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Aerosol source apportionment from 1-year measurements at the CESAR tower in Cabauw, the Netherlands

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Abstract. Intensive measurements of submicron aerosol particles and their chemical composition were performed with an Aerosol Chemical Speciation Monitor (ACSM) at the Cabauw Experimental Site for Atmospheric Research (CESAR) in Cabauw, the Netherlands, sampling at 5 m height above ground. The campaign lasted nearly 1 year from July 2012 to June 2013 as part of the EU-FP7-ACTRIS project (Q-ACSM Network). Including equivalent black carbon an average particulate mass concentration of $9.50\,\mu g\,m^{-3}$ was obtained during the whole campaign with dominant contributions from ammonium nitrate (45 %), organic aerosol (OA, 29 %), and ammonium sulfate (19 %). There were 12 exceedances of the World Health Organization (WHO) $PM_{2.5}$ daily mean limit ($25\,\mu g\,m^{-3}$) observed at this rural site using PM_1 instrumentation only. Ammonium nitrate and OA represented the largest contributors to total particulate matter during periods of exceedance.

Source apportionment of OA was performed season-wise by positive matrix factorization (PMF) using the multilinear engine 2 (ME-2) controlled via the source finder (SoFi). Primary organic aerosols were attributed mainly to traffic (8–16 % contribution to total OA, averaged season-wise) and biomass burning (0–23 %). Secondary organic aerosols (SOAs, 61–84 %) dominated the organic fraction during the whole campaign, particularly on days with high mass loadings. A SOA factor which is attributed to humic-like substances (HULIS) was identified as a highly oxidized background aerosol in Cabauw. This shows the importance of at-

mospheric aging processes for aerosol concentration at this rural site. Due to the large secondary fraction, the reduction of particulate mass at this rural site is challenging on a local scale.

1 Introduction

Atmospheric aerosols have large impacts on the climate directly by scattering and absorbing shortwave radiation. Besides the resulting influence on the visibility (Ramanathan et al., 2007; Romanou et al., 2007), this can have a cooling or heating effect on the atmosphere (IPCC, 2013). The indirect effect refers to the impact of particles on cloud formation and their properties.

In addition, particles can adversely impact human health by, e.g., increasing the probability of cardiopulmonary and lung cancer mortality (Pope et al., 2002). The World Health Organization (WHO) recently estimated that, globally, 3.7 million deaths were attributable to ambient air pollution in both cities and rural areas in 2012 (EU, 2008). This mortality is reported to be due to exposure to small particulate matter (PM_{10}), which can cause cardiovascular and respiratory disease and cancers. Particles with smaller diameters such as $PM_{2.5}$ or PM_1 are reported to have enhanced toxicological effects since they can deposit more deeply in the respiratory system and remain suspended for longer periods of time (Pope and Dockery, 2006). Therefore, a num-

ber of institutions established several air quality standards for different particle sizes to limit aerosol mass. The WHO air quality guideline (global update 2005, WHO, 2006) defines a $PM_{2.5}$ daily mean limit of $25 \, \mu g \, m^{-3}$ and a $PM_{2.5}$ annual mean limit of $10 \, \mu g \, m^{-3}$. The European Union Air Quality Directive 2008/50/EC provides only a target value of the annual mean limit of $PM_{2.5}$ of $25 \, \mu g \, m^{-3}$ (EU, 2008).

Air quality and climate effects are not only dependent on the particle number concentration and size but also on their chemical composition. This information is not only relevant to investigate the nature and magnitude of each effect but also for the identification and quantification of aerosol sources and mitigation strategies for a potential reduction of aerosol mass concentrations. Major inorganic components of PM_1 consist mainly of ammonium nitrate (NH_4NO_3) and ammonium sulfate ((NH_4)$_2SO_4$), formed in the presence of ammonia (NH_3), nitrogen oxides (NO_x is NO plus NO_2), and sulfur dioxide (SO_2), respectively (Seinfeld and Pandis, 2006). Therefore, NH_4NO_3 and (NH_4)$_2SO_4$ are strongly attributed to anthropogenic sources (Finlayson-Pitts and Pitts, 2000). Since inorganic aerosols are formed by gaseous precursors in the atmosphere, the condensed products are considered as secondary aerosols, while primary aerosols like black carbon (BC) are emitted directly.

In contrast, organic aerosols (OAs), which can also be of primary (POA) or secondary (SOA) origin, consist of up to hundreds of thousands of different molecules (Goldstein and Galbally, 2007), where SOA contributes on average 70 % to organic aerosol mass (Hallquist et al., 2009). SOA is formed by gas-to-particle conversion of atmospherically oxidized semi- and low-volatile organic compounds (VOCs). Guenther et al. (1995) estimated a global VOC budget in the order of 1150 Tg carbon per year. Biogenic VOCs (BVOCs) contribute approximately 90 % of total VOC, including isoprene (50 % of total BVOCs), monoterpenes (15 %), and sesquiterpenes (3 %) (Guenther et al., 2012). In turn, 10 % of emitted VOCs are of anthropogenic origin, including, e.g., alkanes, alkenes, benzene, and toluene.

The investigation of the aerosol composition is critical for the development of climate models, since the composition influences important particle properties. However, its determination is still challenging, especially in case of OAs, which contribute significantly to atmospheric particulate matter (Jimenez et al., 2009). In fact, the lack of knowledge on particle composition is a key contribution to the large uncertainty for the determination of the total anthropogenic radiative forcing (IPCC, 2013).

The development of online aerosol mass spectrometric techniques during the last decades enhanced the possibilities to investigate aerosol chemical composition in real time (De-Carlo et al., 2006; Jayne et al., 2000; Jimenez et al., 2003). The Aerosol Mass Spectrometer (AMS, Aerodyne Research Inc., Billerica, MA, USA) is a powerful instrument to quantitatively measure the chemical composition of the nonrefractory fraction of PM_1 (NR-PM_1) with high time and mass res-

olution. Due to the high amount of maintenance from skilled and trained personnel needed for continuous operating measurement, campaigns using an AMS usually do not exceed 1 or 2 months (Sun et al., 2012). Since the variation of aerosol composition is very high depending on measurement site and season (Jimenez et al., 2009), long-term measurements (≥ 1 year) are clearly needed. At a European level, this effort is supported by the Aerosols, Clouds and Traces gases Research InfraStructure network (ACTRIS) program that aims at pooling high-quality data from state-of-the-art instrumentation, such as the Aerodyne Aerosol Chemical Speciation Monitor (ACSM). The quadrupole ACSM (Q-ACSM) is specially designed for long-term continuous and real-time measurements of mass concentrations and composition of NR-PM_1 species (Ng et al., 2011b).

In this study, an ACSM was used to measure the submicron aerosol chemical composition from 11 July 2012 to 3 June 2013 at the CESAR tower in Cabauw, the Netherlands, as part of the EU-FP7-ACTRIS project. A collocated multi-angle absorption photometer (MAAP, Thermo Scientific Model 5012) provided equivalent black carbon (eBC) data. Organic aerosol data were further analyzed by positive matrix factorization using the multilinear engine 2 (ME-2) (Paatero, 1999) via the source finder (SoFi, Canonaco et al., 2013). This data set shows the long-term variability of particle composition and is used for source apportionment of atmospheric aerosols at this northwestern European rural site, with the focus on periods where air quality standards were violated. Information about aerosol composition and their sources especially during pollution events can be further used to establish strategies for the reduction of particulate matter.

2 Methodology

2.1 Site description: CESAR

The CESAR tower is 220 m high and managed and operated by the Royal Netherlands Meteorological Institute (KNMI, the Netherlands). It is located in a rural site (51.970° N, 4.926° E) near Cabauw, the Netherlands, about 20 km southwest of the city of Utrecht and about 45 km southeast of the Dutch North Sea coast. The site conditions are typical for northwestern Europe. They can either be maritime or continental, depending on the wind direction. The surface elevation changes in the surrounding area are, at most, a few meters over 20 km. The tower ground is approximately 0.7 m below sea level; the diurnal variation of the temperature is relatively stable (Vermeulen et al., 2011). The direct surroundings of the tower have a relatively low population density. The nearby region is used mainly by agriculture, with a mixture of intensively and extensively managed grassland. These are used also for keeping animals like cattle and sheep beside nearby chicken farms.

