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Development of Science and Policy Related to Acid Deposition in East Asia Over 30 Years

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1. Abstract

Scientific and public interest in acid deposition and its ecological impacts have increased throughout 1990s in East Asia (Northeast and Southeast Asia). After being established in 2001, the Acid Deposition Monitoring Network in East Asia (EANET) celebrates the 20th anniversary in 2021, and is now being expanded in scope reflecting the shifting social concern from acid deposition to broader air quality and climate change in recent years. This paper reviews the past 30 years of development of scientific research and policy related to acid deposition in East Asia. Since the onset of the twenty-first century, East Asia has had the highest SO2 and NOx emissions in the world by continents, with substantial economic developmental inequality among countries. An overview of studies on sulfur and nitrogen deposition, the acidification of inland water and forest soil, and forest decline reveal that although limited acidification of inland water and forest soils have been documented, no decline in the populations of fish and other aquatic biota has been reported in East Asia. After a review of policy-oriented modeling studies on source receptor relationships and the critical loads of sulfur and nitrogen in East Asia, the history of EANET and its success and challenges are discussed. Finally, the importance of epistemic communities as the interface between science and policy in the region is discussed. Regional governance and cooperation are essential for reducing the emission of greenhouse gases, especially short-lived climate pollutants and atmospheric pollutants to realize the co-benefits of global climate change mitigation and improved air quality.

2. Introduction

Acid rain was considered to be one of the most pressing environmental issues in Europe as early as the 1970's. Because of its serious large-scale impacts on ecosystems and its transboundary nature, acid rain has received wide- spread scientific and public attention, leading to coordinated policies in Europe and North America, such as the Convention on Long-Range Transboundary Air Pollution (CLRTAP). [34] recently provided an overview of the development of acid deposition research and policy particularly in Europe, based on a symposium held in Stockholm in 2017.

In early 1980s, Asian countries were informed about the environmental issues of fish extinction in Scandinavian lakes and rivers, forest dieback across Europe and the decline of Canadian maple tree populations, raising concern among scientists, policy makers, and the public. Throughout the 1980s and early 1990s, scientific studies on acid rain chemistry were prevalent in China [103, 123, 124], Japan [38, 39], Korea [17], and India [49]. These early acid rain studies in Asia were reviewed by Bhatti et al. (1992). Meanwhile, the early 1990's ushered in negotiations for establishing the Acid Deposition Monitoring Network in East Asia (EANET) across both Northeast and Southeast Asia. The European Monitoring and Evaluation Program (EMEP), which had been conceived with the aim of monitoring acid deposition in Europe [34] was used as a template for EANET. EANET was formally established in 2001, marking 2021 as its 20-year anniversary.

As discussed at the end of this paper, negotiations are ongoing for the expansion of the scope of EANET from acid deposition alone to more general air quality and climate change. At this stage of the life cycle of EANET, it is worthwhile to reflect on the development of acid deposition research and policy in East Asia over the past 30 years of the development of science. In this paper, we present a review of the acid deposition issue in East Asia during 1990–2020 including the emissions of precursors, deposition of acidic species, its apparent ecosystem impacts, science and policy related to EANET, and the development of an epistemic community.

In this review, we limit our discussion to acid deposition, excluding O3 and particulate matter smaller than 2.5 and 10 lm in diameter (PM2.5 and PM10, respectively). We also provide an overview of the development of EANET, focusing on its achievement as well as challenges. Additionally, while this review focuses on both research and policy, the recent paper by Duan et al. (2016) and Chapter 4 in the Review on the Status of Air Pollution in East Asia (TFRC/ SAC/EANET 2015) are the most useful purely scientific reviews of acid deposition in Asia.

3. Characteristics of Sulfur and Nitrogen Emissions

Figure 1a and b show the historical trends of anthropogenic emissions of SO2 and NOx in East Asia between 1970 and 2015 compared to those in western and central Europe and North America. The data for Europe and North America are based on the Emissions Database for Global Atmospheric Research (EDGAR) v5.0 (EDGAR 2021), and those for East Asia are based on the Regional Emission Inventory in Asia

3.2 (REAS) 954]. As can be seen in Figure 1, both SO2 and NOx emissions were prominent in Europe and North America during the 1970s. East Asian SO2 and NOx emissions were less than half of those in Europe and North America in the early 1970’s, but increased to be comparable levels in the 1980’s and equaled the European and North American levels in the late 1980s and mid-1990s, respectively. East Asia became the predominant source region of SO2 and NOx emissions in the world by the 2000’s. By contrast, SO2 and NOx emissions in Europe began to decrease throughout the 1980’s and 1990’s, respectively, owing to the legally-binding protocols under CLRTAP [34]. North American SO2 and NOx emissions began to decrease throughout the 1990s and 2000s, nearly 10 years later than those in Europe, owing to the amendment of the Clean Air Act in 1990 in the United States [11]. According to REAS, East Asian SO2 and NOx emissions began to decrease sharply from the mid-2000s and early 2010s, respectively, mainly reflecting the decreasing emissions development between countries. Figure 2 traces SO2 and NOx emissions of EANET countries based on REAS in relation to GDP (World Bank 2021) from 1990 to 2015 at 5-year intervals. Over the last 30 years, most East Asian countries have experienced rapid economic development accompanied by a rapid increase in regional SO2 and NOx emissions. While the rate at which SO2 and NOx emissions increased was paralleled by GDP growth in low-GDP countries, the emissions tended to decrease despite continuous GDP growth in high-GDP countries, such as Thailand, Russia, China, Republic of Korea (R. O. K.), and Japan. The recent decrease in SO2 and NOx emissions in East Asia, as shown in Figure 1, largely reflects trends in China shown in Figure 2, as confirmed by satellite observation [111].

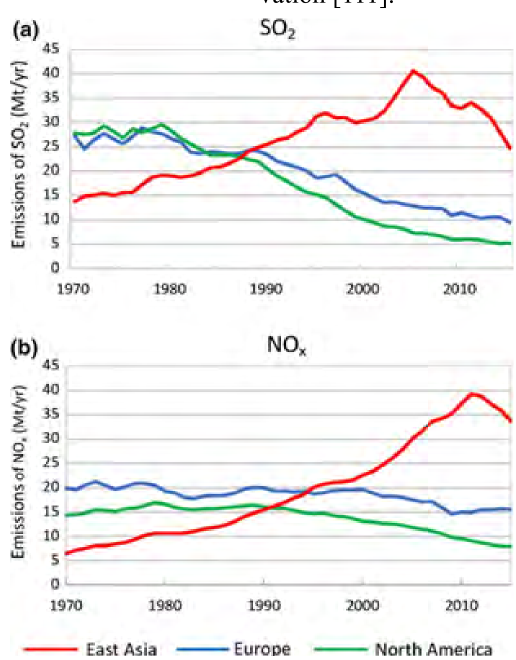

Figure 1: Historical emissions of a SO and b NO in East Asia compared.

