollutant levels and compositions, sequentially it is expected that the impacts of air pollution on vegetation will be different from present. This paper gives a brief introduction of the air pollution, acid precipitation and their effects on vegetation in China in the past and present, the same time gives suggestions for the future research in this field in China.

Key Words:* *Acid precipitation, Air pollution, China, Effect, Fluorides, NOx, O_3, Particulate, SO_2, Vegetation

Introduction

China, a country with a population of 1.22 billion people - one fifth of the world's population, has undergone a rapid economic development over the last two decades. This rapid expansion in the economy and in the population has inevitably resulted in a growing demand for energy. In comparison with most of the developed countries, China has a higher rate of increase in energy consumption (mainly coal) but is short of efficient air pollution control measures. This has resulted a greater air pollutant (SO_2, NO_x and CO_2) emission than the average rate of increase in developed countries (Kato, 1996). The rise in ambient air pollution levels has already caused an alarming deterioration in the air quality in many Chinese cities. Indeed, a recent report indicates that five of the world's 10 worst cities for air pollution are located in China (Yunus *et al.*, 1996) and the air pollution problem is so severe that it is believed to be responsible for more than 1 million deaths per year across the country (i.e. about one in every eight deaths nation-wide; Florig, 1997). Moreover, it has been estimated that in 1995, the total economic losses resulting from acid rain and SO_2 amounted to 110×10^9 Chinese Yuan (13×10^9 US$) - equivalent to 2% of the gross national product (Editing Committee of China Environmental Yearbook [ECCEY], 1998).

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Air pollutants emission and concentration

In China, more than 75% of primary energy utilisation is supplied by domestic coal. The total coal consumption in 1990 was 1052 million tons and in 1995 reached 1280 million tons. It is estimated that by the year 2000, total coal consumption will possibly reach 1500 million tons per year (ECCEY, 1998). Most of the coal used to supply energy has high sulphur and dust content - the national average sulphur content is 1.35% and in some provinces, such as Sichuan, the provincial average attains levels as high as 3.19%, Yunnan, 3.09%, Guizhou, 2.95%, Guangxi, 2.22%. This reliance on coal as the principle source of energy has resulted in an air pollution climate dominated by SO_2, particulates, NO_x and acid rain. It is predicted that even in the next century (2025), the bulk primary energy will still be coal (68%) and oil (25%) (Drennen and Erickson, 1998).

1. Sulphur dioxide

Over the last 2 decades, China's SO_2 emission have grown by more than a factor of 3. In 1995, annual SO_2 emissions amounted to *c.* 23.7 million tons and based on previous increases in the rate of emissions it is forecast that total SO_2 emissions will reach 27.3 million tons by the year 2000 and will reach 33 million tons by the year 2010. Recent estimates show that Chinese SO_2 emissions account for *c.* 69% of the total SO_2 emissions from Asia (Kato, 1996).

In 1997, the ground-level annual daily mean SO_2 concentration was 66 $\mu g\ m^{-3}$ (1 $\mu g SO_2\ m^{-3} \approx 0.391\ nl\ l^{-1}$), ranged from 3 to 248 $\mu g\ m^{-3}$ with 52% of the northern cities' exceeding the Chinese secondary air quality standard (SAQ) for SO_2 (60 $\mu g\ m^{-3}$), in comparison with 38% of the southern cities'. Annual mean SO_2 concentration in the northern cities was 72 $\mu g\ m^{-3}$, while it was 60 $\mu g\ m^{-3}$ in the southern cities. In the northern region, the highest levels of SO_2 were monitored in Taiyuan and Jinan; annual mean SO_2 concentrations reached 248 $\mu g\ m^{-3}$ and 173 $\mu g\ m^{-3}$, respectively, in 1997. In the southern region, the highest levels of SO_2 were monitored in Yibin and Chongqing, where the annual mean SO_2 concentration reached 216 and 208 $\mu g\ m^{-3}$, respectively, in 1997 - although levels were even higher in 1996 (ECCEY, 1997; 1998).

2. Acid precipitation

In the last two decades, acid rain area in China has increased dramatically. In 1993, the acid rain covered an area accounted for more than 40% of the total land area of China. At present, acid rain observed mostly in the southeast part of China and it is predicted that in the next century the area affected by acid rain may continue to expand and may move toward the northern and western parts of China (Wang and Wang, 1995).