The tower is equipped with external platforms and booms at 2, 10, 20, 40, 80, 140, and 200 m. At all these levels, meteorological observations of standard parameters like wind speed, wind direction, dew point temperature, and ambient temperature are routinely performed (Ulden and Wieringa, 1996). These data sets are available at the CESAR database (KNMI, 2013). In addition, other meteorological data like precipitation, radiation, and remote sensing, including lidar, radar, and radiometer techniques, are acquired at the tower and submitted to the CESAR database. Also concentrations of gaseous compounds, including greenhouse gases, are monitored at CESAR (Russchenberg et al., 2005; Vermeulen et al., 2011). The National Institute for Public Health and the Environment (RIVM, the Netherlands) provides hourly data from gaseous CO, NO, NO_2, SO, SO_2, and O_3, but also daily PM_{10} measurements. Data of ambient CO_2, CH_4 (both measured at 20, 60, 120, and 200 m height), and radon 222 (^{222}Rn, measured at 20 and 200 m height) are determined by ECN on a timescale of 30 min. A map indicating the land use of the Netherlands is given in the Supplement (Fig. S1).

2.2 ACSM sampling and data analysis

The ACSM measures the NR-PM_1 fraction, including the organic fraction (Org), ammonium (NH_4), nitrate (NO_3), sulfate (SO_4), and chloride (Chl), using mass spectrometric information on a timescale of approximately 30 min. The ACSM used in this work is equipped with a commercial grade quadrupole residual gas analyzer (RGA); thus, it is also called a Q-ACSM. This instrument is described in detail by Ng et al. (2011b). The RGA provides unit mass resolution (UMR) mass spectra. Fractions of measured UMR signals were assigned to individual aerosol species using the fragmentation table introduced by Allan et al. (2004). Due to the automated zeroing system used for the ACSM, adjustments of the fragmentation table entries concerning interferences from air beam molecules are not needed. All ACSM data processing and analysis within this work was performed using software provided by Aerodyne Research (ACSM Local, version 1.531, ARI, 2012) within IGOR Pro version 6.2.3.

Recent studies showed good agreements of ACSM data with data from other aerosol instruments like the high-resolution time-of-flight AMS (HR-ToF-AMS) in ambient measurement campaigns, even in highly polluted areas such as Beijing (Sun et al., 2012, 2011). An intercomparison of two collocated ACSMs resulted in strong correlations ($R^2 = 0.8$) and agreements within 27 % (Budisulistiorini et al., 2014). Crenn et al. (2015) reported similar results from the intercomparison of 13 ACSMs, a ToF-ACSM, a ToF-AMS, and other collocated instruments in the region of Paris, France. There, the same ACSM instrument (S/N A140-145) as the one used for this study was tested. Those results indicate that the ACSM can be used as a suitable and cost-effective alternative to the AMS for aerosol composition

measurements due to its capability of stable and reproducible operation.

Mass calibrations were performed approximately every month and were based on determining the instrument response factor (RF) (Ng et al., 2011b) using monodisperse NH_4NO_3 (320 nm) as calibration substance and a condensation particle counter (CPC, TSI 3022a) as reference instrument (Jayne et al., 2000; Jimenez et al., 2003). An average RF_{NO_3} of $2.74 \pm 0.45 \times 10^{-11}$ was obtained and used for the calculation of aerosol mass concentrations. Instead of performing a mass calibration for every aerosol species, relative ionization efficiencies (RIEs), compared to those of nitrate, were used. The RIEs of NH_4 and SO_4 were determined directly during the mass calibrations by measuring dry NH_4NO_3 and $(NH_4)_2SO_4$ particles after another. Averaged over all calibrations, a RIE_{NH_4} of 7.53 ± 0.21 and a RIE_{SO_4} of 0.81 ± 0.10 were found and used for the whole data set. Calibration results gained from this particular instrument during an intercomparison in Paris as described by Crenn et al. (2015) were within the uncertainty or only slightly different ($RF_{NO_3} = 2.34 \times 10^{-11}$, $RIE_{NH_4} = 6.54$, $RIE_{SO_4} = 0.62$) considering the large differences between individual instruments at the intercomparison. RIE values of 1.4 and 1.3 for organics and chloride, respectively, were taken from the literature (Alfarra et al., 2004; Canagaratna et al., 2007).

A site-specific, time-resolved particle collection efficiency (CE) correction algorithm (equations are given in the Supplement) was applied, which was developed by Mensah et al. (2012), using SMPS data as reference. In contrast to the commonly used constant value of 0.5, this CE correction accounts for the high ammonium nitrate mass fraction (ANMF) found at this site and is thus more suitable for the data presented here. Another algorithm for composition-dependent CE determination (Middlebrook et al., 2012) was also tested for its validity. It uses a threshold ratio of measured-to-predicted NH_4 to switch between two different equations to determine the CE. The threshold value of 0.75 is close to the observed ratio of measured-to-predicted NH_4 of this data set, resulting in large discontinuities of CE values and in consequence, discontinuous changes in aerosol mass concentrations. Therefore, the Middlebrook algorithm was not used for this data set, which showed, at the same time, low ratios of measured-to-predicted NH_4 and high ANMFs. According to Ng et al. (2011b) and Sun et al. (2012), the variability of the instrument performance was corrected based on the inlet pressure and N_2 signal, as well as the signals derived from the internal naphthalene source were taken to correct for the mass-dependent ion transmission efficiency of the RGA.

The detection limits for each species were not determined within this work but are reported to be 0.148, 0.284, 0.012, 0.024, and 0.011 µg m^{-3} (3σ) for organics, ammonium, nitrate, sulfate, and chloride, respectively, at an averaging time of 30 min (Ng et al., 2011b).

During the whole campaign, the ACSM was located inside the CESAR tower building. Its inlet was placed on the roof

of the building at approximately 5 m height above ground. The inlet head was equipped with a PM$_{2.5}$ cyclone (URG-2000-30EN, URG Corporation, Chapel Hill, USA). The sample air was pulled through a 10 m copper line (ID = 6.5 mm) at a flow rate of 9 L min^{-1}. From this flow, a subsample of 1 L min^{-1} was diverted to a Nafion dryer (RH < 40 %) of which approximately 80 mL min^{-1} entered the ACSM. This resulted in a total residence time of 18 s.

2.3 Collocated aerosol measurements

The following collocated aerosol instruments were used for cross validation of the ACSM data: (i) a scanning mobility particle sizer (SMPS, TSI 3034), operated by the Netherlands Organization for Applied Scientific Research (TNO, the Netherlands); (ii) a monitor for aerosol and gases (MARGA, Applikon Analytical BV), operated by ECN; and (iii) a HR-ToF-AMS, which was operated by Forschungszentrum Jülich during the first 6 days of the ACSM campaign. In addition, BC data obtained by a MAAP instrument (TNO, the Netherlands) were included into the analysis.

The MAAP instrument has been introduced by Petzold and Schönlinner (2004) and Petzold et al. (2005). It is designed for the determination of the black carbon (BC), which is a product of incomplete combustion. There is a general consensus in the scientific community about the properties of black carbon (Bond et al., 2013). The MAAP measures the strong visible light absorption property of BC by simultaneous measurements of the radiation penetrating through and scattered back from a particle-loaded fiber filter. According to Petzold et al. (2013), optical BC determined by MAAP is to be referred to as equivalent black carbon (eBC). One property of BC is that it is highly refractory with a vaporization temperature near 4000 K (Schwarz et al., 2006); thus, BC is not vaporized at 600 °C and cannot be measured by the ACSM. The MAAP achieves a time resolution of 5 min with an uncertainty of 12 % (Petzold and Schönlinner, 2004). A mass absorption cross section (MAC) of 6.6 m^2 g^{-1} for a wavelength of 637 nm (Müller et al., 2011) was chosen to convert the measured particle absorption coefficient to eBC mass concentrations. Although the MAAP has no size-selective inlet beside the PM$_{10}$ heads described below, it can be assumed that eBC-containing aerosols generally fall into the submicron size range (Bond et al., 2013). Thus, eBC mass concentrations are considered as part of the PM$_1$ fraction from here on. As seen later, the eBC fraction is rather low throughout the campaign, meaning that the overall error of this assumption is not significant.

