Acid gases and aerosol measurements in the UK (1999–2015): regional distributions and trends

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Abstract. The UK Acid Gases and Aerosol Monitoring Network (AGANet) was established in 1999 (12 sites, increased to 30 sites from 2006), to provide long-term national monitoring of acid gases (HNO₃, SO₂, HCl) and aerosol components (NO₃⁻, SO₄²⁻, Cl⁻, Na⁺, Ca²⁺, Mg²⁺). An extension of a low-cost denuder-filter pack system (DELTA) that is used to measure NH₃ and NH₄⁺ in the UK National Ammonia Monitoring Network (NAMN) provides additional monthly speciated measurements for the AGANet. A comparison of the monthly DELTA measurement with averaged daily results from an annular denuder system showed close agreement, while the sum of HNO₃ and NO₃⁻ and the sum of NH₃ and NH₄⁺ from the DELTA are also consistent with previous filter pack determination of total inorganic nitrogen and total inorganic ammonium, respectively. With the exception of SO₂ and SO₄²⁻, the AGANet provides, for the first time, the UK concentration fields and seasonal cycles for each of the other measured species. The largest concentrations of HNO₃ and SO₂, and aerosol NO₃⁻ and SO₄²⁻ are found in southern and eastern England and smallest in western Scotland and Northern Ireland, whereas HCl are highest in southeastern, south-western, and central England, that may be attributed to dual contribution from anthropogenic (coal combustion) and marine sources (reaction of sea salt with acid gases to form HCl). Na⁺ and Cl⁻ are spatially correlated, with largest concentrations at coastal sites, reflecting a contribution from sea salt. Temporally, peak concentrations in HNO₃ occurred in late winter and early spring attributed to photochemical processes. NO₃⁻ and SO₄²⁻ have a spring maxima that coincides with the peak in concentrations of NH₃ and NH₄⁺, and are therefore likely attributable to formation of NH₄NO₃ and (NH₄)₂SO₄ from reaction with higher concentrations of NH₃ in spring. By contrast, peak concentrations of SO₂, Na⁺, and Cl⁻ during winter are consistent with combustion sources for SO₂ and marine sources in winter for sea salt aerosol. Key pollutant events were captured by the AGANet. In 2003, a spring episode with elevated concentrations of HNO₃ and NO₃⁻ was driven by meteorology and transboundary transport of NH₄NO₃ from Europe. A second, but smaller episode occurred in September 2014, with elevated concentrations of SO₂, HNO₃, SO₄²⁻, NO₃⁻, and NH₄⁺ that was shown to be from the Icelandic Holuhraun volcanic eruptions. Since 1999, AGANet has shown substantial decrease in SO₂ concentrations relative to HNO₃ and NH₃, consistent with estimated decline in UK emissions. At the same time, large reductions and changes in the aerosol components provide evidence of a shift in the particulate phase from (NH₄)₂SO₄ to NH₄NO₃. The potential for NH₄NO₃ to release NH₃ and HNO₃ in warm weather, together with the surfeit of NH₃ also means that a larger fraction of the reduced and oxidized N is remaining in the gas phase as NH₃ and HNO₃ as indicated by the increasing trend in ratios of NH₃ : NH₄⁺ and HNO₃ : NO₃⁻ over the 16-year period. Due to different removal rates of the component species by wet and dry deposition, this change is expected to affect spatial patterns of pollutant deposition with consequences for
sensitive habitats with exceedance of critical loads of acidity and eutrophication. The changes are also relevant for human health effects assessment, particularly in urban areas as NH$_4$NO$_3$ constitutes a significant fraction of fine particulate matter (<2.5 µm) that are linked to increased mortality from respiratory and cardiopulmonary diseases.

1 Introduction

Monitoring the atmospheric concentrations of acid gases and their aerosol reaction products is important for assessing their effects on human health, ecosystems, long-range transboundary transport and global radiative balance. Concentration data are necessary for quantifying long-term trends and spatial patterns, understanding gas–aerosol phase interactions, and estimating the contributions of different pollutants to dry deposition fluxes (ROTAP, 2012; AQEG, 2013a; Colette et al., 2016), as well as to provide data for testing the performance of atmospheric models (e.g. Chemel et al., 2010; Vieno et al., 2014, 2016). Acid gases in the atmosphere include sulfur dioxide (SO$_2$), nitrogen oxides (NO$_x$), nitric acid (HNO$_3$), hydrochloric acid (HCl) and nitrous acid (HONO). Secondary inorganic aerosols (SIA) include sulfate (SO$_4^{2-}$), nitrate (NO$_3^-$), chloride (Cl$^-$) and nitrite (NO$_2^-$) that are formed from reactions of SO$_2$ and NO$_x$ (and HNO$_3$, a secondary product of NO$_x$) with ammonia (NH$_3$) in the atmosphere. These aerosols make an important contribution to concentrations of particulate matter (PM) in the UK (15%–50 % of the mass of atmospheric PM) and constitute a significant fraction of fine particles that are less than 2.5 µm in diameter (PM$_{2.5}$) implicated in harming human health (AQEG, 2012, 2013b). In addition, base cations in aerosol are also of interest to estimate the extent to which acidity is neutralized and to estimate the contribution of marine influences (ROTAP, 2012; Werner et al., 2011).

Anthropogenic emissions of SO$_2$, NO$_x$, HCl, and NH$_3$ in the UK declined by 81 %, 51 %, 87 %, and 13 %, respectively, over the period 1999 to 2015 (NAEI, 2018). Despite the success in mitigating SO$_2$ emissions however, sulfur still remains a pollutant of national importance, because reduction in sulfur deposition in remote sensitive areas have been more modest than close to major sources (ROTAP, 2012). HCl was also recently identified as another important acidifying pollutant for sensitive habitats (Evans et al., 2011). Emissions of HCl (from coal burning in power stations) have however declined to very low levels (from 74 kt in 1999 to 9 kt in 2015), although it could still pose a threat to habitats close to these sources. For NO$_x$, the more modest decrease in emissions reflects difficulties in their abatement, while for NH$_3$, the decrease to date is largely a result of changes in animal numbers (NAEI, 2018).

With the decline in SO$_2$ emissions and deposition, the large number of reactive nitrogen compounds in the atmosphere are assuming greater importance owing to the complexities of the global N cycle and associated challenges in their abatement. These include the gas phase components NH$_3$, with over 80 % estimated from agricultural emissions (EEA, 2017) and nitrogen oxides (NO, NO$_x$) from combustion, the secondary gas phase reaction products HNO$_3$, HONO and PAN (peroxycetyl nitrate), and particulate phase components ((NH$_4$)$_2$SO$_4$, NH$_4$HSO$_4$, and NH$_4$NO$_3$) formed by the reaction between NH$_3$ and acid gases (AQEG, 2012). Ammonia and the N-containing aerosols are known to cause nitrogen enrichment and eutrophication, as well as contributing to acidification processes (Sutton et al., 2011). Oxidized nitrogen species (NO$_x$) are precursors to ground-level O$_3$ formation, while the production of acids (HNO$_3$, HONO) and PAN in the atmosphere affects air quality and is damaging both to human health and to vegetation (Cowling et al., 1998; Bobbink et al., 2010).

In Europe, air pollution policies regarding acidification and nitrogen eutrophication apply the “critical loads approach” (Bull, 1995; Gregor et al., 2001), which requires that atmospheric deposition inputs be mapped at an appropriate scale for the assessment of effects. In parallel, the “critical levels” of concentrations addresses the direct impacts of concentrations of nitrogen components in the atmosphere (Bull, 1991; Gregor et al., 2001; Cape et al., 2009). Quantifying the dry deposition of reactive nitrogen compounds is a major challenge and a key source of uncertainty for effects assessment (Dentener et al., 2006; Flechard et al., 2011; Schrader et al., 2018; Sutton et al., 2007). While deposition may be estimated using atmospheric transport and chemistry models (e.g. Dore et al., 2015; Flechard et al., 2011; Smith et al., 2000), air concentration data at sufficient spatial resolution are needed, both to assess the atmospheric models and provide input data for estimating deposition using inferential models.

In light of policies to reduce atmospheric emissions, e.g. the amended 2012 Gothenburg Protocol (UNECE, 2018) and the revised National Emissions Ceilings Directive (NECD, EU Directive 2016/2284) (EU, 2016), it is important to assess long-term trends in the measured pollutants, since this provides the only independent means to assess the effectiveness of any abatement policies. Both these international agreements set emissions reduction commitments for SO$_2$, NO$_x$ and NH$_3$, of 59 %, 42 %, and 6 %, respectively, by 2020 (with 2005 as base year) and includes PM$_{2.5}$ for the very first time. Under the 2016 NECD, further reduction commitments of 79 % (SO$_2$), 63 % (NO$_x$), and 19 % (NH$_3$) are also set for the EU 28 countries from 2030. Since emissions of these gases come from different sources, emissions controls require very different strategies, making it important to monitor and assess the relative concentrations and deposition of nitrogen and sulfur compounds.

The spatial and temporal patterns of gases and particulate phases of these pollutants differ substantially. Although it is widely acknowledged that speciation between reactive gas
and aerosol measurement is critical, there are few national long-term monitoring programmes dedicated to measuring their concentrations and dry depositions separately at high spatial resolution (Tørseth et al., 2012). Across Europe, the European Monitoring and Evaluation Programme (EMEP, 2014) continues to recommend using a daily filter pack sampling method to measure oxidized nitrogen (total inorganic nitrate, TIN) and reduced nitrogen (total inorganic ammonia, TIA) (Tørseth et al., 2012; Colette et al., 2016). The filter pack method is generally considered as robust for measuring SO$_2$ and SO$_4^{2-}$ concentrations (EMEP, 2014; Hayman et al., 2006; Sickles et al., 1999). However, many papers have shown that there are potential artefacts in filter-pack sampling for HNO$_3$ and HCl, due to interactions with NH$_3$ and the volatility of NH$_4$NO$_3$ and NH$_4$Cl aerosol (Pio, 1992; Sickles et al., 1999; Cheng et al., 2012). Results from EMEP filter pack measurements are therefore reported as TIN and TIA, due to phase uncertainties in the method (Tørseth et al., 2012). This has been complemented by daily measurements of HNO$_3$ and NO$_3^-$ using annular denuders (Allegreni et al., 1987; EMEP, 2014) that are made at a restricted number of sites because of the resources required. In North America, filter pack sampling is also used in weekly measurements of sulfur and nitrogen species in the CASTnet (Clean Air Status Trends Network) national monitoring network of 95 sites across the contiguous USA, Canada, and Alaska (https://www.epa.gov/castnet, last access: 25 October 2018). At a small number of CASTnet sites, hourly measurements of water-soluble gases and aerosols are made with the Monitor for AeRosols and GAses in ambient air (MARGA) system (Rumsey and Walker, 2016).

High time-resolution measurements of gases and aerosols are useful at selected locations for detailed analysis and model testing, but the high costs and resources required for these measurements make them unsuitable for the assessment of long-term trends at many sites, particularly where spatial patterns are required. To achieve this, a larger number of sites operated at lower time-resolution is needed. In the UK, the Eutrophying and Acidifying Atmospheric Pollutants (UKEAP) network provides long-term measurements for the UK rural atmospheric concentrations and deposition of air pollutants that contribute to acidification and eutrophication processes (Conolly et al., 2016). UKEAP is comprised of two EMEP supersites and four component networks: precipitation network (Precip-net), NO$_2$ diffusion tube network (NO$_2$-net), National Ammonia Monitoring Network (NAMN), and the Acid Gases and Aerosol Network (AGANet). At the two EMEP supersites (Auchencorth and Harwell – relocated to Chilbolton in 2016), semi-continuous hourly speciated measurements of reactive gases and aerosols are made with the MARGA system (Twigg et al., 2016). These measurements are contributing to the validation and improvement of atmospheric models, such as FRAME (Dore et al., 2015) and EMEP4UK (Vieno et al., 2014, 2016) that are used to develop and provide the evidence base for air quality policies, both nationally and internationally.

The long-term dataset of monthly speciated measurements from the AGANet (1999–2015) are analysed in this paper to provide a comprehensive assessment of the spatial, temporal, and long-term trends in atmospheric concentrations of the acid gases HNO$_3$, SO$_2$, HCl and related aerosol components NO$_3^-$, SO$_4^{2-}$ and Cl$^-$ (and also base cations Na$^+$, Ca$^{2+}$, and Mg$^{2+}$) across the UK, together with an assessment of the DELTA denuder-filter pack sampling method (Sutton et al., 2001b; Tang et al., 2009) as compared with other sampling techniques. To aid interpretation of the relative changes and trends in the acid gases and aerosols, NH$_3$ and particulate NH$_4^+$ data from the NAMN (Tang et al., 2018) are included, since atmospheric NH$_3$ is a major interacting precursor gas in neutralization reactions with the acid gases.

2 Methods

2.1 Acid Gases and Aerosol monitoring Network (AGANet)

The UK Acid Gases and Aerosol Network (AGANet), known previously as the nitric acid monitoring network, was started in September 1999 under the Acid Deposition Monitoring Network (ADMN, Hayman et al., 2007) to deliver for the very first time, long-term monthly speciated measurement data on gaseous HNO$_3$ and particulate NO$_3^-$ across the UK. Other acid gases (SO$_2$, HCl) and aerosols (SO$_4^{2-}$, Cl$^-$, plus base cations Na$^+$, Ca$^{2+}$, Mg$^{2+}$) are also measured and reported.

AGANet and NAMN are closely integrated, with AGANet established at a subset of NAMN sites to provide additional speciated measurements of the acid gases and aerosol components. To improve on national coverage, the number of sites in AGANet was increased in 2006 from 12 to 30 (Fig. 1, Table 1). At the same time, the Rural Sulfur Dioxide Monitoring Program ceased, replaced by SO$_2$ and SO$_4^{2-}$ measurements made under the expanded AGANet (Hayman et al., 2007). A broad spatial coverage of the UK is provided by the AGANet sites, with a focus on sites providing parallel information on other air pollutants (e.g. collocation with the Automatic Urban and Rural Network that provides compliance monitoring against the Ambient Air Quality Directives (https://uk-air.defra.gov.uk/networks/, last access: 25 October 2018) and ecosystem assessments (e.g. Environmental Change Network, http://www.ecn.ac.uk/, last access: 25 October 2018) (Monteith et al., 2016).