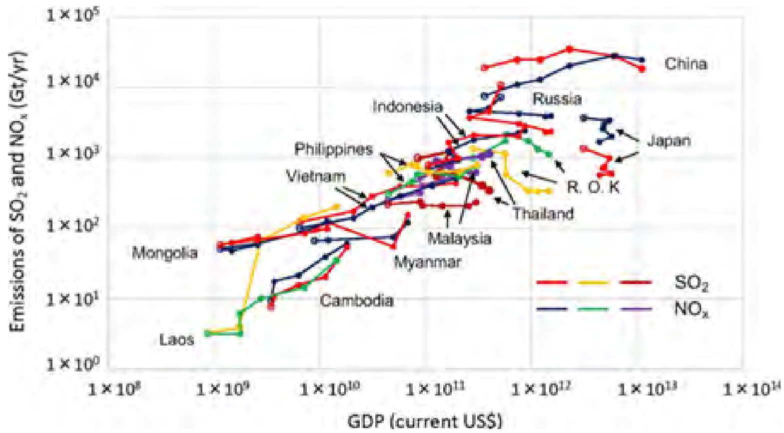

Figure 2: Emissions of SO2 and NOx based on REAS as a function of GDP for the countries participating in EANET during 1990–2015 in 5-year intervals (data for Cambodia and Myanmar are from 1995 and 2000, respectively). The first year of each plot is shown by an open circle.

4. Acid Deposition and Environmental Impacts

4.1. Sulfur and Nitrogen Deposition

Because of the relatively higher SO2 and NOx emissions in East Asia compared to those in Europe and North America in the 2000s (Figure 1), it was anticipated that acid with those in Europe and North America (Based on EDGAR v.5 for Europe and North America and on REAS v.3 for East Asia) deposition in East Asia would also exceed that in Europe and North America. Figure 3 compares the annual mean concentrations and wet deposition of non-sea-salt SO42- (nss-SO42-) and NO3-, and the pH of rainwater in East Asian countries (obtained from EANET) compared those in Europe (EMEP) and North America (National Atmospheric Deposition Program, NADP) during 2014–2019 in relation to the annual precipitation [69]. As can be seen in Figure 3a-c, the annual wet deposition of nss-SO42-, NO3- and H? in eastern and southern China, Japan, R.O. K., Malaysia, Indonesia and Vietnam was higher than that in Europe and North America. Although there were not enough EANET monitoring sites to obtain the spatial distribution of acidity, the annual average precipitation pH at some monitoring sites in China, Japan, R. O. K, Malaysia, Indonesia and Russia were between 4.0 and 5.0, being slightly lower than that in Europe and North America as shown in Figure 3c. The reason why the pH at several sites in China was lower than in Europe and North America is that nss-SO42- and NO3- concentrations exceeded the cations, Ca2? and NH4?, even though they may neutralize acids. It should be noted that EMEP and NADP sites are located in rural or remote areas, whereas some EANET sites are located in urban area.

Temporal variation in the spatial distribution of average precipitation pH in China are shown in Figure 4 based on domestic long-term records [21]. Figure 4a shows that in 1995, acidic rainwater was prominent in mid- western (e.g., Sichuan Province) and southern China. Between 1995 and 2005, an overall trend of decreasing pH was observed, and areas with the highest rainfall acidity shifted eastward (Figure 4b). Accompanying to the reduction in SO2 emissions after 2006, the national average precipitation pH increased, and the overall area experiencing severe acid rain decreased (Figure 4c).

A significant characteristics of acid deposition in Northeast Asia is the effect of buffering capacity of car- bonates/oxides of calcium and magnesium in natural soil dust (yellow sand) originating from the Gobi Desert, Taklamakan Desert and Loess Plateau, characterized by high Ca2? concentrations (Larssen and Carmichael 2000). These high Ca2? concentration originate not only from the natural sources but also from anthropogenic activities, such as cement production and metal smelting [59]. The higher precipitation pH in northern and western China, as shown in Figure 4 reflects the influence of soil dust. Wang et al. (2002) successfully reproduced the spatial distribution of precipitation pH in Northeast Asia in the late 1990s using a chemical transport model. This demonstrated the increase in acidity during the long-range transportation of acid oxides from the coastal areas of the Bohai Sea and Yellow Sea in China (pH 5.6–7.0) to the Korean Peninsula (pH 5.0–5.6) and finally to Japan (pH 4.5–5.0) due to the loss of calcium carbonates/oxides. In recent years, studies on dust- and sandstorms in Northern China and Mongolia related to the buffering effect on acidification have been conducted under the Tripartite Environment Minister Meeting in R. O. K., Japan, and China (Chu 2018). These will be discussed later.

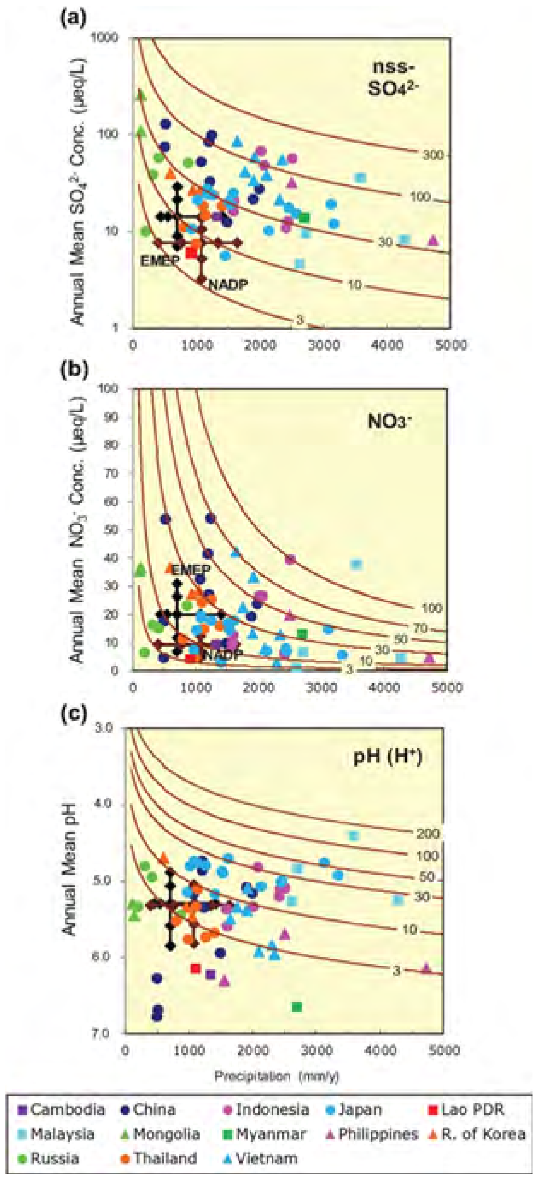

Figure 3: Annual mean ion concentrations or pH, vs. annual precipi- tation, and annual wet deposition amounts of a nss-SO 2-, b NO - and c H? in East Asia and comparison with those in Europe (EMEP, n = ca. 200) and North America (NADP, n = 80–90) in 2014–2019 (Ohizumi et al. 2021). The isolines and numerical values in the figure denote the amount of annual wet deposition (Unit: meq/m2). The dotes for EMEP and NADP denotes 10, 25, 50 75 and 90 percentile values.

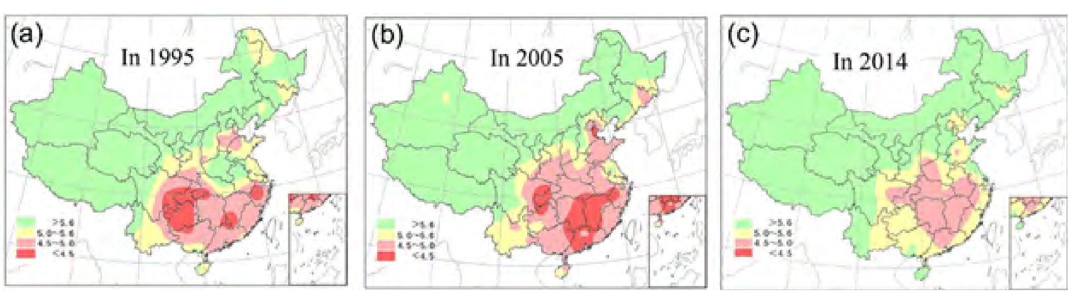

Figure 4: Spatial distribution of annual average precipitation pH in China in a 1995, b 2005 and c 2014 (adapted from Duan et al. 2016)

4.2. Acidification of Inland Water

Inland water and soil water acidification are strongly influenced by bedrock geology, which regulates the acid- neutralizing capacity through mineral weathering. When limestone (rich in CaCO3), dolostone (rich in CaMg(CO3)2) and easily weatherable aluminosilicate minerals are prevalent in the surface geology of the region, streams draining in the area have a high acid-neutralizing capacity, thus, the acidification of inland water and soil do not occur, even with high deposition rates of SO42- and NO3-. This is the case for large portions of East Asia (Yu et al. 2017b), in contrast to northern and central Europe and North America (Larssen and Carmichael 2000).