The chemistry composition of rain in China is quite different from other developed countries; in particular the ratio of sulphate to nitrate of rain is much higher in China, the annual average sulphate/nitrate can be 7.5 in Chongqing and higher than 17 in Guiyang (Xu and Zheng, 2000). The rain pH values can be as low as 3.01 and acid rain frequency can be as high as 97.3% (Table 1).

Table 1 The pH values of rain fall in China in 1997. Acid rain frequency means the percentage acid rain (defined as pH values below 5.6) of the total rainfall (Adapted from (ECCEY, 1998).

District		pH	Acid rain (pH<5.6) frequency (%)
North	Tumen	3.91 – 7.08	64.0
	Qingdao	3.46 – 8.00	62.5
East	Hangzhou	3.33 – 7.36	75.8
	Nanjing	3.80 – 7.98	40.9
	Xiamen	3.03 – 6.72	83.5
South	Guangzhou	3.62 – 6.88	56.9
	Guiling	3.50 – 7.19	61.6
	Nanning	3.95 – 7.04	53.8
	Wuzhou	3.56 – 6.84	71.9
Southwest	Chongqing	3.28 – 8.70	54.3
	Yibin	3.32 – 5.26	81.0
Central	Nanchang	3.26 – 6.84	75.1
	Ganzhou	4.03 – 7.37	69.3
	Changsha	3.01 – 5.74	97.3
	Hengyang	3.52 – 6.60	73.0

3. Nitrogen oxides

From the early 1980s to the present-day, vehicle numbers in Beijing city have increased dramatically from 0.3 million to over 1 million. Because of the growing industrial activity and the rapid increase in vehicle numbers, NO_x emissions in China increased 33-fold between 1950 and 1990 (Fig. 1).

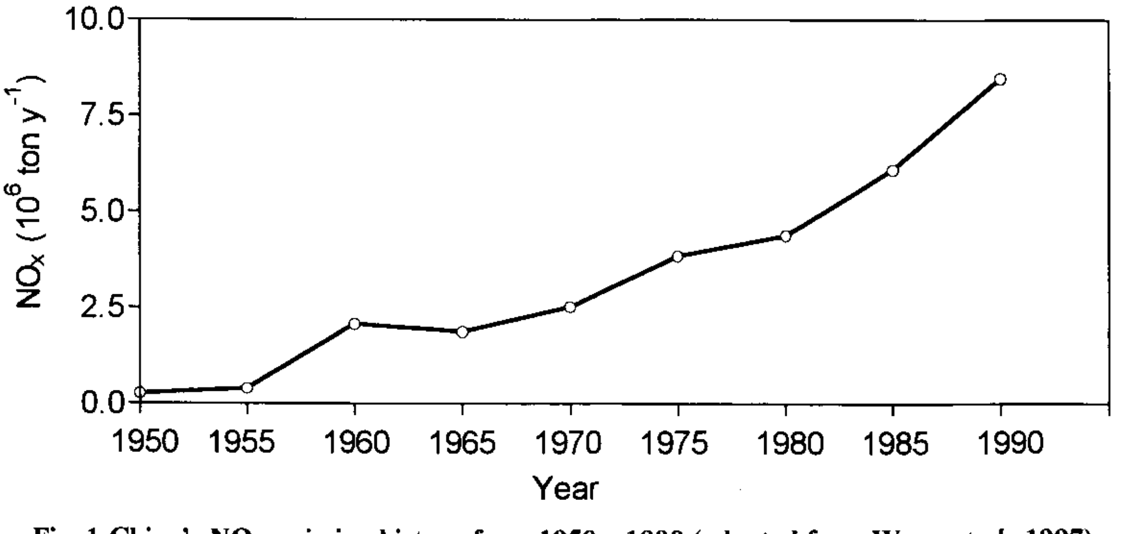

Fig. 1 China's NO_x emission history from 1950 – 1990 (adapted from Wang *et al.*, 1997).