2.2 Extended DELTA methodology for sampling acid gases and aerosol in AGANet

A low-cost manual denuder-filter pack method, DELTA (DE- nuder for Long-Term Air sampling) implemented in the
NAMN for measurement of NH$_3$ gas and aerosol NH$_4^+$ (Sutton et al., 2001a, b; Tang et al., 2018) is extended to provide additional simultaneous monthly time-integrated average concentrations of acid gases (HNO$_3$, SO$_2$, HCl) and particulate phase NO$_3^-$, SO$_4^{2-}$, Cl$^-$, Na$^+$, Ca$^{2+}$, and Mg$^{2+}$ for the AGANet (Conolly et al., 2016; Tang et al., 2015).

The DELTA method used in AGANet has also been applied in an extensive European-scale network of 58 sites to deliver 4 years of atmospheric concentrations and deposition data for reactive trace gas and aerosols from 2006 to 2009 (Tang et al., 2009; Flechard et al., 2011). Detailed descriptions of the DELTA method are provided by Sutton et al. (2001b) and by Tang et al. (2009, 2015). In brief, a small air pump is used to provide low sampling rates of 0.2–0.4 L min$^{-1}$, and air volumes are measured by a high-sensitivity diaphragm gas meter. By sampling air slowly, the method is optimized for monthly measurements, with sufficient sensitivity to resolve low concentrations at clean background sites (e.g. LOD = 0.05 µg m$^{-3}$ for HNO$_3$ for monthly sampling; see Tables S1, S2). In addition, the power requirement is very small, and low voltage versions (using 6 and 12 V micro-air pumps) of the system powered by wind-solar energy operate at some remote sites.

An extended denuder-filter pack sampling train is used to provide speciated sampling of reactive gases and aerosols (Fig. S1) (Tang et al., 2009, 2015). A Teflon inlet (2.8 cm long) at the front end ensures development of a laminar air stream (Table S3), followed by a first pair of K$_2$CO$_3$ and glycerol coated denuders to collect HNO$_3$, SO$_2$, and HCl, a second pair of citric acid coated denuders to collect NH$_3$ and a 2-stage filter pack at the end to collect aerosol components. Stage 1 of the filter pack is a cellulose filter impregnated with K$_2$CO$_3$ and glycerol to collect NO$_3^-$, SO$_4^{2-}$, Cl$^-$, Na$^+$, Ca$^{2+}$, and Mg$^{2+}$, with evolved aerosol NH$_4^+$ from this filter collected on the stage 2 citric acid impregnated filter. The separation of gases and aerosol is achieved by higher diffusivities of reactive gases and aerosol NH$_3$, Na$^+$, Ca$^{2+}$, and Mg$^{2+}$, HCl) and particulate phase NO$_3^-$, SO$_4^{2-}$, Cl$^-$, Na$^+$, Ca$^{2+}$, and Mg$^{2+}$ for the AGANet (Conolly et al., 2016; Tang et al., 2015).

### Table 1. List of sites in the UK Acid Gas and Aerosol Network (AGANet) with details of locations, start dates, and UK-AIR ID (https://uk-air.defra.gov.uk/networks/network-info?view=aganet, last access: 23 October 2018).

| Site name                  | UK-AIR ID | Latitude | Longitude | Start        |
|----------------------------|-----------|----------|-----------|--------------|
| Barcombe Mills             | UKA00069  | 50.9191  | 0.0486    | Apr 2000     |
| Bush OTC                   | UKA00128  | 55.8623  | -3.2058   | Sep 1999     |
| Cwmystwyth                 | UKA00325  | 52.3524  | -3.8053   | Sep 1999     |
| Eskdalemuir                | UKA00130  | 55.3153  | -3.2061   | Sep 1999     |
| Glensaugh                  | UKA00348  | 56.9072  | -2.5594   | Sep 1999     |
| High Muffles               | UKA00169  | 54.3349  | -0.8086   | Sep 1999     |
| Lough Navar                | UKA00166  | 54.4395  | -7.9003   | Oct 1999     |
| Rothamsted                 | UKA00275  | 51.8065  | -0.3604   | Sep 1999     |
| Stoke Ferry                | UKA00317  | 52.5599  | 0.5061    | Sep 1999     |
| Strathvaich                | UKA00162  | 57.7345  | -4.7766   | Sep 1999     |
| Sutton Bonington           | UKA00312  | 52.8366  | -1.2512   | Sep 1999     |
| Yarner Wood                | UKA00168  | 50.5976  | -3.7165   | Sep 1999     |

New sites added from January 2006

| Site name                  | UK-AIR ID | Latitude | Longitude | Start        |
|----------------------------|-----------|----------|-----------|--------------|
| Auchencorth Moss           | UKA00451  | 55.7922  | -3.2429   | Jan 2006     |
| Caenby                     | UKA00492  | 53.3979  | -0.5074   | Feb 2006     |
| Carradale                  | UKA00389  | 55.5825  | -5.4962   | Jan 2006     |
| Detling                    | UKA00481  | 51.3079  | 0.5827    | Feb 2006     |
| Edinburgh St Leonards      | UKA00454  | 55.9456  | -3.1822   | Jan 2006     |
| Goonhilly                  | UKA00056  | 50.0506  | -5.1815   | Jan 2006     |
| Halladale                  | UKA00314  | 58.4124  | -3.8758   | Jan 2006     |
| Harwell                    | UKA00047  | 51.5711  | -1.3253   | May 2006     |
| Hillsborough               | UKA00293  | 54.4525  | -6.0833   | Jan 2006     |
| Ladybower                  | UKA00171  | 53.4034  | -1.7520   | Feb 2006     |
| Lagganlia                  | UKA00290  | 57.1110  | -3.8921   | Jan 2006     |
| Lerwick                    | UKA00486  | 60.1392  | -1.1853   | Jan 2006     |
| London, Cromwell Road 2    | UKA00370  | 51.4955  | -0.1787   | Jan 2006     |
| Moorhouse                  | UKA00357  | 54.6901  | -2.3769   | Jan 2006     |
| Narberth                   | UKA00323  | 51.7818  | -4.6915   | Mar 2006     |
| Plas y Brenin              | UKA00493  | 53.1018  | -3.9179   | May 2006     |
| Roseaund                   | UKA00491  | 52.1214  | -2.6363   | Jan 2006     |
| Rum                        | UKA00276  | 57.0100  | -6.2718   | Feb 2006     |
active gases to the denuder walls where they react with the chemical coating and are retained, whereas aerosol components pass through and are retained by post-denuder filters (Ferm, 1979). In this approach, potential artefacts caused by phase interactions associated with filter packs and bubblers are avoided (e.g. Sickles et al., 1999). A particle size cutoff of around 4.5 µm was estimated for the DELTA air inlet (Tang et al., 2015). The DELTA will therefore also sample fine mode aerosols in the PM$_{2.5}$ fraction, as well as some of the coarse mode aerosols < PM$_{4.5}$.

Na$_2$CO$_3$ is reported to be an effective sorbent for acid gases, allowing simultaneous collection of HNO$_3$, SO$_2$, and HCl on denuders (e.g. Ferm, 1986), but since the measurement of aerosol Na$^+$ is also of key interest in AGANet, a K$_2$CO$_3$ coating is used instead to eliminate possible Na$^+$ contamination from Na$_2$CO$_3$. Glycerol is added to increase adhesion, stabilize the base coating (Ferm, 1986; Finn et al., 2001) and to minimize potential oxidation of nitrite that is also collected on the denuder to nitrate in the presence of atmospheric oxidants such as ozone (Allegrini et al., 1987; Perrino et al., 1990). The lengths of denuders (borosilicate glass tubes 10 and 15 cm long to capture > 95% of NH$_3$ and acid gases, respectively) in the sampling train was calculated according to the procedures described by Sutton et al. (2001b), based on the calculations derived by Gormley and Kennedy (1948) and Ferm (1979); see Table S3. All sites were set up as “outdoor” systems sampling directly from the atmosphere, avoiding potential adsorption losses (in particular HNO$_3$, which is highly surface active) and artefacts in air inlet lines. The sampling train is installed inside a simple watertight housing (Fig. S1), which is mounted on a steel post in the desired location. A low density polyethylene funnel (89 mm aperture) is placed at the inlet as a rain shelter, and sampling height is approx. 1.5 m.

2.3 Chemical analysis

K$_2$CO$_3$ / glycerol-coated denuders and aerosol filters are extracted into 5 mL of deionized H$_2$O for analysis. Anions (NO$_3^-$, SO$_4^{2-}$, and Cl$^-$) in the denuder and filter extracts are analysed by ion chromatography (IC). Base cations Na$^+$, Mg$^{2+}$, and Ca$^{2+}$ from the filter extracts were analysed by IC between 1999–June 2008 and by inductively coupled plasma-optical emission spectroscopy (ICP-OES/ICP-AES) from July 2008. Citric acid coated denuders and filter papers are also extracted into deionized H$_2$O (3 and 4 mL, respectively), with analysis of NH$_4^+$ performed on a high sensitivity ammonia flow injection analysis system, as described in Tang et al. (2018).

Up to June 2009, analyses were carried out at Harwell Laboratory (Hayman et al., 2007) and from July 2009 at CEH Lancaster (Conolly et al., 2016). The limit of detection (LOD) for the DELTA method for the different components are calculated by analysing a series of laboratory blanks. The mean and standard deviation of the results are calculated and the LOD is calculated as 3 times the standard deviation divided by 15 m$^3$, the typical volume of air sampled over a month by the DELTA system. Details of changes in laboratory, analytical methods, and LODs for the gases and aerosols are summarized in Tables S1 and S2, respectively.

2.4 Calculation of air concentrations

The air concentration ($\chi_a$) of a gas or aerosol is calculated according to Eq. (1) (see Sutton et al., 2001b, Tang et al., 2018):

$$\chi_a = \frac{Q}{V},$$

(1)

where $Q$ is the amount of a gas or aerosol collected on a denuder or aerosol filter, and $V$ is the volume of air sampled (from gas meter, typically 15 m$^3$ in a month).

The denuder capture efficiency for each of the gas is calculated by comparing the concentrations of the individual gases in the denuder pairs (Eq. 2) and are applied in an infinite se-
ries correction on the raw data to provide corrected air concentrations \( (\chi_{a}(\text{corrected})) \) according to Eq. (3) (see Sutton et al., 2001b; Tang et al., 2018).

\[
\text{Denuder capture efficiency (\% CE)} = \frac{100 \times \text{Denuder 1}}{\text{Denuder 1} + \text{Denuder 2}}
\]

\[
\chi_{a}(\text{corrected}) = \chi_{a}(\text{Denuder 1}) \times \frac{1}{1 - \chi_{a}(\text{Denuder 2}) \chi_{a}(\text{Denuder 1})}
\]

Sutton et al. (2001b) and Tang et al. (2003) have shown that this procedure provides an important quality control, flagging up occurrences of poorly coated denuders and/or sampling issues. With denuder capture efficiency better than 90 %, the correction represents < 1 % of the corrected air concentration of the gas. Below 60 %, the correction is large (> 50 %) and is not applied, and the air concentration is then calculated as the sum of concentrations of the denuder pair. The amount of correction for gas not captured that is added to the corrected gas concentration, is subtracted from the estimated aerosol concentrations of matching anions and cations (see Tang et al., 2018).

2.5 Data quality control

The following data quality checks are applied to the network data, as part of the network quality management system (Tang and Sutton, 2003; Conolly et al., 2016).

i. Air flow rate (0.2–0.4 L min\(^{-1}\)) – where this is below the expected range for a sampling period, the data are flagged as valid but failing the QC standard.

ii. Denuder capture efficiency – where this is less than 75 % for a sample, the data are flagged as valid but less certain.

iii. Ion balance checks – close agreement expected between NH\(^+_4\) and the sum of NO\(^-_3\) and 2 × SO\(^-_4\)\(^2\), as NH\(_3\) is neutralized by HNO\(_3\) and H\(_2\)SO\(_4\) to form NH\(_4\)NO\(_3\) and (NH\(_4\))\(_2\)SO\(_4\), respectively (Conolly et al., 2016), and for Na\(^+\) and Cl\(^-\), as these are marine (sea salt) in origin.

iv. Screening the whole dataset for sampling anomalies and outliers, e.g. due to contamination or other issues.

2.6 HNO\(_3\) measurement artefact and correction

Tang et al. (2009, 2015) have identified that HNO\(_3\) concentrations (NO\(_3\)\(^-\) on denuders assumed to be from HNO\(_3\)) may be overestimated on carbonate coated denuders, due to partial co-collection of other oxidized nitrogen components such as nitric acid (HONO). In the case of HONO, this collects on the denuder carbonate coating as nitrite (NO\(^-_2\)), but oxidizes to nitrate (NO\(^+_3\)) in the presence of oxidants such as ozone (Bytnerowicz et al., 2005) which can result in a positive interference in HNO\(_3\) determination (Tang et al., 2009, 2015). Other oxidized nitrogen species present in the atmosphere such as peroxyacetyl nitrate (PAN) and nitrogen oxides (NO\(_x\)) can also potentially contribute to a further small interference (Allegrini et al., 1987; Bai et al., 2003). Based on the tests of Tang et al. (2015), raw HNO\(_3\) data are corrected with an empirical factor of 0.45 which is estimated to be uncertain by ±30 %. Apart from where stated, all HNO\(_3\) data reported in this study have the 0.45 correction factor applied.

2.7 Time series trend analyses

Statistical trend analyses using both parametric linear regression (LR) and non-parametric Mann–Kendall (MK) (Gilbert, 1987; Chatfield, 2016) tests were performed on annually averaged data from AGANet, and on a subset of annually averaged data from NAMN made at the same AGANet sites. The datasets are considered sufficiently long-term (> 10 years) and produced by a consistent method for effective statistical trend analyses. Both the LR and MK approaches are widely adopted for trend analyses in long-term atmospheric data (e.g. Meals et al., 2011; Colette et al., 2016; Jones and Harrison, 2011; Marchetto et al., 2013; Hayman et al., 2007; Conolly et al., 2016), and were used in a recent trend assessment of atmospheric NH\(_3\) and NH\(^+_4\) data (1998–2014) from the NAMN (Tang et al., 2018). As described in Tang et al. (2018), LR tests were performed using R, and MK tests used the R “Kendall” package (McLeod, 2015), with estimation of the MK Sen’s slope (fitted median slope of a linear regression joining all pairs of observations) and confidence interval of the fitted trend using the R “Trend” package (Pohlert, 2016). Results from both tests provides an indication of uncertainty associated with the choice of approach. Since there was no difference between either tests, MK results only are presented and discussed in the paper. A comparison of trend analyses from both approaches is however provided in supplementary materials (Figs. S6 and S7 and Tables S4–S6).