In southwestern China, where precipitation acidification was the most prominent in the 1990's (Figure 4a), Xue and Schnoor (1994) surveyed 16 lakes and concluded that lake water acidification had not occurred. They suggested that the water chemistry was buffered by cation exchange and carbonates derived from chemical weathering in water- sheds. In the same region, Larssen et al. (1998) observed that Ca2? was the major cation in streams and that water acidification was neutralized by bedrock. In many tributaries of the upper Yangtze River, Duan et al. (2011) observed a slight decrease in pH during the 1990's associated with an increase in SO42- and NO3- concentrations. With a decrease in SO2 emissions after 1998, no further decrease in pH was observed in the 2000s. More recently, Qiao et al. (2016) reported that the surface water pH at 65 sites spanning north to south China ranged from 6.5 to 9.0 over the past decade (2004–2014), which satisfied the water quality standards in China. For the decade, pH decreased significantly at 31 of the 65 monitoring sites, which were mainly located in the Haihe River, Tiahu Lake, and Yangtze River basin, where NOx emissions continuously increased. Notably, the aforementioned studies were mainly based on large water bodies such as rivers and lakes, excluding small, forested headwater streams, which may be more sensitive to acidification. A very limited number of acidic streams have been found via long-term monitoring or regional surveys in southwest and south China [118, 119].

In Japan, acid deposition was not considered to be likely to lead to marked increases in acidity of surface waters because of the high acid-neutralizing capacity of most catchments [70]. Yamada et al. (2007) found that the pH of lake water had decreased since the mid- 1990's in Lake Ijira, an EANET site in central Japan, where the geology of the catchment is dominated by chert. The reduction in nitrogen retention triggered by climate anomalies has been suggested as a cause of acidification for the Lake Ijira catchment [67]. Similarly, Matsubara et al. (2009) observed a decrease in water pH (ranging from 6.5 to 7.5) in several rivers in parts of central Japan with granite rocks. In addition to Northeast Asia, a decrease in stream water pH associated with the increasing concentrations of SO42- and most of other major cations and anions has also been observed in a tropical forest in Thailand [79].

Recently, with the reduction in sulfur deposition, decreasing inland water acidification has been reported at 22 monitoring sites along rivers in China [75] and in forest catchments in Japan, including Lake Ijira [77, 78]. However, nitrogen deposition [75] has been noted to potentially delay acidification recovery [119]. Despite these studies on the acidification of inland water, to our knowledge, the decline in fishes and other aquatic biota populations due to acidification has not been reported in East Asia.

4.3. Acidification of Forest Soil

In southern China, Dai et al. (1998) found that the forest soil pH on the Zhurong Feng peak of Mt. Heng in Hunan Province and the Wuming District of the Guangxi Zhuang Autonomous Region decreased from approximately 5.5 to 4.5 from 1959 to 1994. Acidification was the highest in the topsoil. The decreased soil pH was not related to changes in the soil organic matter content, suggesting that soil acidification may be related to the high acid deposition in this area. In the Dinghushan Biosphere Reserve in Guangzhou, subtropical China, Liu et al. (2010) reported that the forest soil pH at a depth of 0–20 cm decreased from 4.60 to 4.75 in 1985 to 3.84 to 4.02 in 2005. Zhu et al. (2016) further reported a general decrease in the soil pH from 6.10 to 5.74 (and as low as 5.47 in more severe cases) in forested areas in southwest China from 1881–1885 to 2006–2010. They concluded that atmospheric acid deposition was the major driver of the forest soil acidification with a minor contribution from forest harvesting. More recently, Yu et al. (2020) found that in subtropical China, the soil pH decreased across the whole soil profile (0–150 cm) over a period of 60 years, as illustrated in Figure 5. The decrease in pH was more pronounced in the surface layer than that in deeper layers, reflecting the findings of a previous study [18]. In that period, exchangeable Al3? increased in the topsoil and decreased in deeper soil layers. In addition to forest soil, the acidification of grassland soil in China from 1980 to 2010 has been studied by Yang et al. (2012). The Al/ (Ca ? Mg) and Al/(Ca ? Mg ? K) ratios in soil water reported in China [125] were relatively small mainly because to high calcium concentration, which have to date reduced the risk of forest decline due to soil acidification. Thus, changes in the balance between sulfur and calcium deposition and their effects on the acid–base balance in soil are of great concern in East Asia [21, 55].

In Japan, Nakahara et al. (2010) reported that the mean pH of surface soils decreased from 4.5 in 1990 to 3.9 in 2004 in the Lake Ijira catchment, where the atmospheric deposition of H?, SO42- and NO3- was among the highest in the country. This area is covered by brown forest soil derived from chert. However, the 3rd Periodical Report on the State of Acid Deposition in East Asia (EANET 2016a) highlighted that soil acidification has not progressed in the Lake Ijira catchment since the mid-2010s.

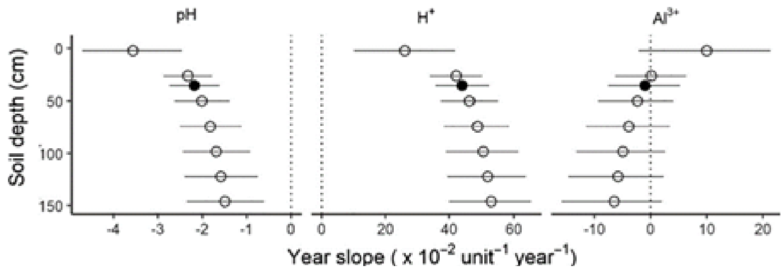

Figure 5: Temporal trends of soil pH, and contents of exchangeable H? and Al3? associated with soil depth over the last 60 years. Values (means and 95% bootstrapped confidence intervals) represent the temporal trends for a given depth. Solid symbols are the mean yearly slopes across all observations. (Adapted from Yu et al. 2020)

4.4. Forest Decline

In Europe, forest decline, as observed in the Black Triangle (the interface between Poland, East Germany and Czech Republic) has been a concern since the 1980's. The threat posed to forests by high sulfur deposition and the release of toxic inorganic aluminum in soil water was first pointed out by Ulrich et al. (1980). However, vegetation damages may also be caused by direct exposure to air pollutants, such as SO2 and O3. Furthermore, acidic fog can also reduce the tolerance of certain tree species to cold weather. Forest decline in Europe has mainly been ascribed to a combi- nation of anthropogenic pressures and natural stresses such as drought, frost and pests [66].

