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Near real-time PM₁ chemical composition measurements at a French urban background and coastal site under industrial influence over more than a year: Temporal variability and assessment of sulfur-containing emissions

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1 **Abstract**

2 Near real-time measurements of submicron particulate matter (PM₁) were carried out at an
3 industrial and coastal site in Dunkirk (Northern France) over a 14-month period (July 2013-
4 September 2014). This site is surrounded by various industrial plants (metallurgy,
5 petrochemistry, food processing, power plant, etc.) and is characterized by intense ship traffic
6 (~700-800 per day along the English Channel) in harbour surroundings. The non-refractory
7 (NR) submicron particles (organics, sulfate, nitrate, ammonium and chloride) and black
8 carbon were measured by an Aerosol Chemical Speciation Monitor (ACSM) and an
9 Aethalometer, respectively. Concomitant monitoring of CO₂, SO₂, and meteorological
10 parameters was also performed. Both the seasonal (five seasons including two summers) and
11 spatial (four identified sectors of emissions: marine, urban, industrial-urban and industrial)
12 variabilities were investigated.

13 We present a descriptive analysis of the PM₁ composition, whose ambient concentrations
14 ranged from less than 1 μg m⁻³ up to approximately 100 μg m⁻³ during a few pollution events.
15 Gaseous SO₂ and particulate SO₄ were systematically observed at high concentrations (up to
16 310 and 48.2 μg m⁻³, respectively) when industrial plumes reached the monitoring site. The
17 conversion ratio of particulate (S_p) to total (S_{tot}) sulfur is relatively constant at 0.1 when RH
18 ranges from 30-70% but reaches an average value of 0.3 at high RH (90-100%). This reflects
19 an enhancement of SO₂-to-SO₄ gas-particle conversion processes resulting in an increase of
20 aerosol acidity as shown by a comparison between measured and predicted NH₄
21 concentrations. An impact of the vertical mixing on the SO₂-to-SO₄ conversion was also
22 observed using vertical turbulence (σ_w) as a descriptive parameter. Indeed, the conversion
23 ratio (S_p/S_{tot}) was found to be reduced under high turbulence conditions due to dilution
24 effects.

25

26 **Keywords:** ACSM, Aethalometer, Submicron particulate matter, Industrial area

27 **1 Introduction**

28 Atmospheric aerosols from both natural and anthropogenic sources have been widely studied
29 during the last decades to assess their influence on climate. Indeed, their optical properties
30 (light scattering and absorption) and their ability to act as cloud condensation nuclei
31 (Flossmann et al. 1985, Ghan and Schwartz 2007) can significantly impact the Earth's
32 radiative balance. In addition, aerosols can strongly impair human health (Pope et al. 2002,
33 Kelly and Fussell 2012, Leclercq et al. 2017) and the wellbeing of ecosystems (Bouwman et
34 al. 2002, Niyogi et al. 2004). Based on their ability to deposit in the respiratory tract, aerosols
35 are classified as PM₁₀, PM_{2.5}, and PM₁, characterized by aerodynamic diameters lower than
36 10, 2.5 and 1 micrometers, respectively. The World Health Organization (WHO) estimates
37 that PM_{2.5} contributes to approximately 0.8 million deaths per year and ranked it as the 13th
38 cause of global mortality (WHO 2002, Cohen et al. 2005, Elder et al. 2009). In the urban
39 environment, PM₁ exhibits a stronger impact on human health than PM_{2.5} since (i) they can
40 penetrate further into the alveolar region, (ii) their chemical composition is dominated by
41 anthropogenic sources and (iii) they lead to a more intense pro-inflammatory response (Pérez
42 et al. 2008, Ramgolam et al. 2009, Kelly and Fussell 2012, Mazzearella et al. 2012).

43 Submicron non-refractory particulate matter (NR-PM₁) can be characterized at a high time
44 resolution using Aerosol Mass Spectrometers (AMS) during short field campaigns over a few
45 weeks. Details about both the chemical composition (organics, nitrate, sulfate, ammonium and
46 chloride) and the size distribution at temporal resolutions of a few minutes have been obtained
47 worldwide (Jayne et al. 2000, DeCarlo et al. 2006, Canagaratna et al. 2007, Zhang et al. 2007,
48 Jimenez et al. 2009). However, despite the unique qualities of an AMS to provide chemical
49 information on PM₁, this type of instruments exhibits several drawbacks for long-term
50 monitoring, such as the need for frequent calibrations and highly skilled operators.

51 More recently, the Aerosol Chemical Speciation Monitor (ACSM) (Aerodyne Research, Inc.)
52 was designed and proposed for long-term monitoring (Ng et al. 2011). In contrast to an AMS,
53 ACSMs do not provide the particle size distribution and exhibit a lower mass resolution,
54 which in turn leads to fewer details about the organic composition. However, ACSM
55 instruments do not require frequent calibrations, are less expensive, and are easier to use
56 during field measurements. It is interesting to note that AMS and ACSM instruments have
57 been already compared several times. A 3-week campaign performed at Queens College (New
58 York, USA) in 2010 showed good correlations (R^2 of 0.81–0.91, slopes of 0.76–1.01)
59 between the two techniques for all measured species (organics, nitrate, sulfate, ammonium
60 and chloride) (Ng et al. 2011). A larger scale intercomparison exercise involving 13 Q-ACSM

61 and one HR-ToF-AMS in Paris (Crenn et al. 2015) showed good agreements between the
62 different instruments with R^2 values higher than 0.9, except for chloride, and slopes ranging
63 from 0.62 to 1.43 depending on the species considered.

64 Industrial activities are important sources of anthropogenic Particulate Matter (PM) (Taiwo et
65 al. 2014, Riffault et al. 2015). However, previous sampling sites where AMS and ACSM
66 instruments have been deployed were usually located in urban, suburban or rural background
67 regions and only a few were impacted by industrial emissions (El Haddad et al. 2013,
68 Budisulistiorini et al. 2015). In Europe, industrial processes represent the third (9.1%) and
69 second (28.6%) primary sources of $PM_{2.5}$ and PM_{10} (EEA 2015), respectively. In France itself,
70 industrial sources accounted for 22% and 29% of $PM_{2.5}$ and PM_{10} in 2012, respectively
71 (CGDD 2014). Of interest for the present study, the Hauts-de-France (HdF) region in France
72 is still heavily industrialized, with industries contributing to 68 and 61% of $PM_{2.5}$ and PM_{10}
73 emissions, respectively, in the Dunkirk conurbation (AtmoHdF 2012).

74 Dunkirk, located in northern France, is impacted by industrial activities such as metallurgy,
75 petrochemistry, food processing, power plant, etc., and is also the first energy platform in
76 HdF. This industrial activity results in massive emissions of gaseous and particulate
77 pollutants, including for the year 2011 Volatile Organic Compounds (VOCs; 1,556 tons),
78 oxides of nitrogen (NO_x , 8,195 tons), sulfur dioxide (SO_2 , 11,752 tons), and particulate matter
79 (10 nm-100 μm ; 3,246 tons) (DREAL 2012).

80 Previous studies focusing on particulate matter in Dunkirk have investigated concentrations in
81 heavy metals (Alleman et al. 2010, Mbengue et al. 2014) and inorganic species (Rimetz-
82 Planchon et al. 2008) in PM_{10} . The organic fraction has also been studied in $PM_{2.5}$ using
83 offline filter methods (Cazier et al. 2011, Crenn et al. 2017, Crenn et al. 2018) and in PM_{10}
84 using an Aerosol Mass Spectrometer (AMS) (Crenn et al. 2017, Crenn et al. 2018, Setyan et
85 al. 2019). PAHs (Polycyclic Aromatic Hydrocarbons) and sulfate in NR- PM_{10} have been found
86 almost exclusively in winter for air masses coming from the industrial zone. However, the
87 filter methods required long sampling times (typically 24-72 hours), precluding any
88 investigation of fast changes in aerosol composition and levels. The two campaigns
89 employing an AMS were only performed for short periods (summer and winter,
90 approximately 1 month each), with only a few wind occurrences from the industrial sector.

91 Volatile Organic Compounds (VOCs), considered as potential precursors of organic aerosols,
92 have also been investigated in Dunkirk (Badol et al. 2008, Badol et al. 2008, Roukos et al.
93 2009). Badol et al. (2008) carried out a measurement campaign of about one year and
94 identified 53 VOCs (Badol et al. 2008). Daily profiles, seasonal variations and pollution roses

95 observed for these compounds indicated strong impacts of traffic, solvent evaporation and
96 industrial emissions on the measured concentrations. In addition, a Chemical Mass Balance
97 (CMB) source-receptor model was applied to estimate the contributions of various emission
98 sources, including 6 urban profiles and 7 industrial profiles (Badol et al. 2008). Roukos et al.
99 (2009) carried out two measurement campaigns in summer and winter 2007 using passive
100 sampling on adsorbent cartridges (Roukos et al. 2009). This study showed that pollution
101 transported over long distances has a significant impact on VOC concentrations in the
102 Dunkirk area during specific weather conditions. Xiang et al. (Xiang et al. 2012) reported a
103 source apportionment study coupling a Positive Matrix Factorization (PMF) analysis and
104 micro-meteorology observations. This study highlighted that an increase of the turbulence
105 could lead to a lower contribution of ground level sources and an enhanced contribution of
106 elevated sources (such as plumes from chimneys).

107 The present study provides the first near real-time and long-term (> 1 year) monitoring of
108 submicron aerosols at an urban background and coastal site strongly impacted by industrial
109 activities (within a few km). This work aims at better understanding PM₁ temporal variability
110 in terms of mass concentration and chemical composition, and their transformation in this
111 complex environment. Organic PM₁ sources will be further investigated using PMF in a
112 forthcoming paper. The database of PM₁ measurements is analyzed by seasons and wind
113 sectors to investigate the origin of local emissions. Particular attention was paid to the impact
114 of industrial sulfur emissions on the formation of sulfate particles.

115

116 **2 Experimental section**

117 **2.1 Campaign description**

118 The measurement site was located on the eastern side of the Dunkirk harbour (Port-Est, PE:
119 51°3.12' N; 2°21.24' E; 1 m). This site is surrounded, within a few kilometers, by an
120 intensive industrial zone at W-SW and by a residential zone at S-SE as shown in Figure 1.
121 This geographical location allows classifying the origin of local air masses according to four
122 wind sectors: marine (M; 271° -70°), urban (U; 71° - 140°), industrial-urban (IU; 141° -
123 225°), and industrial (I; 226° - 270°). This site can sometimes be under the influence of
124 remote sources from France, Belgium, England, and Germany within a few tens to hundreds
125 of kilometers.