3 Results and discussion

3.1 Performance of the DELTA method

This section presents the performance of the DELTA measurements, including a comparison with other air sampling methods and networks. Replicated sampling with the DELTA method were also made to assess measurement reproducibility and this is shown for example for the Bush OTC site in Scotland (UKA00128). A comparison of the parallel measurements (Fig. 2) showed good reproducibility in the method, with close agreement for all components (e.g. mean difference of < ±3 % for all components and ±6 % for HCl).
Figure 2. Comparisons of parallel measurement of monthly (a) atmospheric reactive gases (HNO₃, SO₂, HCl, and NH₃) and (b) particulate (NO₃⁻, SO₄²⁻, Cl⁻, and NH₄⁺) concentrations from duplicate DELTA sampling at the UK Acid Gas and Aerosol Monitoring Network (AGANet) and National Ammonia Monitoring Network (NAMN) site Bush OTC (UKA00128) in southern Scotland for the period 1999 to 2015. (c) A summary of the regression analyses. Each point represents a comparison between the paired monthly DELTA measurements.

3.1.1 Comparison with daily annular denuder measurements

An assessment of the DELTA method for NH₃ has previously been reported by Sutton et al. (2001b). Following the extension to additionally sample acid gases and aerosols, the modified system was compared with independent daily measurements from an annular denuder system (ADS). The ADS (ChemspecTM model 2500 air sampling system, R&P Co. Inc.) was operated at Barcombe Mills in southern England (UKA00069) alongside the AGANet DELTA monthly measurements for a period of 18 months. Due to significant instrument and local site issues resulting in low data capture with the ADS, only 11 months of data were available for intercomparison. The sampling train used in the ADS consisted of 2 K₂CO₃/glycerol-coated annular denuders (same coating as AGANet DELTA), 2 citric acid-coated annular denuders; a cyclone with 2.5 µm cut-off, followed by a 2-stage filter pack containing a 2 µm PALL Zefluor teflon membrane (collection of NO₃⁻, SO₄²⁻, Cl⁻, and NH₄⁺) and a 1 µm PALL Nylasorb nylon membrane (collection of evolved NO₃⁻), with a sampling rate of 10 L min⁻¹. To compare against the monthly DELTA measurements, daily ADS values were averaged to the corresponding monthly periods, with results summarized in Fig. 3.

In the measurement of gases, HNO₃ determination on DELTA (mean = 1.56 HNO₃ µg m⁻³, n = 11) was on average 23 % higher than the ADS (mean = 1.31 HNO₃ µg m⁻³, n = 11). Since both methods used the same carbonate coating on the denuders to sample acid gases, the HNO₃ data here have not been corrected with the empirical factor described in Sect. 2.6. Nitrous acid (HONO) was found to be close to or below limit of detections for most of the DELTA measurements (mean = of 0.03 HONO µg m⁻³), compared with a significantly higher concentration (mean = 0.41 HONO µg m⁻³) from the ADS. Since the sampling period of the ADS is daily, any HONO collected as nitrite on the ADS is likely to remain as nitrite and not oxidized to
Figure 3. Comparison of HNO₃, HONO, sum (HNO₃ + HONO), SO₂, HCl and aerosol NO₃⁻, SO₄²⁻, Cl⁻, Na⁺, Ca²⁺, Mg²⁺ concentrations by the Acid Gases and Aerosol Network (AGANet) DELTA method with available measurements from the collocated ChemSpec Daily Annular Denuder system (ADS) at Barcombe Mills (UKA00069). Mean concentrations were derived from the average of daily ADS data for the corresponding DELTA sampling periods (monthly). HNO₃ values shown for DELTA and ADS are as calculated from the amount of NO₃⁻ collected on the denuders and have not been adjusted by a bias correction factor (see Sect. 2.6). A summary of the regression analyses is provided in the table below the graphs.

The very low HONO (nitrite on the denuders assumed to be from HONO) concentrations from the DELTA supports the hypothesis of the retention of HONO that is subsequently oxidized to nitrate, resulting in an artefact in HNO₃ determination (Possanzini et al., 1983; Allegrini et al., 1987; Tang et al., 2015). Further corroboration is provided by the improved agreement between both methods (line of fit closer to the 1:1 line) when comparing the sum of HNO₃ and HONO (Fig. 3). Agreement between the DELTA and ADS was within 19 % for SO₂ (mean DELTA = 1.75 µg m⁻³ cf. mean ADS = 2.18 µg m⁻³) and 4 % for HCl (mean DELTA = 0.40 µg m⁻³ cf. mean ADS = 0.41 µg m⁻³). Given the limited data available, it is not clear why SO₂ measured on the ADS is higher than the DELTA, since there was good agreement for HCl.

For the particle-phase components, NO₃⁻ measured by the DELTA method (mean = 2.59 µg NO₃⁻ m⁻³) was on average 2-fold higher than the ADS method (mean = 1.32 µg NO₃⁻ m⁻³), whereas SO₄²⁻ by the DELTA method was on average 23 % lower (DELTA = 2.10 vs. ADS = 2.74 µg SO₄²⁻ m⁻³) (Fig. 3). NO₃⁻ and SO₄²⁻ are both present as fine mode (< 1 µm) NH₄NO₃ and (NH₄)₂SO₄ (Putaud et al., 2010). Some NO₃⁻ can also be present in the coarse mode (> 2.5 µm), likely as calcium nitrate.
Figure 4. Comparison of (a) total inorganic nitrate, TIN (sum of HNO$_3$ + NO$_2^-$) and (b) total inorganic ammonium, TIA (sum of NH$_3$ + NH$_4^+$) concentrations at the Eskdalemuir monitoring station (EMEP station code = GB0002R; UK-AIR ID = UKA00130) measured under the EMEP program with concentrations of the corresponding gas and aerosol from the UK Acid Gases and Aerosol (AGANet, HNO$_3$ and NO$_2^-$) and UK National Ammonia Monitoring Network (NAMN, NH$_3$ and NH$_4^+$). EMEP values (EMEP, 2017a) are means of daily measurements for TIN and TIA by the EMEP filter pack method, matched to the AGANet and NAMN sampling periods (monthly). Filter pack measurements at Eskdalemuir terminated in December 2000. A summary of the regression analyses is provided in the table below the graphs.

3.1.2 Comparisons with filter pack measurements: HNO$_3$/NO$_2^-$ and NH$_3$/NH$_4^+$

The EMEP network (http://www.emep.int/, last access: 17 March 2017) measures atmospheric concentrations and depositions of a wide range of pollutants at rural background sites across Europe (Aas, 2014; Tørseth et al., 2012). A daily filter pack method continues to be implemented at 39 sites across Europe for assessment of oxidized and reduced nitrogen species, with results reported as total inorganic nitrate (TIN: HNO$_3$ + NO$_2^-$) and total inorganic ammonium (TIA: NH$_3$ + NH$_4^+$) (Colette et al., 2016; Tørseth et al., 2012), as these are considered more reliable than reporting for the gas and aerosol components separately. At the UK Eskdalemuir site (EMEP station code GB0002R; UKAIR ID UKA00130), a Scottish rural background site on the border between Scotland and England, daily filter pack measurements of TIN and TIA were made as part of the EMEP network from 1989 to 2000 (EMEP, 2017a). Following installation of the DELTA system in September 1999, both methods were operated in parallel for 14 months at Eskdalemuir, allowing a comparison to be made of TIN and TIA from both systems. Comparison results are shown in Fig. 4 of parallel data from the AGANet (sum of HNO$_3$ and NO$_2^-$) and NAMN (sum of NH$_3$ and NH$_4^+$), demonstrating close agreement between the two independent measurements. The EMEP values shown are daily measurements of TIN and TIA averaged to corresponding monthly
Comparison of gaseous SO$_2$ and particulate SO$_4^{2-}$ concentrations at the Eskdalemuir monitoring station (EMEP station code = GB0002R; UK-AIR ID = UKA00130) measured under the Acid Deposition Monitoring Program (ADMN, Hayman et al., 2007) with the corresponding gas and aerosol from the UK Acid Gases and Aerosol network (AGANet). A summary of the regression analyses is provided in the table below the graphs.

|                          | AGANet DELTA: SO$_2$ | Bubbler: SO$_2$ | AGANet DELTA: SO$_4^{2-}$ | Filter pack: SO$_4^{2-}$ |
|--------------------------|----------------------|-----------------|---------------------------|-------------------------|
| Linear regression: $R^2$| 0.812***             | 0.658***        | 0.670***                  | 0.011*                  |
| Slope                    | 0.866***             | 0.866***        | 0.670***                  | 0.44                     |
| Intercept                | 0.094*               | 0.28            | 0.28                      | 0.44                     |
| Mean (µg S m$^{-3}$)     | 0.44                 | 0.44            | 0.67                      | 0.44                     |
| No. of observations ($n$)| 26                   | 26              | 87                        | 87                       |

Significance level (slope different from 1, intercept = 0): * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. ns = not significant ($p > 0.05$)

Figure 5. Comparison of gaseous SO$_2$ and particulate SO$_4^{2-}$ concentrations at the Eskdalemuir monitoring station (EMEP station code = GB0002R; UK-AIR ID = UKA00130) measured under the Acid Deposition Monitoring Program (ADMN, Hayman et al., 2007) with the corresponding gas and aerosol from the UK Acid Gases and Aerosol network (AGANet). A summary of the regression analyses is provided in the table below the graphs.

Independent measurements of SO$_2$ and SO$_4^{2-}$ with a daily bubbler and filter pack method, respectively, are also available for comparison with the DELTA method at the Eskdalemuir site. Daily SO$_2$ data with a bubbler method (Hayman, 2005) from December 1977 to December 2001 and daily SO$_4^{2-}$ data with an EMEP filter pack method from December 1977 to April 2009 (Hayman, 2006) were downloaded from the EMEP website (EMEP, 2017b). A close agreement is found between the bubbler and DELTA method for SO$_2$ (slope = 0.86, $R^2 = 0.82$), while there is more scatter between the filter pack and DELTA method for SO$_4^{2-}$ (slope = 0.67, $R^2 = 0.66$) (Fig. 5). Concentrations of SO$_2$ for the 26 month overlap period were comparable (mean of bubbler method = 0.40 µg S m$^{-3}$ of mean of DELTA method = 0.44 µg S m$^{-3}$), whereas the filter pack SO$_4^{2-}$ concentration (mean = 0.44 µg S m$^{-3}$, $n = 87$) is larger than the corresponding monthly DELTA measurement (mean = 0.28 µg S m$^{-3}$, $n = 87$) (Fig. 5). An earlier detailed assessment of the DELTA system against filter pack with a focus on SO$_2$ and SO$_4^{2-}$ in 1999 by Hayman et al. (2006) had shown close agreement between the methods. It is therefore unclear why the DELTA gives a reading lower than the filter pack SO$_4^{2-}$ at Eskdalemuir in this assessment, since the dataset was a continuation of the original inter-comparison. Possible explanations include uncertainties associated with limit of detection of the daily filter pack method at the very low concentrations encountered at this site, or the sampling of coarser particles by this method (due to high flow rate and open-face sampling) with higher concentrations of sea salt sulfate. The DELTA methodology was unchanged for the duration of the AGANet dataset (1999–2015) in this manuscript, which allows a consistent assessment of overall trends in the SO$_4^{2-}$ data.

3.2 AGANet data

Annual data from the AGANet (and also from the NAMN) are submitted to the Department for Environment, Food & Rural Affairs (Defra) UK-AIR database (https://uk-air.defra.gov.uk/, last access: 17 March 2017), in a format consistent with other UK Authority air quality networks and relevant reporting requirements. Every concentration value is labelled with a validity flag and an EMEP flag (see http://www.nilu.no/projects/ccc/flags/index.html,
last access: 23 March 2017). Ratified calendar year data are published from around June the year following collection. Currently, work is also in progress for the data to be made available from the EMEP database (http://ebas.nilu.no/, last access: 23 March 2017). All data used in this paper (up to 2015), except where specified, are accessed from the UK-AIR website (Tang et al., 2017a, b).

3.3 Uncertainties in HNO$_3$ determination

HNO$_3$ data were corrected for sampling artefacts in the DELTA method with an empirical correction factor of 0.45 (see Sect. 2.6). Interferences in HNO$_3$ determination arise through the simultaneous collection of reactive oxidized nitrogen species on the K$_2$CO$_3$ coating that forms nitrate ions in the aqueous extracts of exposed denuders. Potential interfering species include HONO, NO$_2$, N$_2$O$_5$ and PAN, as well as other inorganic and organic nitrogen species. HONO is most likely to contribute to the interference, since it is collected effectively on a carbonate coating and concentrations of HONO have been reported to be comparable to, and in some places exceed HNO$_3$ in the UK (e.g. Kitto and Harrison, 1992; Conolly et al., 2016). Interference from NO$_2$ on the other hand should be small, since the reactivity of a carbonate coating surface towards NO$_2$ is low (Allegrini et al., 1987), with capture of NO$_2$ on carbonate ranging from 0.5% to 5% (Allegrini et al., 1987; Fitz, 2002) and their concentrations are also small at rural AGANet sites (<10 µg NO$_2$ m$^{-3}$; Conolly et al., 2016). Tests by Steinle (2009) on the DELTA K$_2$CO$_3$ / glycerol coated denuders also confirmed low capture (ca. 3%) of NO$_2$.

The correction factor was derived from two years of field intercomparison measurements at five sites across a range of pollutant concentrations across the UK, from a clean rural background site in southern Scotland (Auchencorth) to a polluted urban site (London, Cromwell road) in southern England (Tang et al., 2015). It is recognized that the correction factor to derive the “real HNO$_3$” signal from the carbonate coated denuders will also be dependent on the relative concentrations of HNO$_3$ to interfering species present in the atmosphere and likely to be both site and season specific. The 2 years of data indeed show this variability between sites and between seasons. Given the complexities of atmospheric chemistry of the large family of oxidized nitrogen species, further work is clearly needed to understand what the carbonate denuders are measuring, before an improved correction algorithm for the HNO$_3$ data can be developed with any confidence.

