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Hourly composition of gas and particle phase pollutants at a central urban background site in Milan, Italy

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Abstract

A comprehensive range of gas and particle phase pollutants were sampled at 1-hour time resolution in urban background Milan during summer 2012. Measurements include several soluble inorganic aerosols (Cl^- , NO_2^- , NO_3^- , SO_4^{2-} , Ca^{2+} , K^+ , Mg^{2+} , Na^+ , NH_4^+) and gases (HCl , HNO_2 , HNO_3 , NH_3 , NO , NO_2 , O_3 , SO_2), organic, elemental and black carbon and meteorological parameters. Analysis methods used include mean diurnal pattern on weekdays and Sundays, pollution roses, bivariate polar plots and statistical models using backtrajectories. Results show how nitrous acid (HONO) was mainly formed heterogeneously at nighttime, with a dependence of its

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formation rate on NO_2 consistent with observations during the last HONO campaign in Milan in summer 1998, although since 1998 a drop in HONO levels occurred following to the decrease of its precursors. Nitrate showed two main formation mechanisms: one occurring through N_2O_5 at nighttime and leading to nitrate formation onto existing particles; another occurring both daytime and nighttime following the homogeneous reaction of ammonia gas with nitric acid gas. Air masses reaching Milan influenced nitrate formation depending on their content in ammonia and the timing of arrival. Notwithstanding the low level of SO_2 in Milan, its peaks were associated to point source emissions in the Po valley or shipping and power plant emissions SW of Milan, beyond the Apennines. A distinctive pattern for HCl was observed, featured by an afternoon peak and a morning minimum, and best correlated to atmospheric temperature, although it was not possible to identify any specific source. The ratio of primary-dominated organic carbon and elemental carbon on hourly $\text{PM}_{2.5}$ resulted 1.7. Black carbon was highly correlated to elemental carbon and the average mass absorption coefficient resulted $\text{MAC} = 13.8 \pm 0.2 \text{ m}^2 \text{ g}^{-1}$. It is noteworthy how air quality for a large metropolitan area, in a confined valley and under enduring atmospheric stability, is nonetheless influenced by sources within and outside the valley.

Keywords: $\text{PM}_{2.5}$, hourly ionic composition, gas-phase pollutants, Po valley

1. Introduction

The interactions between gaseous and aerosol phase pollutants have long been studied due to their impact on air quality (Penkett et al., 1979; Rav-

4 ishankara, 1997) and on human health (World Health Organization, 2006).
5 In order to investigate processes leading to atmospheric pollutants forma-
6 tion and ageing in densely populated areas with large emission load (i.e.
7 hotspots), time resolved composition of both gas and particle phase atmo-
8 spheric compounds are needed. Po valley (Northern Italy) is one of the
9 most important hotspot region in Europe (Putaud et al., 2010), with Milan
10 metropolitan area exhibiting one of the poorest air quality within the valley
11 (Bigi and Ghermandi, 2014).

12 Very few 1-hour time resolution campaigns were accomplished in Milan
13 urban area. Three noteworthy studies resulted from the Limitation of Ox-
14 idant Production/Pianura Padana Produzione di Ozono (LOOP/PIPAPPO)
15 campaign held in May and June 1998 in Milan urban background (Neftel
16 et al., 2002). In one of these studies Baltensperger et al. (2002) analysed
17 data by several continuous instruments sampling aerosol physical properties
18 (number size distribution, volatility, hygroscopicity, mass), aerosol chemical
19 composition (BC_E , nitrate, sulphate) and two gases (NH_3 , HNO_3). This
20 same study showed the large contribution to airborne particles smaller than
21 40 nm by primary emissions rich in soot content and the increase in secondary
22 and hygroscopic aerosol for particles larger than 50 nm. The second study
23 within PIPAPO, Putaud et al. (2002) collected samples of size-segregated
24 aerosol with 4- and 7-hour time resolution and analysed them for elemental
25 carbon (EC), organic carbon (OC), particulate organic matter (POM) and
26 major ionic species; their results showed the large contribution ($> 30\%$) to

27 PM mass by POM, by ammonium nitrate (29% of PM mass) and ammo-
28 nium sulphate (22% of PM mass). Putaud et al. (2002) found also a diurnal
29 and weekly pattern for traffic-related pollutants (e.g. EC and resuspended
30 mineral dust), the influence of traffic emissions on nitrate formation and of
31 industrial emissions on sulphate formation. In the third study of PIPAPO
32 Aliche et al. (2002) investigated hydroxyl radical formation by measuring
33 several gas phase pollutants by DOAS (Differential Optical Absorption Spec-
34 troscopy): HCHO, HONO, NO₂, NO, O₃ and SO₂. Their results identified
35 HCHO as the primary source of OH· radicals (up to 40% of total OH· on
36 clear days), while photolysis of nitrous acid and of ozone provides similar
37 contribution to atmospheric OH· (15–30% of total OH· each), with the for-
38 mer compound dominating during early morning and the latter during the
39 afternoon. In addition to the above mentioned studies, the aerosol elemental
40 composition and sources were investigated with hourly resolution in Milan
41 by D’Alessandro et al. (2003, 2004) during wintertime and summertime 2001
42 evidencing quasi-periodical and episodic pollution sources.

43 Several sources influence the sampling site: Bernardoni et al. (2011) used
44 Positive Matrix Factorization to apportion 4-hour PM₁₀ measurements and
45 showed the diurnal pattern in the relative contribution by resuspended dust,
46 construction works and industry, which altogether account for 48% to total
47 PM₁₀ in summer. These results were confirmed by the a detailed source ap-
48 portionment exercise in Milan urban background by Perrone et al. (2012),
49 where three years of daily PM_{2.5} and PM₁₀ samples were analysed. This

50 latter study also showed how 60% of summer daily $\text{PM}_{2.5}$ derives from traffic
51 and secondary inorganic ions (sulphate, nitrate and ammonium) and contri-
52 bution of resuspended dust to summer daily $\text{PM}_{2.5}$ is only to 3.8%. Consis-
53 tently emission inventory for the only municipality of Milan assessed Road
54 Traffic (SNAP 7) to be the main source of NO_x and EC for the city of Mi-
55 lan, SNAP 2 (non-industrial combustion) is the main source of OC and the
56 second most important of NO_x , and SNAP 6 (solvent use) is the main source
57 of NM – VOC. Notwithstanding these studies, in Milan there is no available
58 analysis of simultaneous characterization of atmospheric pollutants in both
59 gas and particle phase sampled at a 1-hour resolution. The present article is
60 based on a thorough analysis of measurements of several gas phase pollutants
61 and main chemical composition of $\text{PM}_{2.5}$ sampled at 1-hour time resolution.
62 Formation process of $\text{PM}_{2.5}$ in Milan will be presented along with the in-
63 fluence of meteorological conditions and air mass trajectories. Observations
64 include HCl, HONO, HNO_3 and NH_3 , i.e. the first published measurement
65 of hydrochloric acid in the Po valley and the first 1-hour resolution measure-
66 ments of nitrous and nitric acids in the last 15 years in Milan. Details on
67 the instrumentation and methods used are presented in Sect. 2. Results and
68 conclusions are found in Sects. 3 and 4 respectively.

69 **2. Data and methods**

70 Milan ($45^\circ 28' \text{N}$; $9^\circ 13' \text{E}$) urban area counts about 1 500 000 inhabitants
71 and is the second largest town in Italy, after Rome, and considering the whole

72 Milan province the population rises up to about 3.1 millions inhabitants.

73 The data here presented were collected on the roof of the Department
74 of Chemistry, University of Milan, at a ~ 10 m a.g.l. within the University
75 campus, a site representative of central urban background conditions for the
76 city. Sampling was performed from June 5th until July 23rd 2012.

77 Hourly resolution composition of $PM_{2.5}$ for soluble inorganic ions compo-
78 sition and for gases was determined using a commercially available Ambient
79 Ion Monitor (AIM) URG-9000D (URG Corp, USA). In particle phase five
80 anions (Cl^- , F^- , NO_2^- , NO_3^- , SO_4^{2-}) and five cations (Ca^{2+} , K^+ , Mg^{2+} , Na^+ ,
81 NH_4^+) were determined. F^- , Ca^{2+} , K^+ , Mg^{2+} and Na^+ were often below the
82 detection limit and therefore not analysed in details. The gases determined
83 include hydrochloric acid (HCl), nitrous acid (HONO), nitric acid (HNO_3),
84 ammonia (NH_3) and sulphur dioxide (SO_2). The AIM consists of a sampling
85 system for both gas and particles, coupled with two ion chromatographies for
86 the analytical determination. Gases are collected by a liquid diffusion denun-
87 der with H_2O_2 5 mM running continuously at $10 mL h^{-1}$ flow rate. Particles
88 are collected in a chamber supersaturated with ultrapure water vapour: wa-
89 ter soluble particles are allowed to grow and then inertially separated and
90 injected into the ion chromatographies. Further instrumental details and
91 the calibration procedure used in this study can be found in Markovic et al.
92 (2012). AIM data were compared to off-line daily data from $PM_{2.5}$ samples
93 collected during 22 days throughout the campaign by denuded filter-pack
94 setup (Vecchi et al., 2009): the system consisted in two dry annular denud-

95 ers removing both acidic and basic gases, followed by a filter pack made of a
96 quartz fibre front filter and a nylon fibre backup filter. Once the campaign
97 ended, AIM blank values for particle-phase aerosol were estimated by insert-
98 ing a quartz fibre filter between the denuder and the filter pack over 5 full
99 days.

100 Comparison of AIM and denuded filter pack showed statistically significant
101 (by ANOVA test) and large coefficients of determination for linear regression
102 models between off- and on-line data for all species (Figure S1), supporting
103 the reliability of the patterns observed by AIM. Regression coefficients were
104 close to unity for $\text{HNO}_2+\text{HNO}_3$ and NH_3 , while some difference occurred be-
105 tween particle-phases compounds, with lower $\text{NO}_2^-+\text{NO}_3^-$ levels observed by
106 AIM and lower values for SO_4^{2-} and NH_4^+ observed by denuded filter-pack.
107 Assuming that the denuded filter pack reported in Vecchi et al. (2009) is
108 artefact-free for nitrogen compounds, some bias likely affects ammonium in
109 AIM measurements, preventing a fully-correct estimate of ion balance for
110 the experimental dataset. Part of the offset between the two measurement
111 sets might also be due to the possibly different transmission efficiency curve
112 between respective size-selective inlets: AIM uses a custom $\text{PM}_{2.5}$ cyclone,
113 while denuded filter pack used a US-EPA equivalent $\text{PM}_{2.5}$ inlet equipped
114 with a PM_{10} sampling head and a WINS $\text{PM}_{2.5}$ impactor downstream.
115 Gaseous precursors levels during AIM blank test resulted similar to their re-
116 spective mean observed during the campaign, indicating an efficient collection
117 of gas and a complete transmission of particles by the denuders. Significant

118 particulate sodium was observed during blanks and ascribed to contamina-
119 tion in the ultrapure water used in the supersaturated chamber during the
120 blank test. The low particulate nitrite, nitrate and ammonium observed
121 during blank testing (i.e. $< 1 \mu\text{g m}^{-3}$) can be considered negligible. Slightly
122 larger blank for particulate sulphate was observed ($\sim 1.5 \mu\text{g m}^{-3}$), but con-
123 sidered sufficiently low to support the reliability of the AIM measurements
124 for this compound.

125 Elemental carbon (EC) and organic carbon (OC) measurements were col-
126 lected by a Model-4 Semi-Continuous ECO-C Field Analyser by Sunset Lab-
127 oratory, USA (Bae et al., 2004). The carbon analyser was provided with
128 a $\text{PM}_{2.5}$ cyclone and operated at a 24 L min^{-1} flowrate. Measurement had
129 1-hour time resolution, comprising 45 minutes for sampling and 15 minutes
130 for thermal-optical analysis. The analysis followed a high temperature ana-
131 lytical protocol featured by 2 steps in inert atmosphere (625°C , 870°C) and
132 3 steps in oxidizing atmosphere (650°C , 675°C , 870°C). During most of the
133 campaign the split point (i.e. the separation between OC and EC) occurred
134 in inert atmosphere: the pre-combustion of EC was attributed to the low
135 carbon amount and to the presence of undesired oxygen or of metal oxides
136 accumulated on the filter. The data analysis included only EC-OC measure-
137 ment having a split point above 650°C (Jung et al., 2011), in accordance
138 with EUSAAR2 protocol for off-line laboratory measurements (Cavalli et al.,
139 2010). Both calibration and filter replacement were performed on a weekly
140 basis; no EC-OC data are available on Sundays.

141 Aerosol light absorption coefficient (σ_{ap}) was measured by a Multi-Angle
142 Absorption Photometer (MAAP, model 5012, Thermo Scientific Corp., USA)
143 equipped with a PM_{2.5} inlet. Atmospheric equivalent black carbon concen-
144 tration (BC_E) is estimated by the instrument using a MAC (mass specific
145 absorption cross section) of $6.6 \text{ m}^2 \text{ g}^{-1}$ (Petzold et al., 2002). BC_E measure-
146 ments had a 5-min resolution and the data were then averaged to 1 hour. To
147 evaluate atmospheric dispersion conditions, ^{222}Rn short-lived decay products
148 measurements were performed using the experimental methodology reported
149 in Marcazzan et al. (2003). Hourly measurements of meteorological param-
150 eters (wind speed and direction, temperature, relative humidity, pressure,
151 global solar radiation and precipitation) were sampled on the roof of the De-
152 partment of Physics, University of Milan, at ~ 10 m a.g.l., within the grounds
153 of the University campus. Concentration of NO, NO₂ and O₃ were provided
154 by an urban background station sited 400 m NE of the campus and within
155 the air quality monitoring network of the Regional Environmental Protection
156 Agency of Lombardy (ARPA).