126 The field campaign was conducted over 14 months from July 15, 2013 to September 10,
127 2014. An ACSM and an Aethalometer (described below) were installed at the PE site,
128 alongside an SO₂ analyzer (AF21M, Environnement SA) managed by the Atmo HdF regional

129 air quality monitoring network and a CO₂ analyzer (VS-3000, HORIBA). The SO₂ analyzer
130 was operated continuously at a time resolution of 15 min and was calibrated once a month
131 using a SO₂ gas standard at 200 ppb (Air Liquide). The CO₂ analyzer was calibrated 3 times
132 during the campaign, using a CO₂ gas standard at 997 ppm (purity > 99.9%, Praxair). An
133 ultrasonic anemometer (uSonic-3 Scientific, METEK) was deployed at a nearby site
134 approximately 1 km away (Figure 1) for measuring horizontal and vertical wind speeds and
135 direction, as well as temperature and turbulence parameters. Standard meteorological
136 parameters such as temperature, relative humidity, precipitation, pressure, and solar radiation
137 were also recorded on the rooftop of a research building of the Université du Littoral Côte
138 d'Opale (51°2.14'N; 2°22.05'E), located at 2 km. From June 23, 2014 to September 10, 2014,
139 an ambient particulate monitor (TEOM-FDMS 1405-F, Thermo Fischer Scientific) was
140 additionally deployed to measure the total PM₁ mass concentration. Section A of the
141 Supplementary Information reports the number of valid data (Table S1) as well as the
142 temporal data coverage (Figure S1) for each instrument. Measurements were averaged over
143 the ACSM time stamp using Coordinated Universal Time (UTC).

144

145 **2.2 PM₁ chemical composition**

146 **2.2.1 Aerosol Chemical Speciation Monitor (ACSM)**

147 The ACSM (Aerodyne Research Inc.) provides measurements of particulate organics, nitrate,
148 sulfate, ammonium and chloride in the non-refractory submicron particles (Ng et al. 2011).
149 The aerodynamic lens inside the ACSM inlet sampled PM₁ at a flow rate of approximately 80
150 cm³ min⁻¹, controlled by a 100 μm diameter critical aperture. Non-refractory components
151 were vaporized at 600°C after impaction on an inverted-cone surface and then ionized by
152 electron impact at 70 eV. A RGA (residual gas analyzer) quadrupole mass spectrometer was
153 used to acquire mass spectra at a mass resolution of one unit and up to 148 amu. During
154 acquisition, the ACSM was operated with a three-way automatic valve to switch between the
155 filter mode (particle-free air) and the sample mode (ambient air). The difference between
156 these two modes is considered as the signal generated by ambient aerosols. The scan speed
157 was set to 200 ms amu⁻¹. 28 scans were acquired for both the filter and the sample modes,
158 resulting in a time resolution of approximately 30 min. The instrument was installed in an air-
159 conditioned room (20°C) for continuous monitoring.

160 Ambient aerosols were sampled through a PM_{2.5} cyclone (URG-2000-30EQ) at a flow rate of
161 3 L min⁻¹ and a stainless-steel tube whose length and inner diameter were 2.1 m and 1.27 cm,
162 respectively (residence time in the sampling tube was approximately 5 seconds). The particle

163 size-dependent loss due to the sampling setup was calculated using the Particle Loss
 164 Calculator (PLC) (von der Weiden et al. 2009), and was found to be negligible (< 1%).
 165 Sampled aerosols were dried using a Nafion dryer (PD-200T-12 MPS, Perma Pure) reaching a
 166 relative humidity (RH) always lower than 30%, which is important to minimize the effect of
 167 RH on the collection efficiency (CE) as discussed later in this section.
 168 The mass spectra acquired by ACSM were analyzed using a fragmentation table (Allan et al.
 169 2004, Canagaratna et al. 2007) to extract distinct chemical species or group of species. The
 170 mass concentration of a species s (C_s) is derived from the measured ion current $IC_{s,i}$ (amperes)
 171 using Equation (1):

$$172 \quad C_s = \frac{CE_s}{T_{m/z}} \times \frac{10^{12}}{RIE_s} \times \frac{Q_{cal} \times G_{cal}}{RF_{NO_3}} \times \frac{1}{Q \times G} \sum_{alli} IC_{s,i} \quad (1)$$

173 where $\sum IC_{s,i}$ is the sum of the ensemble mass spectrum fragment i contributing to species s ;
 174 CE_s is the collection efficiency of a species s ; $T_{m/z}$ the ion transmission efficiency; RF_{NO_3}
 175 (ions/molecule) the response factor for NO_3 ; and RIE_s the relative ionization efficiency of a
 176 species s compared to NO_3 ; Q_{cal} and G_{cal} are the volumetric sample flow rate ($cm^3 s^{-1}$) and
 177 multiplier gain ($\sim 20,000$) measured during calibrations of RF_{NO_3} , respectively. In general,
 178 values of Q and G observed during ambient measurements are similar to the values measured
 179 during calibration experiments (Q_{cal} and G_{cal}) and these parameters cancel out in Eq. (1).

180 The instrument calibration protocols and signal optimization are based on existing procedures
 181 (Ng et al. 2011) and are presented in more details in the Supplementary Information (section
 182 B). The NO_3 response factor (RF_{NO_3}) and the relative ionization efficiency (RIE_s) were
 183 calibrated at the beginning and periodically during the campaign (once per month). Average
 184 values of RF_{NO_3} (3.81×10^{-11}) and RIE_s (NH_4 : 5.67; SO_4 : 0.55; Cl : 2.26) were used for the
 185 whole campaign. For organics, the default value of $RIE(1.4)$ was used (Jimenez et al. 2003).
 186 The RF_{NO_3} and RIE_{NH_4} values observed in the present study are close to the average value of
 187 13 Q-ACSM used during the intercomparison exercise described by Crenn et al. (Crenn et al.
 188 2015), i.e. 3.55×10^{-11} and 6.31, respectively. Most previous studies used the default values
 189 of 1.2 for SO_4 (Ng et al. 2011) and 1.3 for Cl (Alfarra et al. 2004) but $RIE(SO_4)$ has been
 190 found to vary from one instrument to another (Ng et al. 2011).

191 The CE of a species s is mainly influenced by particle humidity (for $RH > 80\%$), acidity, and
 192 the mass fraction of ammonium nitrate (Matthew et al. 2008). Humidity effects are ignored in
 193 the present study because particles were dried at RH below 30%. However, both particle
 194 acidity and the fraction of ammonium nitrate were taken into consideration for the calculation

195 of CE, using the method described by Middlebrook et al. (Middlebrook et al. 2011) and
196 described in the Supplementary Information (section C) where the CE time series can be
197 found (Figure S1). More than 57% of the CE values were at 0.45 and the rest ranged from
198 0.45-0.9 during the campaign.

199 As the ACSM is equipped with a quadrupole mass spectrometer, the m/z -dependent ion
200 transmission efficiency ($T_{m/z}$) is assessed using an internal naphthalene standard whose parent
201 ion is detected at m/z 128 and several fragments are detected at lower m/z . Finally, a
202 reference period was chosen when the signal of air beam ($m/z = 28$, corresponding to N_2^+) is
203 stable and as close as possible to 10^{-7} A. This normalization takes into account the variation of
204 air beam over time, as well as the changes in the detector sensitivity and flow rate.

205 ACSM data were acquired using ACSM_DAQ_v1438 at a time resolution of 30 min and
206 processed using ACSM_Local_v1535 (ARI) with Igor Pro 6.36 (WaveMetrics, Inc., Oregon
207 USA). A total of 15,231 valid observations was recorded, split as follows between the four
208 wind sectors: M – 5,527 (36%), U – 1,257 (8%), IU – 6,590 (43%) and I – 1,857 (12%).

209

210 **2.2.2 Aethalometer**

211 Black carbon was measured with a two-wavelength aethalometer (AE42, Magee Scientific
212 Inc.) at 880 nm. The aethalometer sampled ambient air at a flow rate of 5 liters per minute
213 through a PM_{10} sampling head (SCC-1.197, BGI). Samples were collected at a time resolution
214 of 5 min on a moving tape made of quartz fibers. The tape does not move forward until the
215 spot reaches a certain density (set at 60% in this study). The principle of the aethalometer is to
216 measure the optical attenuation of the collected aerosols and the BC concentration is
217 calculated from the attenuation variation at a wavelength of 880 nm:

$$218 \quad BC_{raw} = \frac{A \times \Delta ATN}{\sigma_{ATN} \times Q \times \Delta t} \quad (2)$$

219 where A is the surface of the sampling area on the filter (1.67 cm^2), Q the volumetric flow rate
220 and ΔATN the variation of the attenuation during the time interval Δt . $\sigma_{ATN} (\text{m}^2 \text{ g}^{-1})$ is the
221 specific attenuation coefficient for BC.

222 However, according to the literature (Virkkula et al. 2007, Park et al. 2010), the relationship
223 between the change in attenuation and BC concentration is not always linear. BC
224 concentrations have been corrected using a procedure proposed by Weingartner et al.
225 (Weingartner et al. 2003) as shown in Eq. (3):

$$BC_{corr} = BC_{raw} \times \frac{1}{C_{ref} \times \left(\frac{1}{m(1 - \omega_0) + 1} - 1 \right) \times \frac{\ln(ATN) - \ln(10\%)}{\ln(50\%) - \ln(10\%)} + 1} \quad (3)$$

where BC_{raw} is the concentration measured by the aethalometer from Eq. (6). C_{ref} and m values of 2.14 and 0.87, respectively, were used in the present study (Petzold et al. 1997, Weingartner et al. 2003). ω_0 , the single scattering albedo (SSA) of the sampled aerosols, was taken from AERONET (AERosol RObotic NETwork, <http://aeronet.gsfc.nasa.gov>) measurements in Dunkirk (N 51°2.1', E 02°22.07') and an average value of 0.9 was used.

2.2.3 Wind and trajectory analyses

The wind and trajectory analyses combine the observed concentrations with wind and air mass conditions, in order to understand the origin of some pollutants, as described in more details in (Boichu et al. 2019) and (Roig Rodelas et al. 2019). Briefly, these analyses were performed using the Zefir tool v.3.70 (Petit et al. 2017). We used (i) the non-parametric wind regression (NWR) model to study the regional vs. local origin of a given pollutant based on local wind speed and direction provided by the ultrasonic anemometer; and (ii) the Concentration-Weighted Trajectory (CWT) method to determine the regions of emissions for transported ammonium and nitrate and/or their precursors, using backtrajectories calculated with HYSPLIT 4 and meteorological data from the Global Data Assimilation System (GDAS) at 1°×1° spatial resolution (Stein et al. 2016). 3-day backward trajectories arriving at 500 m above ground level were calculated every 6 hours over the whole campaign.

