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Photochemical smog in China: scientific challenges and implications for air-quality policies

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INTRODUCTION

Severe air-pollution events in many parts of China pose a major threat to health and ecosystems [1]. China's air pollution is concentrated to economically developed areas, such as Beijing–Tianjin–Hebei (BTH) and Pearl–River–Delta (PRD) [2,3]. The situation has received considerable attention in international and national media, including its secondary societal and economic impacts such as lowered productivity, reduced investments and loss of professionals who have the choice of residing elsewhere. Large efforts are today underway from the government to improve the situation by measures to reduce primary emissions (see Airborne Pollution Prevention and Control Action Plan (2013–2017), available at http://www.gov.cn). This will also affect secondary pollutants such as ozone (O_3) and particulate matter (PM) but how and to what magnitude are uncertain. The photochemically induced secondary pollutants will add to any severe local urban air pollution but require a very different approach for abatements. In this perspective view, we will address the complexity of photochemical smog while acknowledging the urge for similar descriptions on local urban air pollution as described elsewhere [1–4].

The scale of the problem is in itself a challenge and the necessary measures will affect many societal sectors [4]. In addition to reduction of primary pollutants, a long-term development of abatement strategies requires a sound scientific understanding of emissions, transformations and impacts of air pollution—*the source-to-impact relationships*—as well as technical and non-technical measures for cost-effective actions.

Air pollution has been a focus of research for decades. Two distinct types of smog have been described: the classical wintertime 'London smog', mainly caused by emissions of sulphur dioxide (SO_2) and carbonaceous particles (soot) from combustion of coal; and 'photochemical smog', where emissions of volatile organic compounds (VOCs) and nitrogen oxides (NOx), mainly from traffic, yield secondary air pollutants such as O_3. These types of smog have been extensively researched and updated tools describing the source-to-impact relationships are applied to support policies and abatement strategies. China now exhibits a mixture of these two smog types where high levels of O_3, soot, SO_2 and organic particles are key components [5]. Thus, air pollution in China is different in comparison to Europe and the USA, in terms of the specific pollutant mixtures and the processes involved in pollutant transformation. It is therefore clear that our current knowledge may not be applicable under the Chinese conditions, neither regarding photochemical air-pollution description, nor how to address this in policy–science relationships.

SCIENTIFIC CHALLENGES

Secondary chemistry transforming primary pollutants is of high relevance for Chinese photochemical smog [2,6]. In particular, formation of O_3 and PM, including Secondary Organic Aerosols (SOA), are of major concern for effects on health and ecosystems including crop yield and quality. Furthermore, the dynamics of key radicals (HO_X, RO_X, NO_3 and Cl) and their impact on O_3 and SOA are not well understood. Ozone and PM are Short Lived Climate Pollutants (SLCP), i.e. air pollutants that also influence climate. Here, there are large uncertainties about the interactions of the complex mixtures of soot–SOA–O_3 under different levels of SO_2, NO_X and VOCs [5,7]. To improve the understanding of the Chinese photochemical smog, five scientific areas are identified:

- secondary chemistry, contributing to the formation of O_3 and PM via oxidation of other pollutants (e.g. VOCs, NO_2, SO_2): the source/formation of key species such as HONO and $ClNO_2$ and their role as 'new' radical sources; secondary heterogeneous chemistry could during Chinese photochemical smog be significantly enhanced by the large surface area of PM;
- aerosol properties, including light absorption, as a result of interactions between primary particles (e.g. soot) and

co-emitted gases such as SO_2, NH_3, NO_X and VOC which are e.g. producing organic nitrates or sulphates; specifically, one needs to address synergy effects and unknown mechanisms that lead to anthropogenic enhancement on the formation of SOA;

- adaptation of chemical transport models with specific secondary chemistry to describe the Chinese photochemical smog, source apportionment and application of models on scenarios of abatement with corresponding emission changes on air pollutants, SLCP and influence on secondary photochemistry; appropriate and accurate descriptions of photochemical processes in such modelling systems are essential for valuable cost–benefit analysis, especially regarding impact on health and ecosystems;
- understanding perceptions of risks associated with secondary pollutants to address the policy–science relationship.

A common major first step to address the challenges described above is to have access to high-quality, detailed and updated emission inventories [8]. This will enable using combined model-measurement approaches to scrutinize the processes important for formation of O_3 and secondary PM. Here, chemical mechanisms cannot be reasonably evaluated in the field if the emissions are substantially in error. In particular, emissions of PM and SOA precursors from combustion and biogenic sources need careful evaluation. To address synergy effects and unknown mechanisms of SOA as well as changes in aerosol properties, the model-field observations need significant input from laboratory studies. This requires effective frameworks where specialized researchers can communicate, share knowledge and interact in a constructive way. Furthermore, the source-to-impact relationship of secondary air pollutants is in many instances indistinct, and possibly underestimated, due to its complexity and variability in time and place.