157

158 A tentative localization and identification of the pollution sources was
159 performed by four techniques: analysis of conditional bivariate probability
160 function (CBPF), bivariate polar plots, air mass origin by simulated back-
161 trajectory and backtrajectory statistical models (BSMs). The former two
162 techniques are headed to identify local sources, while backtrajectories were
163 used to potentially track pollution events due to distant sources. CBPF

164 technique estimates the probability that a specific concentration range is
165 observed within a given wind sector, depending upon wind speed (more de-
166 tails in Uria-Tellaetxe and Carslaw, 2014). Bivariate polar plots (Carslaw
167 and Ropkins, 2012) are a level plot in polar coordinates where colour scale
168 represents pollutant concentration and polar coordinates are a smoothed in-
169 terpolation of wind speed and wind direction by Generalised Additive Model
170 (Wood, 2006). Origin of air masses arriving in Milan at 100 m a.g.l. was esti-
171 mated with 36-hours long backtrajectories simulated by HYSPLIT (Draxler
172 and Rolph, 2013) using 0.5° GDAS meteorological data. 24 trajectories per
173 day were computed, i.e. one per hour. The statistical models applied to back-
174 trajectories include Concentration Field (CF), Potential Source Contribution
175 Function (PSCF) and Gridded Difference (GD) using respective functions in
176 Carslaw and Ropkins (2012). All of the three attempt to combine trajectory
177 path to concentration at the receptor; further details about BSMs can be
178 found in Carslaw and Ropkins (2012) and Fleming et al. (2012). Scheifinger
179 and Kaiser (2007) proved BSMs to provide useful information on potential
180 source areas only for distances within the mean residence time of the inves-
181 tigated compound; for trajectories longer than this period the effect of tur-
182 bulent dispersion and removal processes, neglected by the statistical models
183 and the backtrajectories, may lead to unreliable results (Han et al., 2005). In
184 the present study trajectory statistical models were tested for several species,
185 although significant results occurred only for SO_2 , SO_4^{2-} and NH_3 . For sul-
186 phur dioxide Scheifinger and Kaiser (2007) suggest to limit backtrajectories

187 within 60 hours: in this study a length of 36 hours (i.e. largely within 60
188 hours) was used for all pollutants.

189

190 Few different emission inventories were used to review BSM outputs and
191 to identify realistic potential sources: a plot pairing inventory database and
192 corresponding area is presented in Figure S2. Emission inventories for Lom-
193 bardy, Piedmont and Veneto were consistently compiled (INEMAR, 2015),
194 resulting in a bottom-up inventory with a spatial resolution at the municipal-
195 ity district level. These inventories are compliant to the EMEP-CORINAIR
196 guidebook, the IPCC guidelines and the Good Practice Guidance, they are
197 classified accordingly to SNAP (Selected Nomenclature for Air Pollution)
198 and the emission data used in this study refers to 2010 for Lombardy and
199 2008 for Piedmont and Veneto. Emissions for Aosta Valley, Liguria, Rhône
200 Alpes and Provence-Alpes-Côte d’Azur refers to 2008, have a spatial res-
201 olution at the municipality district level and include point sources for the
202 three former regions. Data were provided by the project AERA (Air Envi-
203 ronnement Regions ALCOTRA) through a Web Map Service (AERA, 2015).
204 Finally, emissions for all other regions involved in this study were provided
205 by the UNECE/EMEP emission inventory for 2012. Accuracy and uncer-
206 tainty of these inventories are different, as well as the procedure to build
207 them, their spatial resolution and their reference year. In this study inven-
208 tories were used to identify areas with significant SO₂ emissions (e.g. large
209 industrial plants) and to approximately compare potential sources, therefore

210 the discrepancies between these databases were considered negligible for the
211 purpose of this analysis. Nonetheless the closer the investigated area is to
212 Milan, the more detailed is the inventory used, allowing a reliable analysis
213 of nearby and directly impacting sources.

214 All statistical data analyses were performed within the software environ-
215 ment R 3.0.2 (R Core Team, 2013). All data are reported in local time, i.e.
216 Central European Summer Time (CEST).

217 **3. Results and discussion**

218 In the following the data will be presented and discussed in specific sub-
219 sections for each common species, including their pattern, formation and
220 removal processes (if known) and their potential source. The analysis inves-
221 tigate the average concentration pattern over the whole sampling period,
222 and the variability of a subset of pollutants under different atmospheric dis-
223 persion conditions.

224 *3.1. Meteorological setting*

225 June and July 2012 were featured by high pressure fields originated by the
226 Azores high and occasionally by subtropical anticyclones (Figure S3). Large
227 values of mean geopotential height at 500 hPa occurred during the sampling
228 campaign, when June was characterized by anticyclonic curvature and mod-
229 erately positive anomaly and July by a slightly cyclonic curvature. This
230 meteorological context determined recurrent atmospheric stability, clear-sky

231 and high temperatures, enhancing photochemical activity and formation of
232 tropospheric ozone. Hot weather conditions occurred between the end of
233 June and the beginning of July and, after a short instability period, since
234 July 8th. Main meteorological events reducing atmospheric pollutants levels
235 occurred at the beginning of June prompted by moderately perturbed fluxes
236 and Atlantic cold fronts, on the 14th and 15th of July due to ventilation orig-
237 inated by a temporary positioning in the North of Alps of the maximum of
238 the Azores high, and on July 21st when a cold front led to intense rainstorms
239 in Milan and surroundings. A mountain-valley breeze regime was present
240 during large part of the campaign, with low N winds at night ($< 2 \text{ m s}^{-1}$),
241 increasing and rotating at daytime until a SW flow is established in the af-
242 ternoon with speeds up to 4 m s^{-1} . These meteorological conditions were
243 typical of summers in Northern Italy, therefore notwithstanding measure-
244 ments did not cover the whole summer, they are well representative of the
245 entire season. A summary of statistical values for the measured parameters
246 is presented in table B.1. In Figure B.1 the diurnal pattern during weekdays
247 and Sundays are presented for several observed pollutants, along with tem-
248 perature (T), Radon concentration (Rn), global radiation (GR) and relative
249 humidity (RH).

250 *3.2. Nitrogen oxides and ozone*

251 Nitric oxide shows sharp peaks at mornings only on weekdays, indicating
252 a source from rush hour traffic, along with relatively low atmospheric mixing;

253 NO rapidly declines due to vertical mixing and its oxidation to NO_2 by O_3
254 and by the available radical groups.

255 Nitrogen dioxide peaks at morning rush hours. The afternoon decrease
256 in NO_2 is driven by 3 processes: dilution due to the rise in mixing layer
257 height, photolysis to NO under intense solar radiation and reaction with $\text{OH}\cdot$
258 to nitric acid. The difference between weekdays and Sundays is noteworthy
259 for both NO and NO_2 , strongly indicating an anthropogenic source, most
260 likely traffic, of these compounds.

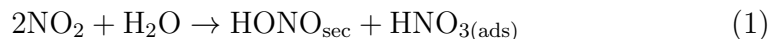
261 Ozone shows an afternoon peak generated by the large availability of
262 $\text{OH}\cdot$ radicals and possibly by the re-entrainment of O_3 within the residual
263 layer (Zhang and Trivikrama Rao, 1999). Indeed the sum of NO_2 and O_3
264 shows a slightly higher peak on weekdays indicating the small contribution
265 by fresh emissions to oxidant species. Ozone exhibits a very mild weekend
266 effect (Cleveland et al., 1974; Jiménez et al., 2005; Tonse et al., 2008; Pollack
267 et al., 2012; Wang et al., 2014), e.g. higher mixing ratios on Sundays due
268 to lower NO_x emissions over the weekend and therefore reduced titration
269 of O_3 , as shown for Milan by Vecchi and Valli (1999). Since 1700 CEST
270 mixing decreases leading to an increase in NO_x and a drop in ozone. Diurnal
271 pattern for NO_x and O_3 are highly similar to mean diurnal pattern for long
272 term measurements in Modena, 160 km SE of Milan, Po valley (Bigi et al.,
273 2012), where ozone exhibits also similar levels to those observed in Milan,
274 consistently with the atmospheric homogeneity across the valley.

275 *3.3. Nitrous acid and nitrites*

276 Gas-phase nitrous acid (HONO) shows a pattern similar to past obser-
277 vations in Milan by Stutz et al. (2002) and Febo et al. (1996), featured by
278 lower mixing ratios at daytime, due to photolysis, and peak at nighttime,
279 most likely by heterogeneous formation. Particle-phase nitrites (NO_2^-) ex-
280 hibit a diurnal pattern highly similar to HONO, as observed in Marseille
281 (Acker et al., 2005), suggesting that part of atmospheric HONO is formed
282 heterogeneously onto particle surfaces under high RH conditions. Traffic
283 emissions are another possible source of HONO: Kurtenbach et al. (2001)
284 found large variability in the ratio HONO/NO_x from road tunnel measure-
285 ments depending on fuel, emission control systems and motor load. A ratio
286 for HONO/NO_x of 1% was used in this study, similarly to the ratio used
287 by Michoud et al. (2014) for Paris and higher than 0.65% as used by Stutz
288 et al. (2002) for Milan. In order to remove HONO direct emissions from total
289 HONO, secondary HONO was estimated as $\text{HONO}_{\text{sec}} = \text{HONO} - 0.01 \cdot \text{NO}_x$,
290 resulting in a $\sim 80\%$ of total HONO from secondary origin.

291 Main formation mechanism of HONO_{sec} in urban atmosphere is expected
292 to proceed heterogeneously onto surfaces, following reactions 1 and 2, with
293 1 being the most likely process (Kleffmann, 2007). It is still unclear whether
294 the contribution of heterogeneous formation of HONO_{sec} onto soot particles
295 (Kalberer et al., 1999) is a significant source to atmospheric concentrations
296 (Kleffmann, 2007; Ziemba et al., 2010) or not (Finlayson-Pitts and Pitts,

297 2000).



298



299 Latest published measurements of nitrous acid in Milan urban background
300 were collected in May and June 1998 by Alicke et al. (2002) and Stutz et al.
301 (2002). The latter investigated HONO formation by DOAS-resolved vertical
302 profiles and found that HONO_{sec} was formed at ground. Stutz et al. (2002)
303 estimated that 1 molecule of HONO_{sec} was released each 25–40 molecules of
304 NO_2 and assessed the NO_2 conversion efficiency to be an order of magnitude
305 lower than expected according to reaction 1.

306 In Alicke et al. (2002) HONO mixing ratios by DOAS exhibited a night-
307 time and daytime mean of 0.92 ppb and 0.14 ppb respectively. In summer
308 2012 the mean mixing ratios observed were of 0.6 ppb at nighttime and
309 0.5 ppb at daytime, with the latter being largely higher than in 1998 and
310 potentially biased by artefacts. Maximum nitrous acid levels were similar in
311 both studies: 4.4 ppb and 4.2 ppb in 1998 and 2012 respectively.

312 In order to assess whether a long term trend in HONO is present and
313 2012 daytime data are artefact-freefall, an analysis of NO_2 levels from 1998
314 through 2012 was performed. Mean daytime NO_2 concentration during the
315 two sampling campaigns were 18.3 ppb and 4.9 ppb in 1998 and 2012 re-
316 spectively, while mean nighttime NO_2 concentration was 33.2 ppb in 1998
317 and 8.8 ppb in 2012. These observations are consistent with an estimated

318 long term trends for deseasonalized monthly mean concentration of NO_2 of
319 $-1.47 \pm 0.17 \text{ ppb yr}^{-1}$ (trends are estimated by Generalised Least Squares
320 method as in Bigi and Ghermandi (2014)). Given the large NO_2 concentra-
321 tions in 1998, Stutz et al. (2002) found a large contribution of direct emissions
322 to atmospheric HONO, notwithstanding they made use of a low HONO/ NO_x
323 ratio, i.e. 0.65%. The large decrease in NO_2 from 1998 to 2012 should cause
324 a significant decrease in HONO over the same period, although, according to
325 2012 HONO data, a decrease occurred only at nighttime. Several intercom-
326 parison studies of HONO measurements by DOAS and chemical instruments
327 (e.g. wet denuder and LOPAP) showed how results from different techniques
328 agreed well at nighttime, while HONO by standard wet denuders might suf-
329 fer from positive artefacts at daytime (Kleffmann et al., 2006): these might
330 occur by reaction on denuder surface either of semivolatile diesel exhausts
331 with NO_2 (Gutzwiller et al., 2002), either of pure NO_2 (Kleffmann et al.,
332 2006), either of NO_2 and S(IV) (Spindler et al., 2003). The high HONO and
333 $\text{HONO}_{\text{sec}}/\text{NO}_2$ ratio in 2012 at daytime suggest that 2012 daytime HONO
334 mixing ratios might be biased by a chemical interference in the denuder, since
335 are not consistent with the observed decrease in NO_2 from 1998 to 2012 and
336 the corresponding decrease in nighttime HONO over the same period.