3 Results and discussion

3.1 Meteorology

Time series of meteorological parameters are displayed in Figure 2 and their seasonal variability is reported in Table 1. The maximum solar radiation varied roughly from 300 (winter) to 800 $W m^{-2}$ (summer). The seasonal ratio of maximum to mean radiation indicates the impact of cloud coverage. Cloud coverage is on average similar in spring-summer and autumn-winter periods, but the ratio increases by about 50% in autumn and wintertime. Over the whole campaign, pressure and temperature time series are also displayed, with average values of 1013 hPa and 14°C, varying from 978 to 1037 hPa and from 1.4 to 32.2 °C, respectively. The average wind speed was 4.9 $m s^{-1}$, with higher values during fall (5.3 $m s^{-1}$) and winter (5.9 $m s^{-1}$), and lower values during spring (4.7 $m s^{-1}$) and summer

257 (4.2 m s⁻¹). Seasonal mean and maximum pressures indicate clearly an increase of the
 258 frequency of low pressure systems in autumn and winter, corresponding to south-
 259 southwesterly winds from the North Atlantic Ocean.

260 The winter season was mild in 2013-2014, with no frost, but high wind speed events
 261 (>14 m s⁻¹) were observed during 12 days from October to February, including 5 remarkable
 262 storms (Christian, Godehard, Xaver, Dirk and Qumara). The summer was also mild with
 263 moderate temperatures (20-25°C) due to the proximity of the North Sea, and the development
 264 of 18 sea-breeze events. During low wind speed periods, 14 fog events occurred. Rain (> 1
 265 mm/hour) was observed on 51 days, enhancing wet deposition mechanisms of particles.

266 Figure S3 (section D of the SI) shows the seasonal variation of the distribution (normalized to
 267 100%) of sectorial winds according to previously defined ranges of wind directions. The
 268 marine sector exhibited the strongest seasonal variation, with a contribution to the wind
 269 distribution during summers 2013 and 2014 higher than 50% but reduced to ~20% during
 270 winter. The urban sector exhibited a small contribution of ~10% during the whole campaign.
 271 However, even with such a small contribution, air masses originating from this sector can
 272 bring high PM concentrations to the measurement site (see next section). The industrial sector
 273 also exhibited a constant contribution of approximately 10%, independent of the season. The
 274 industrial-urban sector exhibited a larger contribution ranging from 30 to 70% all along the
 275 campaign, especially in winter when southern winds were observed. It is important to note
 276 that the seasonal variability of the wind distribution will be one of the main parameters
 277 driving PM₁ mass and composition variations.

278

279 Table 1. Mean, maximum and minimum values of the main parameters and chemical species
 280 for each season during the entire campaign

		Summer 2013	Autumn 2013	Winter 2013-2014	Spring 2014	Summer 2014
Wind speed (m s ⁻¹)	mean±σ	4.2 ± 2.0	5.3 ± 2.5	5.9 ± 2.5	4.7 ± 2.3	4.2 ± 2.0
	max	12.8	19.1	17.6	12.6	13.7
Radiation* (W m ⁻²)	mean;	296;	132;	149;	306;	291;
	max	968	629	706	982	912
Pressure (hPa)	mean±σ	1018 ± 7	1016 ± 11	1006 ± 12	1015 ± 8	1013 ± 5
	min-max	993 - 1032	980 - 1037	978 - 1032	999 - 1036	995 - 1023
RH (%)	mean±σ	79 ± 11	83 ± 7	83 ± 8	78 ± 11	79 ± 11
	min-max	36 - 96	55 - 98	34 - 100	35 - 97	29 - 96
T (°C)	mean±σ	18.4 ± 2.5	11.8 ± 3.8	9.8 ± 2.4	13.7 ± 3.1	18.4 ± 2.2
	min-max	11.3 - 32.2	1.4 - 21.2	2.6 - 19.2	4.5 - 25.1	11.9 - 32.1
Precip. (mm/hour)	mean±σ	0.01 ± 0.14	0.03 ± 0.15	0.03 ± 0.13	0.01 ± 0.11	0.02 ± 0.27
	max	5	3.4	2.6	4.8	16
SO₂	mean±σ	11 ± 26	8 ± 19	11 ± 22	12 ± 20	10 ± 20

($\mu\text{g m}^{-3}$)	max	310	218	206	206	220
CO ₂	mean $\pm\sigma$	430 \pm 15	428 \pm 16	414 \pm 11	416 \pm 14	415 \pm 12
(ppm)	min-max	391 - 557	364 - 555	367 - 548	370 - 537	388 - 549
BC**	mean $\pm\sigma$	0.7 \pm 0.8	/	0.8 \pm 0.8	0.6 \pm 0.9	0.5 \pm 0.6
($\mu\text{g m}^{-3}$)	max	15.3	/	27.7	28.3	21.7
Org	mean $\pm\sigma$	2.0 \pm 1.8	3.8 \pm 5.1	3.3 \pm 3.2	3.1 \pm 3.0	2.4 \pm 2.1
($\mu\text{g m}^{-3}$)	max	11.8	51.9	70.0	25.2	45.8
SO ₄	mean $\pm\sigma$	2.9 \pm 3.1	2.2 \pm 2.8	1.8 \pm 3.6	2.5 \pm 2.8	2.5 \pm 2.5
($\mu\text{g m}^{-3}$)	max	28.4	23.9	48.2	21.7	31.6
NO ₃	mean $\pm\sigma$	1.3 \pm 2.4	2.7 \pm 4.0	2.5 \pm 3.7	4.1 \pm 4.7	1.6 \pm 2.6
($\mu\text{g m}^{-3}$)	max	15.0	23.0	27.3	21.5	16.2
NH ₄	mean $\pm\sigma$	1.1 \pm 1.1	1.4 \pm 1.7	1.2 \pm 1.7	1.9 \pm 1.6	1.1 \pm 1.1
($\mu\text{g m}^{-3}$)	max	6.1	10.0	21.4	8.0	8.0
Cl	mean $\pm\sigma$	0.03 \pm 0.07	0.09 \pm 0.16	0.06 \pm 0.10	0.06 \pm 0.12	0.02 \pm 0.04
($\mu\text{g m}^{-3}$)	max	1.5	3.2	2.3	2.0	0.7

281 * Radiation represents daytime values computed using only data after sunrise and before sunset.

282 ** BC values are not available in autumn.

283

284 3.2 PM₁ mass concentration and chemical speciation

285 To check the aerosol measurements quality, the total mass concentration of PM₁ measured by
286 the ACSM (NR-PM₁) and the aethalometer (BC) was compared to TEOM-FDMS
287 measurements from June 23 to September 10, 2014 (Figure 3). Figure 3 displays time series
288 of PM₁ from both (i) the ACSM and aethalometer and (ii) the TEOM-FDMS. This figure
289 shows that PM₁ concentrations ranged from less than 1 $\mu\text{g m}^{-3}$ to approximately 50 $\mu\text{g m}^{-3}$
290 during the 2.5 months of the comparison period, with both traces showing similar temporal
291 variations. A scatter plot of the ACSM + aethalometer and TEOM-FDMS measurements is
292 shown in Figure 3b. This figure also displays 24-hour averaged values for both measurements,
293 which indicate an excellent agreement (slope: 0.94; $r^2 = 0.94$), suggesting that (i) most of the
294 PM₁ mass ($7.9 \pm 7.1 \mu\text{g m}^{-3}$ measured by the TEOM-FDMS) is captured by the combination
295 of ACSM and aethalometer ($8.1 \pm 6.3 \mu\text{g m}^{-3}$); (ii) the RF/RIE calibrations and the CE
296 correction applied for the ACSM data analysis are correct within measurement uncertainties.
297 Time series of Cl, NH₄, NO₃, Organics and SO₄ measured by ACSM, and black carbon
298 measured with the aethalometer, are displayed in Figure 4. Each species exhibits different
299 temporal variations and concentration levels. Chloride is a minor fraction of PM₁ and its
300 concentration is lower than 3 $\mu\text{g m}^{-3}$ over the entire observation period. NH₄ is relatively
301 constant throughout the whole year at levels lower than 5 $\mu\text{g m}^{-3}$, with the exception of spring
302 2014 (March-May) where levels up to 15 $\mu\text{g m}^{-3}$ were observed. NO₃, Organics, SO₄ and BC
303 are much more time-dependent, with concentrations rising to 30-50 $\mu\text{g m}^{-3}$.
304 Some PM₁ species seem to exhibit significant seasonal variations (Figure S4, section D of the
305 SI; and Table 1). Average seasonal chloride concentrations were very low ($< 0.1 \mu\text{g m}^{-3}$).

306 Sulfates did not exhibit seasonal variations due to its industrial origin and showed similar
307 contributions from this sector (wind occurrences) for all seasons. Nevertheless, the NWR plot
308 (section E of the SI, Figure S5a) for sulfate shows a strong contribution from local sources,
309 highlighting the industrial sector, and a minor one from the marine sector, which could be the
310 result of ship emissions. This pattern is even more pronounced for its precursor SO₂ (Figure
311 S5b) whose high concentrations are clearly pointing at the industrial area. For the other
312 species, PM₁ measured during both summers 2013 and 2014 exhibit similar chemical
313 compositions due to similar wind patterns, which in turn likely imply the impact of similar
314 sources on the measurement site. However, the relative contribution of NO₃ is higher during
315 cold seasons (autumn, winter and spring) compared to the two warmer summers (Table 1).
316 Especially in spring 2014, NO₃ exhibits the highest average concentration (4 µg m⁻³) and
317 contribution (33%) when NH₃ emissions – required for the formation of ammonium nitrate –
318 start to peak in North-Western Europe due to fertilizer spreading. Besides, whereas 96% of
319 NH₃ emissions is attributed to the agricultural sector when considering the entire Hauts-de-
320 France region, it decreases down to 50% in the Dunkirk area, where local emissions from the
321 “manufacturing industries, waste treatment and construction” account for half according to
322 the latest available inventory (AtmoHdF 2012). The levels of ammonium and nitrate could
323 indeed be related to both a local and regional origin through their NWR plots (section E of the
324 SI, Figure S5c and d). The regional influence was further investigated through Concentration-
325 Weighted Trajectory (CWT) maps (section E of the SI, Figure S6), and pointed out a
326 significant contribution from medium- to long-range transport of aged ammonium nitrate
327 aerosols or their precursors from Central Europe (Germany, Poland, and Austria) (Figure S5).
328 Organics concentrations and contributions are also higher during cold seasons (autumn and
329 winter), which seems due to both local and regional sources (Figure S5e) coming from the
330 industrial-urban sector (55-63% of wind occurrences), such as emissions from urban sources
331 such as residential heating and traffic, while an additional contribution from industrial
332 emissions in this direction (storage facility of petrochemical products and food processing
333 industries) cannot be excluded. BC represents 5 to 9% of PM₁ mass but no clear observation
334 is found for its seasonal variation.