In the Nanshan Mountains of Chongqing, China, 35–40% of the Masson pine forest has died since the beginning of the 1980s [120]. Pine trees close to the city were the most severely damaged, while those far from the city near Nanshan Park were no visibly damaged. The average pH values of rainwater in severely damaged areas and no-damaged sites were in the range of 4.2–4.4, and no difference were observed in the soil pH between damaged and non-damaged sites. Based on these observations, it was concluded that the pine forest damage was a result of the direct effect of SO2 and HF emissions from the city on the canopy [10, 120], while others argued that the chief causes of forest decline were pests [122] and acid rain [31, 62]. Studies have also revealed that forests can be severely damaged by a combination of pests and air pollutants, and that SO2 is more harmful to plants in the presence of acid rain and fog at pH 4 than pH 5–7 [10].

During the 1980s, forest decline was also observed in China amongst the firs of Mt. Emei in Sichuan province, in the suburb of Liuzhou City in the Guangxi Province, and amongst Masson pine in Sichuan and Guizhou provinces. It has been speculated that this forest decline was due to a combination of acid rain and other types of air pollutants ([57] and references therein). Between 2000 and 2004, the Masson pine forest near Chongqing City and Guiyang City sustained considerable damage [107]. However, it could not be determined that the defoliation was due to air pollution or soil chemistry; instead, it was attributed to insects and climate stress [107]. Huan et al. (2014) proceeded to lime highly acidic soil in the Masson pine forest near Chongqing to distinguish between the ameliorating effects of Ca2? and Mg2?. Both the introduction of calcite and magnesite to the soil led to a significant increase in pH and decreased in dissolved inorganic aluminum in soil water. However, the Masson pine growth rate did not increase, which was ascribed to the nutrient imbalance due to phosphorous deficiency [43].

In Japan, concern about the decline of Japanese cedars in the Kanto region was first raised by Sekiguchi et al. (1986) suggesting acid rain and oxidants as the causes. This led to increasing interest in the phenomenon across the country ([51] and reference therein). How- ever, the causes of forest decline were not established in most studies because of a focus only on a single potential cause of the author's concern. Among them, Kume et al. (2000) elucidated the relationship between air pollution and the severe pine tree decline on the seaward side of Mt. Gokurakuji (693 m above sea level) in the Seto Inland Sea area in Hiroshima Prefecture in the 1990's. The pine forest decline was due to the decrease of photosynthesis and needle longevity stemming from poorer stomatal conductance, which was strongly correlated with atmospheric NO2, as opposed to SO2 and O3, emitted from urban and industrial areas. The soil pH, nitrogen content and C/N ratio did not differ significantly between declining and non- declining areas. Similarly, beech tree decline on the sea- ward side of the Tanzawa Mountains in Kanagawa Prefecture was found to be correlated not with the O3 concentration, but with the O3 advection flux (Kohno et al. 2007). Simultaneous exposure to acid fog and O3 has been suggested as the primary stressor for the growth and physiology of the beech trees [83]. Ito et al. (2011) suggested that the decline of Japanese cedar and cypress trees surrounding two Kyoto shrines was related to acidified soil with pH 4.35 and (Ca ? Mg ? K)/ Al molar ratios less than 10. To our knowledge, this is the only international report of field research on tree decline and soil acidification in Japan.

Visible forest decline has also been documented in South Korea ([58] and references therein). Lee et al. (2005) reported the following extents of 1st- and 2nd- degree decline (initial and moderate phases, respectively) of Japanese red pine in several mountainous areas in 1996: 68, 62 and 51% on Mt. Nam in urban Seoul, Doowang in the industrial area of Ulsan (Southeastern Korea), and

Mt. Kyebang in Hongchon (Northeastern), respectively. By 2001, forest decline at the three sites had progressed to 74, 70 and 55%, respectively. Although the primary cause has not been identified, the direct effect of air pollution and indirect effect of soil acidification via acid deposition have been suggested.

To our knowledge, international journal papers on forest decline due to air pollution and acid deposition have not been reported in East Asia except in China, Japan, South Korea, and the Siberian region of Russia. At EANET monitoring sites, forest conditions have been mostly healthy during the 2000s and 2010s [26]. A more comprehensive review of forest decline in East Asia has recently been published by Takahashi et al. (2020) including papers written in regional languages and published in domestic journals. It was suggested that the primary causes of tree decline have recently shifted from SO2 deposition and acidification to O3, coarse PM, and climate change.

5. Source-Receptor Relationship and Critical Loads

5.1. Deposition Model Simulation and Transboundary Source–Receptor Relationship

The quantification of the transboundary fluxes of sulfur between countries played an important role in the development of EMEP in Europe [34]. The Swedish initiative within the Organisation for Economic Co-operation and Development (OECD) resulted in a collaborative project that investigated the nature and magnitude of the transboundary transport of SO2 in Western Europe. The national budgets so-called ''blame matrices'', prepared as part of the project represented the first bridge between scientist and policy makers [34]. However, as will be further described in the historical review of EANET, the quantification of transport fluxes has yet to extend to East Asia. Nevertheless, scientific studies on the source–receptor relationship for sulfur and nitrogen have been conducted, particularly in Northeast Asia. This subsection provides a short review of these studies.

Early studies on model simulation of transboundary source–receptor relationships were conducted in the mid- 1990's [5, 42, 44-46]. These studies estimated the source– receptor relationships of sulfur deposition between north- east Asian countries, results of which showed substantial discrepancies. For example, the Chinese contribution to total sulfur deposition in Japan was estimated at 3.5%, 10%, 25% and 25% by Hung et al. (1995), Carmichael and Arndt (1995), Ikeda and Higashino (1997), and Ichikawa et al. (1998), respectively. The RAINS-ASIA model [82], which will be discussed later, estimated that China was responsible for 16% and 11% of the sulfur deposition in South Korea and Japan, respectively. Thus, there was little consensus among researchers on the source–receptor relationship of acid deposition between different East Asian countries in the early stage of inter- national cooperation in the 1990s (Kim 2007).

The Model Inter-Comparison Study for Asia (MICS- Asia) was initiated in 1998 by American and Japanese scientists to evaluate the performance and shortcomings of the chemical transport model (CTM) in East Asia. To date, MICS-Asia has comprised four phases [13-15, 47]. In Phase I (1998–2002), sulfur served as the target species. In Phase II (2003–2008), the study was expanded to include reactive nitrogen, NH3, and O3. New themes of air quality and climate change were incorporated into the study in Phase III (2008–2020) [32]. The work of Phase IV began in 2021 (ACAP 2021).

Figure 6 highlights an example of the results of MICS- Asia II, comparing the wet deposition of SO 2- in seven regional models across 37 EANET sites in March and July 2001 [106]. Predictions vary substantially between models as well as between models and observations. The ensemble means values obtained for a given period of time tend to better reflect observations than individual projections. For example, in March 20 the ensemble mean values was consistent with observations at most sites in China and Japan, with a correlation coefficient of 0.73. In July, the ensemble mean was also close to observations at most sites. A recent study by Itahashi et al. (2020) as part of MICS-Asia III recorded a correlation coefficient of 0.47 between the ensemble mean and observed wet deposition of SO42- in 2010. The incorporation of precipitation into the mode played a key role in accurately predicting wet deposition amount, resulting in correlation coefficient of 0.76.

Several papers have studied source-receptor relation- ships of oxidized sulfur and nitrogen in Northeast Asia after 2008 ([76] and references therein). How- ever, the results of these studies have still varied substantially; for example, the Chinese contribution to sulfur deposition to Japan ranged from 15% [61] to more than 50% [2]. In addition to the inherent uncertainty of chemical transport model projections, substantial differences in estimated emissions from volcanos, which have large interannual fluctuations, also affect the calculation of transboundary sulfur deposition in Japan. Although the deposition of acidic species has been compared between models using identical input parameters during MICS-Asia Phase III [47], these comparisons did not report the source-receptor relationship of acidic species.

