

PCCP

Implications for New Particle Formation in Air of the Use of Monoethanolamine in Carbon Capture and Storage

Journal:	Physical Chemistry Chemical Physics
Manuscript ID	CP-ART-01-2024-000316.R1
Article Type:	Paper
Date Submitted by the Author:	22-Feb-2024
Complete List of Authors:	Perraud, Veronique; University of California, Chemistry Roundtree, Kanuri; University of California, Chemistry Morris, Patricia; University of California, Chemistry Smith, James; University of California Irvine, Department of Chemistry Finlayson-Pitts, Barbara; University of California, Irvine, Department of Chemistry

SCHOLARONE[™] Manuscripts

Implications for New Particle Formation in Air of the Use of Monoethanolamine in Carbon Capture and Storage

Véronique Perraud,* Kanuri Roundtree, Patricia M. Morris, James N. Smith* and B. J. Finlayson-

Pitts

¹Department of Chemistry, University of California Irvine, Irvine, CA 92697

For submission to: PCCP (Special Issues, 25th Anniversary)

*Authors to whom correspondence should be addressed. Email: vperraud@uci.edu; phone: (949) 824-2673; FAX (949) 824-2420 Email: jimsmith@uci.edu; phone: (949) 824-9518; FAX (949) 824-2420

keywords: monoethanolamine; water; methanesulfonic acid; thermal desorption chemical ionization mass spectrometry; new particle formation

TDCIMS MEA paper_text_Feb20_2024_Revised.docx

1 Abstract

2 Alkanolamines are currently being deployed in carbon capture and storage (CCS) technology 3 worldwide, and atmospheric emissions have been found to coincide with locations exhibiting 4 elevated concentrations of methanesulfonic acid (MSA). It is thus critical to understand the fate 5 and potential atmospheric reactions of these chemicals. This study reports the characterization of 6 sub-10 nm nanoparticles produced through the acid-base reaction between gas phase 7 monoethanolamine (MEA) and MSA, a product of organosulfur compound oxidation in air. 8 using a flow reactor under dry and humid (up to ~60% RH) conditions. Number size distribution 9 measurements show that MEA is even more efficient than methylamine in forming nanoparticles 10 on reaction with MSA. This is attributed to the fact that the MEA structure contains both an -11 NH₂ and an -OH group that facilitate hydrogen bonding within the clusters, in addition to the electrostatic interactions. Due to this already strong H-bond network, water has a relatively 12 13 small influence on new particle formation (NPF) and growth in this system, in contrast to MSA 14 reactions with alkylamines. Acid/base molar ratios of unity for 4-12 nm particles were measured using thermal desorption chemical ionization mass spectrometry. The data indicate that reaction 15 16 of MEA with MSA may dominate NPF under some atmospheric conditions. Thus, the unique 17 characteristics of alkanolamines in NPF must be taken into account for accurate predictions of 18 impacts of CCS on visibility, health and climate.

19

21 Introduction

22 Monoethanolamine (HOCH₂CH₂NH₂, MEA) is a multifunctional amine currently 23 deployed in carbon capture and storage (CCS) technology systems aimed at sequestering CO₂ emissions before release into the atmosphere.¹⁻⁵ The most widely used CCS media is a 30% 24 25 aqueous solution of monoethanolamine (MEA).⁶ Briefly, the solvent medium chemically 26 absorbs CO₂ contained in the flue gas, which leads to a CO₂-depleted gas stream exiting the 27 stack. The solvent is subsequently regenerated and recycled back into the absorber column, while the CO₂ is compressed and captured. A potential drawback from these technologies is the 28 29 likely release of MEA into the air.^{4,5,7-14} For example, concentrations of MEA outside a CCS-30 equipped plant of the order of several ppb have been reported.⁷ MEA is also used as a solvent in various consumer products and industrial processes.¹⁵⁻¹⁹ In air, recognized fates of gas phase 31 MEA to date include its reaction with O₃ and OH.²⁰⁻²⁴ the formation of alkylaminium nitrate salts 32 from its interaction with HNO₃^{21,24} and acid-base reactions with gas phase and particulate 33 sulfuric acid.^{25,26} 34

Methanesulfonic acid (CH₃SO₃H, MSA) is a strong acid formed along with SO₂ (a sulfuric acid precursor) in the oxidation of dimethyl sulfide (DMS) and dimethyl disulfide (DMDS)²⁷⁻³² which have a variety of sources both natural and anthropogenic.³³⁻⁵⁹ Therefore, it is not surprising that the oxidation product MSA is detected in the gas phase and in ambient particles worldwide. Ambient gas phase atmospheric concentrations of MSA range from mid-10⁴ to 10⁷ molecules cm⁻³, and can reach levels similar to that of H₂SO₄, which is considered to be a major source of new particles.⁶⁰⁻⁶⁹ In some instances, the MSA concentration in air can actually surpass that of