The SMPS (TSI, Model 3034) is a sequential combination of several integrated components: an impactor, a neutralizer, a differential mobility analyzer, and a condensation particle counter. It determines the size distribution of particles in a range of 10–487 nm (electromobility diameter). The SMPS aerosol mass concentration was calculated from the measured volume distributions using the particle density deter-

mined by the aerosol composition information derived from the ACSM and the MAAP. Assuming spherical particles, the total density is computed by using the densities of the aerosol species, weighted by their mass fractions. Bulk densities of NH$_4$NO$_3$ (1.72 g cm^{-3}) and (NH$_4$)$_2$SO$_4$ (1.77 g cm^{-3}), and densities of 1.8 g cm^{-3} for BC (Bond and Bergstrom, 2006; Park et al., 2004) of organics (1.4 g cm^{-3}, Hallquist et al., 2009) were taken into account. Considering its low influence on the total particle density at this site, it is acceptable to set the density for chloride to 1 g cm^{-3} (Mensah et al., 2012).

During the presented campaign, the MAAP and the SMPS were connected to the common aerosol inlet which sampled at 60 m height. This inlet consisted of four PM$_{10}$ size-selective heads at the top, followed by a Nafion dryer to keep the relative humidity (RH) of the sample air below 40 %. The stainless steel pipe, ranging from the aperture at 60 m to the basement, has an inner diameter of 0.5′′ (1.27 cm) and ends in a manifold, where the sampled air is distributed to a variety of different instruments, including the MAAP and the SMPS, each with its own sample flow. An overall sample flow of 60 L min^{-1} was adjusted inside the 60 m pipe, assuring laminar conditions.

SMPS data were corrected size dependently for (diffusional) losses in the inlet system and SMPS system itself according to Henzing (2011), who compared theoretical findings with measured losses that are obtained by measuring simultaneously before and after the various parts of the inlet system at the CESAR tower. In addition, particles of different compositions were measured in 2013 simultaneously at the pipe entrance at 60 m height and in the basement (J. S. Henzing, personal communication, 2014). For more than 8000 simultaneous observations, the results showed that aerosol measurements through this 60 m sampling line underestimate PM$_{10}$ eBC by approximately 33 % with an uncertainty of 7 %. Therefore, eBC obtained from the MAAP are divided by a factor of 0.66 to account for these losses. For the inorganic species, penetrations through this inlet line were reported to be 62–73 % for nitrate, 55–64 % for sulfate, and 54–56 % for ammonium. However, these results were not used for corrections in this work.

The MARGA measures water-soluble components of both the gas and aerosol phase simultaneously, including the aerosol species nitrate, ammonium, sulfate, and chloride. It uses a wet annular denuder (WAD) followed by a steam jet aerosol collector (SJAC). The operational and calibration procedures, as well as details of the data analysis are given by Trebs et al. (2004). During the campaign the MARGA collected alternately the PM$_1$ and PM$_{2.5}$ fraction of ambient particles at ambient RH and temperature, each fraction sampled hourly with a measurement error smaller than 10 % (Schaap et al., 2011). The detection limits for the most abundant aerosol species were 0.05, 0.10, 0.08, and 0.01 µg m^{-3} (4σ, Rumsey et al., 2014) for ammonium, nitrate, sulfate, and chloride, respectively.

The MARGA inlet was equipped with a PM_{10} size-selective head (Rupprecht and Pataschnick, R&P), placed on the roof of the tower building next to the ACSM inlet aperture at 5 m height. The sample air was transferred into the instrument within a polyethylene tube with an inner diameter of $0.5''$ (1.27 cm) and a sample flow of $16.7\,L\,min^{-1}$, which is either directed through a PM_1 or a $PM_{2.5}$ size-selective cyclone. A detailed description of the MARGA inlet system at the Cabauw tower was previously described by Schaap et al. (2011). There, wall losses were investigated and found to be less than 2 % for several gaseous and particulate compounds. To compare with the ACSM, only MARGA data containing PM_1 composition are considered within this work.

An Aerodyne HR-ToF-AMS was connected to the MARGA inlet during the first 6 days of the campaign. As the ACSM is built upon AMS, the latter likewise measures the chemical composition of the PM_1 nonrefractory (NR) fraction. Instrumental details are available in Jayne et al. (2000), Jimenez et al. (2003), and DeCarlo et al. (2006).

2.4 Statistical methods of organic aerosol data analysis

Source apportionment of organic aerosol components was performed using positive matrix factorization (PMF, Paatero, 1997; Paatero and Tapper, 1994) via the ME-2 solver (Paatero, 1999). PMF is a bilinear model and assumes that the original data set, containing variable mass spectra over time, is a linear combination of a given number of factors, each with a constant mass spectrum and varying contribution over time. It has been successfully used in AMS ambient studies apportioning the measured organic mass spectra in terms of source- and process-related components (Zhang et al., 2011). With the ME-2 solver it is possible to introduce a priori mass spectral information and hence to reduce the rotational ambiguity, i.e., similar PMF results with the same goodness of fit, of PMF solutions (Paatero and Hopke, 2003).

The extraction of OA data and error matrices as mass concentrations in $\mu g\,m^{-3}$ over time, as well as their preparation for PMF/ME-2 according to Ulbrich et al. (2009), was done within the ACSM software. Only m/zs ≤ 100 were considered here since they represented nearly the whole OA mass (around 98 %) and did not interfere with ion fragments originating from naphthalene (e.g., m/z 127, 128, and 129, see also Sect. 2.2). From these matrices, the m/z 12 was discarded because it showed negative signals, probably due to a too-short delay time of the quadrupole scan (125 ms) after a valve switch (Fröhlich et al., 2015). In addition, the m/z 37 and m/z 38 were also removed from the organic matrices of the whole campaign except for winter 2013. This was done because the signal at these masses showed high interferences with the chloride-related ions ^{37}Cl and $H^{37}Cl$. Including these ions led to unreasonable PMF factors which mainly contained only these two masses and represented the

chloride time series, whereas during winter 2013 no such interferences were observed.

The interface source finder (SoFi, Canonaco et al., 2013) version 4.8 was used to control ME-2 for the PMF runs of the ACSM OA data adopting the source apportionment strategy developed by Crippa et al. (2014). Briefly, unconstrained PMF runs were first investigated with 1–10 factors and a moderate number of seeds (10–15) for each factor number. If primary organic aerosol (POA) factor profiles such as hydrocarbon-like OA (HOA) or OA from biomass burning (BBOA) were found, site-specific POA mass spectra or spectra derived from the database were constrained and a sensitivity analysis performed on the tightness of constraint (a value, Lanz et al., 2008). Since aged OA (or oxidized OA, OOA) factors show more variability between measurement sites in terms of their mass spectra (MS) than POA, it is not appropriate to constrain SOA factor profiles using reference spectra derived from different locations (Canonaco et al., 2015). According to Crippa et al. (2014), HOA MS should be more constrained (a value between 0 and 0.2) than BBOA MS ($a \approx 0.3$), since the BBOA fingerprint shows higher variations depending on the burning type and conditions (Alfarra et al., 2007). In each step, either in unconstrained or constrained approaches, the following criteria for finding a proper solution were used based on the recommendations from Ulbrich et al. (2009) and Canonaco et al. (2013):

- The quality parameter Q/Q_{exp} was minimized.

- Factor profiles have reasonable MS, as expected for the measurement site.

- Factor time series have high correlations with respective external data sets such as gaseous CO, CO_2, NO_x, and particulate nitrate, sulfate, and black carbon.

- When a proper solution is found, 50 seed runs were used to find the global minimum for Q/Q_{exp}.

- Investigation of the rotational ambiguity of the solution space is carried out using the a value approach for the constrained factor profiles.