In 1997, the national annual daily mean NO_x concentration was 45 μg m^{-3} (1 μg NO_2 m^{-3} ≈ 0.543 *nl l*$^{-1}$); ranged from 4 to 140 μg m^{-3}. The annual daily mean NO_x concentration in northern cities was 49μg m^{-3} and 41 μg m^{-3} in the southern cities. In more than 36% of all the big cities the NO_x concentrations exceeded the Chinese SAQ for NO_x (50 μg m^{-3}). The bigger cities, with high vehicle numbers, normally exhibit higher NO_x concentrations. For example, in the northern part of China, the monitoring program in Beijing shows that highest NO_x levels reached 133 μg m^{-3} in 1997 while in the southern part of the country, Guangzhou and Shanghai, exhibited the highest NO_x levels reached 140 and 105 μg m^{-3}, respectively, in 1997 (ECCEY, 1998).

4. Suspended particulate matter

It is estimated that of the total suspended particulate matter (TSPM) concentration of 227μg m^{-3} in Chongqing, 43% resulted from the combustion of coal, *c.* 36% from heavy industry (mostly steel plants) and *c.* 9% originated from roads (Chen *et al.*, 1996). The national annual daily mean TSPM was 291 μg m^{-3}, ranged from 32 to 741 μg m^{-3} in 1997. The national annual mean of dust deposition is 15.3 tons km^{-2} $month^{-1}$. The annual mean in northern cities is 21.48 tons km^{-2} $month^{-1}$, the highest levels recorded in Anshan (53.17 tons km^{-2} $month^{-1}$) while the annual mean in southern cities is 9.29 ton km^{-2} $month^{-1}$, with the highest levels recorded in Wuhan (19.16 tons km^{-2} $month^{-1}$) (ECCEY, 1998).

5. Fluorides

The most serious fluoride problems are associated within the local vicinity of brickyards, ceramic factories, aluminium plants, phosphate fertiliser plants and cement factories. A large amount of fluoride is emitted from the burning of coal, clay and other materials of high fluorides content. It is estimated that in China the average fluoride emission rate from brickyards is 663 kg fluorides per million bricks produced (Liang *et al.*, 1992). Taking the average fluoride content of coal to be 157 mg kg^{-1}, then we calculate the total national annual fluoride emissions associated with coal burning to be c. 0.2 million tons. Monitoring data shows that in July – December 1991, the average fluoride concentration at ground-level in Changchun was 0.77 μg F m^{-3} (Inoue *et al.*, 1995); the average fluoride concentration in early summer of 1998 was 0.218 μg F m^{-3} and this raised in winter to an average of 1.198 μg F m^{-3} in Zhongguanchun (Beijing) (Feng, Y. personal communication). Fluoride levels can attain levels as high as 22.6 μg F m^{-3} or higher around some brickyards in local areas (Sun *et al.*, 1998).

6. Ozone

In China, NO_x pollution is not yet as serious as SO_2. So far, little attention has been paid to O_3 and little information exists on ground-level O_3 concentrations in China - measurements restricted to a few inner-city locations or industrialised areas in Lanzhou, Beijing (Tang *et al.*, 1988) and Shanghai (Xu and Zhu, 1994). Only recently, have data become available for O_3 concentrations in rural areas - and this is restricted to individual studies conducted in and around the city of Chongqing (Zheng *et al.*, 1998) and four other rural sites (Chameides *et al.*, 1999). At this stage, because of the need for nation-wide monitoring data, it is not possible to summarise the O_3 pollution climate in China. However, the limited data available shows that in some places O_3 is potentially high enough to cause adverse effects on vegetation. For example, records show the maximum hourly mean O_3 concentration attained more than 400 ppb (1ppb ≈