42	co-located H ₂ SO _{4.68,70} MSA has also been detected in ambient particles worldwide, including in
43	marine and coastal environments, ⁷¹⁻⁷⁷ in coastal areas affected by biomass burning plumes, ^{78,79}
44	and near agricultural regions as well as near urban centers. ^{78,80,81} This acid has been detected in
45	ultrafine and nucleation mode particles measured in the Arctic, ^{76,82-85} at urban sites ^{86,87} and in the
46	Antarctic ⁸⁸ as well as in a boreal forest. ^{67,89} Particulate MSA concentrations in the Arctic
47	summertime have been observed to correlate well with new particle formation (NPF) ^{85,90-94}
48	suggesting a role for MSA in the earliest stages of NPF and growth. Chen and co-workers ⁹⁵
49	predicted that the total annual MSA budget would be 20 Gg S yr ⁻¹ from DMS oxidation reactions
50	alone. However, climate change is dramatically modifying the extent of ice sheet coverage,
51	exposing more sea water, which increases phytoplankton productivity and DMS emissions and
52	thus MSA in air. ^{94,96-98} The significant contribution of MSA to atmospheric NPF is supported by
53	both laboratory experiments ⁹⁹⁻¹⁰⁷ and quantum chemical calculations. ¹⁰⁸⁻¹¹⁵
54	Amines and MSA are both found in ambient particles. ^{67,80,116-118} This includes MEA,
55	which has been detected as one of the most abundant amines in ambient particles in various
56	locations around the globe, ¹¹⁹⁻¹²⁴ overlapping with sources of both DMS and MSA. MEA has
57	also been detected in biomass burning aerosols collected in St John, Newfoundland, Canada ¹²⁵
58	and in both aerosol and precipitation samples over the North Atlantic Ocean. ¹²⁶
59	It is thought that NPF is responsible for a significant portion of the global cloud
60	condensation nuclei budget. ¹²⁷ In addition to influencing cloud properties, airborne particles are
61	well known to interact with solar radiation, thus playing a critical role in the Earth's
62	climate. ^{128,129} In a recent study, Hodshire et al. ¹³⁰ predicted, using a simplified DMS oxidation

Page 5 of 44

model, that inclusion of MSA formation and its role in aerosol processes (either acting as
condensable non- or semi-volatile species, or participating in NPF) influenced the cloud-albedo
aerosol indirect and the direct radiative effect.

66 While recent theoretical studies predicted that MEA may play an important role in NPF.¹⁰⁸ to date there have been no direct experimental investigations of particle formation from 67 68 MEA and MSA. We present the first measurements of 4-12 nm nanoparticles formed from this 69 reaction, including their size distributions as a function of time and relative humidity, as well as 70 their size-resolved chemical composition. For comparison, some data for the reaction of MSA with methylamine (MA), which is known to efficiently form particles, ^{102,103,105,106} is also 71 72 reported. It is shown that MEA is even more effective in forming new nanometer size particles 73 than MA but surprisingly, is not very sensitive to the presence of water vapor. Such ultrafine 74 particles are of particular concerns as they can be deposited deep into the respiratory tract and even cross cellular membranes to reach other organs.¹³¹⁻¹³⁶ Thus, this study has important 75 implications for the potential impacts of CCS on climate.^{128,129} visibility¹³⁷⁻¹⁴⁰ and 76 health.131,132,136,141 77

78

79 Experimental Methods

Flow Reactor Description. Particles were produced from the reaction of gas phase MSA with
gas phase MEA (or MA) in the presence or absence of water vapor in a 1-m long borosilicate
glass flow reactor¹⁴² described in the Electronic Supplementary Information (ESI; Fig. S1).

83 Clean, dry air was provided by a purge air generator (Parker-Balston; model 75-62), and further 84 purified by passing through carbon/alumina media (PermaPure, LLC) and a 0.01 µm inline filter 85 (Parker Balston, BQ). Most of the air was supplied at the front end of the flow reactor through 86 the perforated ring inlets as indicated in Fig. S1 (rings A, B and C). In experiments where water 87 vapor was present, one or two bubblers filled with nanopure water (18.2 M Ω cm; Barnstead, 88 Thermo Scientific) were used to humidify a fraction of the air introduced into the ring inlets. 89 The bubblers were kept in a water bath to maintain a constant temperature of 22°C (295 K). 90 Experiments were carried out at relative humidities (RH) up to $\sim 60\%$ as indicated by a humidity 91 probe (Vaisala; model HMT 838) located at the end of the flow reactor. The reactants (MSA and 92 MEA or MA) were introduced through the spoke inlets (spoke 2 and 3 respectively) located 60 93 cm downstream of the last ring inlet. The flow reactor was cleaned regularly with nanopure 94 water and dried with clean hot air overnight (T = 343 K). After cleaning, the flow reactor was 95 conditioned with gas-phase MSA for a least two days prior to an experiment. All experiments 96 presented in this work were performed at 1 atm and at room temperature (T = 297 K).97