In contrast to Northeast Asia, simulations of acid deposition in other part of East Asia are scarce. Studying the transboundary transport of anthropogenic sulfur emissions in Southeast Asia, Engardt et al. (2005) found that long-range transport is less efficient in this region, and that sulfur deposits in countries such as Thailand, Indonesia, Malaysia, Singapore and Brunei are largely local in origin. In contrast, 90% of all sulfur deposited in Laos, is estimated to originate from external sources, while Myanmar, Vietnam and Cambodia exhibit a mixture of local and transboundary sulfur deposition [30].

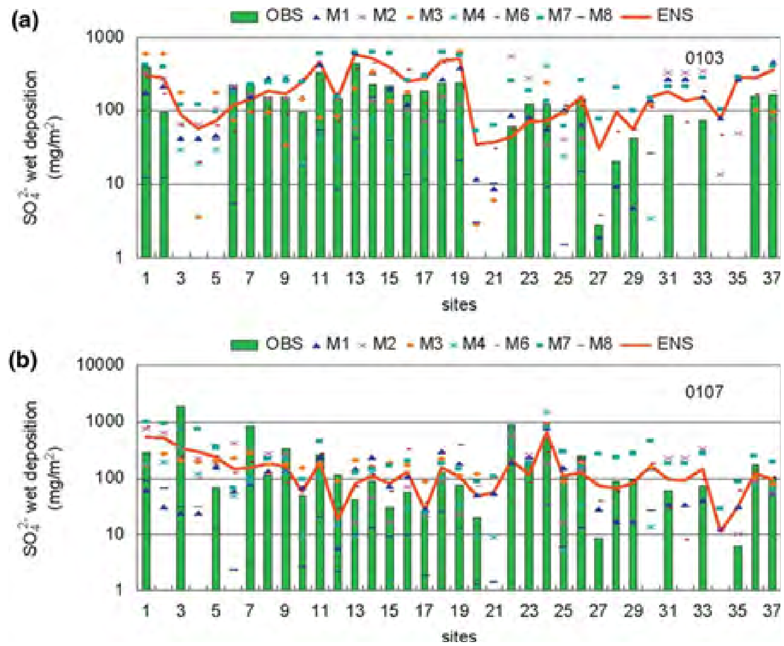

Figure 6: Comparison of wet deposition of SO42- (mg m-2 month-1) by different models at the 37 EANET sies for a March and b July 2001. The thick line represents ensemble mean, the bar represents observation and numbers on the x-axis represent the EANET monitoring site as below (Wang et al. 2008). 1. Guanyinqiao, 2. Jinyunshan, 3. Shizhan, 4. Weishuiyuan, 5. Jiwozi, 6. Hongwen, 7. Xiaoping, 8. Xiang-Zhou, 9. Zhuxian-Cavern, 10. Rishiri, 11. Tappi, 12. Ogasawara, 13. Sado-seki, 14. Happo, 15. Oki, 16. Yusuhara 17. Hedo, 18. Ijira, 19. Banryu, 20. Ulaanbaatar, 21. Terelj, 22. Metro-Manila, 23. Los-Banos, 24. Kanghwa, 25. Cheju, 26. Imsil, 27. Mondy, 28. Listvyanka, 29. Irkutsk, 30. Primorskaya, 31. Bangkok, 32. Samutprakarn, 33. Patumthani, 34. Vachiralongkorn-Dam, 35. Mae-Hia, 36. Hanoi, 37. Hoa-Binh

5.2. Critical Load Approach

The critical loads of acid deposition is defined as ‘‘the maximum deposition of acidifying compounds that will not cause chemical changes leading to long term harmful effects on ecosystem structure and function’’ [68, 88]. Critical loads formed part of the integrated Regional Air Pollution Information and Simulation model (RAINS) developed by the International Institute for Applied System Analysis (IIASA) and were used in international negotiations for sulfur emission reductions in Europe [33, 34]. Critical loads were also included in the impact module of the Asian version of the integrated assessment model, RAINS-ASIA, developed by IIASA [82]. Critical loads for acidity in Asia were calculated in the middle 1990’s by Hettelingh et al. (1995) using the Steady-State Mass Balance Method (SSMB). Although RAINS-ASIA was not used as an international negation tool throughout Asia, critical loads based on the SSMB and other methods were calculated in China [19, 112, 126, 127], Japan [85], South Korea [72, 73], Thailand [7] and eastern Russia [6] using detailed data on soil properties in each country.

In principle, a critical molar ratio of base cations to aluminum in a soil solution, (BC/Al)crit = 1 mol mol-1 has been used as an indicator of impacts on plants. Based on the concept of critical limits, critical loads have been assessed using soil mineralogy depending on specific base cation weathering rates dividing sites into, e.g., five surface geology categories [41, 85]. Soils in the most sensitive class (class 1) were derived from highly siliceous parent rocks such as quartzite and granite, and soils in the least sensitive class (class 5) were derived from parent materials with free carbonates such as limestone and aeolian deposits. Between these extremes soils could be derived from gneiss and related materials (class 2), granodiorite, schist, and related mate- rials (class 3), and gabbro, basalt, and related materials (class 4) [19, 85].

Figure 7 presents a critical load map of the maximum sulfur deposition rate of sulfur (CLmaxS) and nitrogen nutrient (CLnutN) in China (Posch et al. 2015). The most sensitive areas, with a CLmaxS lower than 200 eq ha-1 - year-1, were located in northeastern and southern China because of the low weathering rates and high base cation uptake by plants, while low CLnutN values were commonly found in the northwestern China because of very low precipitation and low net nitrogen uptake by plants. More tolerant areas, with higher CLmaxS and CLnutN values were characterized by higher weathering rates and/or base cation deposition and higher precipitation and/or soil denitrification rates, respectively.

Critical loads of acid deposition have also been calculated in Japan. The lowest values appeared in coastal areas of the Seto Inland Sea, central and northern mainland Japan and the northern Hokkaido island [85]. These results highlight that the different acidification criterion produce quite different results. In South Korea, critical loads of sulfur deposition have been mapped by Park and Lee (2001) and Park and Shim (2002), who determined the minimum and maximum critical loads for sulfur in the southeastern and northern parts of the country, respectively. The calculated values were notably different from those estimated by the RAINS-ASIA model [41]. The differences have been attributed to the more detailed and comprehensive national datasets of geology, soil, vegetation, and meteorology used by Park and Lee (2001). Bashkin and Kozlov (1999) found that the northern Thailand had the lowest critical acid load in the country (\ 200 eq ha-1 year-1). In eastern Russia, the lowest critical loads of sulfur (\ 50 eq ha-1 year-1) was recorded in in the Kamchatka Peninsula and in the area between the Yenisei and Ob rivers in northern Siberia [6].

It is expected that a reduction in SO2 emissions in East Asia will reduce the risk of acidification. However, because calcium deposition is an important indicator of acid neutralizing substances in East Asia, changes in the balance between acid deposition and base cation deposition may also affect the risk of future acidification [127, 21]. It was estimated that PM control in China would increase areas exceeding the critical loads up to 17.9% of mainland China, leading to an increase in acidification [127]. However, it was also estimated that the 15% reduction in NOx emissions in China between 2010 and 2015 would decrease the total area in East Asia that exceeds the critical nitrogen loads by 14.3% [112]. Regional impact assessments using the critical load approach could periodically evaluate the impacts and benefits of SO2 and NOx emission reductions.