3 Results and discussion

3.1 Cross validation of particulate total mass and chemical species concentrations

The particle density during the ACSM campaign was determined using the chemical composition data from the ACSM and the MAAP and resulted in an average of $1.63 \pm 0.12\,g\,cm^{-3}$. The time series of the particle density is given in the Supplement. It was used to calculate the SMPS total mass concentration from its measured volume concentration throughout the campaign. Due to the relatively low

signal-to-noise ratio of the ACSM, the density shows scattering only during periods with low mass loadings. Therefore, it does not influence the cross validation with the SMPS mass significantly. Figures S3 and S4 show the time series of the SMPS mass and the combined mass concentrations measured by ACSM and MAAP and the correlation plot of both data sets, respectively. Using 12 275 common data points for the linear fit, a good qualitative and quantitative agreement (slope: 1.16 ± 0.01, intercept: -1.05 ± 0.06, $R^2 = 0.78$) was observed. Excluding eBC data resulted in a slope of 1.13 ± 0.01, an intercept of -1.14 ± 0.06, and an R^2 of 0.78. The negative offset can be explained by minor influences of sea salt and dust particles, which can be detected well by the SMPS and MARGA but not by the ACSM with a sufficient sensitivity. But the low value of the intercept shows already that the uncertainty introduced by these aerosol components is rather low in general. This can also be explained by the low concentrations of Mg, Na, K, and Ca as measured by the MARGA (see below) and the assumption that the majority of dust particles is most likely found in particles with diameters larger than 1 or even 2.5 μm (Finlayson-Pitts and Pitts, 2000, and references therein).

Since the MARGA routinely measures the water-soluble inorganic aerosol compounds, data from ACSM inorganic species were synchronized and compared to corresponding MARGA PM_1 data for the whole measurement period. The temporal overlap with the collocated high-resolution AMS was between 11 and 17 July 2012. The correlation parameters of individual aerosol species and respective total masses between the ACSM data and the data sets from the MARGA and AMS are given in Table 1, using 1943 and 294 common data points, respectively. The corresponding correlation graphs are shown in the Supplement (Figs. S5 and S6). Except for chloride, high correlation coefficients were achieved. Furthermore, the comparison to both total mass time evolutions shows very high qualitative and quantitative agreement. The quantitative difference between ACSM and AMS organics is also very low, and the discrepancies in case of ammonium and nitrate are within the stated $\pm 30\%$ accuracy of the AMS and ACSM (Ng et al., 2011b) and the $\pm 10\%$ for the MARGA NO_3, respectively (Makkonen et al., 2012). Similar variations were also found by Crenn et al. (2015) as well as Budisulistiorini et al. (2014). The latter reported of a comparison between two collocated ACSMs ($\pm 27\%$, $R^2 = 0.21$ for Chl, $R^2 > 0.8$ for the other species) and between these ACSMs and a continuous tapered element oscillating microbalance (TEOM, $PM_{2.5}$) instrument. Note that the major ions used for nitrate detection in AMS and ACSM (NO^+ and NO_2^+) are produced from both inorganic and organic nitrate (e.g., Farmer et al., 2010). The higher ACSM nitrate compared to MARGA nitrate can therefore also be explained by the presence of organic nitrates. In case of nitrate the ACSM and MARGA comparison cannot be seen as independent. The underestimation of the ACSM in case of sulfate exceeding the uncertainties may arise from calibra-

Table 1. Results of the ACSM to MARGA and ACSM to AMS comparisons for individual species and the respective total masses. Note that for comparison with the MARGA total mass concentrations only the ACSM inorganic species were considered.

		Slope	Intercept (μg m^{-3})	R^2
Chl	MARGA	0.49 ± 0.02	0.11 ± 0.01	0.24
	AMS	0.67 ± 0.01	-0.01 ± 0.01	0.31
NH$_4$	MARGA	0.88 ± 0.01	0.07 ± 0.01	0.93
	AMS	0.82 ± 0.03	0.01 ± 0.02	0.71
SO$_4$	MARGA	0.63 ± 0.01	-0.08 ± 0.01	0.86
	AMS	0.49 ± 0.02	-0.18 ± 0.02	0.76
NO$_3$	MARGA	1.23 ± 0.01	-0.37 ± 0.03	0.96
	AMS	1.17 ± 0.02	-0.04 ± 0.02	0.89
Organics	AMS	1.03 ± 0.04	0.07 ± 0.04	0.73
Total	MARGA	1.05 ± 0.01	-0.70 ± 0.06	0.93
	AMS	0.90 ± 0.02	-0.02 ± 0.07	0.84

tion issues. The mass calibration procedure used in this work was mainly adopted from AMS procedures which may not be directly suitable for the ACSM. The RIE of sulfate might be overestimated due to high observed background signals during the calibration using $(NH_4)_2SO_4$ particles. An overestimated RIE_{SO_4} results in underestimated mass concentrations. This would in turn explain the low SO_4 mass concentrations comparing to the MARGA and AMS. Additionally, the ACSM fragmentation table could not be adjusted for interferences of ions from different aerosol species on the same m/z properly because ACSM software used in this study did not allow plotting time series for specific fragments apportioned by the fragmentation table. Thus, the standard table had to be used. Furthermore, the MARGA measured only low concentrations of Mg, Na, K, and Ca (average sum: 0.08 μg m^{-3}). Thus, contributions of their corresponding sulfate salts, for which the ACSM is less sensitive, are negligible in this context. It should also be noted that chloride concentrations can originate from particulate organic and inorganic chloride components originating from, e.g., sea salt. For the latter, the ACSM is much less sensitive than the MARGA. As described above, influences from sea salt can be considered rather low. In turn, the MARGA might be less sensitive to organic chlorides, as they are likely less water soluble than inorganic chlorides. These explanations would explain the low agreement between the two instruments in case of chloride.

Overall, the comparison of the data measured by the ACSM and MAAP with collocated aerosol instruments showed a good reliability, precision, and in most cases, a good accuracy over the whole campaign, including periods with high and low mass loadings. Therefore, the here obtained chemical composition can be used qualitatively and quantitatively for the source apportionment of aerosol components at this rural site. Major discrepancies to the SMPS especially during some of the pollution events, like 16–27 January 2013 and 5–8 May 2013 (see below) can be ex-
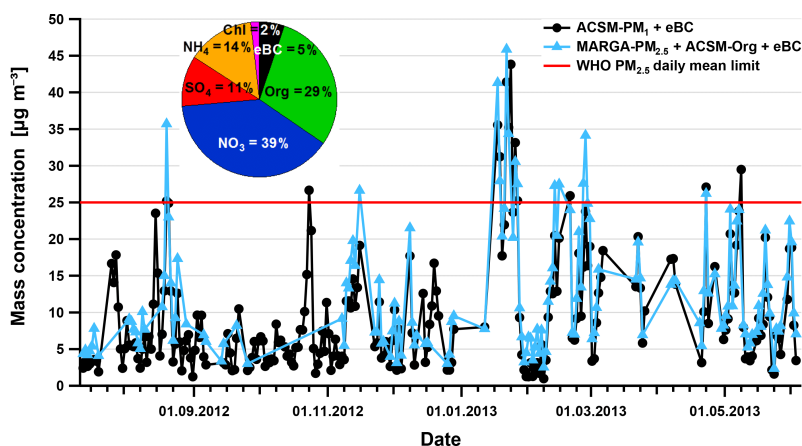
Figure 1. Time series of the daily mean. The black line represents the sum of eBC and all ACSM species, the blue line the sum of eBC, ACSM organics, and all MARGA $PM_{2.5}$ species. The pie chart shows the fractional abundances of individual eBC and ACSM species averaged over the whole campaign. For the determination of the daily means, missing eBC data were filled with zero values, thus deriving lower concentration limits.

plained by the correction of losses through the 60 m inlet line which was done size dependently and did not account for losses of individual species as mentioned in Sect. 2.3. Furthermore, discrepancies can also result from the different sampling heights of the SMPS and MAAP (both at 60 m) and ACSM and MARGA (both 5 m), through which the instruments might have sampled partly within different layers with different mass concentrations. As the quantitative agreements of individual inorganic species as well as of total inorganics between the ACSM and the MARGA during these periods are much higher, the mass loadings determined from these instruments are more reliable than the SMPS data.