The empirical 0.45 HNO$_3$ correction factor is therefore at present a best estimate across a range of pollutant concentrations and seasons encountered in the UK, based on available test data from 5 sites. At the cleanest rural sites (e.g. Eskdalemuir), where a much smaller HONO and NO$_2$ interference of the DELTA HNO$_3$ signal is expected, the HNO$_3$ concentrations may be under-estimated after correction. This may partly explain the slope deviating from unity in the comparison of corrected DELTA TIN with EMEP filter pack TIN data (slope = 0.835, $R^2 = 0.95$) at Eskdalemuir (see Sect. 3.1.2). Conversely, at more polluted sites such as London that are affected by a larger interference from HONO and NO$_2$, the HNO$_3$ determination may be over-estimated after correction. Apart from two urban sites (London and Edinburgh), all other sites in the AGANet are rural, located away from traffic, and the 0.45 correction factor should be more representative.

Since January 2016, the DELTA denuder sample train configuration in AGANet was changed to two NaCl coated denuders (selective for HNO$_3$, e.g. Allegrini et al., 1987), with a third K$_2$CO$_3$ / glycerol coated denuder to collect SO$_2$. At three sites (Auchencorth, Bush OTC and Stoke Ferry), parallel measurements of the old configuration (two K$_2$CO$_3$ / glycerol coated denuders) and new configuration (two NaCl coated denuders + K$_2$CO$_3$ / glycerol coated denuder) were conducted over 12 months in 2016. In the new configuration, nitrates measured on the NaCl denuders are reported as HNO$_3$, whereas nitrate on the K$_2$CO$_3$ denuder are assumed to come from other oxidized nitrogen species and are not reported. Comparing the sum of nitrate concentrations from the new (2 × NaCl + 1 × K$_2$CO$_3$) with the old (2 × K$_2$CO$_3$) configurations indicated matching capture of total nitrate by the two parallel systems (new/old nitrate ratio = 0.95). A comparison of nitrate concentrations on the 2 × NaCl denuders only (new configuration) with the 2 × K$_2$CO$_3$ denuders (old configuration) yielded an average ratio of 0.42, lending further support to the 0.45 empirical factor. Additionally, the new sample train configuration is providing an extensive dataset which will allow the magnitude of HNO$_3$ interference at each site to be quantified, by comparing the amount of nitrate measured on the 2 × NaCl and K$_2$CO$_3$ coated denuders. Initial analysis of 2016 data (unpublished data) showed that the mean ratio of nitrate on NaCl : K$_2$CO$_3$ of all sites was 0.44, ranging from 0.31 (Bush OTC) to 0.59 (Moorhouse). Seasonally, the average monthly ratio (taken as the mean across all sites for each month) was lowest in winter (0.25 in December and 0.27 in January) and highest between May to June (0.59, 0.56 and 0.57). It may therefore be possible to derive an improved correction algorithm that is both site and season specific, and work is ongoing to make this assessment. A detailed assessment of sampling artefacts and uncertainties in the DELTA method and the effects of a method change in the AGANet forms the subject of a next paper that is currently in preparation.

3.4 Spatial patterns in relation to pollutant sources and transport

In Fig. 6, the spatial patterns for each of the gas and aerosol components measured are shown in the annual maps for the example year 2013. A gradient in the concentrations of acid gases HNO$_3$ and SO$_2$, and related aerosols NO$_3^-$ and
SO\textsubscript{4}\textsuperscript{2–} can be seen across the UK, highest in the south and east (combustion or vehicular sources and long-range transboundary pollutant transport from Europe) and lowest in the north and west of the UK (fewer sources, furthest from influence of Europe). The ranges in site-annual mean concentrations (µg molecule m\textsuperscript{-3}) in 2013 for the gases were as follows: HNO\textsubscript{3}: 0.12–1.2; HCl: 0.15–0.52; SO\textsubscript{2}: 0.10 – 1.08, while those for aerosol were as follows: NO\textsubscript{3}: 0.33–3.1; Cl\textsuperscript{−}: 0.54–3.3; SO\textsubscript{4}\textsuperscript{2–}: 0.35–1.2; Na\textsuperscript{+}: 0.35–1.8; Ca\textsuperscript{2+}: < lod–0.11; Mg\textsuperscript{2+}: 0.03–0.19.

The largest HNO\textsubscript{3} concentrations were measured at the London Cromwell site (2013 site annual mean = 1.3 µg HNO\textsubscript{3} m\textsuperscript{-3} cf. 2013 mean of 30 sites = 0.40 µg HNO\textsubscript{3} m\textsuperscript{-3}). London and Edinburgh are the only two urban sites in the AGANet, with the other 28 sites all in rural environments. HNO\textsubscript{3} concentrations in Edinburgh, the capital of Scotland with a population that is 18 times smaller than London (0.5 million vs. 8.8 million), is about 2 times lower than London, but larger than the national average (2013 annual mean = 0.58 µg HNO\textsubscript{3} m\textsuperscript{-3}). For SO\textsubscript{2}, the highest concentrations were recorded at Sutton Bonington due to close proximity to the 2000 MW capacity coal-fired Ratcliffe-on-Soar power station (2 km North). A peak monthly concentration of 10.9 µg SO\textsubscript{2} m\textsuperscript{-3} was recorded in May 2000 at this site, with an annual mean concentration of 5.9 µg SO\textsubscript{2} m\textsuperscript{-3} for that year that was also 3 times higher than the national average (mean of 12 sites = 1.9 µg SO\textsubscript{2} m\textsuperscript{-3} cf. mean of 11 sites (excl. Sutton Bonington = 1.5 µg SO\textsubscript{2} m\textsuperscript{-3})). At remote sites further away from sources, concentrations of HNO\textsubscript{3} and SO\textsubscript{2} are smaller, e.g. Lough Navar in Northern Ireland (2013 annual mean: 0.15 µg HNO\textsubscript{3} m\textsuperscript{-3} and 0.21 µg SO\textsubscript{2} m\textsuperscript{-3}) and Strathvaich Dam in north-western Scotland (2013 annual mean = 0.17 µg HNO\textsubscript{3} m\textsuperscript{-3} and 0.18 µg SO\textsubscript{2} m\textsuperscript{-3}). NO\textsubscript{3} and SO\textsubscript{4}\textsuperscript{2–} as secondary aerosols have longer residence times in the atmosphere and are expected to be more spatially homogeneous than their precursor gases. The spatial distribution in concentrations of particulate NO\textsubscript{3} (0.33–3.1 µg m\textsuperscript{-3}) and SO\textsubscript{4} (0.35–1.2 µg m\textsuperscript{-3}) are however similar to that of HNO\textsubscript{3} (0.12–1.3 µg m\textsuperscript{-3}) and SO\textsubscript{2} (0.10–1.1 µg m\textsuperscript{-3}), with no clear differences in the main regional patterns from only 30 sites.

HCl are mostly emitted from coal combustion and the highest concentrations are in the source areas in the south and east and south-west, and also in central England (north of Ratcliffe-on-Soar power station). There is also a marine source for HCl formed by the reaction of sea salt with HNO\textsubscript{3} and H\textsubscript{2}SO\textsubscript{4} (Roth and Okada, 1998; Ianniello et al., 2011) that may contribute to additional enhancement of local to regional HCl concentrations. The spatial distributions of Cl\textsuperscript{−} and Na\textsuperscript{+} were similar, with largest concentrations at the coastal sites Goonhilly in south-western England and Lerwick Shetland in the Shetland Isles, highlighting the importance of marine sources to the sea salt (NaCl) aerosol. Further away from the coast and influence of marine aerosol, the smallest concentrations of Cl\textsuperscript{−} and Na\textsuperscript{+} are measured in the west of the country (Lough Navar in Northern Ireland and Cwmystwyth in mid-Wales) and most of Scotland (with the exception of Shetland). Mg\textsuperscript{2+} is also seen to show a similar spatial distribution to Na\textsuperscript{+} and Cl\textsuperscript{−}, which suggests that it may be in the form of MgCl\textsubscript{2}, although the range of concentrations at sites are small (0.03–0.19 µg m\textsuperscript{-3}). There is however no clear spatial pattern for Ca\textsuperscript{2+}, but since concentrations are mostly at or below LOD, any assessment of this component is highly uncertain.

In the case of NH\textsubscript{3}, the extensive spatial heterogeneity seen is related to large variation in emission sources at ground level across the UK (Tang et al., 2018). Aerosol NH\textsubscript{4}\textsuperscript{+}, as expected for a secondary component, show a less variable concentration field. The spatial distribution of NH\textsubscript{4}\textsuperscript{+} is similar to SO\textsubscript{4}\textsuperscript{2–} and NO\textsubscript{3} over the UK (Fig. 6), due to the close coupling between species from the formation of particle phase (NH\textsubscript{4})\textsubscript{2}SO\textsubscript{4} and NH\textsubscript{4}NO\textsubscript{3} (see next section).

3.5 Correlations between gas and aerosol species

Correlations plots between the gas and aerosol phases of the different components are shown in Fig. 7, with a summary of the regression results provided in Table 2. The comparison of gas phase concentrations show that gaseous NH\textsubscript{3} is poorly correlated with either SO\textsubscript{2} or HNO\textsubscript{3}, as might be expected since the emission sources of these pollutants are different. In the case of the acid gases however, the significant correlations between HNO\textsubscript{3} : SO\textsubscript{2} (R\textsuperscript{2} = 0.35), HNO\textsubscript{3} : HCl (R\textsuperscript{2} = 0.25), and SO\textsubscript{2} : HCl (R\textsuperscript{2} = 0.21) may be related to similarity in the regional distribution of their emissions. These comparisons show that there is on average 5 times more NH\textsubscript{3} than SO\textsubscript{2} and 13 times more NH\textsubscript{3} than HNO\textsubscript{3} at the AGANet sites (on a molar basis), and that SO\textsubscript{2} concentration is nearly 3 times larger than HNO\textsubscript{3} (on a molar basis).

In the aerosol components, there is very high correlation between NO\textsubscript{3}, SO\textsubscript{4}\textsuperscript{2–}, and NH\textsubscript{4}\textsuperscript{+}, and between Na\textsuperscript{+} and Cl\textsuperscript{−}, but no discernible relationship between NH\textsubscript{4}\textsuperscript{+} and Cl\textsuperscript{−} (Fig. 7). The near 1:1 relationship in the scatter plot of the sum of NO\textsubscript{3} and SO\textsubscript{4} (neq m\textsuperscript{-3}) vs. NH\textsubscript{4} (neq m\textsuperscript{-3}) (slope = 0.91, R\textsuperscript{2} = 0.93), in the absence of any correlation between NH\textsubscript{4} and Cl\textsuperscript{−}, suggests that H\textsubscript{2}SO\textsubscript{4} and HNO\textsubscript{3} in the atmosphere are fully neutralized by NH\textsubscript{3} to form (NH\textsubscript{4})\textsubscript{2}SO\textsubscript{4}, NH\textsubscript{4}H\textsubscript{2}SO\textsubscript{4} and NH\textsubscript{4}NO\textsubscript{3} (Aneja et al., 2001). For Cl\textsuperscript{−}, the high correlation with Na\textsuperscript{+} (slope = 1.04, R\textsuperscript{2} = 0.8) lends support that the Cl\textsuperscript{−} measured in the DELTA are derived mainly from sea salt (NaCl). Similar to the relative concentrations of gases, NH\textsubscript{4}\textsuperscript{+} concentrations (on a molar basis) are larger than SO\textsubscript{4}\textsuperscript{2–} and NO\textsubscript{3}, but NO\textsubscript{3} is in molar excess over SO\textsubscript{4}\textsuperscript{2–}. The correlations between NH\textsubscript{4}\textsuperscript{+} and sum (NO\textsubscript{3} + 2× SO\textsubscript{4}\textsuperscript{2–}), and for Na\textsuperscript{+} and Cl\textsuperscript{−} forms the basis of ion balance checks in data quality assessment, as discussed in Sect. 2.5 and shows that robust data are obtained.
Figure 6. Annual mean monitored acid gas (HNO$_3$, SO$_2$, HCl) and aerosol (NO$_3^-$, SO$_4^{2-}$, Cl$^-$, Na$^+$, Ca$^{2+}$, Mg$^{2+}$) concentrations from the UK Acid Gas and Aerosol Monitoring Network (AGANet) across the UK from annual averaged monthly measurements made in 2013. NH$_3$ and NH$_4^+$ measured at the same time from the UK National Ammonia Monitoring Network (NAMN, Tang et al., 2018) are also shown alongside for comparison.
Sea salt aerosol, derived from sea spray, has essentially the same composition as seawater (Keene et al., 1986). The marine aerosol comprises two distinct aerosol types: (1) primary sea salt aerosol produced by the mechanical disruption of the ocean surface and (2) secondary aerosol, primarily in the form of non-sea salt (nss) sulfate and organic species, formed by gas-to-particle conversion processes such as binary homogeneous nucleation, heterogeneous nucleation and condensation (O’Dowd and de Leeuw, 2007). It has been shown that the ratio of the mass concentrations of SO$_4^{2-}$ and Cl$^-$ to the reference Na$^+$ species in seawater may be used to estimate mass concentrations of non-sea salt SO$_4^{2-}$ (nss_SO$_4$) and non-sea salt Cl$^-$ (nss_Cl) in aerosol, according to Eqs. (4) and (5), respectively (Keene et al., 1986; O’Dowd and de Leeuw, 2007).

\[
[nss\text{-}SO_4] = [SO_4^{2-}] - (0.25 \times [Na^+]) \tag{4}
\]

\[
[nss\text{-}Cl] = [Cl^-] - (1.80 \times [Na^+]) \tag{5}
\]

Applying Eq. (4) to the SO$_4^{2-}$ data in Fig. 7, nss_SO$_4$ is estimated to comprise on average 25\% (range = 3\%–83\%, n = 187) of the measured total SO$_4^{2-}$ aerosol. Regression of nss_SO$_4$ vs. NH$_4^+$ (slope = 0.18, intercept = 0.47, \(R^2 = 0.71\)) (Fig. S3) was not significantly different from the regression of total SO$_4^{2-}$ vs. NH$_4^+$ (slope = 0.18, intercept = 2.4, \(R^2 = 0.73\)) (Fig. 7). Sources of nss_SO$_4$ are (i) biological oxidation of dimethylsulfide and (ii) oxidation of SO$_2$ (O’Dowd and de Leeuw, 2007). This analysis demonstrates that sea salt SO$_4^{2-}$ aerosol makes up a significant and variable fraction of the total SO$_4^{2-}$ measured, consistent with observations of the contribution by sea salt SO$_4^{2-}$ to the total SO$_4^{2-}$ in precipitation in the UK (ROTAP, 2012). The improved intercept from the nss_SO$_4$ regression (Fig. S3) suggests that nss_SO$_4$ are mainly associated with NH$_4^+$.