98Reactants. Liquid monoethanolamine (NH2CH2CH2OH, Sigma Aldrich, >99.5%) was99contained in a small 2-mL glass vial with a septum cap. Approximately ~1 cm of PEEK tubing100(0.007" inner diameter) was inserted into the septum so that the MEA from the headspace101diffused slowly into a stream of air. For comparison, parallel experiments were performed using102MA (CH3NH2) with a commercial permeation tube (VICI Metronics). The amine vial (or103permeation tube) was inserted into separate U-shaped glass tubes immersed into a water bath104maintained at room temperature (T = 295 K). Glass beads were placed in the upstream arm of

105 the U-shaped glass tubes to provide high surface area to keep the gas flow at a constant 106 temperature. Air flowed through each tube at a rate of 215 cm³ min⁻¹ for MEA and 93 or 211 107 cm³ min⁻¹ for MA. For MSA, air (53 to 216 cm³ min⁻¹) flowed directly over the pure liquid 108 (Sigma Aldrich, > 99.0%) contained in a glass trap which was maintained at room temperature 109 using a water bath. Further details regarding the sampling, analysis and quantification of the gas 110 phase reactants are given in the ESI (Text S1 and Fig. S2). The initial concentrations of the reactants after dilution in the flow reactor were (1.7-6.8) \times 10¹⁰ molecules cm⁻³ for MSA (0.7-111 2.8 ppb), $(3.7-8.1) \times 10^{10}$ molecules cm⁻³ for MEA (1.5-3.3 ppb) and (11.8-26.6) $\times 10^{10}$ 112 molecules cm⁻³ for MA (4.8-10.8 ppb). Note that these concentrations represent upper limits as 113 114 they do not account for potential wall losses.

115 **Particle size distribution measurements.** Particle size distributions were continuously 116 measured using a moveable stainless steel sampling line (O.D. 0.635 cm) located inside the flow 117 reactor along the centerline and placed at distances ranging from 3 to 43 cm away from spoke 2 118 (i.e., the MSA addition port). All particle size distributions reported in this study are number 119 size distributions, unless stated otherwise. These distances correspond to reaction times in the 120 reactor ranging from 0.3 to 4.5 s (total flow rate 23.4 L min⁻¹) or 0.5 to 7.7 s (total flow rate 10.7 L min⁻¹) based on a conversion factor determined in previous studies.⁹⁹ Note that the amine 121 122 addition port is introducing the reactant backward into the flow stream so that the reaction of 123 MSA with MEA (MA) is occurring in between spoke 2 and 3, and we chose the MSA addition 124 port as our t = 0 reaction time. It is expected that the residual reactants present in the stream 125 exiting the flow reactor are lost to the walls of the small (ID 0.18 inches) sampling line. 126 Therefore, the reaction times reported are those in the flow reactor, but these could be

127 underestimated if the reaction continues in the sampling line (residence time $\sim 0.3-0.4$ s) and the 128 connection to the SMPS (residence time ~ 0.8 s). Particle losses through the sampling lines to 129 the SMPS were accounted for as described in the ESI (Text S2; Fig. S3). The shortest reaction 130 time accessible (i.e. 0.3 s or 0.5s depending on the total flow in the flow tube, that was 23.4 or 131 10.7 L min⁻¹ respectively) is expected to be the most vulnerable to residence time artefacts. This 132 was tested by sampling at different flow rates through the sampling line (2.4 to 4.8 L min⁻¹), for a 133 given experiment conducted at 0.5 s. Results presented in Fig. S4 show no change in the size 134 distribution measured at all flow rates and suggest that reaction in the sampling line is not 135 significant. 136 Size distributions were measured using a scanning mobility particle sizer (SMPS) consisting of a ²¹⁰Po radioactive source (10 mCi; NRD LLC; model P-2021), an electrostatic 137 138 classifier (model 3080; TSI Inc.) equipped with a nano-differential mobility analyzer (nano-139 DMA; model 3085; TSI, Inc.), and a butanol-based ultrafine condensation particle counter 140 (UCPC; model 3776; TSI, Inc.). To prevent buildup of the reactants in the SMPS during 141 sampling, the sheath air inside the DMA was not recirculated, but instead air was provided by the 142 purge air generator (15 L min⁻¹) and a vacuum pump connected to the sheath air flow pulled the 143 sheath air out of the DMA. The aerosol flow was set to 1.5 L min⁻¹, which provided

144 measurements of the size distributions over a mobility diameter range of 2.5 to 64 nm. The

software AIM v9 (TSI, Inc.) was used to record and process the data. Particles were observed to

146 be stable for long periods of time (Fig. S5), allowing for size-resolved measurements that took

147 up to 20 min per scan to yield enough mass for mass spectrometric analysis.