3.2 Aerosol chemical composition

A meteorological overview of this campaign, including wind direction, precipitation, radon 222 measurements, and ambient temperature and relative humidity (RH) is provided in Fig. S7. Table S1 in the Supplement shows temperature and RH values averaged over selected periods (see below). If not stated otherwise, all data shown in this chapter are in UTC (local time minus 1 or 2 h, respectively) and averaged and synchronized to the ACSM data resolution of 30 min. Summing up the ACSM and MAAP data, a total mass concentration of 9.5 µg m^{-3} was measured on average, with a maximum of 78.4 µg m^{-3} and a minimum of 0.2 µg m^{-3}.

Data coverage of 75 % for 1 day and 90 % for 1 calendar year, respectively, is defined mandatory for a proper risk assessment according to the WHO air quality guideline. The last requirement could not be achieved with the ACSM nor with the MARGA between July 2012 and June 2013 (70 and 71 % coverage, respectively). Nevertheless, the average total mass derived from the combination of the MARGA $PM_{2.5}$ data (including all water-soluble inorganic components NO_3, NH_4, SO_4, Cl, Na, K, Mg, and Ca), MAAP

eBC and ACSM PM_1 organics resulted in 12.5 µg m^{-3} during this time, clearly exceeding the WHO $PM_{2.5}$ annual mean limit (10 µg m^{-3}) by 25 %. Even the PM_1 concentration inferred from ACSM plus MAAP data (campaign average 9.5 µg m^{-3}) approached the WHO $PM_{2.5}$ limit.

The time series of the daily mean total mass derived from both combinations (MARGA $PM_{2.5}$ plus ACSM organics plus eBC, and total ACSM plus eBC), where the required data coverage of 75 % was reached, are shown in Fig. 1. The WHO $PM_{2.5}$ daily mean limit of 25 µg m^{-3} was exceeded on 17 and 12 days, respectively. There were 8 and 7 exceedances observed during a period with high pollution from 16 to 27 January 2013, respectively.

In Fig. 2, the average contributions of individual species are shown as (a) pie charts and (b) stacked time series. A technical problem of the MAAP instrument was responsible for the lack of BC data from 15 February to 25 April 2013. To properly determine average species contributions for each pie chart, the campaign was therefore not divided strictly by season, but into five periods. The first two sections represent the summer (July–August–September) and autumn (October–November–December) 2012, while the first half of 2013 was divided into periods with and without BC data. Larger gaps in ACSM data occurred mainly due to problems with the RGA detector, in addition to minor measurement gaps for maintenance and calibrations.

Overall, particulate nitrate and organics were the dominant species, representing 39 and 29 % of the total aerosol, respectively. Both compounds show similar contributions in summer (period 1) and autumn 2012 (period 2), whereas in winter (January–February–March, period 3 and beginning of period 4) and spring (April–May–June, periods 4 and 5) the NO_3 fraction increased up to an average of 46 % of the total particulate mass, and the organic and eBC fractions de-
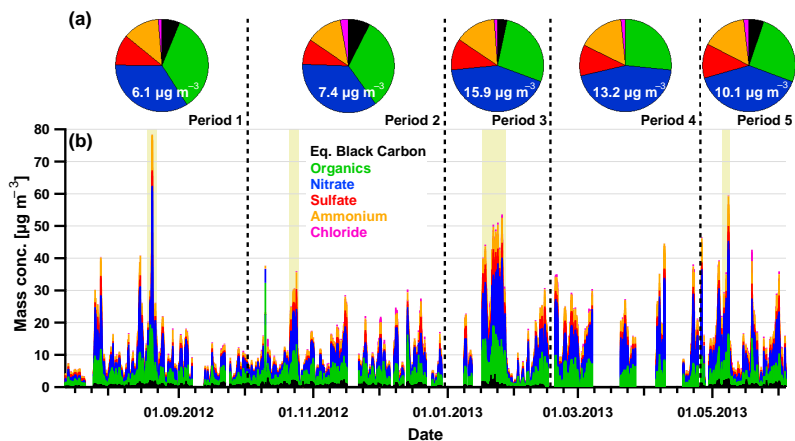
Figure 2. Overview of the ACSM campaign. **(a)** Pie charts of average fractional abundances of aerosol species, separated in five periods. The respective average total mass concentration is written inside the pie chart. **(b)** Stacked time series of mass concentrations of aerosol species. Pollution events are indicated by shaded areas.

creased. The contributions of the other components showed only small variations between the seasons (see Table S2).

The most significant pollution events (17–21 August 2012, 21–25 October 2012, 16–27 January 2013, and 5–8 May 2013) are highlighted with shaded backgrounds in Fig. 2. During these events, MARGA $PM_{2.5}$ data showed up to 33 % higher (e.g., from 17 to 21 August 2012) total inorganic masses than ACSM PM_1 inorganics as implicated in Fig. 1. As mentioned above, the number and proportions of exceedances of the WHO $PM_{2.5}$ daily mean limit is therefore even higher than when only PM_1 data are considered. During the last three events, northerly and northeasterly winds dominated. The period 16–27 January 2013 showed also the lowest temperatures (average: $-4\,°C$) with respect to the whole campaign and a temperature inversion between 2 and 40 m height on the morning of 25 January 2013, where both instruments showed total mass concentrations of ca. $20\,\mu g\,m^{-3}$. On the other hand, no temperature inversion was seen during times when very high ($> 40\,\mu g\,m^{-3}$) aerosol mass loadings were observed, even in wintertime. Many sudden drops of the particulate mass can be either explained by changes in wind direction and/or precipitation events, like in case of the two latter pollution events (16 to 27 January 2013, and 5 to 8 May 2013).

Figure 3 shows the diurnal patterns of each individual species and the total particulate mass for the whole campaign. Corresponding plots with data averaged separately for the five chosen periods can be found in the Supplement (Fig. S8). Overall, NO_3 showed the largest diurnal variation, with a maximum during the night/morning hours, reflecting its nighttime production and a minimum during the day due to the volatility of NH_4NO_3. This pattern is more pronounced in the warmer periods 1 and 5. Since the majority of ammonium is originated from NH_4NO_3, NH_4 has a similar pattern to that of NO_3. SO_4, which is mainly formed photochemi-
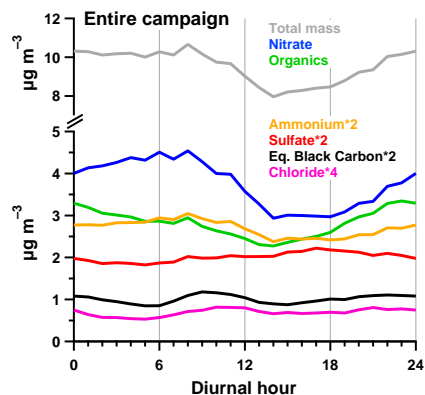

Figure 3. Diurnal variation (local time, LT) of individual species and the total mass, averaged over the whole ACSM campaign.

cally during the day from gaseous SO_2, showed peaks during daytime, although its overall variation is rather low. The maxima of BC can be attributed to direct emissions from traffic (morning and evening rush hours) and biomass burning events (domestic heating in the evenings/nights). Finally, OA showed peaks at the evening hours during the colder periods and a daytime minimum during the summer. More detailed discussion of the diurnal patterns of individual OA factors is given below.

An ion balance of all inorganic compounds indicates that too little NH_4 was measured to neutralize all NO_3 and SO_4 to their corresponding ammonium salts. The measured NH_4 mass concentration against the predicted NH_4 from the ion balance is plotted in the Supplement (Fig. S9) and resulted in a slope of 0.83 ± 0.00. One uncertainty of the ion balance is introduced by the use of RIEs for the inorganic species. As shown below, we consider uncertainties of RIEs to be of minor importance in explaining the observed low particulate

NH_4 concentration. It should be noted that for the calculation of the ion balance all measured NO_3 is considered. In addition to NH_4NO_3, organic nitrates give rise to a nitrate signal in the AMS, albeit with distinct difference in relative ion abundance (Alfarra et al., 2004; Fry et al., 2011, 2009). It was not possible to distinguish the organic and inorganic nitrate fraction based on their mass spectra with the ACSM. Since a relatively high amount of particulate organic nitrates was found in previous campaigns at CESAR using an AMS (Mensah, 2011), and the region is characterized by high NH_3 emissions (Derksen et al., 2011), the potential contribution of organic nitrates to the gap in the ion balance was explored as follows.