Estimated nss_Cl concentrations according to Eq. (4) was however negligible (mean = $-0.09$ µg m$^{-3}$, n = 188), compared to the total Cl$^-$ (mean = 1.3 µg m$^{-3}$, n = 188). Studies have shown that part of the chloride of sea salt can be substituted by SO$_4^{2-}$ and NO$_3^-$ through a reaction with H$_2$SO$_4$ and HNO$_3$, known as the Cl$^-$ deficit (Ayers et al., 1999). The close coupling between Cl$^-$ and Na$^+$ (near 1 : 1 relationship) presented here suggests that the measured Cl$^-$ in the aerosol are mostly sea salt in origin, with no evidence of depletion of Cl$^-$ from sea salt aerosols.

### 3.6 Seasonal variation in acid gases and aerosols

The average seasonal cycles for all gas and aerosol components derived from the mean of monthly data of all sites for the period 2000 to 2015 are compared in Fig. 8. Clear differences are observed in these seasonal cycles, influenced by local to regional emissions, climate, meteorology and photochemical processes.

HNO$_3$ is a secondary product of NO$_x$, but NO$_x$ emissions are dominated by vehicular sources which are not expected to show large seasonal variations. Seasonal changes in chemistry and meteorology are therefore more likely to be a source of the observed variations in HNO$_3$ and NO$_3^-$ (Fig. 8). A weak seasonal cycle is observed in HNO$_3$, with slightly higher concentrations in late winter and early spring that may be attributed to photochemical processes with elevated ozone in spring (AQEG, 2009) leading to formation of HNO$_3$ during this period (Pope et al., 2016). As discussed in Sect. 3.3, a constant correction factor was applied to all HNO$_3$ data, which does not take into account seasonal dependency. The concentrations in HNO$_3$ may therefore be over-estimated in winter (less HNO$_3$ formed from photochemical processes) and under-estimated in summer (larger HNO$_3$ concentrations due to increased -OH radicals for reaction with NO$_2$ to form HNO$_3$), masking the true extent in the seasonal profile.

In contrast, the seasonal cycle for particulate NO$_3^-$ is more distinct with a large peak in concentrations that occur every spring, together with a second smaller peak in autumn (Fig. 8). NH$_3$, the main neutralizing gas in the atmosphere that reacts with HNO$_3$ to form NH$_4$NO$_3$, has a correspondingly large peak in concentration in spring, a second smaller peak in autumn, but with elevated concentrations in summer and lowest in winter (Fig. 8). Although particulate NO$_3^-$ formation is dependent upon the availability of NH$_3$ for reaction with HNO$_3$, its concentration is also governed by the

| Gas | HNO$_3$ | HCl | SO$_2$ | NO$_3^-$ | Cl$^-$ | SO$_4^{2-}$ | NH$_4^+$ | Na$^+$ |
|-----|---------|-----|--------|----------|-------|------------|--------|-------|
| HNO$_3$ | 1.00 | 0.25*** | 0.39*** | 0.45*** | 0.07*** | 0.54*** | 0.49*** | 0.02* |
| HCl | – | 1.00 | 0.21*** | 0.14*** | 0.01** | 0.24*** | 0.19*** | 0.04** |
| SO$_2$ | – | – | 1.00 | 0.30*** | 0.00** | 0.47*** | 0.37*** | 0.01** |
| NO$_3^-$ | – | – | – | 1.00 | 0.00** | 0.61*** | 0.90*** | 0.02** |
| Cl$^-$ | – | – | – | – | 1.00 | 0.04** | 0.01** | 0.79*** |
| SO$_4^{2-}$ | – | – | – | – | – | 1.00 | 0.73*** | 0.00** |
| NH$_4^+$ | – | – | – | – | – | – | 1.00 | 0.00** |
| Na$^+$ | – | – | – | – | – | – | 1.00 | – |

Significance level: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$, ns = not significant ($p > 0.05$).
Figure 7. Scatter plots between concentrations of (a) gaseous species HNO$_3$, SO$_2$, and NH$_3$, and (b) particulate species NO$_3^-$, SO$_4^{2-}$, NH$_4^+$, Cl$^-$, and Na$^+$ from mean monthly measurements (1999–2015) from the 12 sites in the UK Acid Gas and Aerosol Monitoring Network (AGANet) that were operational over the whole period. NH$_3$ and NH$_4^+$ data are from the UK National Ammonia Monitoring Network (NAMN, Tang et al., 2018) made at the same time. Each data point represents a single monthly DELTA measurement.

librium that exists between gaseous HNO$_3$, NH$_3$, and particulate NH$_4$NO$_3$, the latter of which is appreciably volatile at ambient temperatures (Stelson and Seinfeld, 1982). Partitioning between the gas and aerosol phase is therefore also a key driver for their atmospheric residence times and concentrations. HNO$_3$ and NH$_3$ that are not removed by deposition may react together in the atmosphere to form NH$_4$NO$_3$ aerosol, when the concentration product [NH$_3$]·[HNO$_3$] exceeds equilibrium values. Since NH$_4$NO$_3$ is semi-volatile, any that is not dry or wet deposited can potentially dissociate to release NH$_3$ and HNO$_3$, effectively increasing their residence times in the atmosphere. The formation and dissociation in turn are strongly influenced by ambient temperature and humidity.

Warm, dry conditions in summer promote dissociation, increasing gas-phase HNO$_3$ relative to particulate-phase NH$_4$NO$_3$. This process accounts for the minima in NO$_3^-$ concentrations (Fig. 8) and the highest ratio of HNO$_3$ to NO$_3^-$ seen in July (Fig. 9). Cooler conditions in the spring and autumn sees a larger fraction of the volatile NH$_4$NO$_3$ remaining in the aerosol phase. The largest peak in NO$_3^-$ concentrations (Fig. 8) and the lowest HNO$_3$ : NO$_3^-$ ratio in springtime (Fig. 9) is thus a combination of increased NO$_3^-$ formation from reaction between higher concentrations of the precursor gases HNO$_3$ and NH$_3$, and increased partitioning to the aerosol phase in cooler, more humid climate. Import from long-range transboundary transport of particulate NO$_3^-$, e.g. from continental Europe into the UK, as discussed in Vieno et al. (2014, 2016) adds to the elevated NO$_3^-$ concentrations. In winter, low temperature and high humidity also shifts the equilibrium to formation of NH$_4$NO$_3$ from the gas-phase HNO$_3$ and NH$_3$. Since NH$_3$ concentrations are also lowest in winter, with less NH$_3$ available for reaction, NH$_4$NO$_3$ concentrations are correspondingly smaller in winter than in spring or autumn.

SO$_2$, by contrast, are highest in the winter, with concentrations exceeding summer values on average by a factor of 2 (Fig. 8). Higher emissions of SO$_2$ from combustion pro-
Figure 8. Average annual cycles for HNO$_3$, SO$_2$, HCl and aerosol NO$_3^-$, SO$_4^{2-}$, Cl$^-$, Na$^+$, Ca$^{2+}$ and Mg$^{2+}$ from the UK Acid Gases and Aerosol Monitoring Network (AGANet). The NH$_3$ and NH$_4^+$ concentrations measured at the same time in the UK National Ammonia Monitoring Network (NAMN, Tang et al., 2018) are also shown for comparison. Each data point in the graphs represents the mean ± SD of monthly measurements of all sites in the network.

Figure 9. Average annual cycles in the ratios of gas:aerosol component concentrations. HNO$_3$, SO$_2$, HCl and aerosol NO$_3^-$, SO$_4^{2-}$, Cl$^-$ data (annual mean, µg m$^{-3}$) are from the UK Acid Gases and Aerosol Monitoring Network (AGANet). NH$_3$ and NH$_4^+$ data (annual mean, µg m$^{-3}$) that are measured at the same time for the UK National Ammonia Monitoring Network (NAMN, Tang et al., 2018) are also shown for comparison. Each data point in the graphs represents the mean ± 95% confidence interval (CI) of monthly measurements of 12 sites operational in the network over the period 2000 to 2015.

cesses (heating) during the winter months, coupled to stable atmospheric conditions resulting in build-up of concentrations at ground level contributes to the winter maximum. Since the reaction of SO$_2$ with NH$_3$ to form (NH$_4$)$_2$SO$_4$ is effectively irreversible (Bower et al., 1997), the ratio of the concentrations of SO$_2$ and SO$_4^{2-}$ (Fig. 9) is largely governed by the availability of SO$_2$ and NH$_3$ to form (NH$_4$)$_2$SO$_4$. The temporal profile of SO$_4^{2-}$ has a peak in concentrations in spring, although not as pronounced as the NO$_3^-$ peak (Fig. 8). This may be attributed to enhanced formation of (NH$_4$)$_2$SO$_4$, since peaks in concentrations of NH$_3$ and NH$_4^+$ also occur in spring (Fig. 8) and from the import of particulates from long-range transboundary transport. Unlike SO$_2$, aerosol SO$_4^{2-}$ concentrations are higher in summer than in winter, due to increased photochemical oxidation of SO$_2$ to H$_2$SO$_4$ and subsequent formation of sulfate aerosols in sunnier and warmer conditions (Mihalopoulos et al., 2007). In winter, lower SO$_2$ oxidation rates limits H$_2$SO$_4$ formation and therefore also the formation of (NH$_4$)$_2$SO$_4$.

Na$^+$ and Cl$^-$ also have highest concentrations during winter, highlighting the importance of marine sources (more stormy weather) in winter for sea salt aerosol. The seasonal trends in Mg$^{2+}$ are similar to Na$^+$, with maxima during winter and minima in summer (Fig. 8). While sea salt aerosols comprise mainly of NaCl, other chemical ions are also common in seawater, such as K$^+$, Mg$^{2+}$, Ca$^{2+}$ and SO$_4^{2-}$ (Keene et al., 1986). Some of the sea salt aerosol may therefore be
A second, but smaller pollutant episode that was captured by the AGANet occurred in September 2014, with elevated concentrations of SO$_2$, HNO$_3$, SO$_4^{2-}$, NO$_3^-$, and NH$_4^+$ that came from the Icelandic Holuhraun volcanic eruptions (Twigg et al., 2016). The elevated SO$_2$ concentration in September 2014 led to a modest increase in annual concentrations in SO$_2$ for 2014 (0.58 µg m$^{-3}$, cf. annual mean = 0.54 and 0.27 µg m$^{-3}$ for 2013 and 2015, respectively). For the other components (HNO$_3$, particulate SO$_2$, NO$_3^-$ and NH$_4^+$), the spikes in concentrations were smaller than for SO$_2$ and did not noticeably elevate their annual mean concentrations for that year. These pollution events together illustrate very clearly how short pollutant episodes can have a major influence on the measured annual concentrations in the UK, and that changes in meteorological conditions, cou-
that contrary to the reported decrease in UK NH$_3$ in 2015). This is consistent with findings by Tang et al. (2018) exploring a large downward trend in particulate NH$_3$ over the period 2000–2015 shows that the decrease in TIA (sum of NH$_3$ and NH$_4^+$) by the EMEP filter pack method (April 1989–November 2000), matched to the AGANet and NAMN sampling periods (monthly) where the measurements overlap. The AGANet and NAMN data are for gaseous HNO$_3$ and NH$_3$ and for the sum of (HNO$_3$ + NO$_2$) and sum of (NH$_3$ + NH$_4^+$), respectively, by the DELTA method. The AGANet HNO$_3$ values shown here includes the bias correction (Sect. 2.6).

Figure 11. Long-term time series of (a) oxidized nitrogen (HNO$_3$ and NO$_2^-$) and (b) reduced nitrogen (NH$_3$ and NH$_4^+$) concentrations at Eskdalemuir (EMEP station code = GB0002R; UK-AIR ID = UKA00130). EMEP values (EMEP, 2017a) are monthly means of daily measurements for total inorganic nitrogen, TIN (sum of HNO$_3$ and NO$_2^-$) and total inorganic nitrogen, TIA (sum of NH$_3$ and NH$_4^+$) by the EMEP filter pack method (April 1989–November 2000), matched to the AGANet and NAMN sampling periods (monthly) where the measurements overlap. The AGANet and NAMN data are for gaseous HNO$_3$ and NH$_3$ and for the sum of (HNO$_3$ + NO$_2$) and sum of (NH$_3$ + NH$_4^+$), respectively, by the DELTA method. The AGANet HNO$_3$ values shown here includes the bias correction (Sect. 2.6).

3.7 Long-term trends at Eskdalemuir

At the Eskdalemuir rural background site, EMEP filter pack data in TIN (sum of HNO$_3$ and NO$_2^-$) and TIA (sum of NH$_3$ and NH$_4^+$) are available since 1989 (Sect. 3.1.2). In Fig. 11, the EMEP filter pack TIN and TIA time series (April 1989 to December 2000) is extended with AGANet (HNO$_3$ and NO$_2^-$) and NAMN (NH$_3$ and NH$_4^+$) DELTA data (September 1999 to December 2015), with an overlapping period of 14 months. The combined time series shows that the annual concentrations of TIN has halved in 26 years between 1990 to 2015, from 0.36 to 0.16 µg N m$^{-3}$, compared with a 3-fold reduction in NO$_3^-$ emissions (from 928 to 302 kt NO$_2$–N) (NAEI, 2018) over the same period. For TIA, the 52 % decrease between 1990 to 2015 (from 0.93 to 0.45 µg N m$^{-3}$) is larger than the corresponding 13 % reduction in NH$_3$ emissions (from 265 to 231 kt NH$_3$–N) (NAEI, 2018). Speciated NH$_3$ and NH$_4^+$ data from NAMN over the period 2000–2015 shows that the decrease in TIA is mainly driven by NH$_4^+$, which decreased by 59 % between 2000 (annual mean = 0.62 µg NH$_4^+$ m$^{-3}$) and 2015 (annual mean = 0.25 µg NH$_4^+$ m$^{-3}$), compared with no change in NH$_3$ (annual mean 0.32 µg NH$_3$ m$^{-3}$ in 2000, unchanged in 2015). This is consistent with findings by Tang et al. (2018) that contrary to the reported decrease in UK NH$_3$ emissions, NH$_3$ concentrations at background sites (defined by 5 km grid average NH$_3$ emissions < 1 kg N ha$^{-1}$ yr$^{-1}$) are showing an indicative increasing trend, while at the same time, a large downward trend in particulate NH$_4^+$ is observed. Together, the AGAnet and NAMN are thus providing an important long-term dataset that distinguishes between the gas and aerosol phase, allowing gas–aerosol phase interactions to be explored.