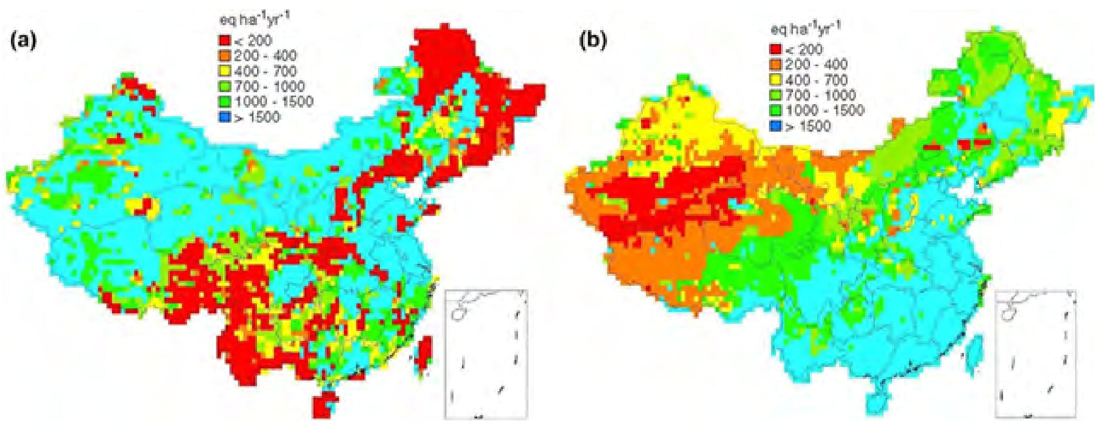

Figure 7: Gridded (0.5° 9 0.5°) 5-th percentile of critical loads of a CLmaxS and b CLnutN in China. (adapted from Posch et al. 2015)

6. Acid Deposition Policy in East Asia

6.1. History of EANET

To prevent future increases in regional SO2 and NOx emissions (cf. Figure 1a and b), efforts were made to establish an international cooperative framework for tracking regional air pollution between East Asian countries since the early 1990s. The founding of EANET was initially proposed by the Japan Environmental Agency (JEA), later renamed the Ministry of the Environment of Japan (MOEJ). Between 1993 and 1997, four conferences were held, inviting specialists and representatives from East Asian countries as well as representatives from the United States National Acid Precipitation Assessment Program (NAPAP), EMEP, United Nations Environment Pro- gramme (UNEP), World Bank and other organizations [89]. Discussions were held on the state of acid deposition in the region, its effects on ecosystems, and regional cooperation. Because monitoring methods and analytical accuracy varied significantly between East Asian countries, it was difficult to compare and evaluate the measurement data at a regional scale. Thus, specialists agreed to establish a regional monitoring network and proposed guidelines for standardized monitoring methods and analytical techniques.

These efforts led to the founding of EANET. Its objectives were to establish a common understanding of the state of acid deposition in East Asia, provide input for decision-making at national and regional levels to prevent or reduce the adverse impacts of acid deposition, and facilitate cooperation among participating countries in service thereof. The first session of the Intergovernmental (IG) Meeting of EANET was held in 1998, and initial activities involved the participation of ten countries, namely China, Indonesia, Japan, Malaysia, Mongolia, Philippines, R. O. K., Russia, Thailand, and Vietnam. The second IG Meeting was held in 2000, and initiated the regular operational phase of EANET in January 2001, based on guidelines and technical manuals for monitoring wet deposition, soil and vegetation, and inland aquatic environments. Additionally, the IG Meeting, the highest decision- making body of EANET, saw the establishment of the Scientific Advisory Committee (SAC). The UNEP Regional Office for Asia and the Pacific located in Bangkok, Thailand, was designated as the Secretariat, and the Acid Deposition and Oxidant Research Center (ADORC), later renamed the Asia Center for Air Pollution Research (ACAP), located in Niigata, Japan, was designated as the Network Center. Cambodia, Laos, and Myanmar joined EANET in 2001, 2002, and 2005, respectively, after which the organization has continued operating with 13 member countries.

As of 2019, the acid deposition monitoring network comprised 66 sites, including 26 urban, 19 rural and 21 remote sites. Data on ecological impact were collected from 19 inland aquatic sites (lakes/rivers), 31 soil and vegetation sites, and two catchment-scale sites

[28]. The Network Center operates a central data management system to compile, store and manage monitoring data in addition to conducting quality assurance/quality control (QA/QC) activities. Reports are published annually, and the compiled data are available to public upon request. To date, Periodic Reports on the State of Acid Deposition (PRSAD) in East Asia have been published in 2005, 2010, and 2015 [24].

Funding has been a matter of concern throughout the initial and formal operational phase of EANET. Dating back to discussions held at the organization's inception, participating countries are encouraged to make voluntary financial contributions, based on their economic circum- stances, in accordance with the respective national laws and regulations, and within the limits of their respective national budgets.

In April 1998, EANET's interim secretariat and interim network center were located in MOEJ and in ADORC in Niigata, Japan, respectively. During the preparatory phase, the Japanese government covered all operational costs on a voluntary basis [89]. Thereafter, the funding was still provided largely by Japan during the first several years of the operational phase, except for the local operational costs of monitoring and hosting IG and SAC Meetings, reflecting the lack of economic development in East Asia during this time (Figure 2). By 2010, the participating countries reached an agreement to share the costs more evenly based on the UN system.

6.2. Background of EANET

The Agenda 21 adopted at the 1992 UN Conference on Environment and Development (UNCED) in Rio de Janeiro, stated that the successes of CLRTAP needed to be expanded to other regions of the world. In response, the MOEJ proposed the establishment of EANET, which was expected to serve as a common knowledge basis and to enhance regional collaboration [89]. How- ever, this does not mean that EANET has followed the same trajectory as European models. This is because the process of establishing such an organization is influenced not only by environmental concerns but also including economic, social and political factors. Thus, although EANET was conceived as the counterpart of EMEP, it has characteristics that are unique to East Asia.

Because of larger intraregional geographical, economic, and social differences, it is understandable that it was more difficult to reach a consensus in East Asia than in Europe. To compensate for the large economic inequalities between East Asian countries at the organization's inception (Figure 2), incentives to participate in EANET were provided via two financial channels: direct funding provided by EANET and Official Development Assistance (ODA) provided by Japan through various existing schemes. Additionally, monitoring equipment was provided and experts were dispatched to facilitate the participation of some developing countries in EANET activities.

In addition to funding difficulties due to economic and developmental inequality, the scope of EANET was also subject to extensive debate because of geographical disparities. There are also gaps among in the science–policy inter- face and in multilateral cooperation between Northeast Asian countries [71]. Scientists have become increasingly aware of transboundary air pollutants in the region, including aerosols, photochemical oxidants, and persistent organic pollutants [71]. Therefore, Japan made a proposal to expand the scope of EANET to monitor not only acidic species but to monitor and model other air pollutants as well [89]. How- ever, no consensus was reached. Thus, EANET has been slow to grow beyond its original goal restricted to the mandate of reducing acid deposition through monitoring, joint research, and capacity building by the ‘‘core’’ budget provided by the participating countries. Activities such as emission inventory and model analysis have been discussed within the frame- work of research and training for capacity-building, which have been designated as ‘‘additional’’ activities supported by voluntary funding mainly provided by the MOEJ.

In 2000, R. O. K. initiated the Joint Research Project on Long-Range Transboundary Air Pollutants in Northeast Asia (LTP), involving R. O. K., China, and Japan. To date, the work of LTP has included the joint monitoring and modeling of transboundary air pollution, with the National Institute for Environmental Research (NIER) of R. O. K. acting as the secretariat. Given this activity, there has been debates between R. O. K. and Japan on how EANET and LTP could best collaborate.