148	Size-resolved chemical composition measurements. Nanoparticles with diameters ranging
149	from 4 to 12 nm were sampled using a thermal desorption chemical ionization mass spectrometer
150	(TDCIMS) ^{105,143-146} which was connected to the same sampling line as the SMPS. The particle
151	stream was sampled through two inlets, each equipped with a ²¹⁰ Po unipolar charger (UPC) ^{143,147}
152	to generate negatively charged particles. At each inlet, particles were subsequently size-selected
153	using a radial nano-DMA (rDMA) running in either high resolution mode with a sheath flow of
154	10 L min ⁻¹ and an aerosol flow of 1.0 L min ⁻¹ through each nano-rDMA, or low resolution mode
155	with a sheath air flow of 5.0 L min ⁻¹ and an aerosol flow rate of 1.6 L min ⁻¹ through each nano
156	rDMA. ¹⁴⁸ For both conditions, instead of recirculating the sheath gas within the nano-DMA,
157	gaseous N_2 produced from the headspace of a liquid N_2 dewar was used as the sheath flow to
158	prevent the accumulation of gas-phase MSA or MEA, and a vacuum pump was used at the DMA
159	sheath flow outlet. The particles were collected on the tip of a Pt filament by electrostatic
160	precipitation (applied high voltage of $+3.5$ kV). The filament was continuously flushed with an
161	additional 1.25 L min ⁻¹ flow of N_2 to minimize sampling artifacts from gas-phase species. To
162	select particles with a defined mobility diameter, the voltage on each rDMA was varied from 30
163	to 325 V. Note that the use of two separate inlets, which merged at the collection wire region,
164	increases the flux and mass of particles that are collected on the wire without sacrificing the
165	rDMA resolution.

166 The TDCIMS was run in positive ion mode to measure MEA with $(H_2O)_nH^+$ as the reagent 167 ions (n=0–3), and in negative ion mode to measure MSA with $(H_2O)_nO_2^-$ as the reagent ions from 168 the presence of trace amounts of H_2O and O_2 , respectively, in the carrier N_2 gas.

169 Monoethanolamine was detected as two major ions in the mass spectra, the parent [M+H]⁺ ion

170 (m/z 62) and a fragment ion corresponding to $[M+H-H_2O]^+$ (m/z 44). The fragmentation of the parent [M+H]⁺ ion of MEA is consistent with early experimental and theoretical studies^{149,150} 171 172 showing that although the amino group is the favored protonation site due to its higher proton affinity compared to the alcohol group,¹⁵¹ rearrangement and the loss of H₂O dominates over the 173 174 loss of NH₃. The corresponding fragment ion associated with the loss of NH₃ (m/z 45) was not 175 observed in any of the mass spectra. MSA was detected in negative ion mode as the parent 176 deprotonated [M-H]⁻ ion (m/z 95) followed by a major fragment ion at m/z 80 (SO₃⁻), with 177 additional minor ions at m/z 64 (SO₂⁻), m/z 96 (SO₄⁻), m/z 97 (HSO₄)⁻ and m/z 112 (SO₅⁻). Both 178 positive and negative mass spectra are presented in Fig. S6. From the desorption profiles 179 presented in Fig. S7, it is evident that MEA (and MA; data not shown) desorbs first from the 180 filament followed by MSA, consistent with the differences in their respective saturation vapor pressures (P_{sat}) at 298 K: P_{sat} (MEA) = 3.4 × 10⁻⁴ atm¹⁵² and P_{sat} (MSA) = 7.4 × 10⁻⁷ atm.¹⁵³ 181 182 Additional details on the TDCIMS analysis are described in the ESI (Text S3-S5; Fig. S6-S11).

183

184 **Results and Discussion**

Figure 1 represents the size distributions of particles from the MSA+MEA reaction under dry conditions, with each panel (A-F) representing a different reactant concentration condition. Varying reactant concentrations was achieved by either increasing or decreasing the flow of the reactant that was introduced into the flow tube, or by changing the total flow rate in the flow tube (23.4 L min⁻¹ for Panels (A-C); 10.7 L min⁻¹ for panels (D-F)). Clearly, mixing gas phase MSA and MEA at low ppb levels results in rapid formation of particles. Corresponding plots of the

191	evolution of the total number concentration as well as the geometric diameter as a function of the
192	reaction time are presented in Fig. S12. Even at the smallest reactant concentrations (Fig. 1A,
193	1.5 ppb MEA, 0.68 ppb MSA), particles measured at the shortest distance ($t = 0.3$ s; total flow
194	rate 23.4 L min ⁻¹) are formed at a number concentration of 6.3×10^6 particles cm ⁻³ with a
195	geometric mean mobility diameter (GMD) of ~ 4 nm. At 2.4 s, the number concentration
196	increases by a factor of two with little change in size. At longer times, there is no further
197	increase in the particle number concentration while the particles continue to grow to a GMD of
198	\sim 4.6 nm, suggesting that under these conditions there is a balance between nucleation, growth by
199	addition of the reactants onto particles, and coagulation. Particle losses inside the flow tube were
200	estimated using the particle loss calculator tool developed by von der Weinder et al. ¹⁵⁴ (using a
201	density of 1 g cm ⁻³), and was found to be small for all diameters (e.g. for a particle diameter of
202	2.5 nm, particle transmission is predicted to be 92 or 95% for a total flow rate inside the flow
203	tube of 10.7 or 23.4 L min ⁻¹).