An extended time series illustrating the continued decline in SO$_2$ and SO$_4^{2-}$ has also been constructed by combining historic SO$_2$ and SO$_4^{2-}$ measurement data at the Eskdalemuir site going back to December 1977 (see Sect. 3.1.3) with AGANet SO$_2$ and SO$_4^{2-}$ data (September 1999 to December 2015) (Fig. 12). A substantial decline in SO$_2$ is observed, falling by 98 % from 4.5 µg S m$^{-3}$ in 1978 to 0.07 µg S m$^{-3}$ in 2015, in good agreement with similarly large reduction in UK SO$_2$ emissions over the same period of 95 % (from 2570 to 126 kt SO$_2$–S) (NAEl, 2018). The decrease in SO$_4^{2-}$ is of a smaller magnitude, declining by 88 % from an annual mean concentration of 0.89 µg S m$^{-3}$ in 1978 to 0.11 µg S m$^{-3}$ in 2015, highlighting the non-linearity in relationship between the atmospheric gas and aerosol phase of sulfur at this background site.

3.8 Assessment of trends in relation to UK emissions

The long-term time series in annually averaged concentrations of the gas and aerosol components are compared in Fig. 13a and b, respectively. Since there was a change in the number of sites during the operation of the AGANet, annually averaged data from the original 12 sites for the period 2000–2015 (1999 data excluded since AGANet started in September 1999) and from the full network (30 sites) for the period 2006–2015 are plotted alongside each other for comparison. From 2006–2015, the decreasing trends for all gas and aerosol components from the expanded 30 sites are seen to be similar to those from the original 12 sites. The annual mean concentrations in gas and aerosol components derived from the expanded 30 sites (2006–2015) or from the original 12 sites over the same period are also in general comparable (Table 3). The exceptions are Na$^+$ and Cl$^-$ that have higher mean concentrations from the 30 sites than the original 12 sites (Table 3), due to the addition of two coastal sites (Shetland and Rum), with larger contribution from sea
the sea salt origin of Cl\(^{-}\) concentrations (Fig. 13b), illustrating very clearly the close phase precursors, the trends of these aerosol components decreased in concentrations with time, but unlike their gas and of NH\(^{4+}\) the parallel monitoring of acid gases and aerosols in AGANet time.

![Figure 12. Long-term time series of SO\(_2\) (December 1977–July 1993) and SO\(_4^{2-}\) (December 1977–December 2001) concentrations measured in the UK Acid Deposition Monitoring Network (ADMN) (Hayman et al., 2007) and the AGANet DELTA measurements (October 1999–December 2015) at the Eskdalemuir monitoring station (EMEP station code = GB0002R; UK-AIR ID = UKA00130). ADMN values (EMEP, 2017b) are monthly means of daily measurements for SO\(_2\) and SO\(_4^{2-}\) by a daily bubbler and filter pack method, respectively, matched to the AGANet sampling periods (monthly) where the measurements overlap.

In the gas phase, SO\(_2\) decreased 7-fold from an annual mean concentration of 1.9µg SO\(_2\) m\(^{-3}\) in 2000 to 0.25µg SO\(_2\) m\(^{-3}\) in 2015 (n = 12), compared with more modest reductions in HNO\(_3\) (from 0.35 to 0.21µg HNO\(_3\) m\(^{-3}\)), NH\(_3\) (from 1.4 to 1.0µg NH\(_3\) m\(^{-3}\)) and HCl (from 0.31 to 0.20µg HCl m\(^{-3}\)) over the same period (Fig. 13a). Particulate SO\(_4^{2-}\), NO\(_3\), and NH\(_4^{+}\) also decreased in concentrations with time, but unlike their gas phase precursors, the trends of these aerosol components track each other closely, differing only in the magnitude of concentrations (Fig. 13b), illustrating very clearly the close coupling between these components. On the other hand, the absence of a trend in the particulate Cl\(^{-}\) is likely to reflect the sea salt origin of Cl\(^{-}\) which is not expected to vary over time.

Important changes in the chemical climate is captured by the parallel monitoring of acid gases and aerosols in AGANet and of NH\(_3\), NH\(_4^{+}\) in NAMN. It is clear from the long-term data that there is substantial intra- (Fig. 10) and inter-annual variability in the annual mean concentrations of both the gas and aerosol phases (Fig. 13), in particular the spike in concentrations in 2003 (see Sect. 3.6) that buckles the trend. An interpretation of the direct relationship between emissions and concentrations in the atmosphere is therefore not straightforward, as the concentrations are also influenced by other factors such as variations in meteorological conditions and long-range transboundary import into the UK.

In Fig. 14, the relative trends in UK NO\(_x\), SO\(_2\), HCl and NH\(_3\) emissions (NAEI, 2018) are compared with the annually averaged gas and particulate concentrations measured in the AGANet and NAMN for (i) original 12 sites for the 16 year period from 2000 to 2015, (ii) original 12 sites for the 15 year period from 2001 to 2015 (because annual mean concentrations in 2000 for all components were smaller than in 2001–2006), and (iii) expanded 30 sites and also original 12 sites for the 10 year period from 2006 to 2015. All data were normalized to 0 for the start years in each of the comparison.

The long-term trends in HNO\(_3\), SO\(_2\), HCl and particulate NO\(_3\), SO\(_4^{2-}\), Cl\(^{-}\), based on MK statistical trend analysis (Sect. 2.7) of annual mean measurement data are compared in Fig. 15 and summarized in Table 4 for the two time series: (i) the original 12 AGANet sites for the 16 year period from 2000 to 2015, and (ii) the expanded 30 AGANet sites for the 10 year period from 2006 to 2015. This approach avoids introducing bias as a result of changes in the sites and ensures site continuity for the long-term trend assessment. NH\(_3\) and NH\(_4^{+}\) concentrations from the NAMN that were measured at the same time at the AGANet sites were included for comparison and to aid interpretation of the acid gas and aerosol data.

To quantify changes in measured concentrations over time, annual trends (e.g. µg HNO\(_3\) m\(^{-3}\) yr\(^{-1}\)) are estimated from

| Mean concentration (2006–2015), µg m\(^{-3}\) | 12 sites (mean ± SD) | 30 sites (mean ± SD) |
|--------------------------------------------|----------------------|----------------------|
| HNO\(_3\)                                  | 0.31 ± 0.06          | 0.34 ± 0.06          |
| SO\(_2\)                                   | 0.66 ± 0.24          | 0.70 ± 0.25          |
| HCl                                        | 0.25 ± 0.04          | 0.28 ± 0.04          |
| NO\(_3\)                                   | 1.40 ± 0.31          | 1.41 ± 0.26          |
| SO\(_4^{2-}\)                               | 0.73 ± 0.25          | 0.74 ± 0.23          |
| Cl\(^{-}\)                                  | 1.33 ± 0.09          | 1.54 ± 0.09          |
| Na\(^{+}\)                                 | 0.75 ± 0.07          | 0.84 ± 0.08          |
| Ca\(^{2+}\)                                | 0.04 ± 0.03          | 0.04 ± 0.02          |
| Mg\(^{2+}\)                                | 0.06 ± 0.01          | 0.07 ± 0.01          |
| NH\(_3\)                                   | 1.18 ± 0.16          | 1.40 ± 0.16          |
| NH\(_4^{+}\)                               | 0.62 ± 0.18          | 0.61 ± 0.16          |

| www.atmos-chem-phys.net/18/16293/2018/ | Atmos. Chem. Phys., 18, 16293–16324, 2018 |

Table 3. Comparison of mean concentrations from the original 12 Acid Gases and Aerosol Network (AGANet) sites vs. the expanded 30 AGANet sites for the different gas and aerosol components. NH\(_3\) and NH\(_4^{+}\) measured at the same time in the UK National Ammonia Monitoring Network (NAMN, Tang et al., 2018) are also included for comparison. Each data point are the mean ± SD of annual mean concentrations over the period 2006 to 2015.
the regression results of the MK tests. This is considered as providing a more reliable estimate of trend than comparing measured annual concentrations at the beginning and end of the time series, which is subject to bias due to substantial variability in annual concentrations between years (Tang et al., 2018). Changes in measured concentrations over time (MK percentage median change) in the time series are estimated from the MK Sen’s slope and intercept (Eq. 6). MK annual trends and percentage median change are summarized in Fig. 15 and Table 4.

Table 4. Summary of Mann–Kendall (MK) time series trend analysis on annually averaged gas and aerosol concentrations from the UK Acid Gases and Aerosol Monitoring Network (AGANet) for (i) 12 sites that were operational over the period 2000 to 2015 and (ii) 30 sites that were operational over the period 2006 to 2015. NH$_3$ and particulate NH$_4^+$ concentrations measured at the same time in the UK National Ammonia Monitoring Network (NAMN, Tang et al., 2018) are also included for comparison. The 95% confidence interval (CI) for the median trend and relative median change (%) are also estimated.

| Annual mean data (µg m$^{-3}$) | Mann–Kendall (MK): 12 sites (2000–2015) | Mann–Kendall (MK): 30 sites (2006–2015) |
|---------------------------------|------------------------------------------|------------------------------------------|
| HNO$_3$                         | Median annual trend & [95% CI] (µg yr$^{-1}$) | Median annual trend & [95% CI] (µg yr$^{-1}$) |
|                                | Relative median change & [95% CI] (%)   | Relative median change & [95% CI] (%) |
| HNO$_3$                         | −0.0135** [−0.067, −0.0180] | −0.0167* [−0.0075, −0.0200] |
|                                | −45** [−26, −55] | −36* [−18, −41] |
| SO$_2$                          | −0.1010*** [−0.0729, −0.1255] | −0.0717** [−0.0300, −0.0108] |
|                                | −81*** [−72, −91] | −60** [−33, −73] |
| HCl                             | −0.0057** [−0.0020, −0.0100] | −0.0088* [0.0000, −0.0200] |
|                                | −28** [−11, −42] | −24* [0.0, −47] |
| NH$_3$                          | −0.0300** [−0.0125, −0.0433] | −0.0312ns [0.0033, −0.0625] |
|                                | −30*** [−13, −39] | −18ns [+2.0, −31] |
| NO$_3^−$                        | −0.0810** [−0.0520, −0.1125] | −0.0900** [−0.0580, −0.1300] |
|                                | −52** [−37, −63] | −43** [−30, −56] |
| SO$_4^{2−}$                     | −0.0750*** [−0.0450, −0.0988] | −0.0675** [−0.0233, −0.1167] |
|                                | −69*** [−52, −82] | −54** [−25, −78] |
| nss-SO$_4^{2−}$                 | −0.0733*** [−0.0500, −0.1012] | −0.0575** [−0.0167, −0.1033] |
|                                | −78*** [−64, −92] | −62** [−25, −86] |
| Cl$^−$                          | 0.0079ns [−0.0888, 0.0236] | −0.0075ns [−0.0167, −0.0300] |
|                                | 9.6ns [−9.5, 33] | −4.2ns [+10, −16] |
| NH$_4^+$                        | −0.0500** [−0.0375, −0.0675] | −0.0480** [0.0267, −0.0700] |
|                                | −62*** [−51, −74] | −49** [−33, −64] |

Significance level: * p < 0.05; ** p < 0.01; *** p < 0.001; ns non-significant (p > 0.05).

a Median annual trend = fitted Sen’s slope of Mann–Kendall linear trend (unit = µg yr$^{-1}$). b Relative median change estimated from the annual concentration at the start ($y_0$) and at the end ($y_1$) of time series computed from the Sen’s slope and intercept ($= 100 \times (y_1 - y_0)/y_0$).
Figure 14. Relative trends in UK emissions (NAEI, 2018) and in annually averaged gas and particulate concentrations from the UK AGANet and UK National Ammonia Monitoring Network (NAMN, Tang et al., 2018) for (a) the original 12 sites for the 16 year period from 2000 to 2015, and (b) expanded 30 sites compared with the original 12 sites for the 10 year period from 2006 to 2015.

\[
\text{\% median change} = 100 \cdot \frac{y_i - y_0}{y_0},
\]

where \((y_0)\) and \((y_i)\) are estimated annual mean concentrations at the start and end of the selected time period, estimated from the slope and intercept of the LR or MK tests.

Statistical trend analysis of monthly mean measurement data in the gas and aerosol components are also shown for comparison in Fig. S3 (mean monthly data of 12 sites for period 2000–2015) and Fig. S4 (mean monthly data of 12 sites for period 2006–2015). MK annual trends and percentage median change, based on the monthly data (Tables S7, S8) were similar to the annual test results (Table 4). While not discussed further here, since assessment of long-term trends in this paper focuses on trends in annual mean concentrations for comparison with trends in estimated annual emissions, the monthly plots serves to illustrate the large intra-annual variability of concentrations in gases and aerosols.

3.8.1 Trends in HNO\(_3\) and NO\(_3^-\) vs. NO\(_x\) emissions

The overall downward trends in HNO\(_3\) and NO\(_3^-\) are seen to be broadly consistent with the \(-49\%\) fall in estimated NO\(_x\) emissions (NAEI, 2018) over the 16 year period between 2000 and 2015 (Fig. 14). Reductions in combustion (power stations and industrial) and vehicular sources (fitting of catalytic converters), coupled to tighter regulations are major contributory factors to the decrease in UK NO\(_x\) emissions. The rate of reduction however stagnated in the period 2009
Time series trend analysis by non-parametric Mann–Kendall Sen slope on annually averaged gas and aerosol concentration data from the UK Acid Gases and Aerosol Monitoring Network (AGANet) of (i) 12 sites with complete time series over the period 2000–2015 and (ii) expanded 30 sites with complete times series over the period 2006–2015. NH\textsubscript{3} and NH\textsuperscript{4}+\textsubscript{4} concentrations data measured at the same time in the UK National Ammonia Monitoring Network (NAMN, Tang et al., 2018) are also included for comparison.

It is notable that the first 6 years (2000–2006) of HNO\textsubscript{3} and NO\textsuperscript{−}\textsubscript{3} annual data show substantial variability between years and in particular is dominated by the large 2003 peak in concentrations (see Sect. 3.6). This highlights the sensitivity of the trend assessment to the selection of a reference start for the time series, since the annual mean concentrations of both HNO\textsubscript{3} and NO\textsuperscript{−}\textsubscript{3} in 2000 are in fact smaller than concentrations in the following 6 years. Re-analysis of the same annual data normalized against 2001 instead of 2000 takes the relative trend lines for HNO\textsubscript{3} and NO\textsuperscript{−}\textsubscript{3} much closer to the relative trend line in NO\textsubscript{x} emissions. In the later period between 2006 and 2015, the relative trend lines in HNO\textsubscript{3} and NO\textsuperscript{−}\textsubscript{3} derived from the mean of either 12 or 30 sites were not significantly different, and the relative trend lines in emission and concentrations followed each other closely (Fig. 14).