335 For the sake of comparison with other studies, Figure 5 displays 17 field campaigns
336 performed worldwide to investigate the composition of ambient aerosols. These studies
337 employed ACSM instruments in different environments classified as urban (5 sites), suburban
338 (2 sites), industrial-urban (2 sites) and rural (8 sites). For similar types of sites the studies are
339 sorted by decreasing NR-PM₁ mass concentrations. The durations of these campaigns are

340 highly variable and range from 3 weeks to 7 years. Campaigns performed on a timescale of
341 years can provide seasonal and even annual variations of submicron aerosols. The averaged
342 mass concentration varies from approximately 2-6 $\mu\text{g m}^{-3}$ for background sites (Hyytiälä,
343 Montsec, Look Rock) (Budisulistiorini et al. 2015, Minguillón et al. 2015, Heikkinen et al.
344 2020) up to 30-50 $\mu\text{g m}^{-3}$ for polluted cities (Santiago, Beijing) (Sun et al. 2012, Carbone et
345 al. 2013). Our study indicates an annual concentration of 9.0 $\mu\text{g m}^{-3}$, which is similar to that
346 observed for another industrial-urban site in Atlanta (9.5 $\mu\text{g m}^{-3}$) (Budisulistiorini et al.
347 2015), whose duration was one year.

348 The contribution of each species to NR-PM₁ is shown in the bottom panel of Figure 5.
349 Organics dominate (> 50%) for most of the sites and the contribution can reach up to 70% as
350 seen in Atlanta. In our study, organics account for 32% on average, which is within the lowest
351 contributions observed worldwide and similar to that found at the SIRTA suburban site near
352 Paris (39%) (Petit et al. 2015). Ammonium exhibits a relatively constant fraction at all
353 locations, with an average value of approximately 12%. The sulfate component contributes
354 from 8 to 35% of NR-PM₁, with higher contributions found for rural sites without (Tibetan
355 Plateau: 30%; South Africa: 32%) (Tiitta et al. 2014, Du et al. 2015) or with urban influence
356 (Senegal: 35%) (Rivellini et al. 2017). Interestingly, industrial sites do not exhibit the highest
357 levels of sulfate, whose contribution to the chemical composition does not seem to depend on
358 the nature of the site. Nitrate varies from 6% to 28%, with the highest values observed for
359 Paris (28%) (Petit et al. 2015), Dunkirk (26%) (this work) and Beijing (25%) (Sun et al.
360 2012). Higher levels of NO_x (NO and NO₂) in urban areas are likely the cause for higher
361 nitrate levels in PM₁ since NO_x are oxidized to nitric acid in the atmosphere, which in turn
362 forms secondary nitrate particles (Matsumoto and Tanaka 1996). Chloride is always a minor
363 fraction and represents less than 3% at all sites.

364 Figure 6 shows PM₁ mass concentrations and chemical compositions averaged over the entire
365 campaign for the four wind sectors defined above (marine, urban, industrial and industrial-
366 urban). The average concentration for all sectors is $9.4 \pm 9.1 \mu\text{g m}^{-3}$ with significantly higher
367 concentrations for the industrial ($11.7 \pm 8.8 \mu\text{g m}^{-3}$) and urban ($15.6 \pm 9.5 \mu\text{g m}^{-3}$) sectors, and
368 a lower concentration for the marine sector ($8.0 \pm 8.5 \mu\text{g m}^{-3}$). The organic fraction
369 contributes from 28% up to 38% of the PM₁, with the exception of the industrial sector for
370 which the aerosol composition is enriched in sulfate (only 14% organics). As mentioned
371 before, a higher organic fraction for the industrial-urban sector could be due to a storage
372 facility of petrochemical products or food processing industries in this area. For inorganic
373 species, the ammonium contribution is similar between all sectors ($\approx 14\%$) and non-refractory

374 chloride (therefore non-marine) represents less than 1% of the mass concentration. For
375 chloride, the small but significantly higher contribution from the industrial sector reflects the
376 presence of sources already identified in other campaigns in this area or similar ones (Hleis et
377 al. 2013, Taiwo et al. 2014, Crenn et al. 2017, Setyan et al. 2019), e.g. the formation of KCl in
378 the steelwork sintering process. Nitrate and sulfate contributions are more variable between
379 the different sectors, with contributions ranging from 8-35% for NO₃ and 13-58% for SO₄.
380 The urban and industrial sectors stand out by exhibiting the highest contributions for nitrate
381 (about 35%) and sulfates (about 58%), respectively. As mentioned previously, these higher
382 contributions may be due to higher NO_x and SO₂ emissions in the urban and industrial sectors,
383 respectively. BC contributes about 5% for the marine and industrial sectors, while it reaches 8
384 to 9% for the urban and industrial-urban sectors where traffic and wood burning may be
385 significant sources.

386

387 **3.3 Aerosol ion balance**

388 To investigate the ion balance of the sampled aerosols, a predicted concentration of NH₄ was
389 calculated from the sum of chloride, nitrate and sulfate ion concentrations using Eq. (2).
390 These ions are assumed to be bound to ammonium in a neutral aerosol to form NH₄Cl,
391 NH₄NO₃ and (NH₄)₂SO₄, respectively. The calculated value (NH_{4,predicted}) is then compared to
392 the measured NH₄ concentration. If the two values are similar, the particles are fully
393 neutralized. Although an ion balance cannot be used as a direct measurement of pH in
394 aerosols, it can still be used to distinguish qualitatively between an “acidic” (negative balance;
395 below the 1:1 line) and an “alkaline” (positive balance; above the 1:1 line) aerosol (Zhang et
396 al. 2007, Guo et al. 2015).

397 Particles are considered “more acidic” if the measured concentration is lower than the
398 predicted value, since the imbalance indicates that acidic compounds such as H₂SO₄, HNO₃ or
399 HCl are likely present in the aerosol. Note that as Cl being a minor fraction (<1%) of NR-
400 PM₁, the contribution of HCl to the aerosol ion balance – if present – is negligible.

401 Figure 7 displays the correlation between measured and predicted NH₄ with a color coding
402 based on SO₄ concentrations. All SO₄ concentrations higher than 25 μg m⁻³ are included in the
403 red color. When the symbols are scattered around the 1:1 line, the measured and predicted
404 concentrations are in good agreement, indicating that particulate ammonium is fully
405 neutralized. When the symbols are significantly below the 1:1 line, there is an overestimation
406 of the predicted concentrations. As mentioned above, this overestimation may be due to either

407 a lack of ammonium (thus of NH_3) or the presence of acidic species in the particles, which
408 may be characteristic of "fresh" emissions.

409 Aerosols appear not neutralized or "acidic" (below the 1:1 line) with high sulfate
410 concentrations (Figure 7a) but not with high nitrate concentrations (Figure S7, section F of the
411 SI). This observation suggests that aerosol ion balance is driven by acidic sulfate compounds.
412 The same analysis is displayed for the four wind sectors in Figure 7b-e. For the marine and
413 urban sectors, the aerosols are well neutralized and SO_4 concentrations are low ($4\text{-}5 \mu\text{g m}^{-3}$ on
414 average). NH_4 is therefore mainly bound to NO_3 . The industrial sector shows strong
415 deviations from the 1:1 line, with higher acidity levels correlated to elevated SO_4
416 concentrations (Figure 7e), which suggests that acidic aerosols are due to "fresh" industrial
417 emissions of sulfur compounds. It is interesting to note that the industrial-urban sector shows
418 a mixed behavior between industrial and urban sectors.

419 In order to investigate the influence of acidity on the NR- PM_1 composition, the measured-to-
420 predicted NH_4 ratio was plotted as box plots for the entire campaign and the four sectors in
421 Figure 8a. The aerosol composition for acidic and non-acidic particles is displayed in Figure
422 8b. Figure 8a clearly shows that the industrial sector exhibits the lowest $\text{NH}_{4,\text{measured-to-}}$
423 $\text{NH}_{4,\text{predicted}}$ ratio (0.64 on average), corresponding to the most acidic particles (ratio threshold
424 of 0.75). This ratio can be seen as a rough indicator of an equal number of moles of
425 $(\text{NH}_4)_2\text{SO}_4$ and NH_4HSO_4 (Zhang et al. 2005). The neutralized particles were defined for a
426 ratio ranging from the mean value of 0.85 to the 90th percentile (1.07). The "most acidic" and
427 "neutralized" periods account for 22% and 41% of the total sampling time, respectively. The
428 rest of the observations falls into a mildly acidic regime and are not analyzed separately.
429 Figure 8b displays the average composition of "neutralized" and "most acidic" particles,
430 respectively. Compared to those considered as neutralized, the "most acidic" particles
431 represent only half the mass. However, they contain more SO_4 (58% compared to 17%),
432 which is consistent with the above discussion. The acidic particles contain less nitrate (7%
433 compared to 34%), probably due to the displacement of HNO_3 by H_2SO_4 in the competition
434 for NH_3 (West et al. 1999).

435

436 **3.4 SO_2 -to- SO_4 conversion**

437 In the absence of direct measurements of industrial emissions, it is likely that some of the
438 particulate sulfate is directly produced during industrial processes, in particular as by-products
439 of the coke oven used to feed the three blast furnaces of the main steelworks in the area.
440 Furthermore, NH_4HSO_4 could be formed at the exit of the chimneys through the direct

441 reaction of SO_3 with H_2O and NH_3 in the presence of additional molecules (Chen et al. 2018).
442 Then NH_4HSO_4 can quickly react with NH_3 and form ammonium sulfate. However, a rapid
443 conversion of gaseous SO_2 into particulate sulfate could also contribute to the large sulfate
444 concentrations observed at the measurement site when the wind originates from the industrial
445 sector. The conversion can take place both in the gas phase (Stockwell and Calvert 1983,
446 Atkinson and Lloyd 1984) through an oxidation of SO_2 initiated by OH radicals, and in the
447 aqueous phase through reactions involving H_2O_2 and O_3 (Jacob and Hoffmann 1983,
448 Schwartz 1987). The sulfuric acid thus generated has a strong ability to nucleate in the
449 presence of water molecules and to form new particles by condensation of other low volatility
450 species (Kulmala et al. 2000, Weber et al. 2001). However, the oxidation of SO_2 in aqueous
451 phase is faster than in the gas-phase (Khoder 2002). Eatough et al. (1994) found that the gas-
452 phase oxidation rate can vary from <1% to 10% of ambient SO_2 per hour at high temperature
453 and relative humidity. Meanwhile, in the aqueous-phase the rate can reach 100% of SO_2
454 converted in less than an hour under optimum conditions (Eatough et al. 1994). Other
455 pathways such as the heterogeneous oxidation of S(IV) catalyzed by Fe(III) or Mn(II) (Jacob
456 and Hoffmann 1983) can occur, especially in this area where heavy metals have been
457 evidenced as important tracers of industrial emissions (Alleman et al. 2010, Setyan et al.
458 2019): When sulfuric acid is formed within or transferred to the aerosol, it reacts easily with
459 gaseous ammonia to form ammonium sulfate at the particle surface (Matsumoto and Tanaka
460 1996).