1 *nl l*$^{-1}$) in Xigu, Langzhou, in 1981, and 332 and 254 ppb in 1982 and 1983, respectively (Wang *et al.*, 1997), in Shanghai and Chongqing hourly mean O_3 concentrations attained *c.* 100 ppb (Xu and Zhu, 1994; Zheng *et al.*, 1998), while in Linan (a rural area 60 km away from Hangzhou city) hourly mean O_3 concentrations attained 120 ppb and in Waliguan (a rural area in Qinghai) levels of 130 ppb were attained (Wang, C., personal communication). Research in the USA has established that crop yields were reduced by 10% or more when SUM06 (3-month sum of all daytime, 1-hour-averaged $O_3 > 60$ ppb) is 15-25 ppm.h (1 ppm = 1000 ppb). Monitoring data in Linan indicates that the SUM06 reached 15-31 ppm.h between the end of 1994 and the beginning of 1995 (Chameides, *et al.*, 1999). Most of the time, O_3 levels at the majority of monitoring sites are below levels considered to be phytotoxic to vegetation. However, the rapid growth of industry and increases in the number of motor vehicles are predicted to result in a growing photochemical oxidant problem in the foreseeable future (Zheng *et al.*, 1998, Chameides, *et al.*, 1999).

Effects on vegetation

1. Effects of SO_2 and acid precipitation

Since the end of 1970s and the beginning of 1980s, field evidences of vegetation injury caused by precipitation have been observed in many places in southwest and southeast parts of China. Forest decline in Nanshan, a mountain adjacent to the southeast of Chongqing, is the most well known case in China and has been attracting a lot of public attentions since the beginning of 1980s. Nanshan has *c.* 2000 ha of forest, mainly masson pine (*Pinus massoniana* Lamb.). Since the beginning of 1980s, more than half of its trees were lost and 85% of the pine trees showed some degree of injury (Liu *et al.*, 1988 a; 1988b; Zheng, 1991) (see Photo 1). The symptoms of injury on the pine trees include needle tip necrosis, thin crown, reduced needle length, premature abscission, branch dieback, and reduced radial growth. Broadleaf trees (including *Robinia pseudoacacia, Quercus dentata, Eucalyptus robusta* and *Erythrina variegata*) also displayed necrotic lesions (Yu *et al.*, 1990a; Zheng, 1991). Most of the research results showed that the tree dieback and forest decline in Nanshan is caused by direct damage of SO_2 in combination with acid fog, acid rain and consequently the increased susceptibility to insects, e.g. caterpillar (*Dendrolimus punctatus*) (Liu *et al.*, 1988; Yu *et al.*, 1990b; Zheng, 1991; Shen *et al.*, 1995).

A recent estimation (Feng *et al.*, 1999) showed that 19.05% of the agricultural land in the 7 provinces (Jiangsu, Zhejiang, Anhui, Fujian, Hunan, Hubei and Jiangxi) in Southern China was affected by SO_2 acid rain and pollution and the average crop yield reduction due to the combined effects of SO_2 and acid rain was 4.34% in the middle of 1990s. Vegetable yield was reduced by 7.8%, wheat by 5.41%, soybean by 5.73% and cotton by 4.99%. In the above 7 provinces, 4.2% of the forest are affected by acid deposition, total area is around 1.28 million hm^2, 62% is *P. Massoniana* Lamb. forest. The average volume loss rate of forest is 13.2%, with a total loss of 1.01 million m^3.

Since the beginning of 1980s, many field investigations and laboratory studies about the effects of air pollution on vegetation have been conducted in China. The majority studies have been focused on the impacts of acid rain and SO_2 on vegetation. Few controlled studies have been directed at quantifying the impacts of SO_2 and even less have attempted to investigate the combined effects of SO_2 and other air pollutants (e.g. NO_x, O_3, fluoride and SPM). Most of the researches have been conducted using unrealistically high pollutant concentrations for a short time period, and little work has been directed at investigating the mechanisms underlying effects.

2. Effects of particulate matter

Often, a layer of dust can be seen on the plant leaves in and around the cities or industrial areas. Particulate matter, not only can reduce the yield of vegetable and fruit, but also can increase heavy metal content and worsen the quality (Fu *et al.*, 1996; Wu *et al.*, 1990). Wu *et al.*, (1990) conducted an investigation and results show that dust around a steel plant in Chongqing significantly increased heavy metal contents in leaves of lettuce (Mn by 8.7 times, Cu by 6.5 times, Zn by 3.8 times and Pb by 11.4 times). The Pb content of lettuce leaves exceeded the Chinese National Standard for food by *c.* 6 times. Since dust emission from industry is much easier to be reduced than other gaseous air pollutants, it is likely that the dust effects on vegetation around industrial areas will be reduced in the foreseeable future. However, in and around the city areas because of the continuous building and road construction works everywhere, dust effects still going to be a problem for quite a period of time.