In contrast to the ion balance from the ACSM data, MARGA PM_1 measurements during the whole campaign showed a nearly 1 : 1 correlation of measured-to-predicted NH_4 (slope of the linear regression line: 1.03 ± 0.00, Pearson $R^2 = 0.97$), but with a negative offset of ca. $0.30 \pm 0.01 \, \mu g \, m^{-3}$. This offset, which is at least 3 times higher than the detection limits of the MARGA, cannot be explained by including positive metal ions to the ion balance since the sum of Mg, Na, K, and Ca mass concentrations had low contribution to particulate mass as mentioned in Sect. 3.1. Thus, significant influence of their nitrate salts to total nitrate can be excluded. In addition, as the MARGA is measuring the water-soluble nitrate fraction, the MARGA NO_3 can be considered to be exclusively NH_4NO_3. This assumption is acceptable, as shown by using the MARGA NO_3 instead of the ACSM total NO_3 for the ion balance of ACSM data (including ACSM SO_4, Chl, and NH_4), following a procedure given by Xu et al. (2015), who calculated the organic nitrate fraction by subtracting the inorganic nitrate concentrations measured by a particle-into-liquid sampler (PILS, see Orsini et al., 2003) from ToF-AMS total nitrate concentrations. In the Cabauw data set, the correlation of measured-to-predicted NH_4 resulted in a nearly 1 : 1 regression line without a significant offset (Fig. S10). This is in agreement with the MARGA internal ion balance which also indicates neutralized inorganic aerosols. Therefore, the mass concentration of nitrate groups associated with organic molecules (hereafter called organic nitrate or $OrgNO_3$), can be estimated by subtracting the MARGA nitrate from the ACSM nitrate concentration. The $OrgNO_3$ time series using this approach is plotted in Fig. S11; the respective diurnal variations averaged over for each period and for the entire campaign are presented in Fig. S12. An average mass fraction of 9 % was calculated for $OrgNO_3$ (average concentration: $0.43 \, \mu g \, m^{-3}$) in respect to total ACSM NO_3. Note that the organic nitrate concentrations may be biased low considering that, although the nitrate concentrations of both instruments are considered PM_1, the transmission efficiency of the ACSM decreases for particles with diameters between ca. 800 nm and 1 μm. This may also explain the slightly negative $OrgNO_3$ values seen in Fig. S11. The organic nitrate fraction shows a maximum concentration at night for nearly all individual periods and for the

whole campaign, followed by a decrease during the day. Particulate organic nitrates found in this study may therefore be considered as mainly semi-volatile as the evaporation from particles is dominating during warmer daytime hours. Due to the low temperatures within period 3, evaporation is less important during this time as seen by the daytime maximum. These findings are in agreement with previously reported relatively high AMS organic nitrate fractions by Mensah (2011) in May 2008 ($0.5 \, \mu g \, m^{-3}$, 35 % of total nitrate, 5.2 % of total aerosol mass) and March 2009 ($0.2 \, \mu g \, m^{-3}$, 10 % of total nitrate, 3.6 % of total aerosol mass) in Cabauw.

3.3 Factor analysis of organic aerosols

Prior to PMF analysis, the ACSM data set was subdivided into four data sets, which were explored by PMF separately, mainly due to the operational time of the ACSMs that are in the following referred to as (i) July–August–September 2012: summer 2012; (ii) October–November–December 2012: autumn 2012; (iii) January–February–March 2013: winter 2013; (iv) April–May–beginning of June 2013: spring 2013. A more detailed overview of the selected seasons is given in Table S3.

Figure 4 summarizes the time series of PMF factor classes (two POA factors: HOA and BBOA; and two SOA factors: OOA and a humic-like substance (HULIS) related factor, see below for descriptions) found for all seasons, except for BBOA in summer 2012. A corresponding graph dividing these PMF results into the five periods according to Fig. 2 is shown in Fig. S13. The POA profiles were constrained within ME-2 using the HOA and BBOA mass spectra found by ME-2 operated PMF analyses by Crippa et al. (2014) at the CESAR tower in Cabauw in March 2009. For the HOA profile, a constant a value of 0.1 was found to be most suitable for every season. If observed, the BBOA a value was set to either 0.2 (autumn 2012) or 0.3 (winter and spring 2013). The a values are based on different sensitivity tests for each season, as described Canonaco et al. (2013). Factor profiles including fractional contributions of marker ions (f_{44}, f_{43}, and f_{60}) and diurnal variations for winter 2013 are displayed in Fig. 5 as a representative example. Corresponding graphs for every season are shown in Figs. S14 and S15, respectively. Table S4 gives an overview of the temporal correlations of each factor with external data sets, while scatter plots from winter 2013 profiles against profiles found in May 2008 at CESAR by Crippa et al. (2014) are given in Fig. S16.

As mentioned, no significant BBOA influence was seen in summer 2012. Thus, a three-factor solution was chosen for this season and consequently, no BBOA time series exist at that time. For all other factors, the time series are continuous on 1 October 2012 (i.e., analysis of separate data files leads to a consistent result in terms of both factor profiles and concentrations). The SOA factors showed always higher contribution (54–84 %, averaged by season) to total organics compared to POA (16–46 %). For all PMF factors, no preferential
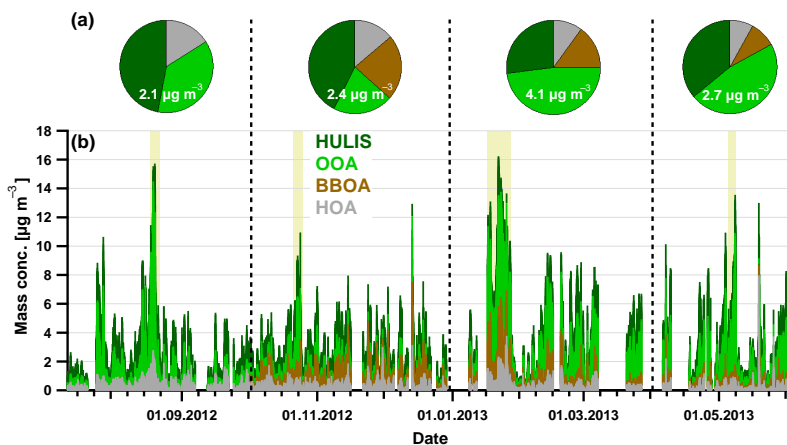
Figure 4. Overview of ACSM PMF factors. **(a)** Seasonal pie charts of the factor contributions. The respective average total organic concentration is written inside the pie chart. **(b)** Stacked time series of mass concentrations. Shaded areas represent highly polluted events.

wind direction was observed over the entire campaign. During the pollution events mentioned above, OOA originated mostly from the directions between 20 and 180° in respect to the tower. This is not the case for HULIS, the origins of which varied throughout all directions also during pollution events.

The seasonal average HOA contribution to total organic mass was highest in summer 2012 and lowest in spring 2013 (16 and 8 %, respectively). All HOA diurnal patterns (Fig. 5b) showed a maximum at 07:00 and 11:00 (LT) and a slight increase in the evening, emphasizing that its main source is related to traffic likewise to BC (see Sect. 3.1). In winter 2013, these maxima were less distinctive comparing to the other seasons. Highest temporal agreements with HOA were seen by the POA tracers BC, NO_x, and CO ($R^2 = 0.38$, 0.47, and 0.47, respectively) over the entire campaign.