337 Assuming that during nighttime most of nitrous acid is formed through
338 reaction 1, which is first order in NO_2 , formation rate between two generic

339 instants t_1 and t_2 was computed according to equation 3.

$$\overline{F_{HONO_{sec}}} = \frac{[HONO_{sec}](t_2) - [HONO_{sec}](t_1)}{(t_2 - t_1)[NO_2]_{night}} \quad (3)$$

340 Formation rate is best estimated for nights when NO and NO₂ levels are
341 low, in order to consider negligible both direct HONO emissions by traffic and
342 positive artefacts, and when HONO formation is steady and lasts throughout
343 the night, in order to average over several hours. These conditions are met
344 between June 18th–19th, since NO and NO₂ concentration were steadily below
345 5 ppb and 15 ppb (Figure B.2), and lead to a formation rate of 0.009 ppb
346 HONO_{sec}/ (h (ppb NO₂)), consistently with the rate of (0.012±0.005) found
347 by Aliche et al. (2002) in Milan by DOAS.

348 3.4. Nitric acid, nitrates and ammonia

349 Nitric acid is formed by reaction of OH· with NO₂ leading to HNO₃
350 characteristic pattern featured by a single afternoon peak, when hydroxyl
351 radicals are abundant. Nitric acid formation is expected to be controlled
352 by the availability of OH· rather than by NO₂, since HNO₃ diurnal pattern
353 exhibits no difference between weekdays and Sundays, contrarily to NO₂,
354 whose peak is above 12 ppb on weekdays and ~5 ppb on Sundays.

355 Atmospheric particle nitrate is expected to be formed by two main path-
356 ways, either through reaction of HNO₃ with NH₃ or through absorption of
357 N₂O_{5(g)} (Lammel and Cape, 1996; Finlayson-Pitts and Pitts, 2000): the for-
358 mer reaction is homogeneous, the latter is heterogeneous and occurring only

359 at slow rates and at nighttime, since N_2O_5 and NO_3 photolyse very rapidly
360 (see Appendix A). The heterogeneous pathway is expected to form signifi-
361 cant amount of aerosol nitrate: Alexander et al. (2009) estimated a similar
362 contribution by the two pathways on an annual basis in Italy, while Michalski
363 et al. (2003) estimated the heterogeneous pathway to contribute up to 50%
364 to total summer nitrate in La Jolla (Southern California).

365 Diurnal pattern of nitrate shows an increase since 0200 to 1000 CEST dur-
366 ing weekdays, whereas on Sundays exhibits steady concentration at mornings
367 and an increase since 0900 to 1400 CEST. Afternoon patterns do not differ
368 both in shape and levels between weekdays and Sundays. Two methods were
369 used to investigate partitioning of ammonium nitrate with its precursors
370 at thermodynamic equilibrium: ISORROPIA 2.1 (Fountoukis and Nenes,
371 2007) simulation model (forward mode and thermodynamically stable state
372 conditions) and dissociation constants for reaction A.2 of ammonium nitrate
373 aerosol assuming a single-component aerosol (see Appendix B for details and
374 calculations). Both methods show how patterns of atmospheric nitrate and
375 theoretical partitioning do not fully match (Figure S4), differently to other
376 continental climate sites in summer (e.g. Melpitz, Germany (Poulain et al.,
377 2011)), suggesting that condensation/evaporation processes do not exclu-
378 sively control aerosol nitrate.

379 At nighttime N_2O_5 is expected to be partly responsible of the nitrate
380 increase, also suggested by the slower formation rate compared to the one
381 required by the thermodynamic equilibrium of the homogeneous reaction ac-

382 cording to ISORROPIA (not shown). The nitrate increase after sunrise is
383 possibly kept up by homogeneous reaction A.2, triggered by the rapid forma-
384 tion of nitric acid from NO_2 rush hour emissions: since there is no evidence of
385 nitric acid rise until 0900–1000, possibly this early morning HNO_3 is rapidly
386 converted to nitrate. On Sundays nitrate increases from 0900–1000, i.e. most
387 likely through the diurnal homogeneous reaction: the lower NO_2 levels over
388 the weekend (Saturday and Sunday) lead to a smaller production of early
389 morning HNO_3 and nighttime N_2O_5 . Nitrate decreases during afternoons
390 due to temperature increase and the shift of ammonium nitrate equilibrium
391 to the gas-phase. Occasional morning increase in nitrate might be due also
392 to re-entrainment of nitrate within the residual layer, which has been shown
393 to provide a significant contribution to ground level concentration in Milan
394 in summer (Curci et al., 2015).

395 Diurnal pattern for ammonia shows a steady profile. This is due to sev-
396 eral reasons: total $\text{NH}_3 + \text{NH}_4^+$ is mainly in gasphase ammonia, therefore the
397 particulate phase is not significantly affecting NH_3 concentration. Variability
398 in aerosol nitrate diurnal pattern is almost negligible compared to ammonia
399 levels, therefore no significant amount of ammonia is expected to be released
400 in the afternoon when ammonium nitrate equilibrium moves towards gas-
401 phase. Besides, in the afternoon HNO_3 increases, leading to ammonium ni-
402 trate formation partly compensating the dissociation of ammonium nitrate.
403 Sulphate shows quite constant concentration and no significant change in
404 NH_3 is expected from ammonium sulphate. Finally NH_3 emissions might

405 be slightly larger during daytime, but compensated by dilution from the
406 enhanced boundary layer.

407 CBPF for nitrate shows how peak concentration of this compound is as-
408 sociated to very low wind speeds, i.e. is formed locally and concentrations
409 increase due to the low dispersion conditions. On the contrary, median to
410 moderate nitrate concentrations are associated to S or SSE winds, i.e. winds
411 associated to moderate–large ammonia content. CBPF for ammonia indi-
412 cates that larger observed levels are associated to distant sources at S, SSE,
413 E and W of Milan, most likely emissions from agricultural activities in the
414 Po valley surrounding the city. The existence of farther sources of ammo-
415 nia were investigated also by BSMs (see Figure S5), which confirm emissions
416 within the Po valley as main responsible for ammonia levels in Milan. This is
417 consistent with 2010 emission inventories, ascribing 97% of all ammonia emis-
418 sions to agricultural activities, and the remainder to traffic and wastewater
419 treatment (Bigi and Ghermandi, under review).

420 During the campaign, significant pollution episodes occurred, both for
421 particulate nitrate (June 8th–9th) and ammonia (June 27th–28th). The former
422 occurred during a 48–hours period of relatively low temperature featured by a
423 blocked atmosphere and an elevated inversion (Figure S6), the latter occurred
424 during a rapid transport of ammonia originated in the Po valley (Figure S7).

425 *3.5. Hydrochloric acid*

426 Known anthropogenic sources of HCl are coal combustion (Lightowers
427 and Cape, 1988), biomass burning and waste incineration (Kaneyasu et al.,
428 1999), whereas main natural source of HCl is sea-salt (Eldering et al., 1991).
429 Few measurements of atmospheric HCl mixing ratios are reported in the
430 literature and the data here presented are the first published measurements in
431 Milan, to authors knowledge. Mean HCl level during the campaign resulted in
432 0.19 ppb. In summer 1999 Bari et al. (2003) observed 0.32 ppb and 0.28 ppb of
433 HCl in two areas within New York City and attributed it to sea-salt. Eldering
434 et al. (1991) observed HCl mixing ratios up to 3 ppb in Southern California in
435 summer 1986 and demonstrated its formation by sea-salt chloride depletion
436 by attack of HNO_3 . Kaneyasu et al. (1999) showed the contribution of waste
437 incineration to both atmospheric HCl and aerosol chloride during winter
438 1991 in the Japanese Kanto Plain, where chloride ranged between $10 \mu\text{g m}^{-3}$
439 and over $70 \mu\text{g m}^{-3}$. The study by Kaneyasu et al. (1999) showed how HCl
440 emissions combined with NH_3 to $\text{NH}_4\text{Cl}_{(p)}$, which finally dissociate to gaseous
441 HCl and NH_3 at daytime with warmer atmospheric temperatures.

442 Observed HCl in Milan exhibits no weekly pattern, but a clear diurnal
443 pattern identically repeated throughout the week and best correlated to at-
444 mospheric temperature (see Figure B.1). This outcome was confirmed by
445 a bivariate polar plot analysis conditioning concentration on atmospheric
446 temperature instead of wind speed, indicating that higher HCl values are
447 associated to higher temperature and do not depend on wind direction (Fig-

448 ure B.3). Atmospheric temperature is higher in the afternoons, when a SW
449 breeze is established, explaining why in Figure B.3 larger HCl is associated
450 to SW winds. The presence of a significant distant source (e.g. sea) for HCl
451 was checked by BSMs, which pointed to a near source, excluding long range
452 transport (see Figure S8). Similarly, the analysis of sodium chloride bal-
453 ance, considering both aerosol and gas phase, resulted in a large excess in
454 Chlorine. These outcomes suggest that HCl is originated by a chlorinated
455 compound free to evaporate in the atmosphere from a non-point source, and
456 sufficiently abundant to generate significant atmospheric HCl levels. Unfor-
457 tunately available data are not sufficient to explain this excess in Chloride
458 and further samplings are needed to unveil the origin of HCl.

459 3.6. Sulphur dioxide and sulphates

460 SO₂ concentration in Milan are similar to other European cities (see
461 Henschel et al., 2013, for an European overview) and its diurnal pattern,
462 along with that of sulphate, shows no distinctive features, besides slightly
463 lower concentrations on Sundays. Few stationary sources of SO₂ have signifi-
464 cant emissions and are sufficiently near Milan to directly impact the city:
465 the largest is an industrial area including a refinery and a carbon black
466 manufacture sited 40 km West of the sampling site and emitting altogether
467 $\sim 6\,900\text{ Mg yr}^{-1}$ of SO₂ (E1 in Figure B.4). Other significant SO₂ sources
468 are: a refinery sited 50 km South-West of the sampling site, with an esti-
469 mated SO₂ emission of $4\,500\text{ Mg yr}^{-1}$ (E2 in Figure B.4) and a refinery sited

470 130 km ESE (annual SO₂ emission of 1 600 Mg) (E3 in Figure B.4).
471 CBPF, bivariate polar plots and trajectory statistical models were used to
472 identify whether any of the above potential source is impacting the sampling
473 site. CBPF analysis associates low SO₂ levels (0.0–0.5 ppb, i.e. below the
474 20th quantile) to Northerly winds (CBPF plots for SO₂ are in Figure S9).
475 Moderate SO₂ concentrations (20th–70th quantile) are associated to a South-
476 ern source (possibly E1 and/or Genoa) and a Western source (possibly E2).
477 Highest concentrations are associated to a source SW of the site, most likely
478 E1. Consistently bivariate polar plot for SO₂ associates mean levels to W
479 and SW winds (Figure B.5e).
480 Potential sources of long-range transported SO₂ identified by BSMs were
481 nearby Venice (250 km E of Milan) and in the Ligurian Sea/Ligurian shore
482 (Figure S10). At the former a large refinery and a power plant are present;
483 at the latter SO₂ emissions are expected by ship traffic and by a power plant
484 (E4 in Figure B.4) which is partly coal fired. Finally some contribution could
485 occur by sources nearby Marseille (E5 in Figure B.4), where an overall SO₂
486 emissions in the order of 50 000 Mg yr⁻¹ is expected.

487 Regarding sulphate, CBPF associates its lowest concentration to Northerly
488 winds (see Figure S11), similarly to SO₂. Low sulphate concentration is as-
489 sociated to low Eastern winds, possibly aged emissions from Venice or E3.
490 Median sulphate levels (SO₄²⁻ ranging between 4.1–5.4 μg m⁻³, i.e. between
491 40th–60th quantile) are associated to moderate SW winds, suggesting a dis-
492 tant origin, eventually E4. Peaks in sulphate are associated to low SW winds

493 and possibly originate from E1 emissions. Backtrajectories models for sul-
494 phate suggest possible sources similar to SO₂ (Figure S12): all three statisti-
495 cal models indicate a source in the Ligurian sea, most likely maritime traffic
496 and E4, a source nearby Venice and possibly some contribution from the area
497 of Marseille. BSMs results consider transport from closer sources, within 36
498 hours travel distance: SO₂ oxidation rate for coal-fired power plant plume in
499 summer was estimated as $\sim 1\% \text{ h}^{-1}$ (Richards et al., 1981). Therefore the
500 presence of other (e.g. more distant) sources of sulphate impacting Milan is
501 possible, although their identification would require backtrajectories longer
502 than 36 hours, better if associated to a particle dispersion model in order to
503 provide reliable results for a BSMs (Han et al., 2005, e.g.) and beyond the
504 aims of this study.

505 *3.7. Elemental, organic and black carbon*

506 EC exhibits a typical diurnal pattern of primary pollutants on weekdays,
507 with a peak at 0800 CEST and a minimum when atmospheric mixing is high-
508 est. OC diurnal pattern on weekdays is featured by a slight afternoon increase
509 due to secondary aerosol formation. The diurnal pattern for OC/EC ratio
510 during weekdays, an index of secondary organic carbon formation, shows a
511 minimum during primary emissions and a maximum during intense photo-
512 chemical activity. No EC and OC data are available for Sundays. Following
513 the rationale of Cabada et al. (2004), the EC tracer method was used to
514 estimate secondary-influenced and primary-dominated OC, OC_s and OC_p

515 respectively. Hours with lower photochemical activity were selected upon
516 ozone levels: for each day the 25th quantile of Ozone was computed and
517 only OC EC data during hours with Ozone below this threshold have been
518 used to estimate the OC_p to EC regression. Some limitations apply to this
519 simplistic rationale for the Po valley, where large levels of SOA are expected
520 even at low ozone concentration. The primary-dominated OC to EC linear
521 fit resulted in a slope of 1.7 and an intercept of $2.9 \mu\text{g m}^{-3}$ (Figure B.6),
522 similarly to the ratio of 1.7 found by Lonati et al. (2007) for summer 2002
523 and 2003 on daily $PM_{2.5}$ samples in Milan. The intercept most likely includes
524 sampling artefacts, non-combustion OC emissions (e.g. biogenic), meatcook-
525 ing operations and background OC. These coefficients were used to estimate
526 secondary organic aerosol (SOA) concentration and finally CBPF was used
527 to estimate potential transport. Results show how peak SOA values are asso-
528 ciated to winds from the SW sector, i.e. to afternoon winds with maximum
529 solar radiation and photochemical activity. Median concentration of SOA
530 are associated to Easterly winds, most likely transported from OC emissions
531 within the Po valley.