204 Similar behavior is seen as the initial MSA concentration is increased, but with larger 205 total particle number concentrations formed (Fig. 1B,C; Fig. S12 A,B). In this case, at longer 206 reaction times the particle number concentrations start to decrease and the GMD increases due to 207 coagulation (Fig. 1C). Similar, but more pronounced, trends are seen at an initial MEA 208 concentration of 3.3 ppb and increasing MSA concentrations (Fig. 1D-F). For approximately the 209 same MSA concentration (Fig. 1B and 1D; Fig. 1C and 1E), doubling the concentration of MEA 210 leads to an increase in total number concentration of a factor of ~1.2-1.4 at 2.3-2.4 s reaction 211 time, with an increase in diameter from 4.9 to 5.6 nm (MSA = 1.4-1.5 ppb) and from 5.3 to 6.3212 nm (MSA = 2.8-3.0 ppb).

213 For the low concentration series, the formation of approximately half of the peak particle 214 concentration at the first measurement time implies that the rate-determining step is fast. There 215 is some uncertainty in the exact reaction time for this first data point since it does not take into 216 account possible continued reaction in the sampling lines. However, a half-life of ~ 0.5 s for the 217 reaction of MSA with excess MEA at 1.5 ppb (Fig. 1A), is consistent with a gas phase bimolecular rate constant for MEA with MSA of approximately 4×10^{-11} cm³ molecule⁻¹ s⁻¹. 218 219 The particle formation rate $(J_{>2.5nm})$ was estimated using the total concentration of particles 220 measured at ~ 2.4 s (peak concentration) for all conditions, and dividing by 2.4 s. Figure 2 shows 221 the resulting J_{>2.5nm} values as a function of the product of the MEA and MSA initial 222 concentrations. There is an initial rapid increase which is approximately linear out to $[MEA] \times$ $[MSA] \sim 2 \text{ ppb}^2$, suggesting that the initial 1:1 cluster formation is the rate-determining step. 223 224 The drop-off at higher concentrations reflects coagulation. This is consistent with the TDCIMS 225 measurements (Fig. 3), which show that the acid/base molar ratios in the particles from 4-12 nm 226 remains within experimental error of one. All measurements were performed at 4.5 s reaction 227 time, and at [MEA] initial concentration of 1.5 ppb, and there were no significant differences in 228 the measured molar ratio across the experiments performed under excess MEA ([MSA] = 0.68229 ppb), pseudo-equimolar MEA/MSA ([MSA] = 1.4 ppb) or excess MSA ([MSA]= 2.8 ppb), thus 230 the data obtained for all [MSA] concentrations was averaged together.

231

MSA concentrations in air can be as high as 10⁷ molecules cm⁻³ (~0.4 ppt)^{60,61,66,68} and MEA in the low ppb range has been recorded outside a CCS facility.⁷ The slope of the line in

234	Fig. 2 at the lowest reactant concentrations is $(5.3 \pm 0.03) \times 10^6$ particles cm ⁻³ ppb ⁻² so a NPF rate
235	from the upper limit atmospheric concentrations of MEA (10 ppb) and MSA (0.4 ppt) of as much
236	as ~21,200 particles cm^{-3} is predicted. This can be compared to a range of formation rates of
237	particles >3 nm diameter (J_3) from sulfuric acid of 0.001 – 10 ⁵ cm ⁻³ observed in different
238	environments around the world. ¹⁵⁵ In short, even for conditions where MEA and MSA
239	concentrations are less than the reported maxima, this single reaction system may contribute
240	significantly to NPF and its importance may increase as MEA use in CCS increases.
241	Efficient particle formation from MEA and MSA is consistent with the excellent stability
242	at room temperature and low vapor pressure of the MSA-MEA salt synthesized by Greaves and
243	co-workers. ¹⁵⁶ Furthermore, MEA-MSA has been reported to have properties of a protic ionic
244	liquid, even though it remains a solid at room temperature. ¹⁵⁶⁻¹⁵⁹ Its properties include a glass
245	transition of -44 °C, melting point of about 100 °C, and a thermal stability up to 286-323 °C for
246	the fused salt. ¹⁵⁶

247 Tropospheric air contains significant amounts of water vapor, hence the impact of 248 relative humidity (RH) on particle formation from MEA + MSA was also examined. 249 Surprisingly, and in contrast to previous results obtained for small alkylamines, the addition of 250 water vapor to the MSA+MEA system did not significantly increase the number concentration at 251 RH below ~ 20% as indicated in Fig. 4. Figure 5A,B shows the evolution of the particle size 252 distributions as a function of time at an RH of ~50% for two different sets of precursor 253 concentrations corresponding to the dry conditions presented in Fig. 1A and 1C respectively. 254 The evolution of the size distributions as a function of time in the flow reactor is similar to that

255	observed under dry conditions. To better compare the dry versus humid case, total particle
256	concentrations and geometric mean diameters measured at 4.5 s over several repeated
257	experiments were averaged and are shown in Fig. S13. The addition of water vapor increased
258	the total number concentration by about only a factor of 1.3-1.5 as indicated by the bars. Note
259	that the enhancement factor (EF) measured at 4.5 s for the high MSA, high RH case is an
260	underestimate as it already includes coagulation (Fig. 5C). At the peak particle concentration (
261	= 1.4 s reaction time), EF = 1.9 . There is only a small increase in size (red squares) at the
262	highest MSA concentrations.