6.3. Success and Challenges of EANET

Since the inception, EANET has accumulated reliable monitoring data on precipitation chemistry (inorganic ions) in East Asia. Various QA/QC activities have been con- ducted by the Network Center to improve the technical capabilities and skills of those involved in managing acid deposition monitoring in participating countries. A stan- dardized set of methodologies for site selection, sampling and chemical analysis has been established to improve technical conformity within the network. A series of QA/QC Guidebooks [25] have been prepared and round-robin tests of standard aqueous samples are con- ducted annually. The Network Center dispatches technical missions annually to all the participating countries to advise, assist, and improve on-site and laboratory monitoring practices. In addition to national workshops, individual training courses and fellowships for researchers are hosted annually at the Network Center.

In addition to precipitation chemistry, the concentrations of gaseous SO2, NO/NO2* (NO2* is a nitrogenous species measured as NOx-NO by a chemiluminescent NOx analyzer with a molybdenum catalyzer, which partially contains HNO3, organic nitrates and particulate NO3- in addition to NO2, since they are also reduced to NO by the catalyzer), and inorganic ions in PM have been monitored and com- piled into datasets, although QA/QC has been less stringent than that for precipitation chemistry. However, while atmospheric mass concentrations of PM2.5 and PM10, and the concentration of O3 are also relevant for acidification, the monitoring of these species is of secondary importance in EANET, and has thus far been voluntarily reported only in Japan and R. O. K.

Published in 2010, the Report on Hemispheric Transport of Air Pollutants (HTAP) emphasized the significance of air pollution from a national, regional and global perspective. Although this report intended to provide scientific motivation for the expansion of the scope of EANET from acid deposition to broader air pollution, the SAC did not make the necessary recommendations because of various considerations, including political concerns. Despite the lack of institutional expansion, an increase in the number of monitoring sites to measure these species has been strongly encouraged in recent years.

Notably, the central focus of EANET has always been acid deposition, and not the monitoring of air quality. One of the reasons for this is that in the mid-1990s, Japan and several other countries considered acid rain to be a more significant long-term threat than air pollution. However, measurements of atmospheric gas concentrations and PM have been authorized in EANET as ‘‘dry deposition monitoring’’ for which EANET adopted the inferential method [22]. In this method, the deposition flux (F) is calculated as F = C 9 Vd, where C is the atmospheric concentration, and Vd is the deposition velocity, deter- mined using meteorological data and deposition surface characteristics [8] as described in the Technical Manual on Dry Deposition Flux Estimation [23]. Evaluating the dry deposition flux requires meteorological data from each monitoring site. However, no meteorological data have been recorded outside of Japan; therefore the 3rd PRSAD includes only dry deposition estimates for Japan [24, 25]. In contrast, EANET's atmospheric concentration data, particularly those on O3, have been widely utilized in scientific research [92].

After several years of discussion, all EANET data were made available to the public, and a gesture of transparency has been highly appreciated internationally [34]. Precipitation chemistry as well as atmospheric concentration data have been widely used in studies conducted by MICS-Asia and other projects [47, 106]. The provision of reliable monitoring data has incentivezed greater collaboration between EANET scientists despite the lack of formal epistemic community within EANET as discussed later.

As previously discussed, the evaluation of monitoring data using emission inventories and modeling has not been formally conducted in EANET. Therefore, the organization has not provided any direct guidance or suggestions to participating countries on the effective management of acid deposition precursors such as SO2 and NOx. To date, EANET has not identified the acidification of forest soil outside of the Ijira Lake site in Japan, although the acidification of inland water has been reported at several sites in China, Japan, and Russia [24, 25]. Furthermore, no clear impact on biota has been recorded. The acidification of lakes and rivers, as reviewed in previous sections focusing on non-EANET data, has not led to the extinction of fishes. Although forest decline has been observed sporadically, it has been ascribed to multiple causes. For these reasons, acid deposition has never become a serious environmental issue in East Asia.

6.4. Sulfur Emissions Control Policy in China

Since the early 1990s, China has been the largest emitter of SO2 in the world [29]; SO2 emission per unit GDP are also significantly higher than those for other East Asian countries (Figure 2). However, China is the only country in East Asia that has taken measures to limit air pollution in response to acid rain [37].

The main air pollution management initiative comprised the designation of prioritized Acid Rain Control Zones and SO2 Pollution Control Zones [37]. The criteria for the designation of the Acid Rain Control Zones were a precipitation pH less than B 4.5 and sulfur deposition above critical loads. Meanwhile, SO2 Pollution Control Zone was characterized as central regional urban areas with annual average SO2 concentrations exceeding the second class (60 lg m-3) or daily concentrations exceeding the third class of the national standard (100 lg m-3) [37]. Cities in southern China with both high SO2 pollution and precipitation acidity were designated as Acid Rain Control Zones. Based on these criteria, Acid Rain Control Zones spanned 14 provinces covering most areas south of the Yangtze River, while the SO2 Pollution Control Zones included 63 cities, mostly east of Gansu Province in northeastern part of China [37]. Through initiatives such as these, China has successfully controlled SO2 emissions. Total emissions began to decrease in 2007, as evidenced by the decrease in tropo- spheric column densities observed by satellites [111]. The reduction in SO2 emissions leveled off between 2010 and 2015 as energy consumption continued to rise [86]. However, more recently, further reductions in national SO2 and NOx emissions have been achieved. In 2020, SO2 concentrations in most Chinese cities were below the national air quality limits, and between 1990s and 2018, the extent of areas experiencing acid rain decreased from over 30% to 5.5% of China's total surface area [113].

Other countries in East Asia, including Japan and R.O. K. have not reduced SO2 or NOx emissions to mitigate acid rain. Instead, significant reductions in SO2 emissions have been made to meet air quality standards based on health impacts (cf. Figure 2).

7. Epistemic Community for Science–Policy Interface in East Asia

As scientific data and knowledge are becoming increasingly important for understanding the mechanisms and processes underlying environmental issues, the role of the epistemic communities in facilitating international cooperation has been emphasized [36]. An epistemic community is defined as a network of professionals with recognized expertise, authority and policy- relevant knowledge in a particular field [35]. Scientific knowledge based on trust and consensus among the members of an epistemic community is considered to be a critical element for successful cooperation between countries in addressing contemporary environmental challenges [117].

The critical role played by epistemic communities has been demonstrated for acid rain in Europe. First, the OECD project performed model calculations to prepare ''blame matrices'', through which the transport of pollutants between countries could be quantified. Next, the concept of ''critical loads'' of ecosystem and the interactive RAINS model [4] allowed the connection of scientific knowledge with policymaking [34, 87]. In effect, the concept of critical loads and the RAINS model functioned as intermediary tools, facilitating close and dynamic interactions between those with scientific expertise and those making political decisions [87, 95].

In contrast to Europe, there was little consensus among researchers in different countries in East Asia, and there was no initiative to reach a consensus among scientists in the early stage of international cooperation in the 1990s [50]. Takahashi (2002) noted that this period did not appear to be the appropriate time for such a community to emerge in Asia. Within the EANET framework, SAC was originally created to facilitate technical discussions based on objective and scientifically valid knowledge. However, given the diverse backgrounds and competences of scientists representing the participating countries, it was not possible to reach a consensus on policy recommendations. Low- and middle-income developing countries within EAENT did not have sufficient experts with professional knowledge and skills, and were instead represented by government officials or laboratory managers at SAC meetings. During the first decade of EANET, experts from China and Vietnam were largely affiliated with their respective Ministries of Environmental Protection, and were thus influenced by political agendas. In contrast, experts coming from Japan, R. O. K., Malaysia and the Philippines largely comprised researchers in academia, who could make more objective scientific and technical recommendations.