3. Effects of fluorides

Many experiments have been conducted in China to study the effects of fluoride on the mulberry tree because of it's importance in silk production, especially in Zhejiang and Jiansu provinces where silk and the manufacture of bricks form the cornerstone of the economy. A ceramic factory in Jiangbei, Chongqing started production in the early 1990s. Two months after this, the local vegetation (bamboo, vegetables and other agricultural crops) downwind of the factory started to show typical fluoride injury - necrosis around the margins and tips of leaves. One year later, almost all the bamboo had been destroyed within a radius of 400 m around the factory (Zheng, 1993 unpublished). Although largely circumstantial, observational evidence of this kind is common in China (Sun *et al.*, 1998; Fluorides Project Research Team 1994). Sun *et al.*, (1998) conducted a field study around an aluminum factory in Zhengzhou to assess the effects of fluoride emitted from the factory on the growth of winter wheat. Their results showed that there were significant reductions in growth and yield of the winter wheat within at least 800 m of the factory. Fig. 2 shows another example of the effects of fluorides on rice yield around a brickyard in the countryside close to Shanghai.

4. Effects of O_3

There is little experimental data available concerning the effects of environmentally relevant O_3 concentrations on vegetation in China. However, the study by Zheng *et al.*, (1998) draws attention to the potential for adverse effects. In this experiment, 11 cultivars of Chinese crops (aubergine, cauliflower, Chinese leaves, tomato, lettuce, wheat, maize, radish, courgette, green pepper, and rice) commonly grown in the Chongqing region were screened for their relative ozone sensitivity under controlled conditions. Over a 4-week period, plants were exposed to a low ozone concentration during the night 15 ppb rising to a midday maximum of 75ppb. The results indicated that, in terms of effects on growth rate, several Chinese cultivars (green pepper, rice, aubergine, cauliflower etc) were as sensitive to ozone as some of the most commonly adopted bioindicators (e.g. 'Cherry Belle' radish and plantain).

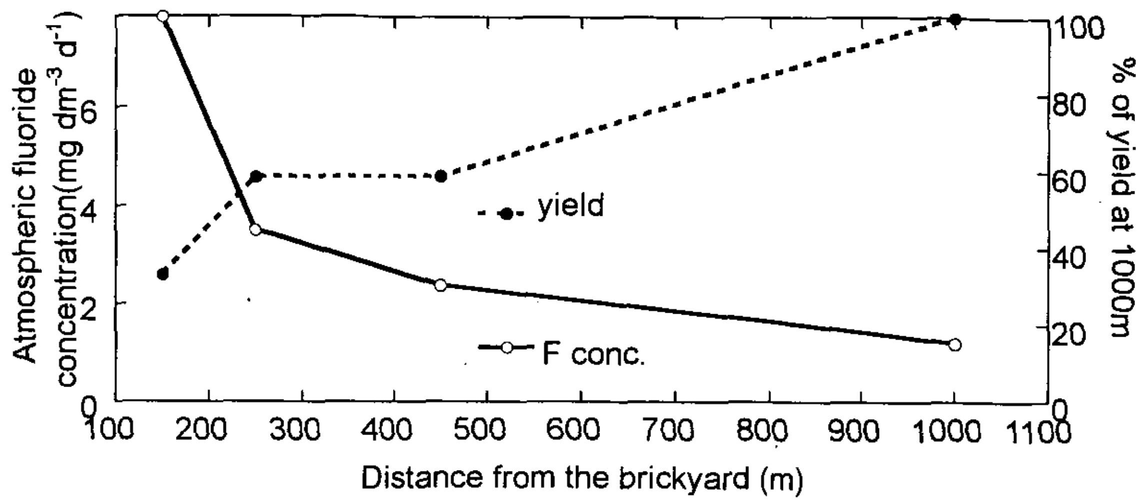

Fig. 2 Ambient air fluoride concentrations and the rice yield around a brickyard in the countryside close to Shanghai (1991). The annual fluorides emission of this brickyard is 24,000kg. (Adapted from Fluorides Project Research Team, 1994).