532 Absorption coefficient by MAAP is well correlated to EC, leading to an
533 average mass absorption coefficient $MAC = 13.8 \pm 0.2 \text{ m}^2\text{g}^{-1}$, in fair agree-
534 ment to previous studies on off-line quartz fibre filters sampled in Milan
535 (Vecchi et al., 2012). MAC is fairly constant throughout the day, with a day-
536 time $MAC_{\text{day}} = 13.5 \pm 0.3 \text{ m}^2\text{g}^{-1}$ and night-time $MAC_{\text{night}} = 13.9 \pm 0.3 \text{ m}^2\text{g}^{-1}$
537 (Figure S13), indicating the same EC source, most likely traffic.

538 BC_E and EC, along with nitric oxide, show similar bivariate polar plots
539 indicating a nearby main source sited SSE, along the direction of the nearest
540 major road (see Figure B.5b for BC_E). CBPF associates lower concentration
541 in BC_E and EC to SW winds, because of the dispersion induced by the stronger
542 SW afternoon winds.

543 **4. Conclusions**

544 The article presented the most recent and complete analysis of 1-hour
545 resolution observations of gaseous pollutants and main chemical composition
546 of $PM_{2.5}$ in Milan. Ozone and nitrogen oxides pattern are consistent with
547 intense photochemical activity under strong solar radiation, heavy emission
548 sources and recirculation of pollutants, along with re-entrainment from the
549 residual layer. HONO mixing ratios, compared to 1998, exhibited a de-
550 crease, consistently with the reduction in NO_2 atmospheric concentration,
551 although with a similar formation rate. Particulate nitrate formed through
552 two pathways, depending upon the meteorological conditions and air mass
553 origin. Backtrajectories and pollution rose models attributed sulphur dioxide
554 and sulphate in Milan to emission sources in the Po valley, in the Ligurian
555 coast and Ligurian sea. Steady high levels of ammonia have been observed
556 throughout the campaign, originated mostly by agricultural emissions within
557 the whole Po valley. A distinct pattern was observed for hydrochloric acid:
558 several potential sources were investigated and results hints to evaporation
559 of a chlorinated compound, although further studies are needed to provide a

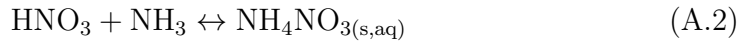
560 clear answer.

561 The atmospheric stability conditions enduring during the campaign al-
562 lowed to investigate in details processing and ageing of several gas phase
563 and aerosol pollutants. Overall variability has been checked by hierarchi-
564 cal cluster analysis, with a distance matrix based on Pearson's correlation
565 coefficient (Bigi and Ghermandi, 2014) and grouping driven by a divisive
566 algorithm (Kaufman and Rousseeuw, 1990). Results showed a strong cor-
567 relation among most pollutants and a modest cluster structure, featured by
568 two groups (Figure S14): the first group includes long range transported
569 pollutants (e.g. SO_2 , SO_4^{2-}) and compounds strongly correlated to radia-
570 tion and temperature. The second group includes locally emitted and lo-
571 cally formed pollutants, e.g. BC and NO_3^- . Implications of these results
572 for local air quality policies are several: a large metropolitan area, although
573 sited in a confined valley and under long-lasting atmospheric stability, can
574 have air quality significantly affected by long-range transported pollutants.
575 Moreover concentration variability of locally emitted pollutants in this same
576 metropolitan area relies on the variability of overall emissions within the val-
577 ley, therefore more attention needs to be paid to wide emissions, e.g. organic
578 aerosol and ammonia. Conventional air quality policies aim to decrease con-
579 centration of regulatory pollutants (e.g. NO_2): we showed how their outcome
580 controls the variation of less frequently monitored pollutants (e.g. HONO),
581 whose influence on local air quality and climate is large and neglected by
582 policymakers. Po valley plume stretches towards several European regions

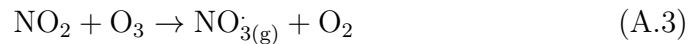
583 surrounding Italy on the East and South (Finardi et al., 2014), including
 584 large parts of the Mediterranean sea, contributing to acidification and eu-
 585 trophication (Jalkanen et al., 2000; Im et al., 2013). Study outlooks include
 586 a source apportionment analysis and deeper investigation of carbonaceous
 587 aerosol and secondary organic formation events.

588 **Appendix A. Nitrate formation pathways**

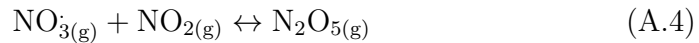
589 Formation of nitrate through homogeneous reaction pathway



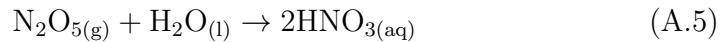
590 Formation of nitrate through heterogeneous reaction pathway with night-
 591 time formation of nitrate radical NO_3 , its reaction to dinitrogen pentoxide
 592 N_2O_5 , the hydrolysis of N_2O_5 on surfaces releasing $\text{HNO}_{3(\text{aq})}$ which is finally
 593 hydrolysed to nitrate, similarly to the reaction A.2.



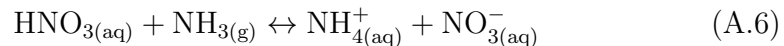
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596



597 **Appendix B. Calculation of equilibrium constants for ammonium**
 598 **nitrate**

599 K_p and K_p^* are the dissociation constants for solid phase and aqueous
 600 ammonium nitrate respectively, estimated according to Mozurkewich (1993).
 601 K_{AN} is the equilibrium constant for reaction A.2, estimated according to
 602 Seinfeld and Pandis (2006) and Poulain et al. (2011).

603 Deliquescence relative humidity for ammonium nitrate (Seinfeld and Pan-
 604 dis, 2006):

$$\ln(\text{DRH}) = \frac{723.7}{T} + 1.6954 \quad (\text{B.1})$$

605 Dissociation constant K_p for solid ammonium nitrate formed through
 606 reaction A.2 (Mozurkewich, 1993):

$$\ln(K_p) = 118.87 - \frac{24084}{T} - 6.025 \ln(T) \quad (\text{B.2})$$

607 Dissociation constant K_p^* for aqueous ammonium nitrate formed through
 608 reaction A.2 (Mozurkewich, 1993):

$$K_p^* = (P_1 - P_2(1 - a_w) + P_3(1 - a_w)^2) \cdot (1 - a_w)^{1.75} \cdot K_p \quad (\text{B.3})$$

609 with

$$\ln(P_1) = -135.94 + \frac{8763}{T} + 19.12 \ln(T) \quad (\text{B.4})$$

610

$$\ln(P_2) = -122.65 + \frac{9969}{T} + 16.22 \ln(T) \quad (\text{B.5})$$

611

$$\ln(P_1) = -182.61 + \frac{13875}{T} + 24.46 \ln(T) \quad (\text{B.6})$$

612

and water activity a_w approximated by RH expressed in the range 0–1.

613

Equilibrium constant K_{AN} of ammonium nitrate at RH above that of deli-

614

quescence ammonium nitrate, in ppb² from Poulain et al. (2011).

$$K_{AN} = k(298) \exp \left\{ a \left(\frac{298}{T} - 1 \right) + b \left[1 + \ln \left(\frac{298}{T} \right) - \frac{298}{T} \right] \right\} \cdot 10^{-18} \quad (\text{B.7})$$

615

with $k(298) = 3.35 \cdot 10^{-16} \text{ atm}^{-2}$, $a = 75.11$, $b = -13.5$

616

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617

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618

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Table B.1: Statistical values of gas, particles and meteorological parameters during the whole campaign. Note: for global radiation the range of maximum diurnal is indicated; for precipitation the total rainfall and the maximum intensity are indicated.

Parameter	Mean $\pm \sigma$	Median
Nitric oxide (NO), ppb	4.58 \pm 2.66	4.00
Nitrogen dioxide (NO ₂), ppb	6.41 \pm 6.46	4.18
Nitrogen oxides as NO ₂ (NO _x), ppb	10.99 \pm 8.10	8.00
Ozone (O ₃), ppb	41.73 \pm 19.73	41.10
Organic carbon (OC), $\mu\text{g m}^{-3}$	6.16 \pm 4.53	4.42
Elemental carbon (EC), $\mu\text{g m}^{-3}$	0.70 \pm 0.61	0.62
Equivalent black carbon (BC _E), $\mu\text{g m}^{-3}$	1.59 \pm 1.16	1.25
Hydrochloric acid (HCl), ppb	0.19 \pm 0.20	0.12
Ammonia (NH ₃), ppb	14.40 \pm 5.25	13.18
Nitrous acid (HONO), ppb	0.55 \pm 0.34	0.49
Nitric acid (HNO ₃), ppb	0.71 \pm 0.51	0.60
Sulphur dioxide (SO ₂), ppb	0.92 \pm 0.50	0.79
Calcium ion (Ca ²⁺), $\mu\text{g m}^{-3}$	0.27 \pm 0.21	0.20
Chloride ion (Cl ⁻), $\mu\text{g m}^{-3}$	0.10 \pm 0.21	0.10
Potassium ion (K ⁺), $\mu\text{g m}^{-3}$	0.08 \pm 0.18	0.10
Magnesium ion (Mg ²⁺), $\mu\text{g m}^{-3}$	0.04 \pm 0.10	0.00
Sodium ion (Na ⁺), $\mu\text{g m}^{-3}$	0.67 \pm 0.79	0.30
Ammonium ion (NH ₄ ⁺), $\mu\text{g m}^{-3}$	5.63 \pm 3.40	5.00
Nitrite ion (NO ₂ ⁻), $\mu\text{g m}^{-3}$	1.01 \pm 0.58	0.90
Nitrate ion (NO ₃ ⁻), $\mu\text{g m}^{-3}$	2.72 \pm 3.15	1.80
Sulphate ion (SO ₄ ²⁻), $\mu\text{g m}^{-3}$	4.82 \pm 2.22	4.60
Atmospheric pressure (p), hPa	996.6 \pm 3.9	996.3
Atmospheric temperature (T), C	26.0 \pm 4.5	25.8
Global radiation (GR), W m ⁻²	396.20–909.67	
Precipitation (r), mm–mm h ⁻¹	134.8–29.4	
Radon (Rn), Bq m ⁻³	6.1 \pm 4.0	4.8
Relative humidity (RH), %	51.9 \pm 16.0	50.4
Wind speed (W), m s ⁻¹	1.2 \pm 0.9	1.1

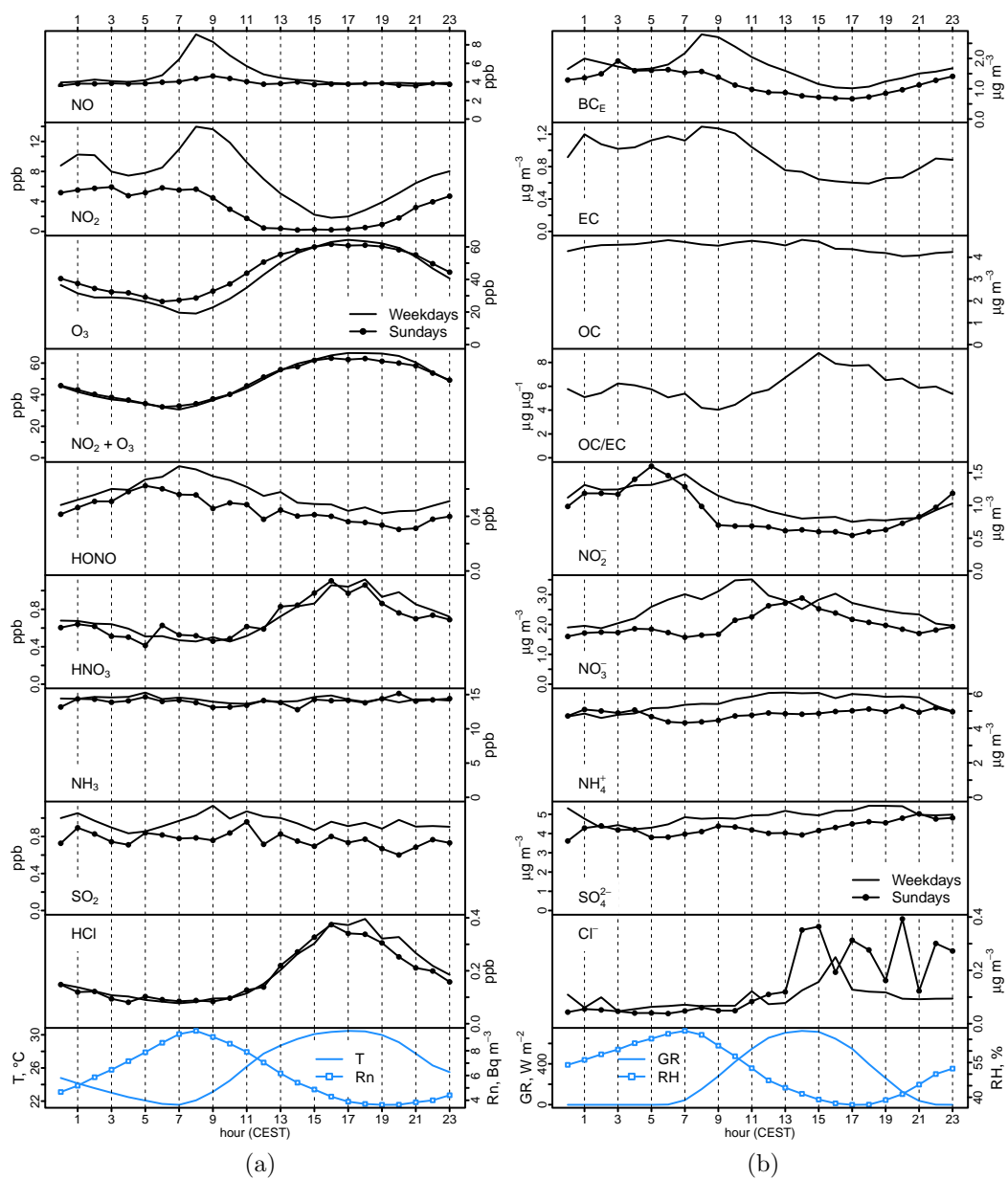


Figure B.1: Diurnal pattern for weekdays and Sundays for several gas phase (panel a) and particle phase (panel b) pollutants. Diurnal pattern of few meteorological variables and parameters are also included: temperature (T), Radon concentration (Rn), global radiation (GR) and relative humidity (RH).