461 The observation of industrial plumes arriving at our ground sampling site requires that two
462 simultaneous conditions are met: (i) wind blowing from the southwest (industrial direction);
463 (ii) unstable atmospheric conditions favoring the dispersion of the plume. The Non-parametric
464 Wind Regression (NWR) plots of gaseous SO_2 and particulate SO_4 (Figure S6, a and b)
465 confirm that high concentrations observed at the receptor site are mainly originating from the
466 industrial sector (226° - 270°). Similar results were observed by a recent study near steel
467 plants in China, which showed important increases in sulfate, ammonium, CO and SO_2 when
468 industrial plumes reached the sampling site (Lei et al. 2020).

469 To assess whether the conversion of SO_2 into SO_4 was of importance during this study, the
470 influence of several microphysical and meteorological parameters on SO_2 , SO_4 and the
471 particulate sulfur fraction has been investigated. These parameters included atmospheric
472 vertical turbulences via the standard deviation σ_w of vertical wind speed w , horizontal wind
473 speed, relative humidity and temperature.

474 We defined the particulate sulfur fraction as the ratio of particulate sulfur to total sulfur
475 (S_p/S_{tot}), which is calculated as:

$$476 \quad \frac{S_p}{S_{tot}} = \frac{[SO_4] \times \frac{32}{96}}{[SO_2] \times \frac{32}{64} + [SO_4] \times \frac{32}{96}} \quad (4)$$

477 S_p/S_{tot} only takes into account the sulfur content and is not influenced by the oxygen content.

478 Two key parameters (RH and vertical turbulence) were found to significantly influence the
479 gas-particle partitioning of sulfur as shown in Figure 9 (left and right, respectively). SO_2 and
480 SO_4 concentrations, as well as S_p/S_{tot} ratios, shown in this figure are only considered within
481 the industrial sector to focus on the SO_2 -to- SO_4 conversion that occurs on a short timescale
482 (4-30 min) during the transport of fresh industrial emissions of sulfur compounds to the
483 sampling site. This data was binned accordingly to the descriptive parameter (RH, σ_w) with
484 the number of observations for each bin shown at the top of the first panel.

485 Figure 9a shows that SO_2 increases linearly with relative humidity until 70-80%. This
486 concomitant increase in RH and SO_2 could be due to industrial plumes passing over the
487 measurement site since these plumes are likely emitted with large concentrations of both
488 water and SO_2 . Interestingly, the SO_2 concentration starts decreasing when RH is higher than
489 80%, but an opposite trend is observed for SO_4 (Figure 9b). Indeed, SO_4 also increases with
490 RH until 80% but does not decrease for higher humidity levels. This behavior could be related
491 to a higher rate of SO_2 conversion at high humidity. In Figure 9c, the S_p/S_{tot} ratio clearly
492 shows that the particulate sulfur fraction is almost constant (approximately 0.1 with a low
493 level of dispersion in the values) for low RH conditions (RH < 70%). However, the sulfur
494 content is shifted from the gaseous to the particulate phase when RH is higher than 70-80%
495 (S_p/S_{tot} increases to 0.3 at 90-100% RH). This is likely due to the gas-particle conversion that
496 is favored under high RH conditions when the fast aqueous phase oxidation dominates. Sun et
497 al. reported a similar impact of RH on sulfate aerosols in Beijing for the winter season (Sun et
498 al. 2013). The authors showed that the average SO_4 -to-total sulfur ratio is less than 0.05 at low
499 RH (< 40%), indicating a very low sulfur oxidation ratio. This ratio quickly increases and
500 reaches 0.23 at high RH (80-90%). Recently, Zhao et al. studied the conversion of SO_2 into
501 sulfates on soot surfaces and found that water promotes sulfate formation for RH ranging
502 from 6%–70%, while RH > 80% inhibits it (Zhao et al. 2017). This trend was not observed
503 here because BC is only a minor contributor (~7%) of PM_{10} .

504 The atmospheric vertical turbulence can be characterized by the standard deviation σ_w of the
505 vertical wind speed w . This metric represents the fluctuation of vertical wind speed in units of
506 $m s^{-1}$ and quantifies the vertical mixing of air masses within the surface layer. Trends in SO_2 ,

507 SO₄ and S_p/S_{tot} on this parameter are shown in Figure 9 (right). For both SO₂ and SO₄, higher
508 σ_w values lead to higher concentrations (Figure 9a-b). As previously discussed, this can be
509 explained by elevated emission sources such as plumes from industrial chimneys, which can
510 reach the ground surface after dispersion only with strong enough vertical mixing. This
511 behavior had already been observed for some VOCs at the same site (Xiang et al. 2012). In
512 contrast, S_p/S_{tot} decreases with σ_w (Figure 9c), which suggests that the SO₂-to-SO₄ conversion
513 is less favored under highly turbulent conditions.

514 To better understand the multi-influence of humidity and vertical turbulence (σ_w) on the
515 conversion process, the examination of the RH dependence of the S_p/S_{tot} ratio was carried out
516 using four bins of σ_w (0-0.5, 0.5-1, 1-1.5 and ≥ 1.5 m s⁻¹) (Figure S7). This shows that the
517 S_p/S_{tot} ratio increases significantly with RH for each σ_w bin, except for the last one ($\sigma_w \geq 1.5$
518 m s⁻¹) where the change is unclear. Interestingly, higher conversion ratios seem to be observed
519 for low values of σ_w , which may be due to lower wind speed values, and as a consequence,
520 longer conversion times between the emission source and the measurement site. These results
521 indicate that high RH (> 70%) and relatively low σ_w values (< 1 m s⁻¹) are the best conditions
522 for an efficient SO₂-to-SO₄ conversion at this site.

523

524 **4 Conclusions**

525 The first long-term and near real-time measurements of the chemical composition of
526 submicron aerosols at an urban background and coastal site impacted by industrial emissions
527 showed a unique chemical signature compared to previous datasets existing in the literature.
528 The aerosol evolution and processing are tightly linked with meteorology and even
529 micrometeorology due to the complex dynamics of coastal areas, and the industrial emissions
530 at various stack heights. The complete dataset has been analyzed according to the season and
531 four different wind sectors. The aerosol ion balance based on measured and predicted NH₄
532 suggests that the majority of the particles are neutralized in the marine and urban sectors,
533 whereas in the industrial sector, “more acidic” particles generally contain significantly higher
534 concentrations of sulfate (~60%). Gas-phase SO₂ is strongly emitted in the nearby industrial
535 area and quickly processed to form secondary SO₄ particles on a timescale shorter than 30
536 min. Both SO₂ and SO₄ were mostly emitted by chimneys and therefore influenced by
537 atmospheric vertical mixing. High RH (>70%) and relatively low σ_w values (< 1 m s⁻¹)
538 provided the best conditions for an efficient SO₂-to-SO₄ conversion rate.