The BBOA profile showed a very high contribution of m/z 60, which is dominated by the $C_2H_4O_2^+$ ion. This fragment is characteristic for anhydrosugars such as levoglucosan (Alfarra et al., 2007), which are established markers of wood combustion processes (Simoneit and Elias, 2001; Simoneit et al., 1999). The fractions of m/z 60 to the BBOA profile in autumn 2012 (3.7 %) and winter 2013 (3.2 %) are higher than in spring 2013 (2.4 %). As mentioned, BBOA was not found in summer 2012. This was verified by the fact that the contribution of m/z 60 to the BBOA profile decreases for higher a values in that season, which is an indication for the nonexistence of BBOA. The highest contributions of BBOA to total organics were seen in the colder autumn (23 %) and winter seasons (15 %). This and the diurnal maximum during the evenings and nights match the expectations for a factor linked with domestic heating activities, together with the fact that this factor was not seen during the warmer summer season. Averaged over the whole campaign, the contribution to total organics was 13 %, including summer 2012, where its fraction was set to 0. In winter, the cor-

relations with eBC and CO were higher ($R^2 = 0.64$ and 0.57, respectively) than over the whole campaign ($R^2 = 0.39$ and 0.49, respectively), meaning that these compounds are reasonably more attributed to domestic heating during the colder periods compared to the contribution of heating to BC and CO during the other seasons in this region.

The OOA profile is similar to a MS pattern as expected for a low-volatile OOA (LVOOA) factor. The correlation coefficients (Pearson R^2) with the OOA and LVOOA spectra given by Ng et al. (2011a) are 0.94 and 0.97, respectively. Similar agreement was found compared to the LVOOA factor observed by Mensah (2011) and Crippa et al. (2014) ($R^2 = 0.97$ and 0.94, respectively) at the CESAR tower in May 2008. The OOA factor showed a nighttime maximum and a daytime minimum. This is rather characteristic for a semi-volatile OOA (SVOOA) behavior, as well as the high agreement with NO_3 over the whole year ($R^2 = 0.63$), as described by Lanz et al. (2007). The correlation to the LVOOA-associated compound SO_4 is less significant ($R^2 = 0.48$). OOA dominated the organic fraction in winter and spring 2013 (47 and 48 % contribution, respectively; 33 % over the whole year). During the defined pollution events the increase of the OOA mass concentration (up to 11.8 μg m^{-3}), relative to the campaign average (1.05 μg m^{-3}), is much stronger compared to the other PMF factors, demonstrating that this factor is mainly responsible the high total OA mass during these periods.

The so-called HULIS factor showed the highest f_{44} values of all factors, increasing from summer 2012 to spring 2013 from 0.23 to 0.35. The HULIS factor class was first observed by Mensah (2011) in previous AMS campaigns at the CESAR tower in May 2008 and March 2009. The identification and characterization of this factor class was done in the 2008 campaign by the comparison with data from an ion-exchange chromatographic method for direct quantification of humic-like substances (HULIS) and from water-soluble
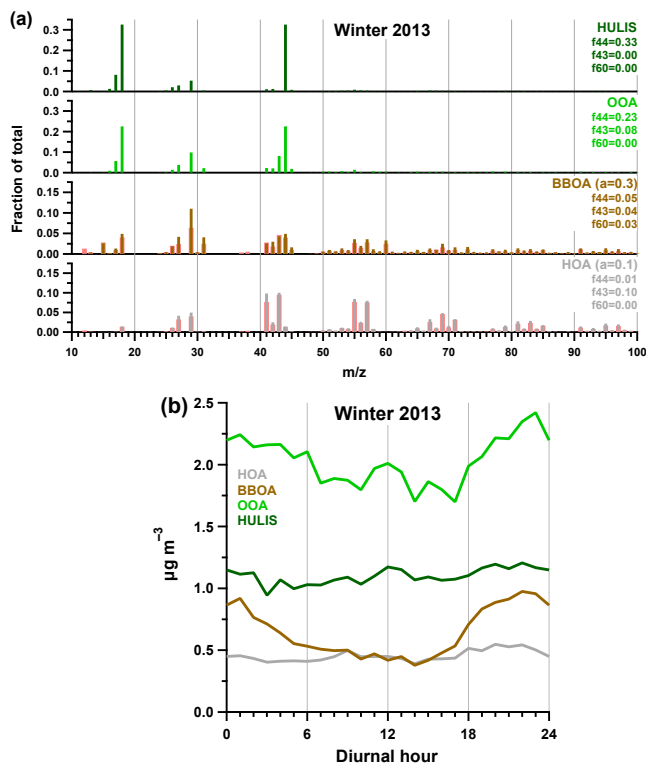
Figure 5. (a) Mass spectra of ACSM PMF factors and **(b)** average diurnal variations (LT) of ACSM factors found in winter 2013. For the constrained profiles HOA and BBOA, the applied *a* value is written in brackets. Corresponding reference spectra are shown by red bars. Note that the *y* axis scales of the POA profiles are zoomed by a factor of 2 compared to SOA profiles.

organic carbon (WSOC) analyzed offline on a set of filters collected in parallel (Paglione et al., 2014). The HULIS factor provided the highest contribution to the total organic mass over the entire campaign (41 %) and was the dominant factor in summer and autumn 2012. Since it had no distinct diurnal variation and preferential wind direction, it can be considered as a regional background aerosol at this rural site. Additionally, the variation between the seasonal average concentrations of HULIS within the ACSM data set is less than ±10 %. Also the comparison to the most important tracers (Table S4 in the Supplement) showed no preferential attribution, either to a low-volatile ($R^2 = 0.41$ with SO$_4$), semi-volatile ($R^2 = 0.39$ with NO$_3$), or primary organic aerosol ($R^2 = 0.47$ with eBC). These characteristics were also reported for the HULIS factor found at CESAR in May 2008 (Crippa et al., 2014; Mensah, 2011). The correlation of the sum of the secondary inorganic species NO$_3$ and SO$_4$ with the sum of the OOA and HULIS time series gives a coefficient of $R^2 = 0.70$, which is slightly higher than with OOA only ($R^2 = 0.67$). This might confirm the SOA character of the HULIS factor. The reason why HULIS was the dominant factor in summer and autumn 2012 is due to the lower

mass concentrations of the other factors compared to the remaining periods. In turn, OA mass increased during pollution events mainly due to the increase of the other SOA factor, namely OOA. A number of studies are published with different theories on the formation and sources of atmospheric HULIS. It shows similarities to terrestrial humic acids (HA) and fulvic acids (FA) due to their (poly-)acidic nature, but with lower molecular weight (< 1000 amu) than HA or FA (Graber and Rudich, 2006; Kiss et al., 2003). On the other hand, solid-phase extraction protocols for HULIS associating them with the more hydrophobic fraction of water-soluble aerosols (e.g., Varga et al., 2001). Graber and Rudich (2006) suggested that the formation of HULIS happens by oligomerization of lighter organic acids in liquid droplets in the atmosphere within timescales of hours to days. This emphasizes that HULIS is an ubiquitous background factor at Cabauw, characterized as a regionally well-mixed aerosol of a long atmospheric lifetime.

Note that the correlation coefficient (Pearson R^2) of the POA factors with eBC data is relatively low ($R^2 = 0.38$ and 0.39 with HOA and BBOA, respectively), while in turn HULIS shows a higher correlation with eBC ($R^2 = 0.47$). Furthermore, correlation coefficients of the POA factors with CO and NO$_x$ time series do not exceed values above 0.47. Low correlations of POA and those tracers, e.g., HOA with CO or BC varying from far below 0.5 to slightly above 0.5 were also seen in a number of previous studies at remote and rural sites (Canonaco et al., 2013; Minguillón et al., 2015; Schmale et al., 2013; Setyan et al., 2012), including Cabauw (Crippa et al., 2014; Mensah et al., 2011). There, low concentrations of POA factors and (e)BC to total aerosol mass were determined and resulted in higher uncertainties for the identification of these factors, likewise in this study. Nevertheless, the correlations of POA factors with CO and NO$_x$ are still by far higher than the correlation of these tracers with the SOA factors, which, together with the very high agreement of m/z 60 with BBOA, underlines the correct identification of HOA and BBOA. The higher correlation of HULIS with eBC can be understood in the light of a study by Andreae and Gelencsér (2006), who pointed out that the fiber filters, as used by the MAAP, loaded with macroscopic amounts of humic-like substance samples appear very dark brown or nearly black. A possible interference of HULIS with BC measurements can therefore not be excluded and would explain why the HULIS factor, in contrast to the POA factors, has a relatively high correlation coefficient with BC data. The use of instruments like an aethalometer or a particle soot absorption photometer as used in other studies would reduce these interferences (Andreae and Gelencsér, 2006; Petzold et al., 2013), but were not available during the campaign. Though given the relatively low contribution of eBC to total aerosol mass, a possible partial interference with HULIS is of minor importance regarding total aerosol masses.