5. Combined effects of air pollutants on vegetation

In most parts of China, as elsewhere, vegetation is exposed to a complex cocktail of air pollutants (SO_2, NO_x, SPM and O_3 etc). For example, Chongqing is one of the heaviest SO_2 polluted cities in China with an annual average SO_2 concentration of 340 $\mu g\ m^{-3}$ and maximum daily average of 940 $\mu g\ m^{-3}$ in 1995 (Jiang and Zhang, 1996), and such high SO_2 concentration induces acidic precipitation in the whole Chongqing region (with a total area of 23,000 km^{-2}). The average pH value of rainfall was 4.30, more than 85% of total rain events were acidic rain from 1987-1989 (Zhao *et al.*, 1994). Another air pollutant is particulate with an annual average total suspended particulate concentration of 320 $\mu g\ m^{-3}$ in 1995; NO_x is relatively low with an annual average range from 50-80 $\mu g\ m^{-3}$ during ten years time (from 1985-1995) (Jiang and Zhang, 1996). Recently, Zheng *et al.*, (1998) reported that in and around Chongqing ozone (O_3) exposure during intermittent O_3 episodes in the summer months commonly approached WHO short-term guidelines for protection of the most sensitive vegetation. Such an air pollution climate has already caused serious ecosystem problems in this area. Many sensitive tree species, like eucalyptus can no longer survive in the city area and since 1960 up to now, the street-tree species in the city area have been changed three times due to heavy air pollution. Some vegetable species, like *Brassica oleracea capitata* L., which were very popular and commonly grown in this area about 30 year ago, can no longer grown around the city any more. The average yield reduction of vegetable induced by air pollution around the city is *c.* 25% (Zheng and Chen, 1991). Photo 2 shows a typical landscape of the countryside around Chongqing city.

Technically it is difficult to investigate the combined effects of this pollutant load in the same experiment under controlled conditions, especially in developing countries where there is a shortage of exposure facilities. Consequently, although some controlled studies on air pollution combinations have been carried out in China, most employed unrealistically high concentrations of air pollutants for short time periods to study physiological and biochemical effects (Dai *et al.*, 1994; Zhou *et al.*, 1993). There are a few field fumigation experiments, but lack of continuous air pollutant monitoring data. For example, Zheng *et al.*, (1996) grew four vegetable species in pots in different locations, 5 km, 10 km and >20 km downwind of the city of Chongqing. Results showed that in comparison with the yield at the 'clean site' (Jieshi, >20 km), yields of all four vegetable species at the two sites closest to the city were significantly decreased by ambient air pollution. It was also found that the number of aphids on some of the vegetable leaves was significantly higher at the sites closer to the city.

Photo 1 Forest decline caused by air pollution and acid precipitation in Nanshan, Chongqing, China. Pictured by Y. Zheng, 1990.

Photo 2 A typical countryside landscape around Chongqing city, China. Pictured by Y. Zheng, 1993.

Some thoughts about the future

To be able to quantitatively assess the risks posed by air pollutants to vegetation in China the following gaps remain to be filled. (1) There is a need for the expansion of air pollutant monitoring networks into agricultural and forest areas. So far, most of the air pollution monitoring stations have been restricted to sites located in the inner city or industrial areas. (2) Monitoring data relating to ground-level ozone and fluoride concentrations is lack in most areas. From the available data it is clear that in some regions the concentrations of both these pollutants are approaching levels that are generally considered to be phytotoxic. (3) Almost all of the air pollution impact studies conducted to date have focused on agricultural crops and forest trees - virtually nothing is known about the impacts of air pollutants on natural (or semi-natural) herbaceous vegetation. Future studies should focus not only on agricultural crops and forest trees, but also consider effects on natural vegetation, especially endangered species. (4) There is a severe shortage of reliable yield-response data applicable to conditions in China or the way in which the presence of other air pollutants may influence these relationships. This will require the application of state-of-the-art technology for the field-based assessment of effects in order that critical levels/SAQ's based on sound scientific principles can be derived for Chinese vegetation types. The studies will require the injection of funding to permit employment of OTCs and/or the development of free air exposure systems in the South and North of China, where the climates are very different.

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