539

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552

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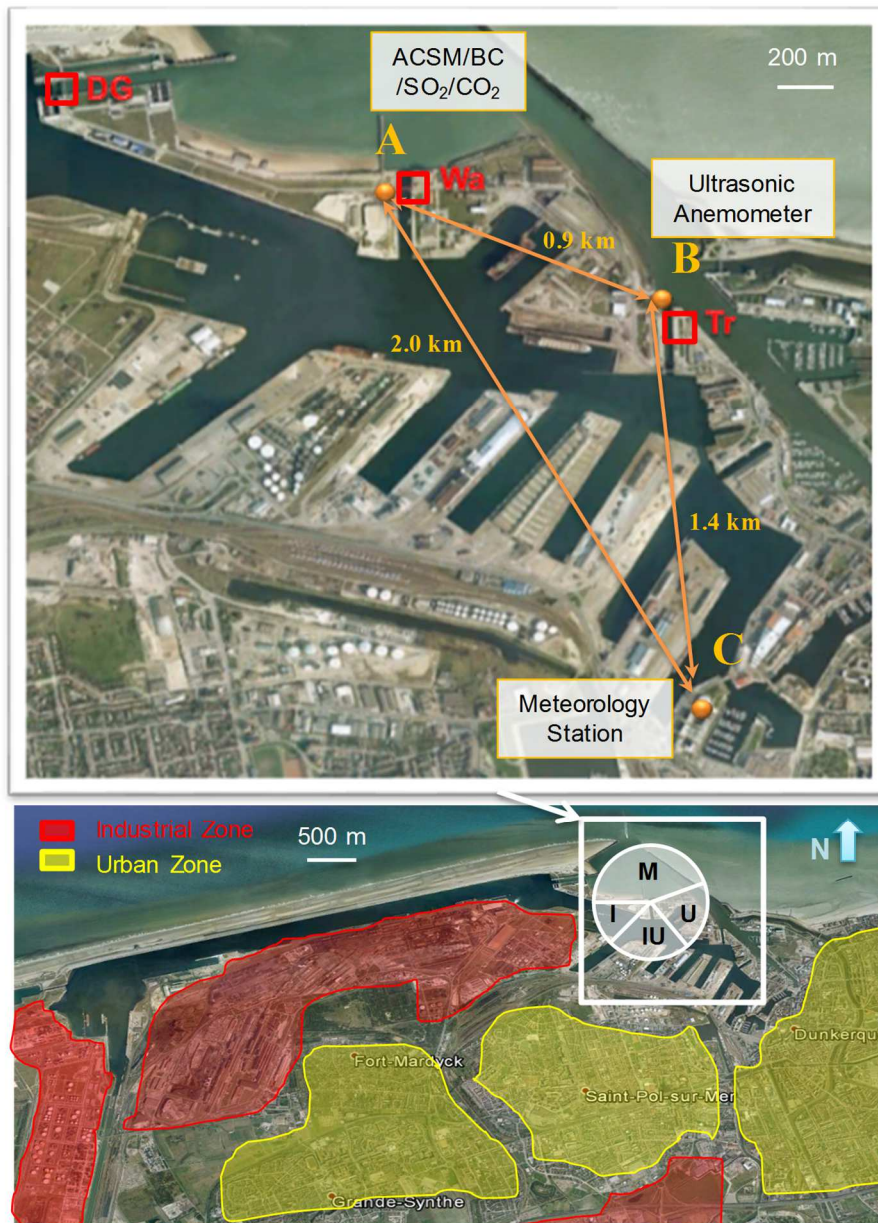
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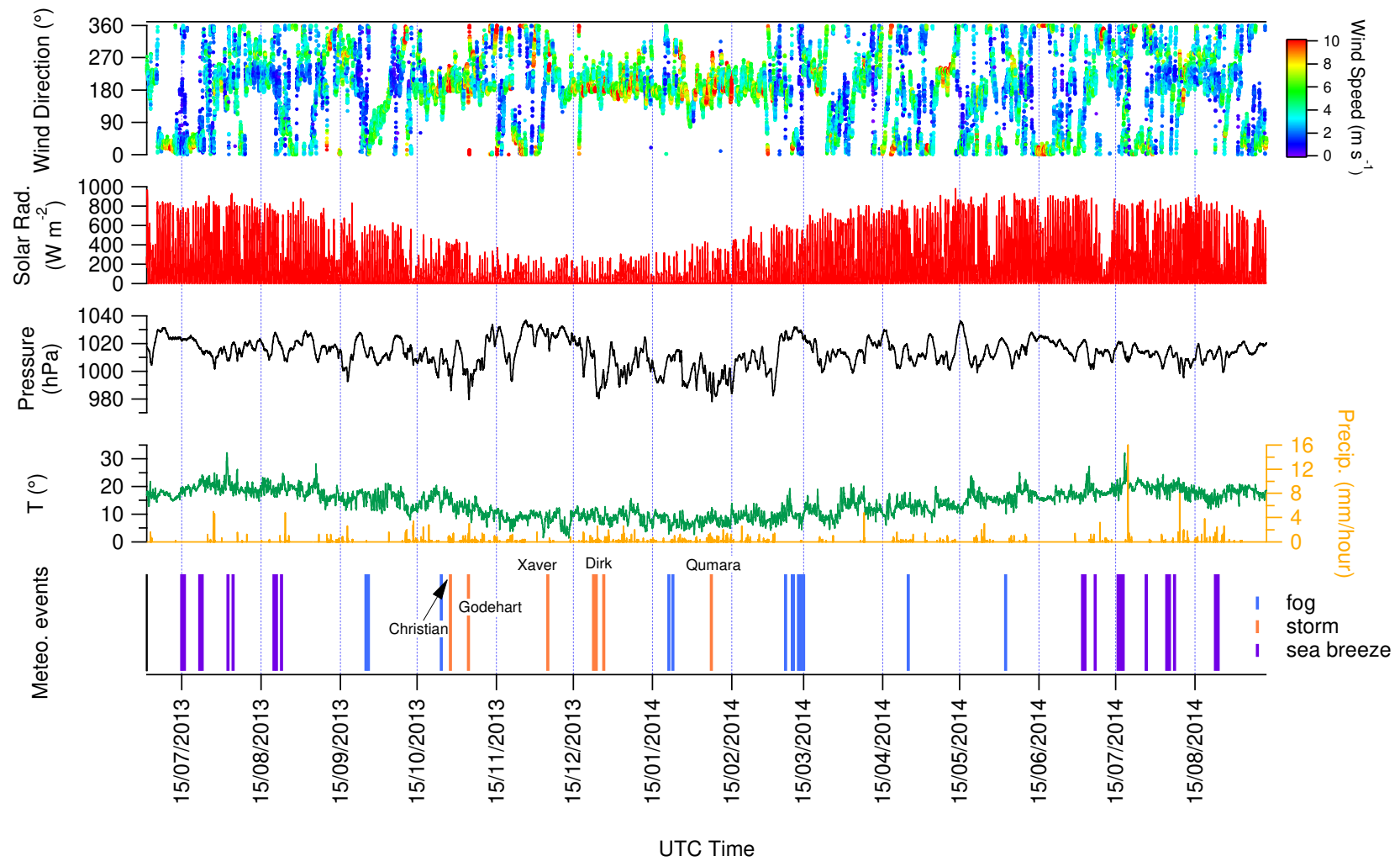
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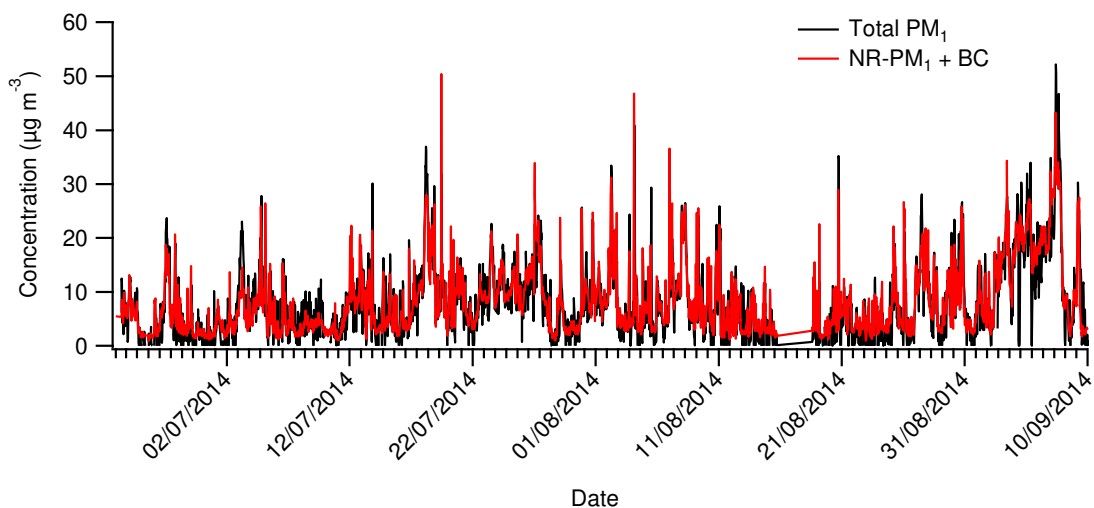


861
 862 Figure 1. (Bottom) Maps of the urban-industrial area of Dunkirk (M: marine, U: urban, IU:
 863 industrial-urban, I: industrial wind sectors); Top Locations of the measurements: (A) Port-Est:
 864 ACSM and other chemical measurements; (B) Ultrasonic anemometer; (C) Meteorology
 865 station. Locks indicated in red (DG: De Gaulle; Wa: Watier; Tr: Trystram) (adapted from
 866 Google Maps)

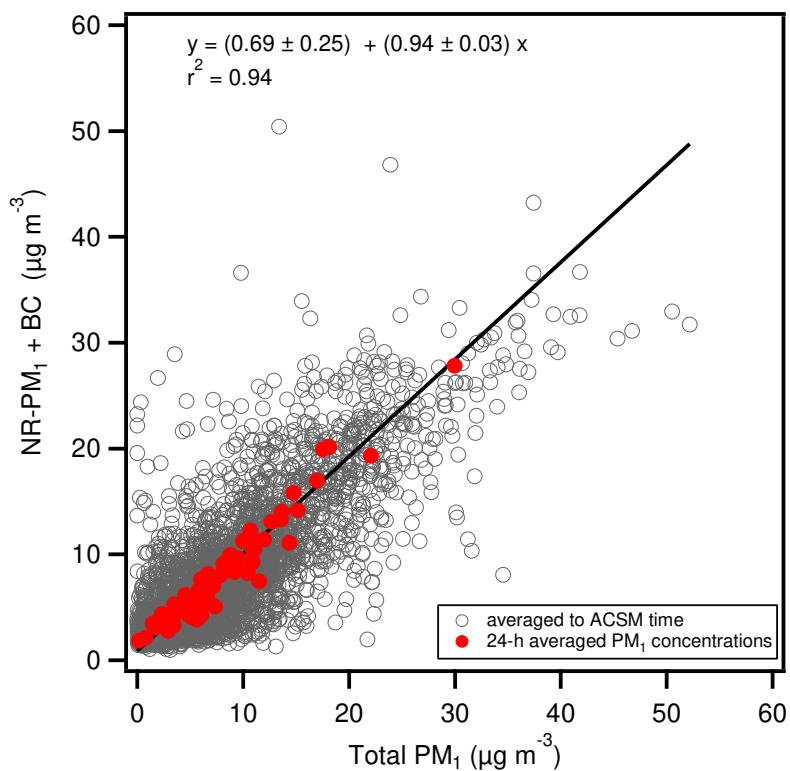


867

868 Figure 2. From top to bottom: wind direction colored by wind speed, solar radiation, pressure, temperature and precipitation, and identified
 869 meteorological events (fog, storm and sea breeze).

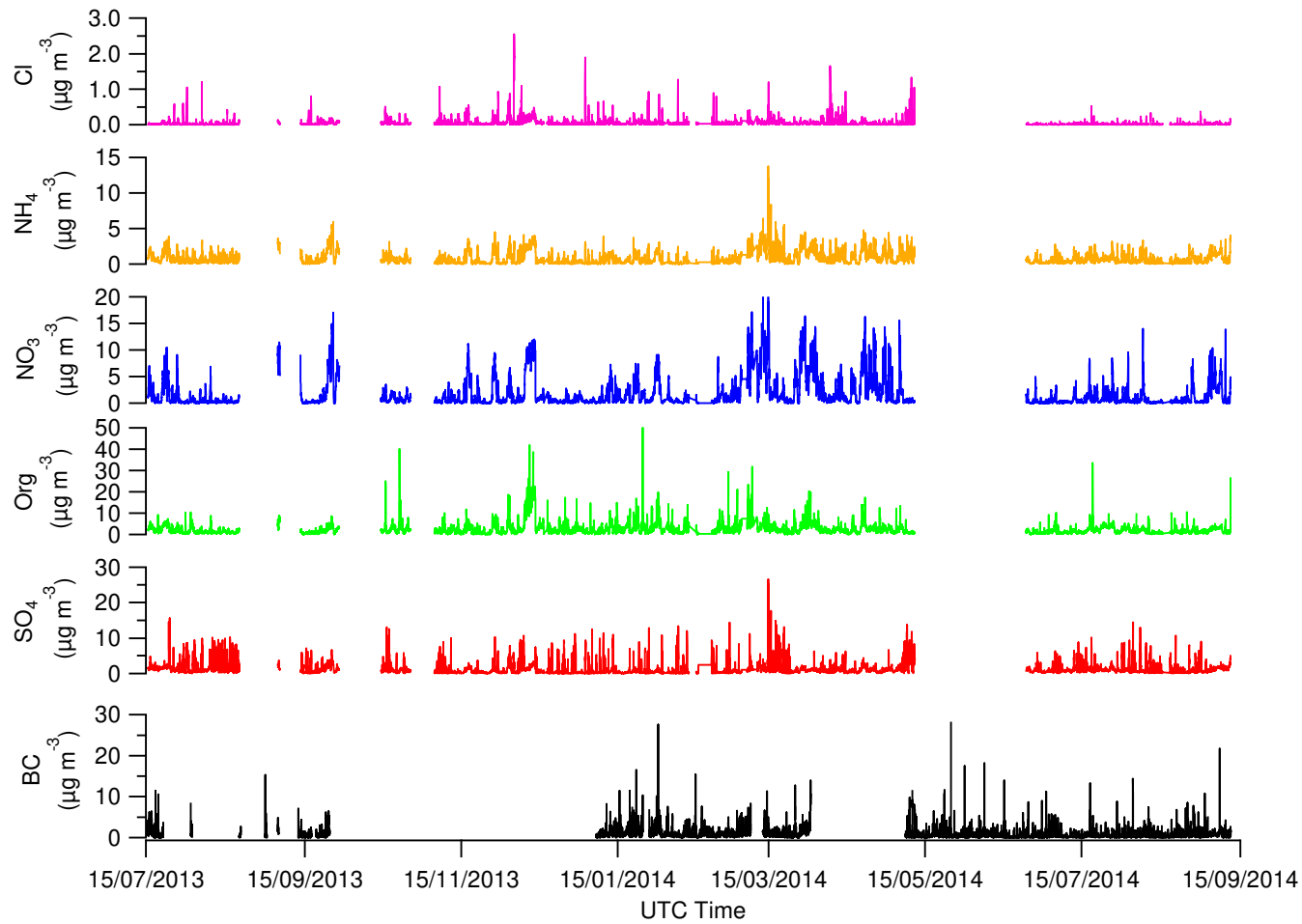


870 (a)