To exemplify the complexity and variability of secondary air pollution, we use the densely populated Beijing (BJ) region and Hong Kong (HK) region, as part of PRD. The air pollutants in BJ consist of a mixture with seasonally varying composition (photochemical smog in summer, haze caused by primary pollution and dust in winter) [5]. The organic aerosol (OA) originates from both primary and secondary sources with a significant SOA fraction, indicated by high amounts of oxygenated organic aerosol (OOA) in the atmosphere (see Fig. 1) [9,10]. The high abundance of acidifying gases (SO_2 and NOx), primary particles, such as soot and SOA precursors can enhance acid-catalysed heterogeneous processing of organics. HK is characterized by persistently high levels of secondary produced O_3 mainly as a result of long-range transport of air pollutants (Fig. 2) [11]. The anthropogenic enhanced levels of oxidants (e.g. O_3) lead to large SOA production. However, in contrast to Beijing, the biogenic contribution to SOA is much higher [12] and the distribution of secondary pollutants is very different (Fig. 1). These differences in effects of secondary processes on the composition of PM urge for different needs in control policies of primary emissions in various parts of China.

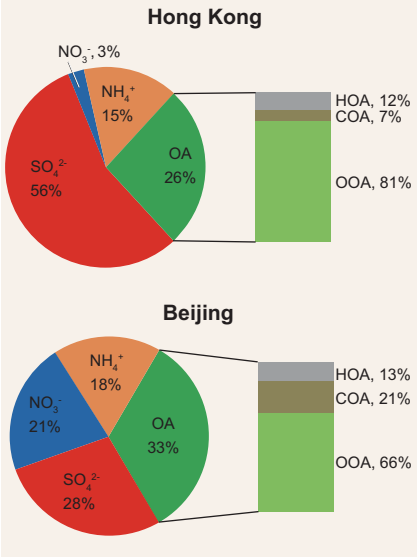

Figure 1. Chemical composition (sulphate, nitrate, ammonium and organics) in the non-refractory components in PM with a diameter less than 1 μm (NR-PM1) as measured by Aerodyne high-resolution time-of-flight aerosol mass spectrometers (HR–ToF–AMS) in Beijing and Hong Kong during summer 2011. The organic specification was done using positive matrix factorization, here described by three fractions: hydrocarbon-like organic aerosol (HOA), cooking organic aerosol (COA) and oxygenated organic aerosol (OOA), where HOA and COA are from primary sources and OOA is a major signature of SOA. There is a clear difference both for inorganic secondary pollutants (e.g. much higher fraction of SO_4^{2-} in Hong Kong) and the organic fraction (e.g. much higher fraction of COA and HOA in Beijing), while the OOA fraction is clearly dominating the organic part of the aerosol both in Beijing and Hong Kong. The figure is based on data presented in Hu *et al.* [9] and Li *et al.* [10].

CHALLENGES FOR SUCCESSFUL SCIENCE–POLICY INTERACTIONS

To ensure that scientific progress on air pollution provides benefits for policymakers, dedicated efforts are needed for a joint understanding of the roles and responsibilities of scientists and policymakers including efficient transfer of knowledge. The UN-ECE Convention

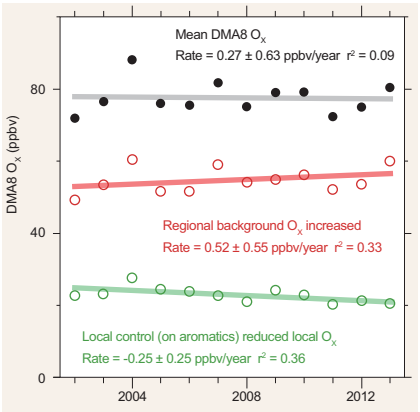

Figure 2. Autumn mean (September–November) daily maximum 8-h average (DMA8) O_X (O_3+NO_2) recorded by 11 air-quality monitoring stations in Hong Kong (black symbols). Increasing background O_X (red symbols) has offset local reduction in O_X (green symbols), illustrating the need to address long-range transport in science–policy interaction especially for secondary pollutants such as O_3. There is a significant (*P*-value 0.05) trend for local reduction while the increasing background was of more marginal statistical significance (*P*-value 0.06). Detailed discussions on trend analysis are found in Xue *et al.* [11]. Figure adapted with permission from Xue *et al.* [11]; copyright (2014) American Chemical Society.

on Long Range Transboundary Air Pollution (LRTAP) was signed in 1979 (www.unece.org/env/lrtap/) and has been important for the development of science and policies related to air pollution in Europe [8]. A key feature of the LRTAP process is the scientist–policymaker interaction in a series of tiers, forming a pyramid of knowledge based on hundreds of scientists covering a wide range of subjects within air chemistry and impacts. Integrated assessment models bring together this information and the costs of emission control to derive least-cost options for emission reductions to meet environmental targets. The strength is that, in each step of the process, scientists are responsible to produce results which are useful to reach policy goals, and the policymakers feel confident that decisions are based upon a consensus from numerous researchers. Although the structure and contents of the LRTAP cannot be directly transferred to China, there is much to gain from the experience accumulated over the 38 years of the convention.

As exemplified, air pollution in BJ and HK has distinctive characteristics but still contains aspects of the unique air-pollution situations frequently occurring in China. Here, the rapidly increasing knowledge from laboratory and field observations should be implemented in models to support assessment and scenario development extending to interdisciplinary knowledge production. Thus, based on the findings, one can further scrutinize the experience from LRTAP in view of a Chinese concept to support the required future integration of science and policy. Such work will include a structured approach for communication of air-pollution risks to human health and ecosystems, and ways to reduce the risks. A reduction of secondary air pollution will potentially affect a wide range of stakeholders in industry, government and the general population. Therefore, the anchoring of common understandings of risks is important, spurring stakeholder engagement in development of required abatement measures for secondary pollutants.

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