263 Classical nucleation theory predicts that the number of water molecules in the critical 264 cluster can, under some conditions, be obtained from the slope of a log-log plot of the formation rate of new particles versus the gas phase water concentration.¹⁶⁰ However, this is highly 265 dependent on a number of assumptions.¹⁶¹ As seen in Figure 6, there is no significant correlation 266 267 with H₂O concentration. This could indicate that water is not a central ingredient in the critical 268 cluster formed from MEA and MSA. Alternatively, it could be due to the absence of an energy 269 barrier in the reaction so the slope simply reflects a lack of particle formation rate on the water 270 concentration.¹⁶¹ Furthermore, no change in the acid/base molar ratio was observed in the 271 TDCIMS measurements in the presence of water compared to the dry case (Fig. 3; blue data 272 points) indicating that the particles remained neutral. This lack of dependence on water is in contrast to previous results obtained for the small alkylamines,¹⁰² where a slope of 1.3-2.3 in the 273 274 log-log plot was observed.

275	Shen et al. ¹⁰⁸ carried out computational studies of cluster formation from MEA and MSA,
276	They showed that the 1:1 cluster was the least stable and hence formation of this cluster is the
277	rate-determining step. This is consistent with the measured rates of particle formation depending
278	on the product of the MEA and MSA concentrations and the 1:1 acid/base ratio of the particles.
279	They demonstrated that the binding of MEA and MSA was determined by a combination of
280	proton transfer from the acid MSA to the nitrogen of the MEA base, along with hydrogen
281	bonding. MEA differs from simple amines in that it has both the $-NH_2$ group as well as the $-OH$
282	group, providing more than one hydrogen-bonding opportunity to MSA. Indeed, in all acid-base
283	clusters, MSA acted as a H-bond donor and in many of the clusters, the -OH group of MEA
284	acted as a H-bond donor to MSA. This results in strongly bound clusters held together by both
285	electrostatic forces and a network of H-bonds, as illustrated in Fig. S14. It is interesting that the
286	resulting structures have the -CH ₃ group of MSA on the edge of the cluster, making the cluster
287	somewhat hydrophobic. The hydrate distribution reported by Shen et al. ¹⁰⁸ predicted that each
288	cluster was predominantly hydrated by only one water molecule even at relatively high RH
289	(80%). They also predicted that if water is present during cluster formation, it will enhance
290	particle formation by about an order of magnitude at 50% RH due to a decrease in the
291	evaporation rate of the initially formed 1:1 cluster. This predicted increase is significantly
292	greater than the factor of 1.5-1.6 measured in these experiments.

In previous studies of NPF from MSA and amines, methylamine (MA) was shown to be the most efficient of the simple alkylamines in forming particles.^{102,103,105,106} Figure 7 compares the size distributions of particles formed from the reactions of 1.4 ppb MSA with 1.5 ppb MEA or 4.8 ppb MA under dry conditions. Even with three times the amine concentration, the total concentration of particles formed from MA is 17 times smaller than from MEA. This is
consistent with previously reported theoretical calculations^{108,114,162} which predict a greater
stability of the clusters with the increased H-bonding capability of MEA and, as a consequence,
particle formation rates that are orders of magnitude higher for MSA+MEA compared to that for
MSA+MA at similar concentrations. The gas phase basicity¹⁵¹ of MEA (896.8 kJ mol⁻¹)
compared to MA (864.5 kJ mol⁻¹) also favors particle formation from MEA, along with the
increased H-bonding opportunities.

304 In previous experimental studies, water had a dramatic effect on NPF from MSA reacting with small alkylamines, 102, 104-106, 163 guite different from MEA. In the case of MA, the presence 305 306 of water during particle formation led to a large increase in both number concentration and size 307 starting at RH < 10% (Fig. S15). Calculations indicated that the 4MSA-4 MA cluster with one 308 water molecule, for example, resulted in a structure that had many potential hydrogen bonding 309 sites available, allowing the cluster to grow via H-bonding with other species.¹⁶³ For MEA, 310 however, the clusters already have strongly hydrogen-bonded internal networks so that 311 opportunities for further interactions with water molecules are reduced.

To compare the relative importance of the MEA and MA reactions with MSA for particle formation under atmospheric conditions, measurements under 10-50% RH were carried out with MA (Fig. S15). Figure 8 shows the NPF rate ($J_{>2.5nm}$) for MEA compared to that of MA as a function of the product of the reactant concentrations. The slope of the linear fit through the data for the MEA reaction is more than four times that of the MA reaction. Thus, although NPF from MA + MSA is greatly enhanced in the presence of water, the MEA reaction is still more efficient

318	under similar conditions.	This highlights the significance of alkanolamines in NPF at low
319	concentrations and points	to H-bonding as a driver for NPF with MSA.