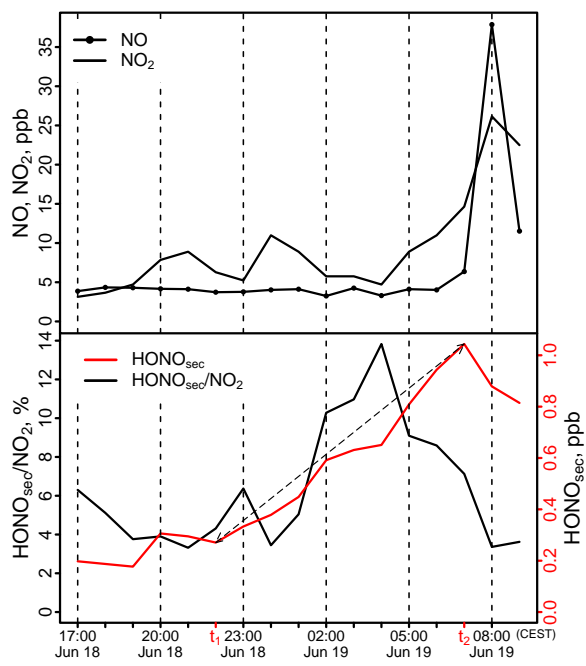


Figure B.2: Formation of HONO_{sec} during the night between June 18th–19th 2012. The arrows indicate the data used to calculate first-order formation rate.

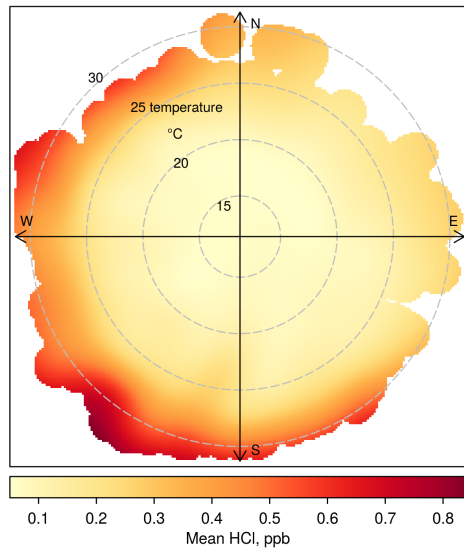


Figure B.3: Bivariate polar plots for hydrochloric acid with atmospheric temperature ($^{\circ}\text{C}$) as radial scale.

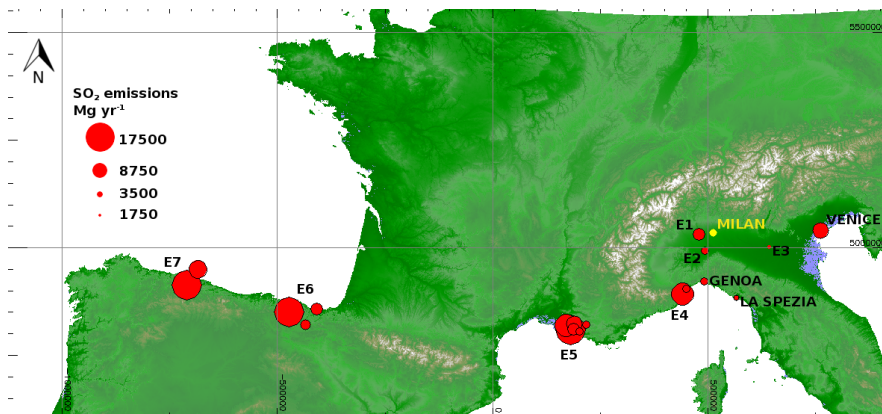


Figure B.4: Annual SO_2 emissions from point sources which potentially impacted on sulphur dioxide and sulphate ambient concentration in Milan in summer 2012 (DEM provided by Jarvis et al. (2008)).

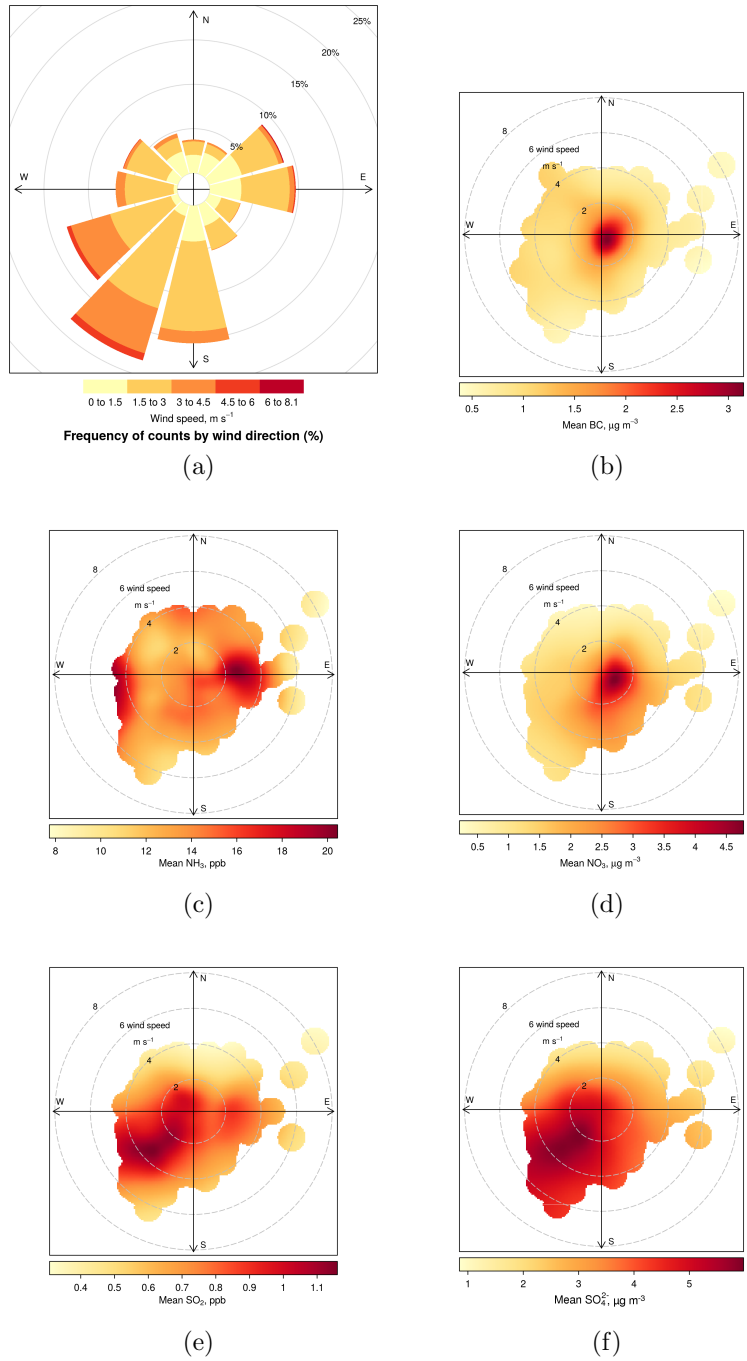


Figure B.5: Wind rose and bivariate polar plot of BC_E , NH_3 , NO_3^- , SO_2 and SO_4^{2-} ; colour codes correspond to concentration for each direction; circles correspond to wind speed.

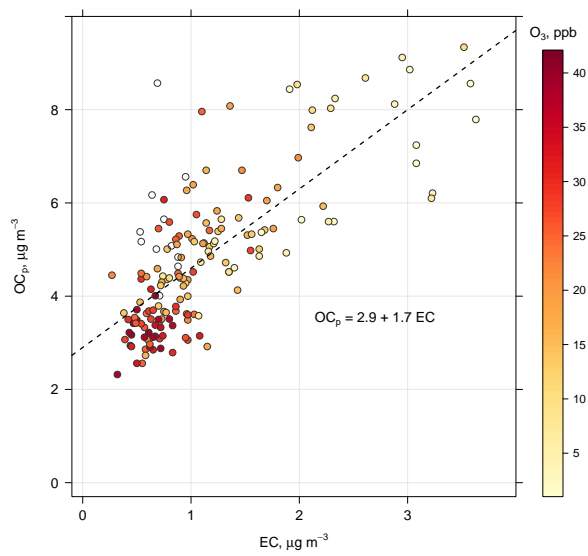


Figure B.6: Scatter plot of primary-dominated OC (OC_p) vs EC data, colour-coded according to O_3 levels. Dashed line represents the linear regression model fit for the estimate of the OC_p/EC ratio.

Supplementary Material

[Click here to download Supplementary Interactive Plot Data \(CSV\): supplementary_material_rev.pdf](#)

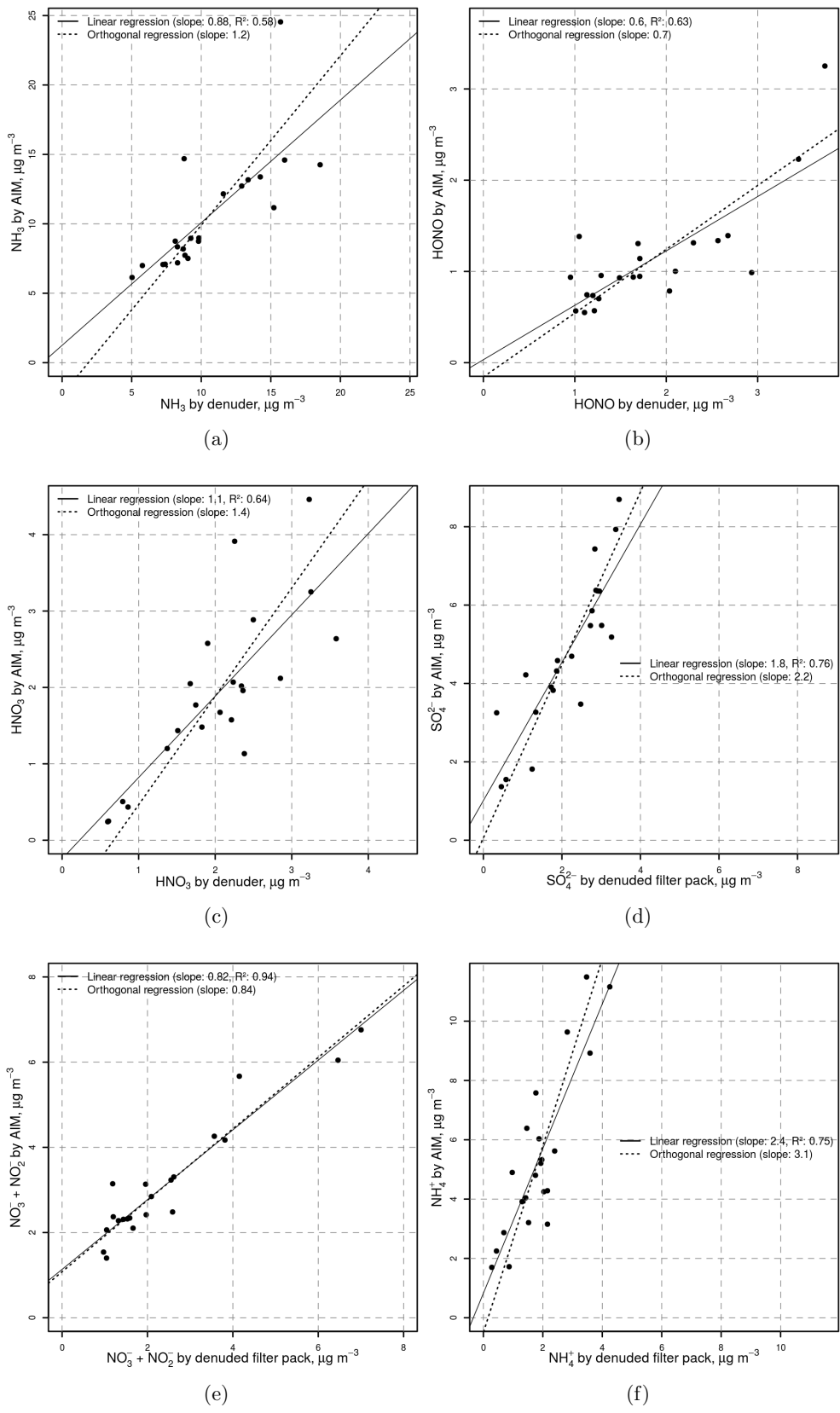


Figure S1: Scatterplot and regression models between on-line and off-line data by AIM data and denuded filter pack respectively.