871 (b)

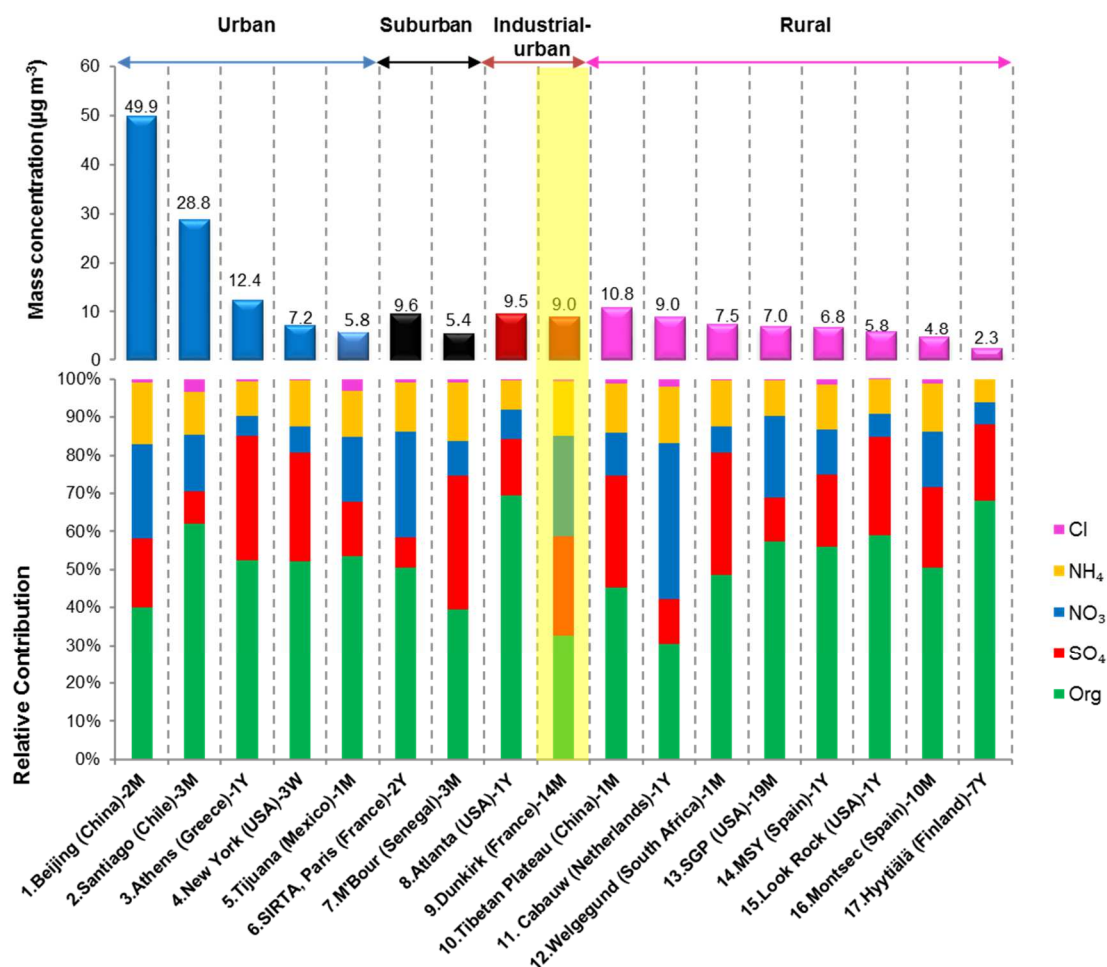
872 Figure 3. (a) Time series of NR-PM₁+BC measured by ACSM+aethalometer vs. Total PM₁
 873 measured by TEOM-FDMS; (b) Scatter plot of PM₁ concentrations (NR-PM₁+BC vs. Total
 874 PM₁): averaged to ACSM time (grey) and 24-h averaged concentration (red). Linear fit
 875 performed on 24-h averaged values.



876

877 Figure 4. Time series of the chemical composition of NR-PM₁ measured by ACSM (Cl, SO₄, NH₄, NO₃, and Org), and Black carbon measured

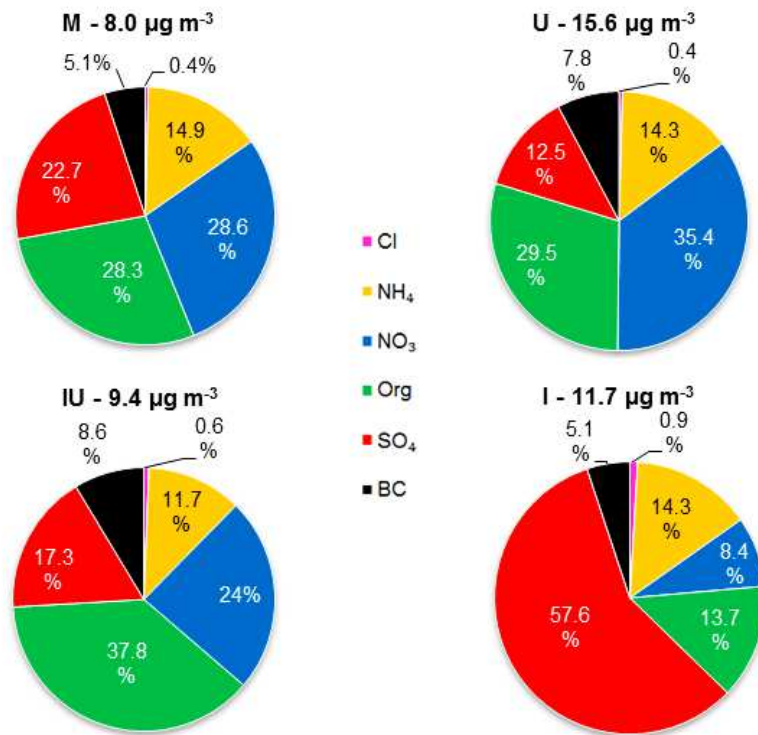
878 by aethalometer.



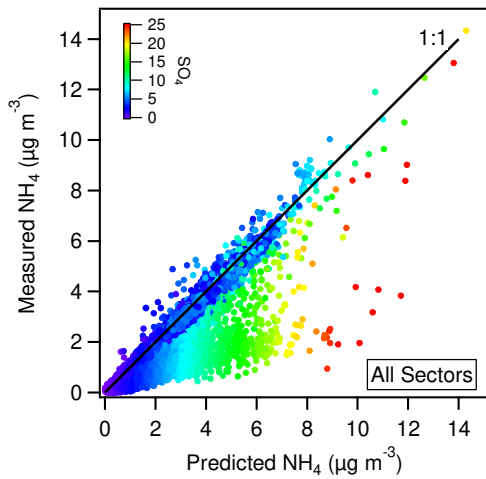
879

880 Figure 5. Comparison of averaged concentrations and chemical compositions of NR-PM₁ for
 881 this study and other ACSM field campaigns. Details on the location and duration of each
 882 campaign are given on the horizontal axis (M: months; W: weeks; Y: year). SGP: Southern
 883 Great Plains, MSY: Montseny.

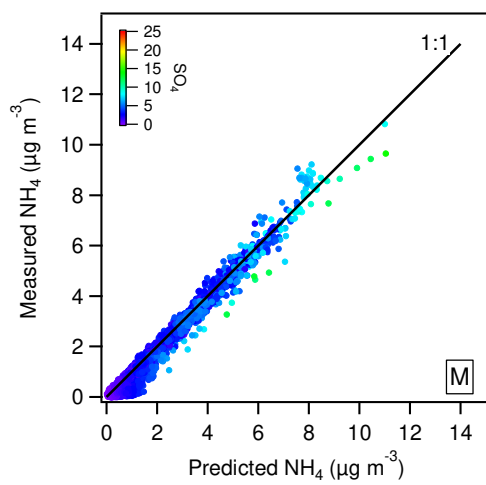
884 1. Beijing (China) (Sun et al. 2012); 2. Santiago (Chile) (Carbone et al. 2013); 3. Athens (Greece) (Stavroulas et
 885 al. 2019); 4. New York (USA) (Ng et al. 2011); 5. Tijuana (Mexico) (Takahama et al. 2013); 6. SIRTA, Paris
 886 (France) (Petit et al. 2015); 7. M'Bour (Senegal) (Rivellini et al. 2017); 8. & 15. Atlanta (USA) and Look Rock
 887 (USA) (Budisulistiorini et al. 2015); 9. This study; 10. Tibetan Plateau (China) (Du et al. 2015); 11. Cabauw
 888 (Netherlands) (Schlag et al. 2015); 12. Welgegund (South Africa) (Tiitta et al. 2014); 13. SGP (USA) (Parworth et
 889 al. 2015); 14. SMY (Spain) (Minguillón et al. 2015); 16. Montsec (Spain) (Ripoll et al. 2015). 17. Hyytiälä
 890 (Finland) (Heikkinen et al. 2020)



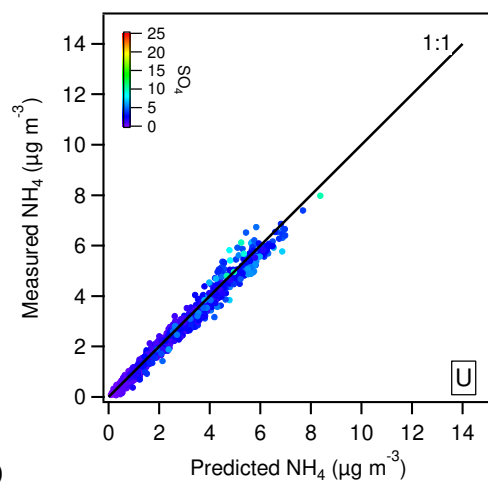
891 Figure 6. Chemical speciation of PM₁ for the four wind sectors (M: marine, U: urban, IU:
 892 industrial-urban, I: industrial).