Results of MK tests showed that the reductions in annual HNO\textsubscript{3} concentrations are statistically significant for both time series (Fig. 15; Table 4). The MK percentage median change in annual mean HNO\textsubscript{3} was −45 % (2000–2015, \(n = 12\)) and −36 % (2006–2015, \(n = 30\)), consistent with the −49 % and −40 % fall in estimated NO\textsubscript{x} emissions over the corresponding periods (Table 5). The decrease in HNO\textsubscript{3} is accompanied by a larger decrease in particular NO\textsubscript{5} (2000–2015: MK = −52 % (\(n = 12\)), 2006–2015: MK = −43 % (\(n = 30\)) (Table 4) and an indicative small increasing trend is observed in the ratio of HNO\textsubscript{3} to NO\textsuperscript{−}\textsubscript{3} with time (Fig. 16), hinting at an increased partitioning to the gas phase. Since HNO\textsubscript{3} is one of the major oxidation products of NO\textsubscript{x}, through reaction with OH\textsuperscript{−} or heterogeneous conversion of N\textsubscript{2}O\textsubscript{5}, it provides an important measure of the fraction of NO\textsubscript{x} emissions that is oxidized and signals any long-term changes in the atmospheric processing timescales of NO\textsubscript{x} over the country. NO\textsubscript{2} is measured at 24 rural sites across the UK in the UKEAP NO\textsubscript{2}-net, with 11 sites collocated with the AGANet (Conolly et al., 2016). The long-term time series in the data also showed a matching decreasing trend in network averaged NO\textsubscript{2} concentrations with NO\textsubscript{x} emissions between 2000 and 2015, with annual mean NO\textsubscript{2} concentrations across the network falling 2-fold to 4 µg NO\textsubscript{2} m\textsuperscript{−3} in 2015 (Conolly et al., 2016). Despite the uncertainty in corrected HNO\textsubscript{3} data (Sect. 3.3), the encouraging agreement between trends in HNO\textsubscript{3} and NO\textsubscript{2} concentrations and NO\textsubscript{x} emissions lends support to a linear response in HNO\textsubscript{3} concentrations to reductions in NO\textsubscript{x} emissions.

3.8.2 Trends in SO\textsubscript{2} and SO\textsubscript{4}\textsuperscript{2−} vs. SO\textsubscript{2} emissions

Unlike NO\textsubscript{x}, there has been a more significant decline in SO\textsubscript{2}, both in emissions and measured concentrations during this period (Fig. 14). Between 2000 and 2009, SO\textsubscript{2} emissions fell substantially by 66 % from 1286 to 432 kt SO\textsubscript{2}. The reduction reflects mitigation measures introduced since the 1980s (fitting of flue gas desulfurization to coal fired power stations and 2012 (improvement in emissions abatement offset by proportionate increase from diesel combustion and increase in vehicle numbers), followed by a 16 % decrease between 2012 and 2015 due to the closure of a number of coal-fired power stations.

The long-term time series in the data also showed a matching decreasing trend in network averaged NO\textsubscript{2} concentrations with NO\textsubscript{x} emissions between 2000 and 2015, with annual mean NO\textsubscript{2} concentrations across the network falling 2-fold to 4 µg NO\textsubscript{2} m\textsuperscript{−3} in 2015 (Conolly et al., 2016). Despite the uncertainty in corrected HNO\textsubscript{3} data (Sect. 3.3), the encouraging agreement between trends in HNO\textsubscript{3} and NO\textsubscript{2} concentrations and NO\textsubscript{x} emissions lends support to a linear response in HNO\textsubscript{3} concentrations to reductions in NO\textsubscript{x} emissions.

3.8.2 Trends in SO\textsubscript{2} and SO\textsubscript{4}\textsuperscript{2−} vs. SO\textsubscript{2} emissions

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Table 5. Comparison of percentage change in estimated UK NO$_3^-$, SO$_2^-$, and NH$_3$ emissions reported by the National Atmospheric Emission Inventory (NAEI, 2018) with % change between 2000 and 2015 (12 sites with complete time series) and between 2006 and 2015 (30 sites with complete time series) in annually averaged HNO$_3$/NO$_3^-$ and SO$_2$/SO$_2^-$ concentrations from the UK Acid Gas and Aerosol Monitoring Network (AGANet), and annually averaged NH$_3$/NH$_4^+$ concentrations from the UK National Ammonia Monitoring Network (NAMN, Tang et al., 2018).

| Components    | 2000–2015 (12 sites) | 2006–2015 (30 sites) |
|---------------|----------------------|----------------------|
|               | UK emissions$^b$     | MK Sen Slope % relative median change$^a$ | UK emissions$^b$ | MK Sen slope % relative median change$^a$ |
| Gas HNO$_3$   | $-49$ (NO$_2^-$)     | $-45^*$               | $-40$ (NO$_2^-$) | $-36^*$               |
| Particulate NO$_3^-$ | $-80$ (SO$_2^-$)    | $-81^*$               | $-85$ (SO$_2^-$) | $-60^*$               |
| Gas SO$_2$    | $-80$ (SO$_2^-$)     | $-78^*$               | $-65$ (SO$_2^-$) | $-54^*$               |
| Particulate SO$_4^{2-}$ | $-9$ (NH$_3$)       | $-78^*$               | $-45$ (HCl)     | $-24^ns$              |
| nss_SO$_4^{2-}$ | $-69^***$           | $-78^*$               | $-30^***$  | $-18^ns$              |
| Gas HCl       | $-28^ns$             | $-62^*$               | $-78^*$       | $-4^ns$               |
| Particulate Cl$^-$ | $-30^***$         | $-54^*$               | $-28^ns$      | $-18^ns$              |
| Gas NH$_3$    | $-30^***$            | $-62^***$             | $-28^ns$      | $-18^ns$              |
| Particulate NH$_4^+$ | $-62^***$          | $-62^***$             | $-30^***$      | $-18^ns$              |

Significance level: $^* p < 0.05$, $^{**} p < 0.01$, $^{***} p < 0.001$, $^{ns}$ non-significant ($p > 0.05$).

$^a$ Relative median change calculated based on the estimated annual concentration at the start ($y_0$) and at the end ($y_t$) of time series computed from the Sen’s slope and intercept ($MK = 100 \times [(y_t - y_0)/y_0]$).

$^b$ UK emissions data from NAEI (2018).

The decrease in both emissions and concentrations of SO$_2$ is also almost twice as large as HNO$_3$ (Table 5), illustrating the greater success in mitigating sulfur than nitrogen and the increasing dominance of N components in the atmosphere compared with S, with larger decline in SO$_2$ than NO$_3^-$.

At the same time, the reduction in SO$_2$ emission and measured concentration is accompanied by a smaller negative trend in particulate SO$_4^{2-}$ (2000–2015: $-69$ % MK; 2006–2015: $-54$ % MK) (Table 5), with concentrations falling 3-fold from an annual mean of 1.2 µg SO$_4^{2-}$ m$^{-3}$ in 2000 to 0.42 µg SO$_4^{2-}$ m$^{-3}$ in 2015. The smaller decrease in particulate SO$_4^{2-}$ compared with its gaseous precursor, SO$_2$, is similar to that observed at Eskdalemuir (Sect. 3.1.3). A similar picture is also seen in Europe, where atmospheric concentrations of gas phase SO$_2$ decreased by about 92 % compared with a smaller reduction of 65 % in particulate SO$_4^{2-}$ in response to sulfur emissions abatement over the 1990–2012 period in the EMEP region (EMEP, 2016). The ratio of SO$_2$ : SO$_4^{2-}$ is also seen to show a decreasing trend over time (Fig. 16), with the largest change occurring between 2000 and 2006 that matches the period of largest decline in SO$_2$ emissions.

Sea salt SO$_4^{2-}$ (ss_SO$_4$) aerosol, as discussed in Sect. 3.5, makes up a significant fraction of the total SO$_4^{2-}$.

From the MK trend analysis, the decrease in annual mean SO$_2$ concentrations of $-81$ % (2000–2015, $n = 12$), and $-60$ % (2006–2015, $n = 30$) (Fig. 15, Table 4) are consistent with the substantial reduction of $-80$ % and $-64$ % in SO$_2$ emissions over the two overlapping periods, respectively (Table 5).
Equation (4) described in Sect. 3.5) and nss\_SO\textsubscript{4} (\(=\) total SO\textsubscript{4}\textsuperscript{2−} - nss\_SO\textsubscript{4}) are compared with the long-term trends in total SO\textsubscript{4}\textsuperscript{2−} in Fig. 17. Overall, there is no apparent trend in the long-term annual mean ss\_SO\textsubscript{4} data, with concentrations in range of 0.16 to 0.21 µg SO\textsubscript{2}−4. Since ss\_SO\textsubscript{4} is derived from an empirical relationship with Na\textsuperscript{+} (Sect. 3.5), the long-term trend data for Na\textsuperscript{+} is also included in the analysis (Fig. 17). Similar to ss\_SO\textsubscript{4}, there is no overall trend in the Na\textsuperscript{+} data, with small inter-annual variability and annual mean concentrations in the range of 0.65–0.85 µg Na\textsuperscript{+} m\textsuperscript{−3}.

3.8.3 Trends in HCl and Cl\textsuperscript{−} vs. HCl emissions

HCl emissions in the UK also decreased substantially by 89% between 2000 and 2015, from 82 to 9 kt in 2015 (NAEI, 2018), contrasting with a smaller, but non-significant decreasing trend in HCl concentrations (Figs. 14, 15, Table 5). The annual mean monitored concentrations in HCl over this period decreased from 0.30 µg HCl m\textsuperscript{−3} in 2000 to 0.19 µg HCl m\textsuperscript{−3} in 2015. Most of the reduction in HCl emissions occurred before 2006 (−79%, from 82 kt in 2000 to 17 kt in 2006), with emissions plateauing since 2006 (NAEI, 2018) (Fig. 14). A corresponding decrease is not seen in the HCl measurement data, where concentrations remained fairly stable at between 0.31 µg m\textsuperscript{−3} HCl in 2000 to 0.33 µg m\textsuperscript{−3} HCl in 2006. Since 2006 however, the relative change in HCl emissions is closely tracked by changes in concentrations of both the annual mean data from the original 12 sites and from the expanded 30 sites in the AGANet, with the small peak in HCl emissions in 2013 also captured in the annual mean data. This part of the time series therefore clearly shows a direct relationship between emissions and concentrations.

So why is the most significant fall in HCl emissions between 2000 and 2006 not captured by the network? HCl are
mainly released as point sources. Coal burning, particularly from coal-fired power stations, is responsible for the majority of UK emissions: 92% in 1990 and 76% in 2015, and reductions in HCl emissions in the UK inventory are largely as a result of declining coal use and the installation of emissions abatement measures at coal-fired power stations (implemented since 1993) aimed at reducing S that also coincidentally reduced HCl emissions. It may be that a network of only 12 sites in the early periods failed to capture peak emissions and changes in source areas. While there is an indicative, but non-significant decreasing trend in HCl (2000–2015: MK = −28%, 2006–2015: MK = −24%), no detectable trend in particulate Cl can be seen (Table 4). Since Cl is mainly associated with Na (sea salt) in the AGANet measurements (Sect. 3.5), the absence of a trend in Cl (Fig. 15) and Na (Sect. 3.8.2, Fig. 17) provides evidence of a constant background in sea salt in the UK atmosphere.

3.8.4 Trends in NH and NH vs. NH emissions

In comparison to the acid gases, there is a more modest decrease of −9% in NH emissions, from 254 kt NH in 2000 to 231 kt NH in 2015 (NAEI, 2018). This is smaller than the decrease seen in the annually averaged NH concentrations at the 12 AGANet sites (2000–2015: −30% MK) over the same period (Figs. 14, 15, Table 5). A recent assessment by Tang et al. (2018) showed that NH trends are highly dependent on site selection and categorization of sites in the analysis. A more comprehensive analysis of a larger number of sites shows smaller reductions over time, whereas a significant decreasing trend in NH concentrations was observed in the grouped analysis of sites in areas classified as dominated by pig and poultry emissions, contrasting with an upward (non-significant) trend for sites in cattle-dominated areas. Therefore there is a large degree of uncertainty in interpreting the trends in NH concentrations from a subset of just 12 sites, since NH emissions are dominated by agricultural emissions (> 80%) that vary hugely on a local to regional scale across the UK.

At the same time, there is a larger decrease in particulate NH concentrations (−62% MK), contrasting with the smaller decrease in NH concentrations over the period 2010–2015 (−30% MK) (Table 4), with the NH : NH ratio also increasing with time (Fig. 16). This provides evidence for a shift in partitioning from the particulate phase NH to the gaseous phase NH in the UK data, discussed in Tang et al. (2018). The change in partitioning from particulate NH to gaseous NH is also occurring in other parts of Europe, where decreases in NH concentrations have been smaller than emission trends would suggest, due to large decreases in SO emissions (Bleeker et al., 2009; Horvath et al., 2009).

3.8.5 Changes in UK chemical climate

Atmospheric SO concentrations in the UK has declined to very low levels over the 16 years of measurements in AGANet, with annual mean concentrations in 2015 (0.25 μg SO m , n = 12) approaching that of the other acid gases HNO (0.21 μg HNO m , n = 12) and HCl (0.20 μg HCl m , n = 12). NH measured at the same time at the AGANet sites also decreased, but to a smaller extent, to a mean concentration of 1.0 μg NH m (n = 12) in 2015. The changes in measured concentrations of SO, HNO, HCl and NH are consistent with the estimated decrease in emissions of SO, NO , HCl, and NH since 2000. SO is therefore no longer the dominant acid gas, with HNO and HCl together contributing a larger fraction of the total acidity in the UK atmosphere.

Past studies have shown that the increasing ratio of NH to SO in the atmosphere leads to increased dry deposition of SO, accelerating the decrease in atmospheric SO concentrations than would be achieved by emissions reduction alone (Fowler et al., 2001, 2009; ROTAP 2012). The dry deposition of SO and NH, by uptake of the gases in a liquid film on leave surfaces, are known to be enhanced when both gases are present in a process termed “co-deposition” (Fowler et al., 2001). Where ambient NH concentrations exceed that of SO, there is enough NH to neutralize acidity in the liquid film and oxidize deposited SO, and maintain large rates of deposition of SO. With changes in the relative concentrations of acid gases in the UK and across Europe however, the deposition rates will increasingly be controlled by the NH /combined acidity (sum of SO, HNO and HCl) molar ratio, rather than based on SO alone (Fowler et al., 2009).