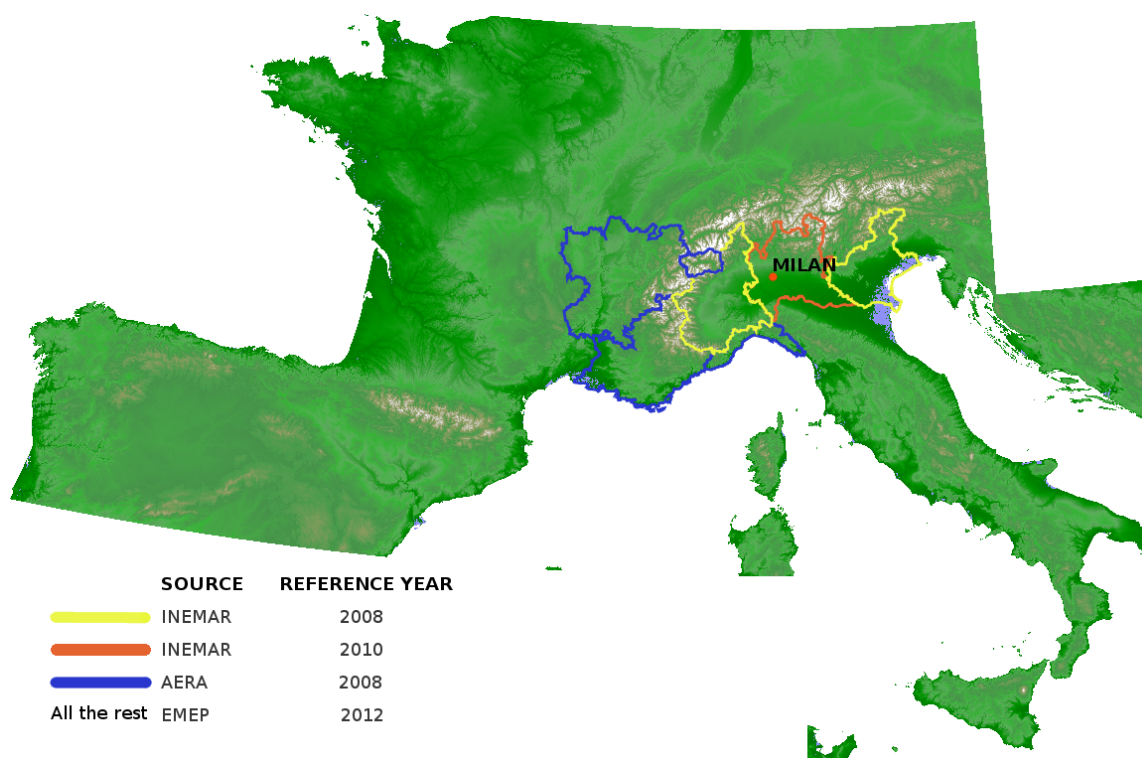


Figure S2: Graphical match between region and the emission inventory used.

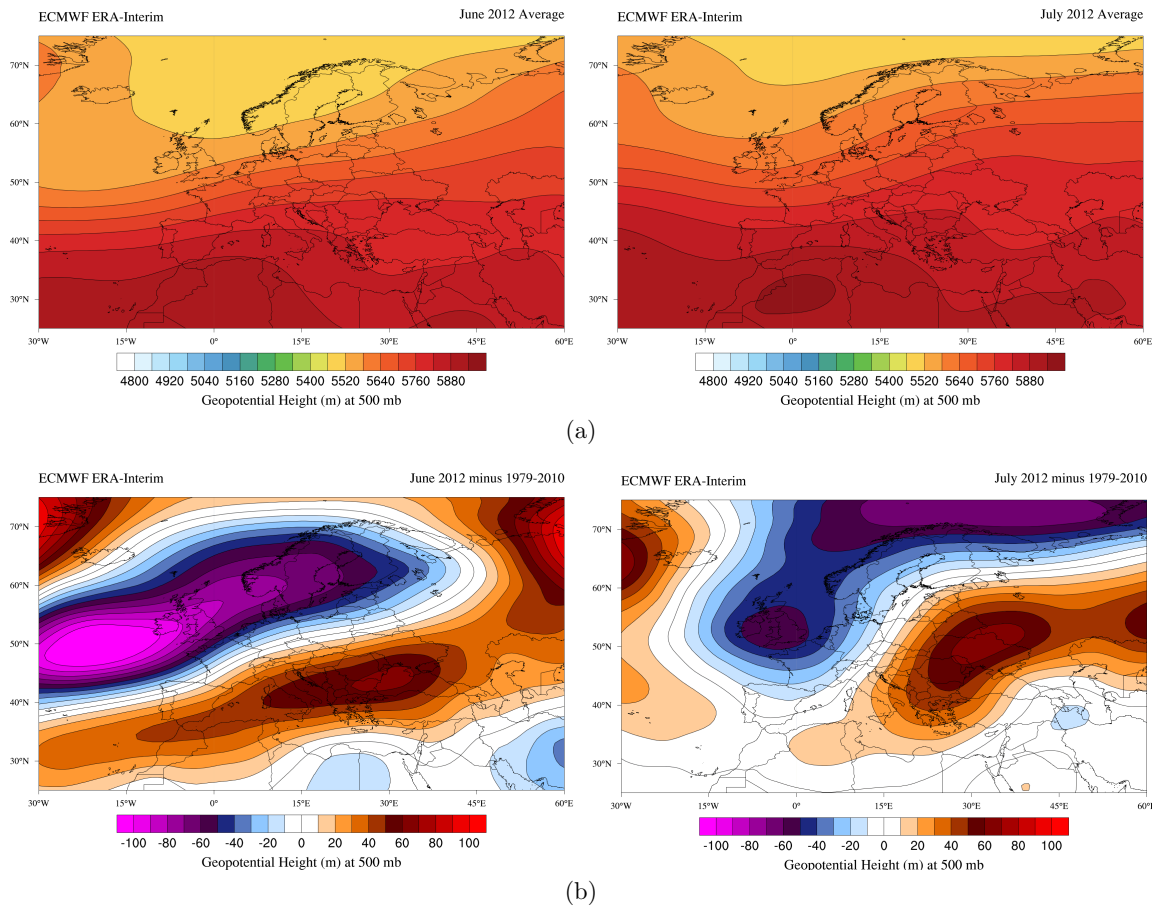
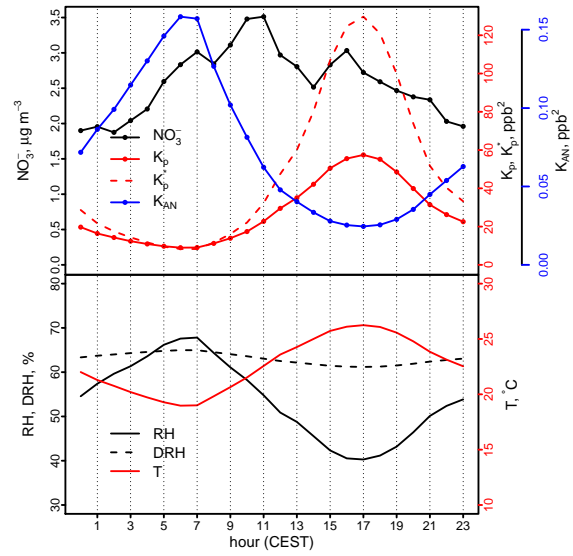
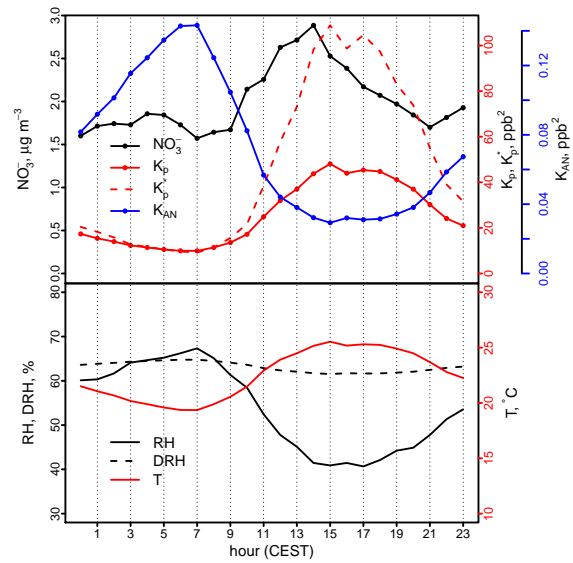


Figure S3: Mean monthly 500 hPa geopotential height (a) and anomaly (b) by ERA-Interim reanalysis. ERA-Interim — ECWMF obtained using Climate Reanalyzer (<http://cci-reanalyzer.org>), Climate Change Institute, University of Maine, USA.



(a)



(b)

Figure S4: Diurnal pattern on weekdays (a) and Sundays (b) for nitrate and ammonium nitrate thermodynamic properties.

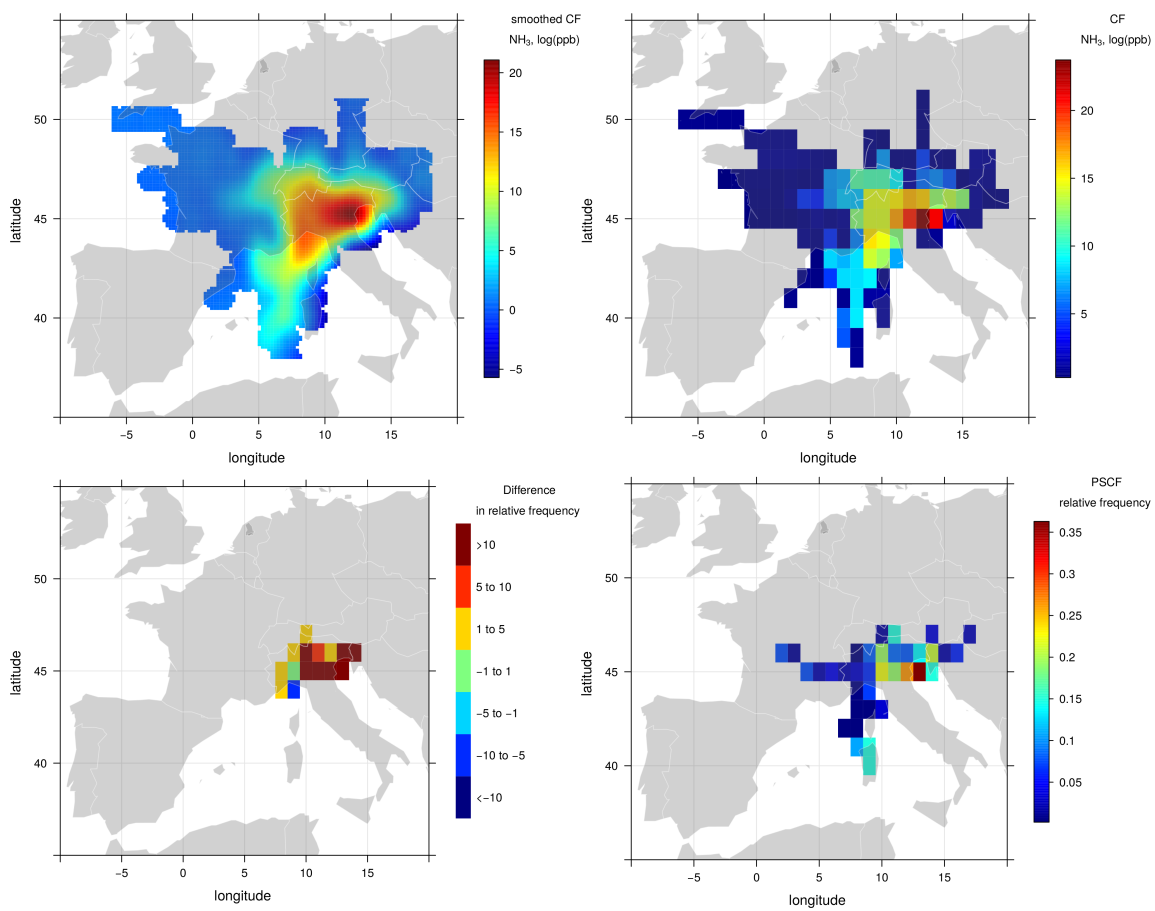


Figure S5: Graphical output of backtrajectory statistical models applied to NH₃ concentration observed in Milan.