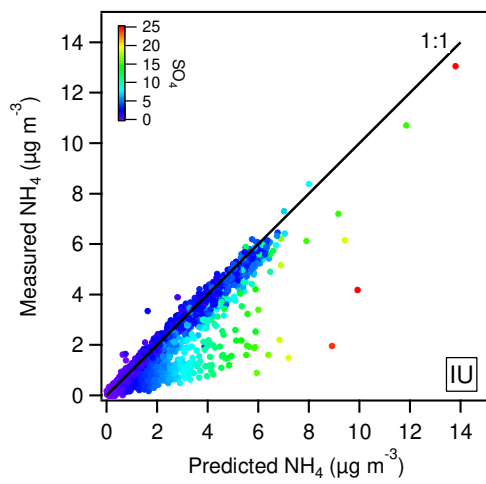
893 (a)



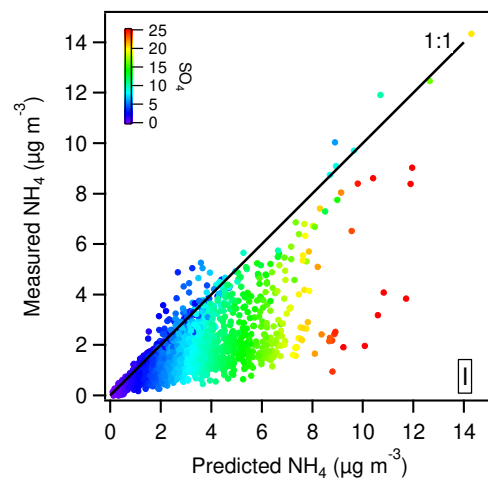
894 (b)



(c)

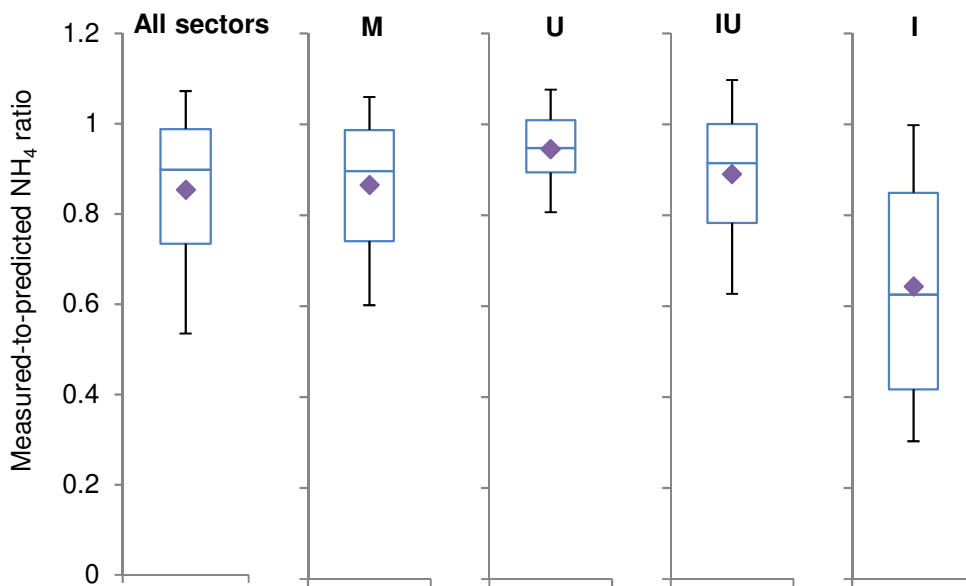


895 (d)

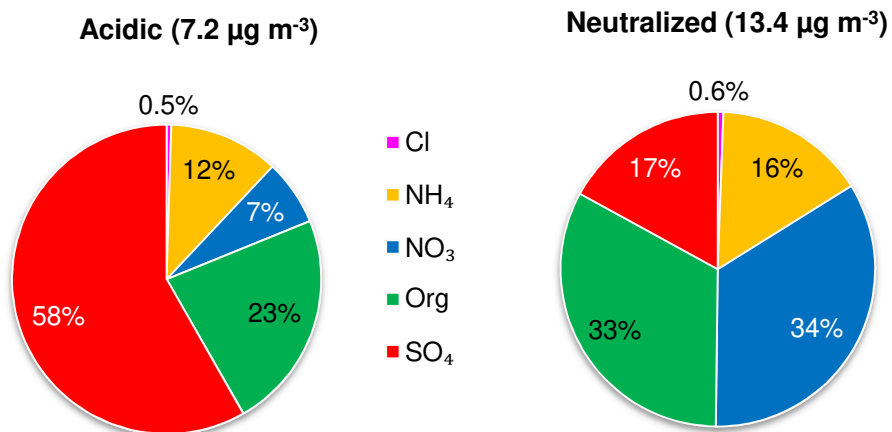


(e)

896 Figure 7. Correlation between measured and predicted NH_4 color coded by the sulfate
 897 concentration ($\mu\text{g m}^{-3}$) (a) for the entire study and (b) – (e) for marine (M), urban (U),
 898 industrial-urban (IU) and industrial (I) wind sectors, respectively. The black line represents
 899 the 1:1 line.

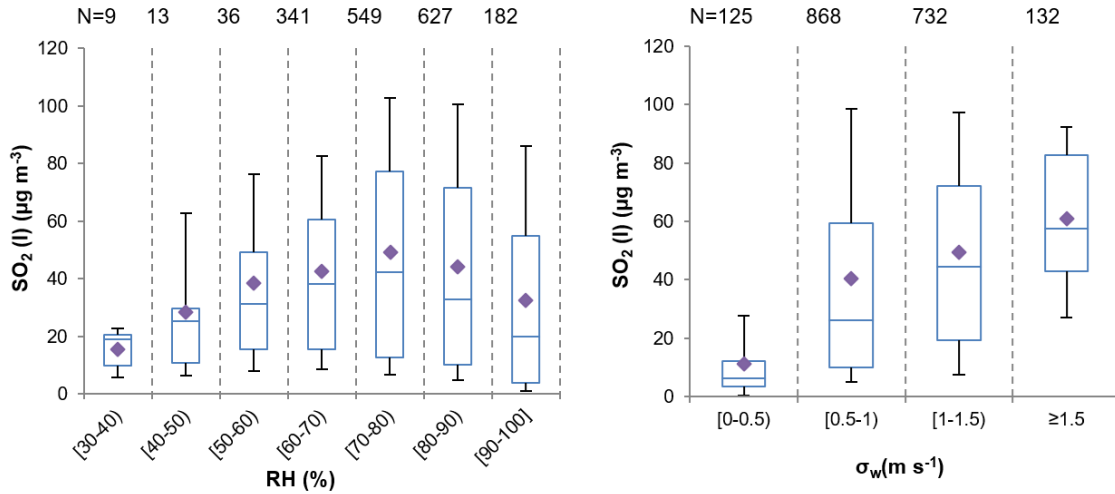


900 (a)

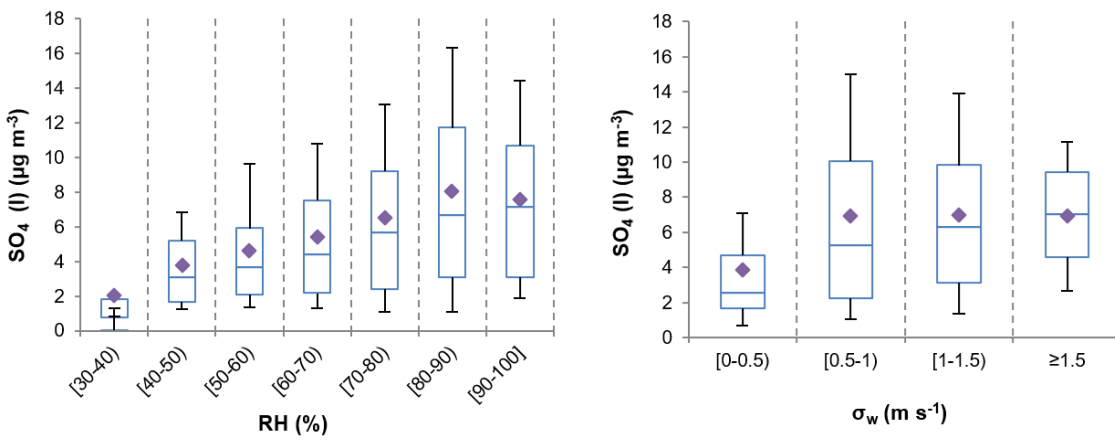


901 (b)

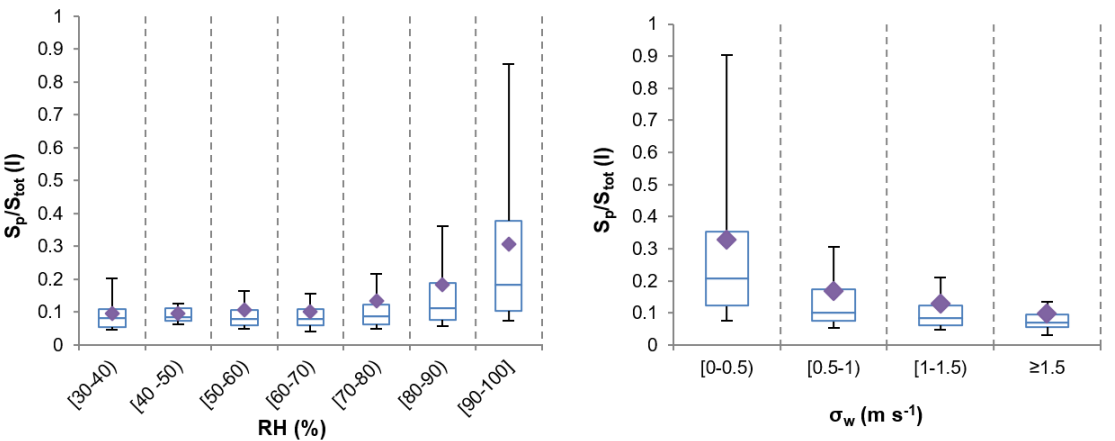
902 Figure 8. (a) Box plots of the measured-to-predicted NH_4 ratio for the entire study and the
 903 four wind sectors. The data correspond to the mean (diamond), median (horizontal line), 25th
 904 and 75th percentiles (lower and upper boxes), and 10th and 90th percentiles (lower and upper
 905 whiskers). (b) Chemical speciation of NR- PM_{10} species for (left) acidic particles (measured-to-
 906 predicted ratio less than 0.75) and (right) neutralized particles (ratio between 0.85 – mean
 907 value – and 1.07 – 90th percentile).



908 (a)



909 (b)



910 (c)

911 Figure 9. Dependences on relative humidity (left) and σ_w (right) of (a) SO_2 , (b) SO_4 and (c)
 912 the particulate-to-total sulfur ratio within the industrial (I) wind sector. The data correspond to
 913 the mean (diamond), median (horizontal line), 25th and 75th percentiles (lower and upper
 914 boxes), and 10th and 90th percentiles (lower and upper whiskers).