The source apportionment, as described here, used a data set which was subdivided into the four seasons prior to PMF
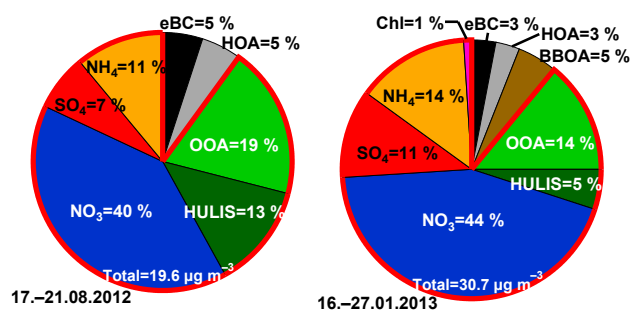
Figure 6. Average contributions of individual aerosol species and PMF factors during selected pollution events. Surrounded red regions represent the secondary aerosol fraction.

analysis. A single PMF analysis of the whole data set with constrained HOA and BBOA profiles led to solutions with a highly overestimated BBOA factor in the summer, compared to the results when the seasons were explored individually (see Fig. S17 and S18 in the Supplement). Furthermore, the contributions of individual factors change significantly in some periods, especially for the OOA factor during pollution events. This is mainly driven by the different OOA f_{44} and f_{43} values. This behavior is independent of applied a values for BBOA and may result from the uncertainty of this statistical tool. Since there was no evidence of BBOA seen in the separate analysis of the summer period (e.g., low fraction of m/z 60 in the organic mass spectrum and no correlation of the constrained BBOA factor with POA tracers, no matter which a value was used), the solutions derived from the single PMF analysis were reasonably rejected.

3.4 Composition and sources of aerosols during pollution events

The investigation of pollution events showed that the majority of aerosol mass was contributed from secondary aerosols. As an example, Fig. 6 displays average abundances during two selected periods (17–21 August 2012 and 16–27 January 2013), where the organic fraction is further divided into the PMF factors. Nitrate and organics are still the dominant species, while the primary aerosol components (BC, HOA, BBOA) have even less influence than during the entire respective season. An exception is seen from BBOA, the contribution of which is reasonably higher during the latter pollution event with very low temperatures, when domestic heating sources most likely increased. But even at this time, primary organics do not exceed 11 % of total mass on average.

Since the major inorganic components NH_4NO_3 and $(NH_4)_2SO_4$ are produced by atmospheric processes, they are considered as secondary aerosols. Together with the high abundance of SOA (OOA, HULIS), chemically aged aerosol components have the largest impact on total particulate mass (up to 90 %) at Cabauw. The largest effect on

a potential reduction of particulate mass can therefore be achieved by reducing NO_x and/or NH_3 emissions, the anthropogenic precursors of particulate NH_4 and NO_3. Indeed, on 16 January 2013, the highest NO_x values were observed compared to the whole campaign (daily mean: 96 μg m^{-3}; seasonal average: 29 μg m^{-3}). Also the highest NH_3 values were obtained between 17 and 19 August 2012 (daily means: 26–28 μg m^{-3}; seasonal average: 10 μg m^{-3}). These ammonia values are higher than previously reported for Cabauw, e.g., by Derksen et al. (2011) in May 2008 (approximately 20 μg m^{-3}). Lolkema et al. (2015) gave an annual average of 5.3 μg m^{-3} for 2013 in this region.

4 Conclusions

This work provides chemical composition data of atmospheric aerosols acquired during 1 year at the CESAR tower in Cabauw, the Netherlands, which is a representative rural site for northwestern Europe. The concentration of submicron particles from combined ACSM and MAAP data showed 12 exceedances from the WHO PM$_{2.5}$ daily mean limit. The respective campaign average of 9.5 μg m^{-3} approached the WHO PM$_{2.5}$ annual mean limit. Taking MARGA PM$_{2.5}$ into account, the number and proportions of these exceedances are even higher, emphasizing the importance of these high ACSM PM$_1$ results shown here which represent lower limits. As carbonaceous compounds are estimated to be 5 times more toxic than inorganic particles (Lelieveld et al., 2015) MARGA data alone would not give sufficiently possible implications regarding adverse health effects. While few people live in the direct vicinity of the measurement site, the high aerosol concentration measured at the site can be considered to represent the regional background. This regional background is adding to local aerosol contributions in high populated urban sites (Pandis et al., 2013), namely the four largest cities of the Netherlands which have a distance of 40 km or less from the CESAR tower.

Particulate mass loadings found at this rural site are dominated by secondary aerosol formation through atmospheric gas phase chemistry and particle phase aging. It is shown that particulate ammonium nitrate is the major aerosol component (39 % on average) and represents the more hygroscopic aerosol fraction. Since the human respiratory system is characterized by high humidity, more hygroscopic aerosols have a higher deposition tendency within the human lung than less water-soluble particle compounds (Asgharian, 2004; Broday and Georgopoulos, 2001). With special regard to adverse health effects, this is very crucial because Asgharian (2004) also found that especially hygroscopic submicron particles can deposit in the entire lung. The high ammonium nitrate fraction also implies that inorganic SA reduction in Cabauw can be most efficiently achieved through the reduction of gaseous ammonia emissions in the area.

As indicated by model results for the southwestern United States from Zhang and Wu (2013), the reduction of NH_3 emissions, in conjunction with already implemented SO_2 and NO_x emission reductions, can further reduce $PM_{2.5}$ than reducing SO_2 and NO_x emissions alone, particularly for regions with high emissions of NH_3 from agricultural sources. The latter is clearly the case for Cabauw with its high amount of animal husbandry and the use of nitrogen-containing fertilizers around the CESAR tower. Banzhaf et al. (2013) derived similar conclusions for PM_{10} using different emission scenarios within domains covering Germany and Europe. Applying a 3-D chemical transport model over Europe, Megaritis et al. (2014) found that a reduction of NH_3 emissions by 50 % would have a much higher effect on reducing $PM_{2.5}$ than decreasing NO_x emissions by 50 %. The latter scenario would even result in negative side effects, such as higher tropospheric ozone concentrations (especially in summertime 4 % over western Europe and up to 40 % in major urban areas) and higher amounts of particulate sulfate and OA by 8 and 12 %, respectively, in winter.

The local mitigation of organic aerosol mass (29 % contribution on average) is more challenging, as secondary organic aerosols are highly abundant at the Cabauw site (74 and 22 % of OA and total PM_1 on average, respectively). The presented data set shows a large and ubiquitous HULIS fraction (37 %) which, based on diurnal patterns and a lack of correlation with wind direction, can be considered as long-range background aerosols formed from atmospheric aging processes. In turn, primary organic aerosols emitted mainly from traffic and biomass burning (12 and 13 % of OA on average) have only minor importance. For a more detailed identification of the SOA sources, compound-specific measurements of OA as well as routine VOC monitoring are needed.

Finally, the presented data set and interpretations provide an important contribution to the EU-FP7-ACTRIS project which supported building of new knowledge as well as policy issues on climate change, air quality, and long-range transport of pollutants on a European scale.

5 Data availability

The observational data in this study from ACSM and SMPS instruments are available on the EBAS database: http://ebas.nilu.no (Schlag et al., 2015; Henzing and Moerman, 2015). MAAP data can be accessed by request to J. S. Henzing (TNO). MARGA data can be accessed by request to Rene Otjes (ECN).

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