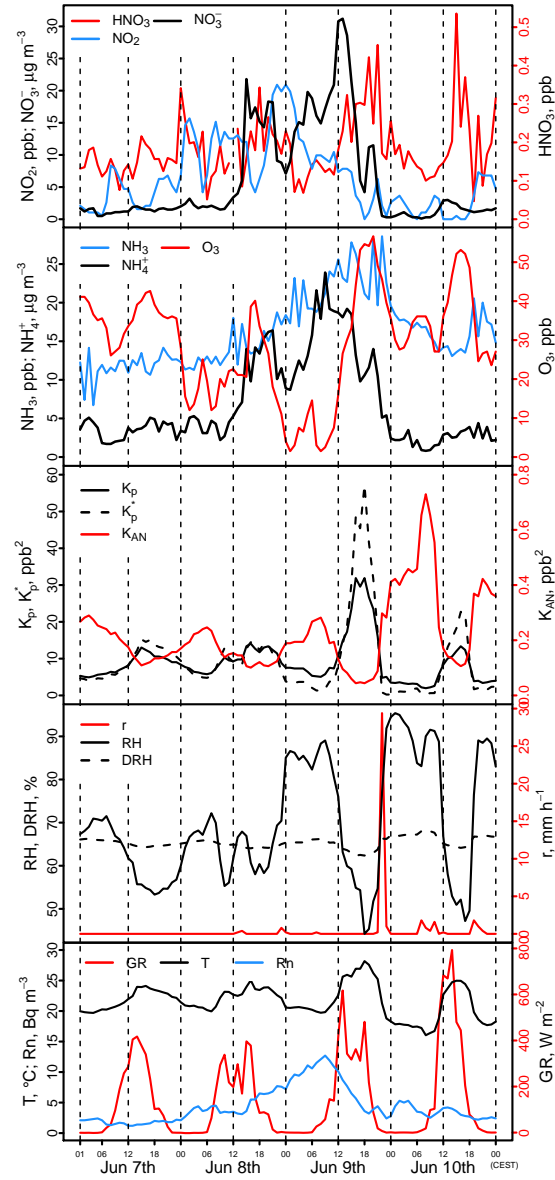


Figure S6: Overview of the nitrate peak event during June 8th-9th 2012

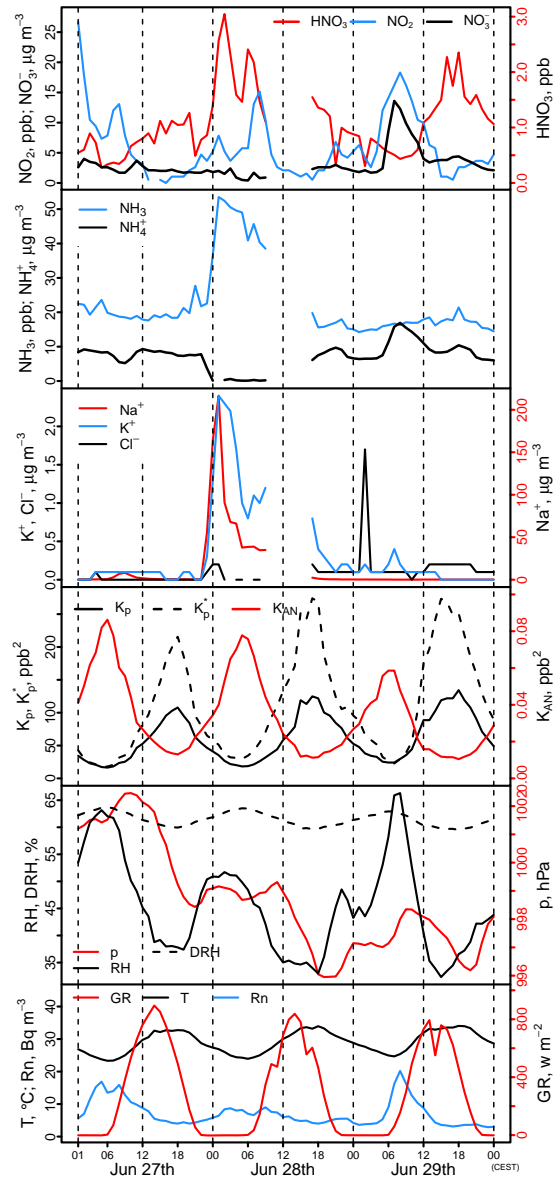


Figure S7: Overview of the ammonia peak event during June 27th–29th 2012

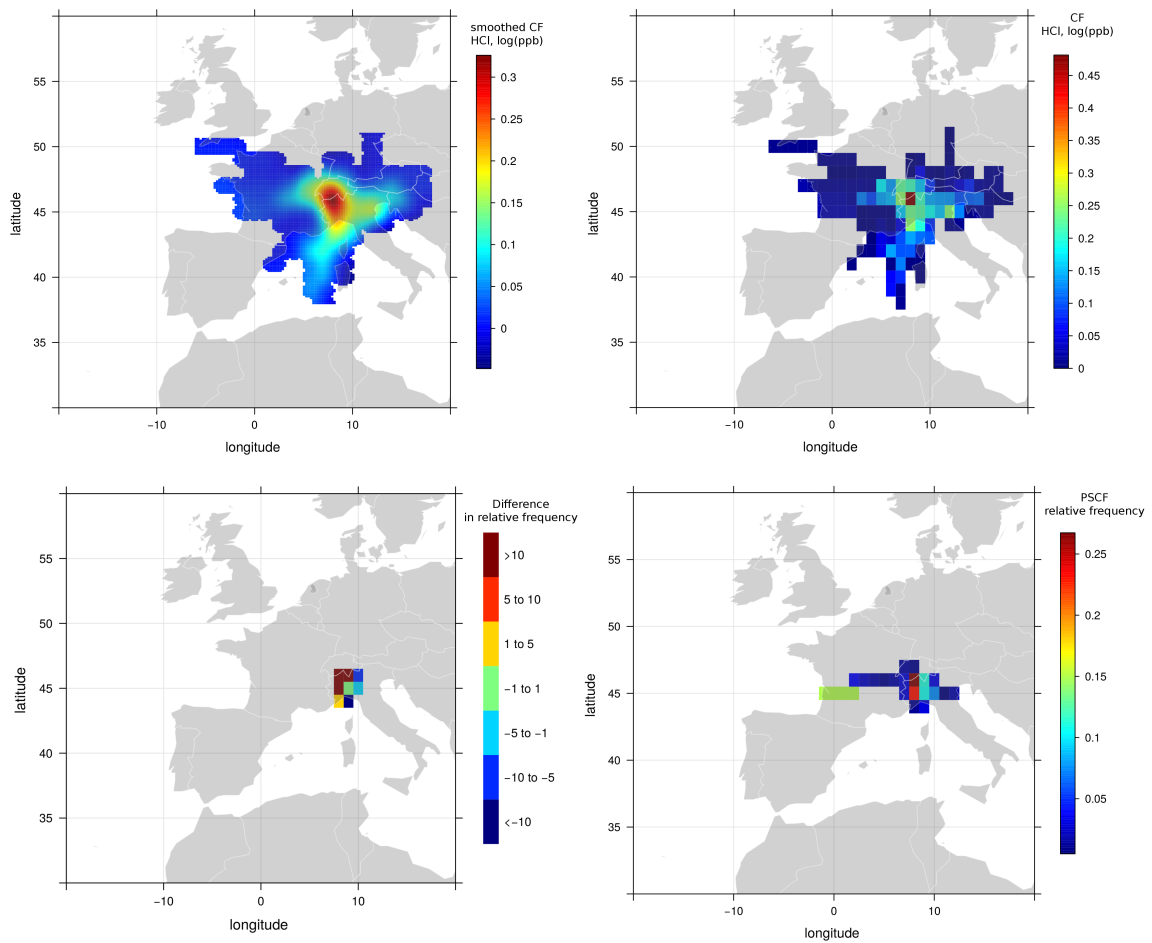


Figure S8: Graphical output of backtrajectory statistical models applied to HCl concentration observed in Milan.

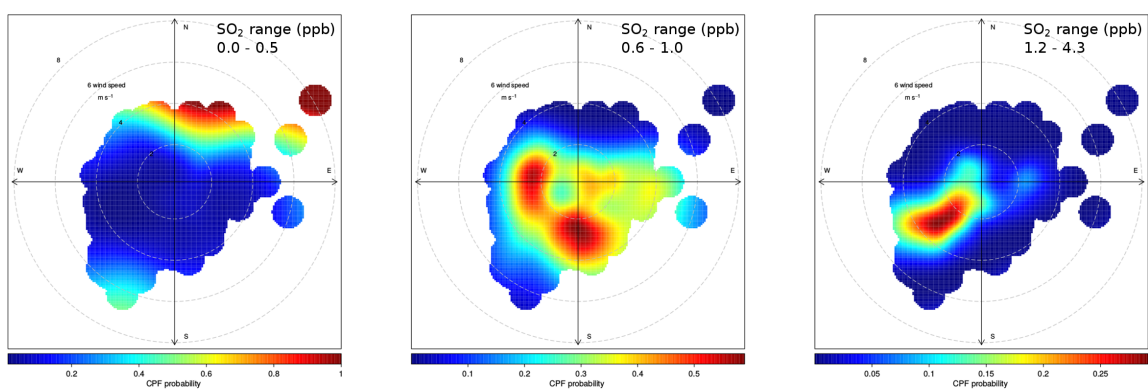


Figure S9: CBPF for SO₂ concentration observed in Milan.

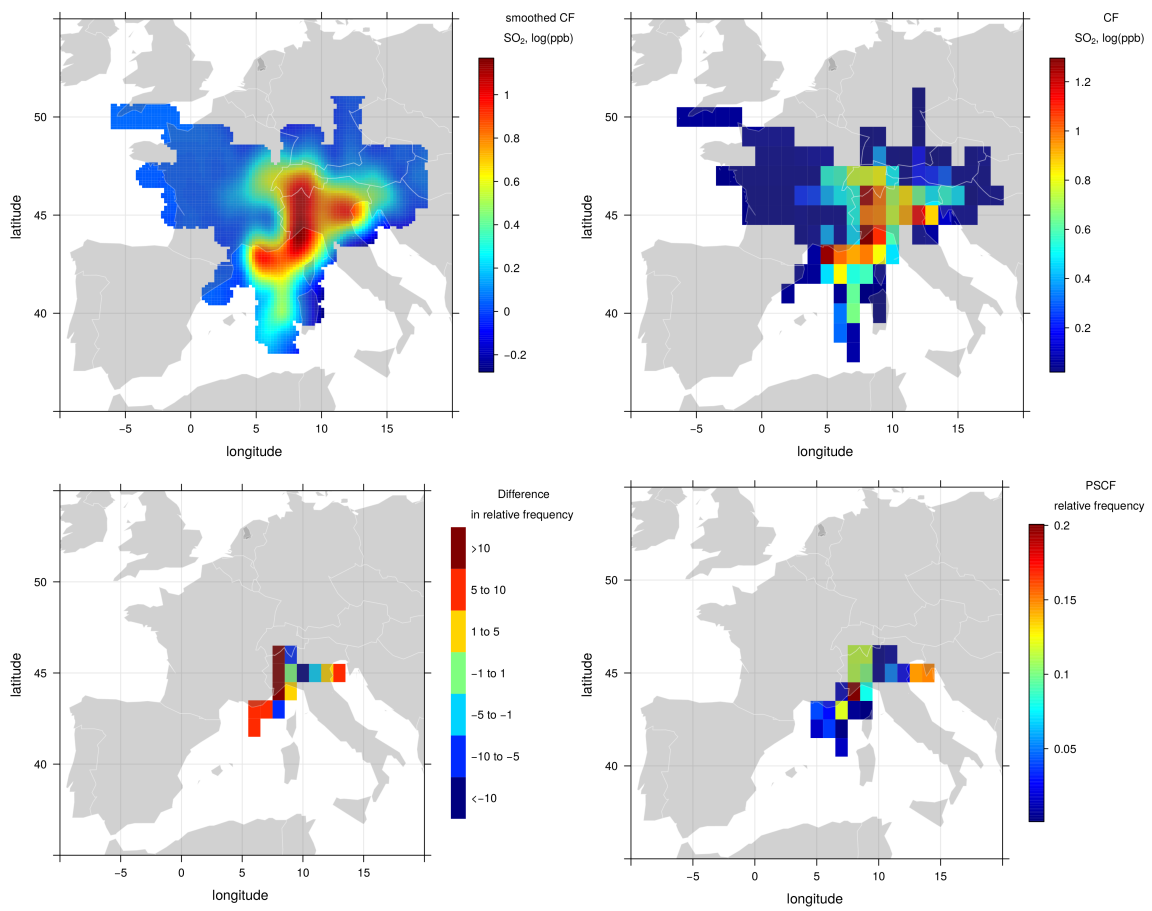


Figure S10: Graphical output of backtrajectory statistical models applied to SO₂ concentration observed in Milan.