It is worthwhile to mention that scientific collaboration had already begun in the early 1980s between scientists at Peking University and the Chinese Research Academy of Environmental Sciences, China, and the National Institute for Environmental Studies, Japan. Among other collaborative studies, a framework between X. Tang and H. Aki- moto produced a joint paper by Hashimoto et al. (1984). This early precedence of scientific collaboration helped establish the more recent epistemic community in East Asia in the field of acid deposition and air pollution.

The fragmentations of past scientific communities and the recent emergence of epistemic communities in the field of air pollution in East Asia have been discussed by Yarime and Li (2018), and Otsuka and Cheng (2020), respectively. As previously discussed, MICS-Asia has played a significant role in the development of an epistemic community in the field of acid deposition and air pollution. MICS-Asia was initiated by non-official scientists who shared common beliefs on the comparison of regional models. Annual workshops held by MICS-Asia have been funded by the MOEJ as an additional activity of EANET. Scientists in developing countries have also been invited within the fund. A substantial number of papers co-authored by researchers from different countries have been published through MICS-Asia (Phase I–III). This has helped cultivate and expand the present epistemic community to include increasing number of researchers from developing countries in the EANET region. Thus, although MICS-Asia is ostensibly aimed at improving scientific understanding, it also contributes to the development of political ties in service of regional policymaking in Asia.

Another initiative intended to foster an epistemic com- munity focused on air pollution in Asia was the Science Panel of Asia Pacific Clean Air Partnership (APCAP), which was first hosted by the UNEP Regional Office for Asia and the Pacific (ROAP) in Bangkok in 2014. APCAP is a mechanism and platform for the promotion of coordination and collaboration among various clean air initiatives in Asia and the Pacific [71, 97, 99]. This was a response to the first UN Environment Assembly, where air pollution was identified as a top priority requiring immediate action. APCAP consists of a Science Panel, which helps create a scientific community for atmospheric science, and a Joint Forum, which seeks to advise policy makers for setting targets to improve air quality in the region. As of 2021, the Science Panel comprises of internationally recognized scientists, from Austria, China, India, Japan, Nepal, Singapore, R.O. K. and Thailand invited by UNEP ROAP. Their fields of expertise include environmental sciences, including atmospheric chemistry, modeling, public health, environmental economics, and mitigation engineering. In 2019, the APCAP Science Panel and Climate and Clean Air Coalition (CCAC) (CCAC 2021) jointly published the first regional assessment report, Air Pollution in Asia and the Pacific: Science-Based Solutions [97, 99].

8. Future Challenges and Prospect

To date, acid deposition in East Asia has not caused serious damage to aquatic or terrestrial ecosystems. In contrast, air quality issues linked mainly to high levels of PM2.5 and O3 have become severe in many countries in the region [108]. The 6th Global Environment Outlook (GEO-6) [98] has described that air pollution is the main environmental contributor to the global burdens, leading to an estimated 6–7 million premature deaths and welfare loss [109]. In response, discussions on the expansion of the scope of EANET to include the monitoring of atmospheric pollutants such as PM2.5 and O3 are now underway. Presentations of monitoring data after evaluation are thought to be a useful basis for developing pollution mitigation policies in participating countries [27].

As for cooperation related to atmospheric policymaking, several frameworks have been established in East Asia. In addition to EANET, the LTP project, led by R. O. K., Japan and China has recently marked its 20th anniversary. The limited study of transboundary air pollution is expected to broaden in scope to include more general air pollution mitigation in Northeast Asia. The Northeast Asia Clean Air Partnership (NEACAP) was launched in 2018 by R. O. K. within the framework of the Northeast Asian Subregional Programme on Environmental Cooperation (NEASPEC) organized by the UN Economic and Social Commission for Asia and the Pacific (ESCAP). NEACAP is expected to promote science-based, policy-oriented cooperation between China, Japan, R. O. K., Russia and other countries. To date, roundtable meetings were held in 2018 and 2019, and the first Science and Policy Committee Meeting was held in Seoul in 2019.

Cooperation focusing on dust and sands storms (DSS) and air pollution transmission between R. O. K., Japan and China under the Tripartite Environment Minister Meeting (TEMM) has played an important role in improving environmental governance in Northeast Asia [16]. The Joint Research Group on DSS began its activities in 2008, establishing Working Group I to improve DSS forecast accuracy and Working Group II to conduct ecological restoration studies in areas undergoing desertification. In 2013, the three member states held the Tripartite Policy Dialogue on Air Pollution. Through this Policy Dialogue, Working Groups I and II were established for scientific research on (1) air pollution prevention and management, and (2) air quality monitoring forecasting, and policy, respectively. EANET, LTP, and NEACAP share some common targets; thus, future initiatives should aim to further coordinate the activities within international governance frameworks.

In terms of related environmental issues, the disturbance of the nitrogen cycle in forest ecosystems due to excess reactive nitrogen inputs from the atmosphere (‘‘nitrogen saturation’’) is still a major concern in Northeast Asia, while reduced acidification is expected with a reduction in sulfur deposition. Even though NOx emissions in Asia have declined since 2011, NH3 emissions are still gradually increasing [54]. Reflecting these emission trends, the observational data recorded at 66 sites in China by a Nationwide Nitrogen Deposition Monitoring Network (NNDMN) showed that oxidized nitrogen deposition decreased after 2010, while reduced nitrogen deposition remained approximately constant, suggesting the importance of further NH3 emission mitigation measures [107]. The reduction of NOx emissions in China seemed to have contributed to a decrease in the exceedance of critical nitrogen loads in Northeast Asia. However, different potential reactions of forest ecosystems to NH3 deposition should also be taken into consideration [112]. Furthermore, even relatively low rates of nitrogen deposition can directly impact sensitive species, such as lichens [102]. Thus, reducing atmospheric NH3 and NOx concentrations is crucial for the conservation of forest ecosystems in East Asia.

Over the past decade, climate change has become an issue of increasing concern in relation to research and policy on air pollution. Simultaneously, controlling CO2, the main contributor to the greenhouse effect, and short- lived climate pollutants (SLCPs) is a prerequisite for mitigating further global warming in the mid- and long- term future [84, 100, 101]. The most prominent SLCPs comprise CH4, O3, black carbon, and hydrofluorocarbon (HFCs). Because O3 and its precursors are air pollutants and black carbon is a component of PM2.5, reducing SLCP emissions results in the co-benefits of mitigating climate change and air pollution in a more cost-effective way than addressing these issues separately [3, 64, 80, 94]. Similarly, while controlling CH4 emissions is essential for mitigating climate change, it also decreases the hemi- spherical background concentration of O3, substantially contributing to improved urban and rural air quality. Because East Asia is the largest emitter of CO2 and SLCPs in the world, the mitigation policies in the region will have a substantial impact on future global climate change. The mitigation of SLCPs in developing countries will especially require international cooperation.

From this review, it is clear that researchers within the field of air pollution and climate change need to work more closely, for example, through joint studies. Capacity- building for monitoring and modeling also needs to be improved to study more complicated mechanisms of atmospheric chemistry and climate change. Science-based policies stemming from a regional epistemic community can go a long way to further mitigate air pollution in East Asia and climate change on a global scale. To that end, EANET and other regional frameworks have helped to foster mutual trust and cooperation between scientists and policy makers. Therefore, despite longstanding economic and political impediments to coordinated research and policy interventions in East Asia, EANET and similar initiatives are expected to play an important role in realizing a clean atmosphere for a low-carbon future in the region, based on strengthened epistemic communities.

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