320

321 Conclusions

322 This study shows that the acid-base interaction of a short chain alkanolamine, MEA, used in 323 CCS with MSA is quite strong and produces sub-10 nm particles extremely efficiently compared 324 to a simple primary alkylamine, methylamine. Water vapor has a limited impact on NPF rates as 325 MEA has OH- groups that already promote strong H-bonding network within the clusters. This 326 is in contrast with previous work on alkylamines where water had a large impact on nucleation 327 and growth of new particles. The particle composition from 4 - 12 nm showed an acid/base 328 molar ratio close to unity, whereas those from the MA reaction contained more acid at the 329 smaller diameters. These findings highlight that there is not a one-size-fits-all when it comes to 330 treating amine interactions with MSA in atmospheric models. 331 The overall contribution of MSA-initiated aerosol chemistry may become increasingly more important in the future.¹⁶⁴ For example, there is a reduction of sea-ice coverage at the 332 333 poles, leading to an increase in DMS emissions^{94,96-98} with an associated increase in MSA. At the 334 same time, there has been a decline in anthropogenic SO₂ emissions over few the past decades,¹⁶⁵⁻¹⁷⁰ with a related reduction in particulate sulfate in ambient particles in the Northern 335 part of the globe.^{167,168,171,172} Thus, MSA acid-base mediated NPF will become increasingly 336

337 more important in air in the near future.

Lastly, alkanolamines are being widely deployed as in CCS technology which may lead to an increase in their abundance in the atmosphere. Thus, assessing and understanding the impacts of this acid-base driven chemistry on new particle formation in air is more important than ever.

342

343 Acknowledgments

344 The authors are grateful to the National Science Foundation (Grants No. 1928252 and CHE-

345 2004066) and the Army Research Office (Grant No. W911NF2010064) for supporting this

346 research. K. R. acknowledges support from the Lucille Foundation and the Chemistry Summer

347 Undergraduate Research Fellowship (Chem-SURF). The authors thank Dr. Paulus Bauer for

348 helpful discussions during the preparation of this manuscript. The authors are also grateful to

349 Dr. Natalia Karimova and Dr. R. Benny Gerber for their assistance in the DFT calculations and

access to the Green-Planet Cluster at the University of California, Irvine.

351

352

353 Electronic Supplementary Information

The supporting information material is available free of charge on the ACS Publications website. It includes supporting texts covering additional details on the experimental methods as well as 15 figures (S1-S15).

ORCID numbers

V. Perraud	0000-0003-1247-9787
K. Roundtree	0009-0001-0013-3394
P. M. Morris	0009-0008-6823-6569
J. N. Smith	0000-0003-4677-8224
B. J. Finlayson-Pitts	0000-0003-4650-168X

361 Figure Captions

362 Figure 1. Size distributions of particles from MEA (1.5 ppb) reacting with (A) 0.68 ppb MSA, 363 (B) 1.4 ppb MSA and (C) 2.8 ppb MSA, and size distribution from MEA (3.3 ppb) reacting with 364 (D) 1.5 ppb MSA, (E) 3.3 ppb MSA and (F) 6.1 ppb MSA. Measurements were conducted at the 365 same sampling ports distributed equally along the length of the flow tube, but experiments displayed in panel (A-C) were performed with a total flow rate of 23.4 L min⁻¹ (resulting in 366 367 reaction times between 0.3 and 4.5 s), while experiments displayed in panel (D-F) were 368 performed with a total flow rate of 10.7 L min⁻¹ (resulting in reaction times between 0.5 and 369 7.7s). All experiments were performed under dry conditions, and size distributions are the 370 average of 3 to 8 replicates (error bars correspond to one standard deviation) for each reaction 371 time. All size distributions were corrected for particle losses through the sampling lines. Total 372 particle concentrations and geometric mean diameters as a function of reaction times are given in 373 Fig. S12.

374

Figure 2. Particle formation rate $(J_{>2.5 nm})$ for the MSA+MEA system under dry conditions as a function of the product of the MSA and MEA mixing ratios in ppb. Each data point represents an average over 3 to 8 individual SMPS scans taken at 2.3-2.4 s reaction time, with error bars representing one standard deviation, and corrected for particle losses through the sampling lines.

The red line is a linear fit to the data ([MSA] × [MEA] $\leq 2 \text{ ppb}^2$) with a slope of (5.3 \pm 0.03) × 10⁶ particles cm⁻³ ppb².