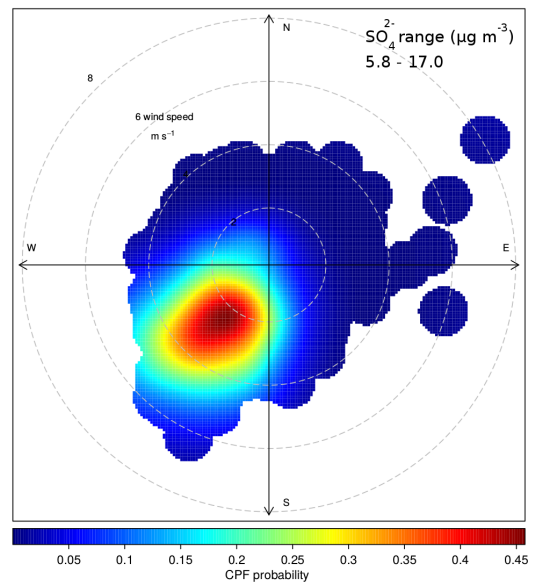
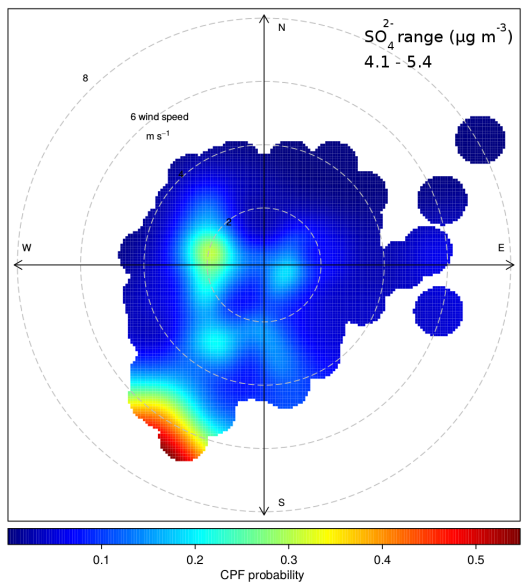
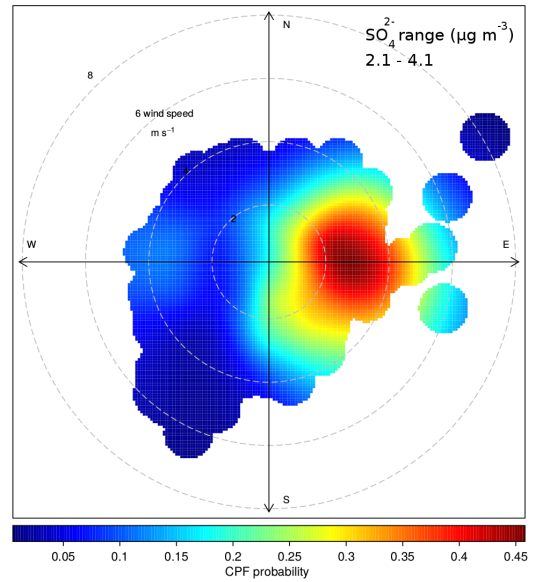
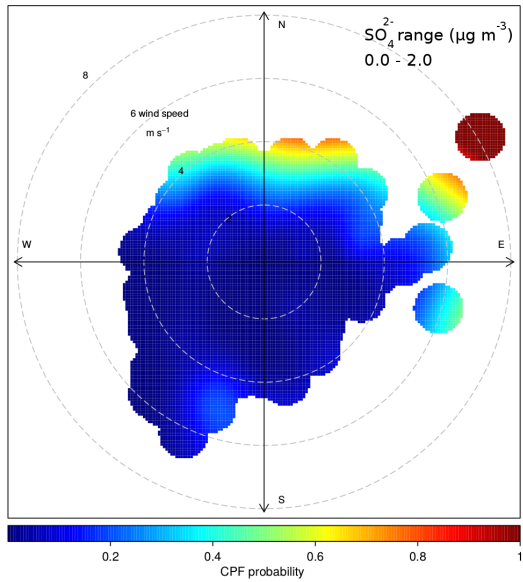


Figure S11: CBPF for SO₄²⁻ concentration observed in Milan.

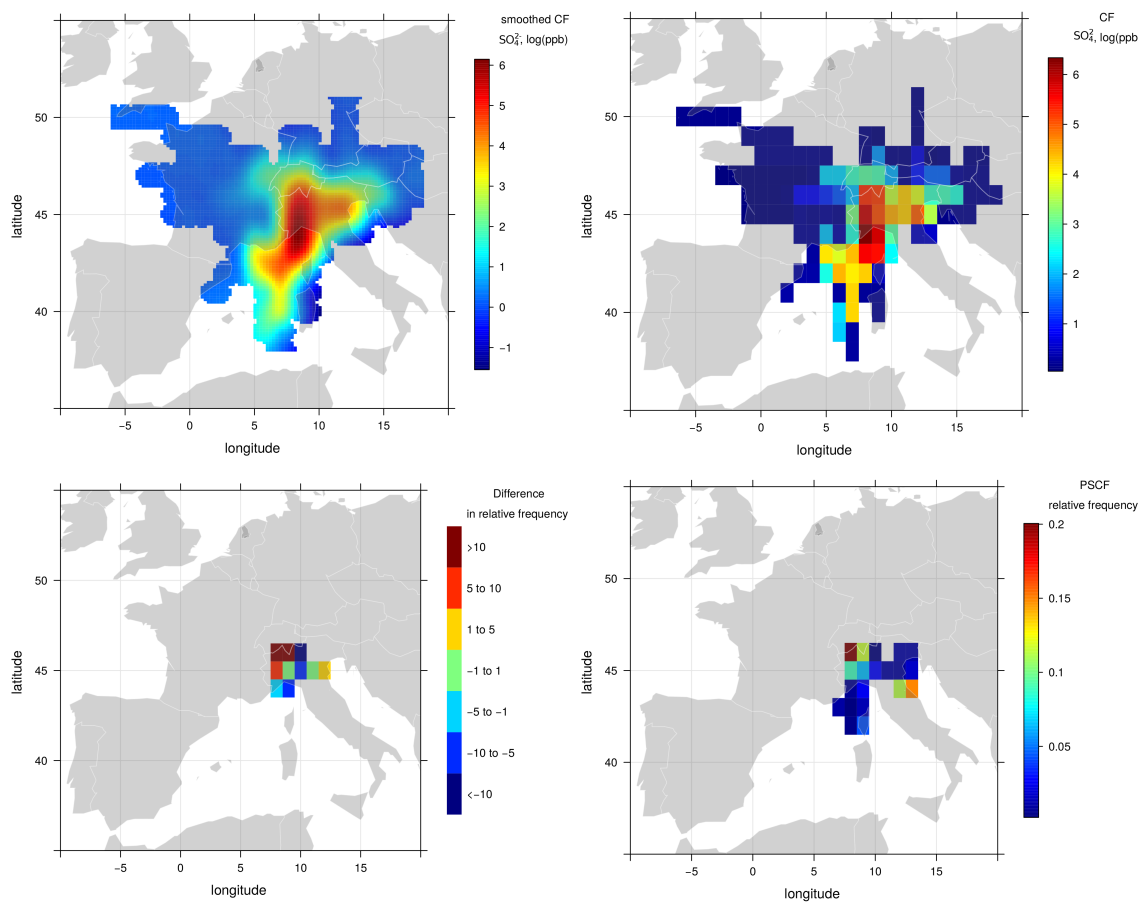


Figure S12: Graphical output of backtrajectory statistical models applied to SO₄²⁻ concentration observed in Milan.

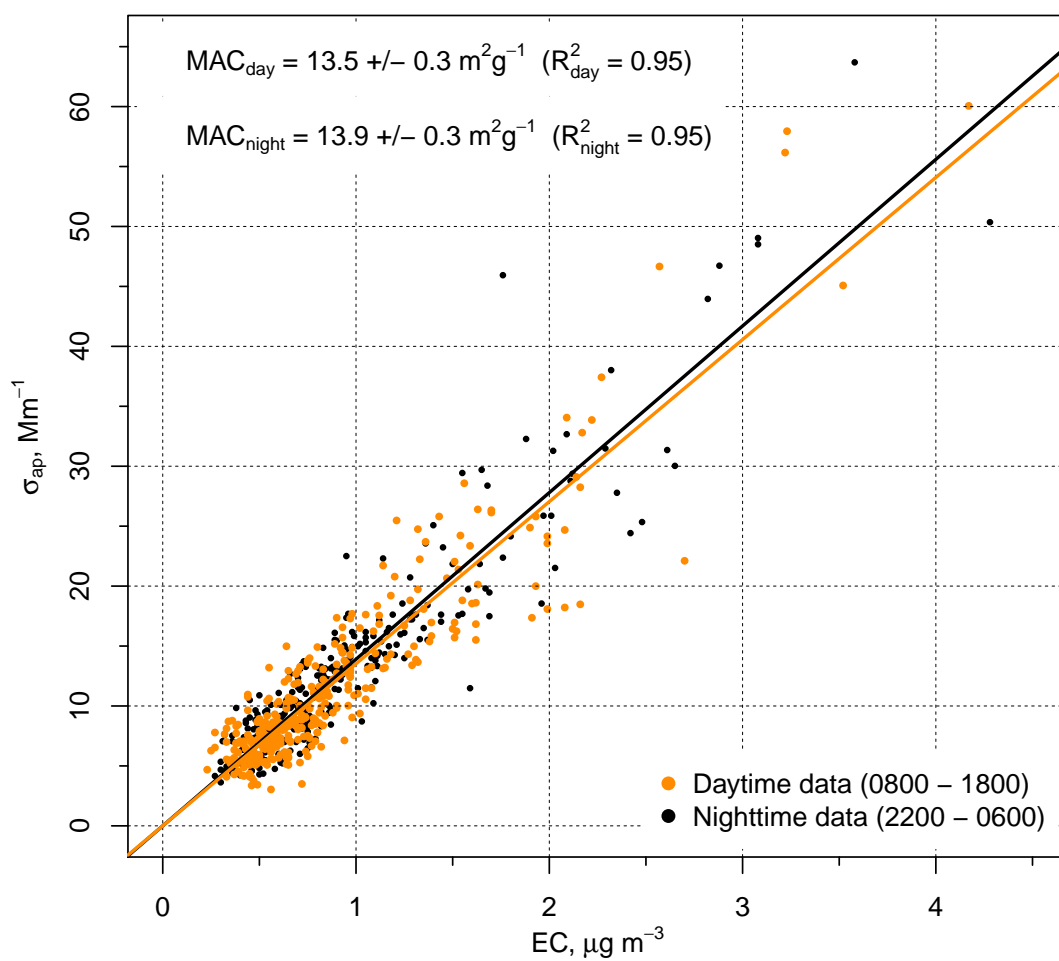


Figure S13: Scatterplot and linear model fit between absorption coefficient by MAAP (σ_{ap}) and Elemental Carbon concentration by ECOC field analyzer (EC) .

Cluster dendrogram (by divisive algorithm) for the full dataset (DC=0.69)

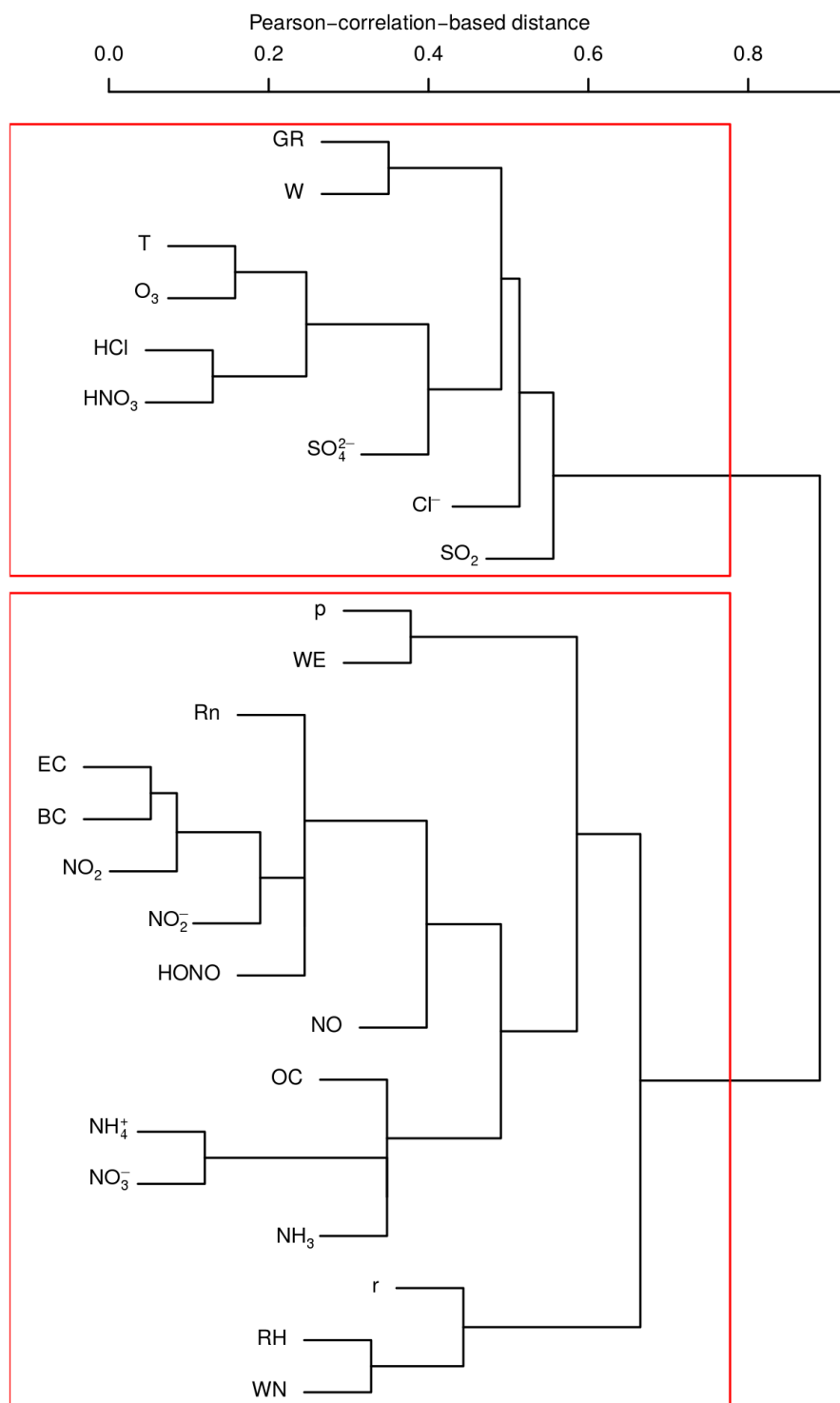


Figure S14: Results of cluster analysis on the complete dataset data using a Pearson-correlation-based distance. Coloured boxes indicate clusters; lookup table for abbreviations can be found in table B.1. WN and WE are the Northern and Eastern component of wind vector, DC is the Divisive Coefficient.