To look at the UK situation, an analysis of the molar ratios of NH to acid gases is presented in Fig. 18a. The molar ratio of NH to acid gases (sum of SO, HNO and HCl) increased with time, from 1.9 in 2000 to 4.7 in 2015, confirming that NH is increasingly in molar excess over atmospheric acidity. The ratio of annual mean concentrations of NH (80 nmol m ) to SO (29 nmol m ) was 2.7 in 2000. By 2015, this ratio had increased to 15 (annual mean concentrations of NH = 58 nmol m cf. SO = 4 nmol m ). Molar concentrations of HNO (4 nmol m ) and HCl (6 nmol m ) were comparable to SO in 2015, highlighting the increasing importance of HNO and HCl in contributing to atmospheric acidity. A larger decrease in SO (−81%) than particulate sulfate (−69%) in the AGANet data (Table 4) would appear at first to suggest that the large NH : SO ratio is contributing to a more rapid decrease in SO concentrations. However, when the sea salt fraction of SO is removed from the sulfurate trend (Sect. 3.8.2), the decrease in nss SO (−78%) is similar to SO (−81%) (Table 4). Since the decreasing trend in the ratio of SO to SO also appeared to stabilize after 2006 (Sect. 3.8.2), this would suggest that maximum deposition rates for SO may
Currently, the critical loads of acidity (sulfur and nitrogen) are exceeded by 44% of the area of sensitive habitats in the UK (based on mean deposition data for 2012–2014), whereas the figure for exceedance of eutrophication (nutrient nitrogen) is even larger, at 62% (based on deposition data for 2012–2014) (Hall and Smith, 2016). Air quality policies have been very successful in abating SO$_2$ emissions (−80%; 2000–2015) and moderately successful with NO$_x$ emissions (−58%; 2000–2015), with both on course to meet the emission reduction targets set out under the 2012 Gothenburg protocol and 2016 NECD. Difficulties in abating NH$_3$ is reflected in the smaller reduction in NH$_3$ emissions (−9%; 2000–2015), with emissions increasing, rather than decreasing since 2013 and it is likely that abatement measures may be required to meet emission reduction targets. In recognizing the need to tackle the ammonia problem, the Code of Good Agricultural Practice (COGAP) was published under the UK government’s Clean Air Strategy (launched in July 2018) as a step towards reducing NH$_3$ emissions from agriculture.

Based on the current emission trends and evidence from AGANet and NAMN long-term measurements, atmospheric N deposition from oxidized N (NO$_x$, HNO$_3$ and NO$_2^-$) and from reduced N (NH$_3$, NH$_4^+$) are likely to continue to exceed critical loads of N deposition over large areas of sensitive habitats, with implications for UK’s commitment to maintain or restore natural habitats (e.g. Natura 2000 sites; Hallsworth et al., 2010) to a favourable conservation status under the EU Habitats Directive (Council Directive 92/43/EEC) and ecosystem monitoring under Article 9 and Annex V of Directive 2016/2284 (NECD). The changes are also relevant for human health effects assessment, since NH$_4$NO$_3$ and (NH$_4$)$_2$SO$_4$ are mainly in the fine mode and constitute a significant fraction of PM$_{2.5}$ that are associated with acute and chronic human health problems. The change in partitioning from (NH$_4$)$_2$SO$_4$ to NH$_4$NO$_3$, coupled to import of NH$_4$NO$_3$ from long-range transport (driven by emissions of NH$_3$ and NO$_x$ from outside the UK) poses policy challenges in protection of human health from effects of air pollution.

Figure 18. Long-term changes between 2000 and 2015 in (a) molar ratio of NH$_3$ to acid gases (SO$_2$, HNO$_3$, and HCl) and (b) molar ratio of particulate NH$_4^+$ to acid aerosols (SO$_4^{2-}$ and NO$_3^-$) from measurements made at 12 sites in AGANet.

have been reached with the smaller SO$_2$ concentrations since 2006.

The substantial decrease in UK SO$_2$ emissions and concentrations, while UK NO$_x$ emissions and concentrations remain relatively high in comparison, set against a much smaller decrease in NH$_3$ emissions and concentrations since 2000 is leading to changes in the respective particulate SO$_4^{2-}$/NH$_4^+$ and NO$_3^-$ concentrations. Since the affinity of H$_2$SO$_4$ (oxidation product of SO$_2$) for NH$_3$ is much larger than that of HNO$_3$ and HCl, available NH$_3$ is first taken up by H$_2$SO$_4$ to form ammonium sulfate compounds (NH$_4$HSO$_4$ and (NH$_4$)$_2$SO$_4$), with any excess NH$_3$ then available to react with HNO$_3$ and HCl to form NH$_4$NO$_3$ and NH$_4$Cl. Analysis of the different particulate components in Sect. 3.5 showed that the ammonium aerosols are mainly made up of (NH$_4$)$_2$SO$_4$ and NH$_4$NO$_3$. With the large reduction in SO$_2$, more NH$_3$ is available to react with HNO$_3$ to form NH$_4$NO$_3$ and concentrations of NH$_4^+$ and NO$_3^-$ are now observed to be in molar excess over SO$_4^{2-}$, providing evidence of a change in the particulate phase from (NH$_4$)$_2$SO$_4$ to NH$_4$NO$_3$ (Fig. 18b).

A change to an NH$_4$NO$_3$ rich atmosphere and the potential for NH$_4$NO$_3$ to release NH$_3$ and HNO$_3$ in warm weather, together with the surfeit of NH$_3$ also means that a larger fraction of the reduced and oxidized N is remaining in the gas phase as NH$_3$ and HNO$_3$. An increased partitioning to the gas phase may account for the larger decrease in particulate NH$_4^+$ (MK −62 % between 2000 and 2015, n = 12) and NO$_3^-$ (MK −52 % between 2000 and 2015, n = 12) than NH$_3$ (MK −30 % between 2000 and 2015, n = 12) and HNO$_3$ (MK −45 % between 2000 and 2015, n = 12) (Table 4) and the increase in gas to aerosol ratios (NH$_3$ : NH$_4^+$ and HNO$_3$ : NO$_3^-$) over the 16 year period (Fig. 16). A higher concentration of the gas-phase NH$_3$ and HNO$_3$ may therefore be maintained in the atmosphere than expected on the basis of the emissions trends in NH$_3$ and NO$_x$. Given the larger deposition velocities of NH$_3$ and HNO$_3$ compared to aerosols, more of the NH$_3$ and HNO$_3$ emitted will have the potential to deposit more locally with a smaller footprint within the UK.
particularly in urban areas where concentrations of the PM$_{2.5}$ precursor gases NO$_x$, SO$_2$ and NH$_3$ are higher.

4 Conclusions

The UK Acid Gases and Aerosol network (AGANet) is delivering, uniquely, a comprehensive UK long-term dataset of speciated acid gases (HNO$_3$, SO$_2$, HCl) and aerosol components (NO$_3^-$, SO$_4^{2-}$, Cl$^-$, Na$^+$, Ca$^{2+}$, Mg$^{2+}$) and also of NH$_3$ and NH$_4^+$ measured within the National Ammonia Monitoring Network (NAMN). Speciated measurements are made with an established low-cost DELTA denuder-filter pack methodology, allowing assessment of atmospheric chemical composition and gas–aerosol phase interactions. Other manual denuder-filter implementations designed for high time-resolution measurements are useful at selected locations for detailed analysis and model testing, but they are resource intensive and expensive. The DELTA monthly measurements on the other hand are cost-efficient for estimating annual mean concentrations, providing sufficient resolution for analysis of temporal trends, and which can be operated at a large number of sites in the network to provide long-term trends and temporal and spatial patterns.

Large regional patterns in concentrations are observed, with the largest concentrations of HNO$_3$, SO$_2$, and aerosol NO$_3^-$ and SO$_4^{2-}$ in southern and eastern England, attributed to anthropogenic (combustion, vehicular) and long-range transboundary sources from Europe, and smallest in western Scotland and Northern Ireland. HCl concentrations are also largest in the south-eastern, south-western and central England, attributed to dual contribution from anthropogenic (coal combustion) and marine sources (reaction of sea salt with HNO$_3$ and H$_2$SO$_4$ to form HCl). For Cl$^-$, this has a similar spatial distribution as Na$^+$, with the highest concentrations at coastal sites, reflecting their origin from marine sources (sea salt).

Distinctive temporal trends are established for the different components, with the seasonal variability influenced by local to regional emissions, climate, meteorology and photochemistry. A weak seasonal cycle is observed in HNO$_3$, with slightly higher concentrations in late winter and early spring, due to formation from photochemical production processes. Particulate NO$_3^-$ and SO$_4^{2-}$ have highest concentrations in spring, coinciding with the peak in concentrations of NH$_3$ and NH$_4^+$, and are therefore likely to be attributed to formation of NH$_4$NO$_3$ and (NH$_4$)$_2$SO$_4$ from reaction with a surplus of higher concentrations of NH$_3$ at that time of year. Conversely, peak concentrations of SO$_2$, Na$^+$, and Cl$^-$ occur during winter, likely from combustion processes (heating) for SO$_2$ and marine sources in winter (more stormy weather) for sea salt generation. Magnesium is a crustal element, but which is also present in sea salt aerosols. The seasonal trend in Mg$^{2+}$ is similar to Na$^+$, with maxima during winter and minima in summer; therefore some of the sea salt aerosol may be in the form of MgCl$_2$.

Enhancement of local to regional concentrations of reactive gases and aerosols in the UK from long-range transboundary transport of pollutants into the UK is highlighted by two pollution events, captured in the long-term AGANet monthly measurements. In 2003, a spring episode with elevated concentrations of HNO$_3$ and NO$_3^-$ was driven by meteorology, with easterly winds transporting NH$_4$NO$_3$ formed in Europe into the UK and a high pressure system over the UK (February–April) that led to a build-up of NH$_4$NO$_3$ and HNO$_3$ concentrations from both local and transboundary sources. A second, but smaller episode of elevated concentrations of SO$_2$ and HNO$_3$, as well as of particulate SO$_2^-$, NO$_3^-$, and NH$_4^+$, in September 2014 was shown to be from transport of pollutant plume from the Icelandic Holuhraun volcanic eruptions at that time.

After more than 16 years of operation, the AGANet is also capturing important long-term changes in the concentrations and partitioning between gas and aerosol of the N and S components in the atmosphere. A significant decrease of $\sim 81\%$ (MK) in annual mean concentrations of SO$_2$ between 2000 and 2015 was in agreement with the estimated $\sim 80\%$ reduction in SO$_2$ emissions, but larger than the accompanying decline in particulate SO$_2^{2-}$ ($\sim 69\%$ MK). A more modest reduction in HNO$_3$ ($\sim 45\%$ MK) and particulate NO$_3^-$ ($\sim 52\%$ MK) are consistent with the estimated $58\%$ decline in NO$_x$ emissions over this same period. The decrease in particulate NH$_4^+$ ($\sim 62\%$ MK) is larger than the precursor gas NH$_3$ (2000–2015 = $\sim 30\%$ MK / LR) and larger than the estimated decline in estimated NH$_3$ emissions of $9\%$. However, it should be noted that NH$_3$ trends are highly dependent on site selection according to an earlier assessment made on a more comprehensive dataset from the UK NAMN.

The substantial decrease in UK SO$_2$ emissions and concentrations, while UK NO$_x$ emissions and concentrations (HNO$_3$) remain relatively high in comparison, set against a much smaller decrease in NH$_3$ emissions and concentrations since 2000 is leaving more NH$_3$ available to react with HNO$_3$ to form the semi-volatile particulate NH$_4$NO$_3$. Particulate NH$_4^+$ and NO$_3^-$ are now in molar excess over SO$_4^{2-}$, providing evidence of a shift in the particulate phase from (NH$_4$)$_2$SO$_4$ to NH$_4$NO$_3$. A change to an NH$_4$NO$_3$ rich atmosphere and the potential for NH$_4$NO$_3$ to release NH$_3$ and HNO$_3$ in warm weather, together with the surfeit of NH$_3$ also means that a larger fraction of the reduced and oxidized N is remaining in the gas phase as NH$_3$ and HNO$_3$. The change in partitioning from particulate NH$_4^+$ to gaseous NH$_3$ is also occurring in other parts of Europe, where decreases in NH$_3$ concentrations have been smaller than emission trends would suggest, due to successful mitigation in SO$_2$ emissions. Higher concentrations of the NH$_3$ and HNO$_3$ in the atmosphere will deposit more locally, exacerbating the effects of local N deposition loads over large areas of sensitive habitats, with implications for UK’s commitment to
maintain or restore natural habitats (e.g. Natura 2000 sites) to a favourable conservation status under the EU Habitats Directive (Council Directive 92/43/EEC) and ecosystem monitoring under Article 9 and Annex V of Directive 2016/2284 (NECD). The changes are also important in terms of human effects assessment since NH$_4$NO$_3$ constitute a significant fraction of PM$_{2.5}$ that are implicated in acute and chronic human health effects and linked to increased mortality from respiratory and cardiopulmonary diseases.

**Data availability.** Ratified data from the Acid Gases and Aerosol Network (AGANet) and the National Ammonia Monitoring Network (NAMN) are publically accessible on the Defra UK-AIR website (https://uk-air.defra.gov.uk/data, last access: 30 March 2018).

**Author contributions.** MAS conceived and proposed the establishment of the NAMN and AGANet. MAS and YST developed the methods (ALPHA and DELTA) and set up the NAMN and AGANet. UD assisted in building the DELTA equipment and conducting the DELTA vs. ADS intercomparison during the initial phase. IS assisted in designing, building, and installing DELTA equipment for the expanded AGANet. CFB provided resources and coordinated and managed the UKEAP network. CC and KV coordinated and managed the Precip-net and NO2-net and also the former ADMN networks. YST, IS, NvD, and CC organized and carried out UKEAP network site servicing. YST, NvD, JP, ST, MP, POK, and HC performed the sample preparation and chemical analyses, POK and AL supervised and performed quality checks on chemistry analyses. MGP provided resources and managed and coordinated chemistry analyses. DL carried out data curation and designed the CEH AAGA database for NAMN and AGANet data. KV was involved in producing and maintaining the Defra UK-AIR website. YST coordinated and managed the NAMN and AGANet (1997–2016), performed the data collection, data analysis, and wrote the paper. MS, MRH, CFB, UD, NvD, and KV provided input on the paper.

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