381

Figure 3. Acid/base molar ratio measured by the TDCIMS for the MSA+MEA system, under dry conditions (red data points) and at 52% RH (blue data points) measured at 4.5 s reaction time. MEA was measured in POS ion mode while MSA was measured in NEG ion mode, and the ion distributions in the MS spectra were similar at all reactant concentrations (Fig. S6). Text S5 provides more detailed information on how the acid/base molar ratios were estimated. All measurements were performed with initial MEA concentration of 1.5 ppb. No significant difference was observed in the measured molar ratio across for experiments performed with

- [MSA] = 0.68 ppb, [MSA] = 1.4 ppb or [MSA] = 2.8 ppb, in either dry or humid conditions; thus is the set of the set of
- the data points represent average values across the [MSA] concentrations range for each RH
- 391 condition. For each data point, the error bars represent one standard deviation. The dashed line
- 392 corresponds to an acid/base molar ratio of unity for reference.

- **Figure 4.** Evolution of the particle size distributions as a function of relative humidity (RH)
- from the reaction of MSA (0.7 ppb) with MEA (1.4 ppb). Panel (A) represents the evolution as a
- 396 function of time while panel (B) represents a snapshot of the size distributions at a given RH
- 397 (each distribution is an average over three SMPS scans with the error bars representing one

398 standard variation). All measurements were performed at 4.5 s reaction time, and particle size 399 distributions were corrected for particle loss through the sampling lines.

400

401 **Figure 5.** Size distributions (A and B) and evolution of the particle total concentrations and 402 geometric mean diameters measured as a function of the reaction time (C and D) in the flow 403 reactor for the MSA+MEA reaction system at 50% RH. All lines in panels C and D are guides to 404 the eye. All data originate from replicate scans (n = 5) and are displayed with one standard

405 deviation. All size distributions were corrected for particle losses through the sampling lines.

406

407 **Figure 6.** Logarithm of the particle formation rate $(J_{>2.5 \text{ nm}})$ for the MSA+MEA system under

408 humid conditions (RH ranging from 8 to 56%) as a function of the log of the water concentration 409 $(1 - 1)^{-3}$ $(1 - 1)^{-3}$

- 409 (in molecule cm^{-3}). Each data point represents an average over 3 to 6 individual SMPS scans
- 410 taken at 4.5 s reaction time, with error bars representing one standard deviation. All data points 411 were corrected for particle loss through the sampling line. Red data are for [MSA] = 0.68 ppb
- 411 were corrected for particle loss through the sampling line. Ked data are for [MSA] = 0.08 pp412 and [MEA] = 1.5 ppb while the blue data are for [MSA] = 2.8 ppb and [MEA] = 1.5 ppb. The
- 412 and [MEA] = 1.5 ppb while the orde data are for [MSA] = 2.8 ppb and [MEA] = 1413 slopes of the lines are 0.04 for 2.8 ppb MSA and 0.2 for 0.68 ppb MSA.

414

Figure 7. Representative averaged size distribution (red trace) from the reaction of MSA (1.4 ppb) with MEA (1.5 ppb). For comparison, a size distribution for MSA (1.4 ppb) reacting with MA (4.8 ppb) is also shown (green trace). Both size distributions were taken at ~4 s reaction time. Each size distribution was averaged over five consecutive scans and the shaded area corresponds to one standard deviation uncertainty. The thick line corresponds to a log normal fit to the averaged data. Both size distributions have been corrected for particle losses through the sampling lines.

422

423 **Figure 8**. Comparison between particle formation rate $(J_{>2.5 \text{ nm}})$ for the MSA+MEA and the

424 MSA+MA systems under humid conditions (RH ranging from 8 to 56%) as a function of the

425 product of the MSA and the amine mixing ratios in ppb. Each data point represents an average

426 over 3 to 6 individual SMPS scans taken at 4.5 s reaction time, with error bars representing one

- 427 standard deviations. The dashed lines are exponential fits to the data and the solid lines are the
- 428 tangent to the fits with slopes of 5.5×10^6 and 1.2×10^6 for the MSA+MEA and MSA+MA
- 429 systems respectively. All data points were corrected for particle loss through the sampling lines.
- 430

432 **References**

- 433 1. G. T. Rochelle, Amine Scrubbing for CO₂ Capture, *Science*, 2009, **325**, 1652-1654.
- 434 2. C. H. Yu, C. H. Huang, C. S. Tan, A review of CO₂ capture by absorption and adsorption, *Aerosol Air Qual. Res.*, 2012, 12, 745-769.
- S. A. Mazari, B. S. Ali, B. M. Jan, I. M. Saeed, S. Nizamuddin, An overview of solvent
 management and emissions of amine-based CO₂ capture technology, *Int. J. Greenh. Gas Con.*, 2015, 34, 129-140.
- 4. L. Cuccia, J. Dugay, D. Bontemps, M. Louis-Louisy, J. Vial, Analytical methods for the
 monitoring of post-combustion CO₂ capture process using amine solvents: A review, *Int.*J. Greenh. Gas Con., 2018, 72, 138-151.
- M. Corsten, A. Ramirez, L. Shen, J. Koornneef, A. Faaij, Environmental impact
 assessment of CCS chains Lessons learned and limitations from LCA literature, *Int. J. Greenh. Gas Con.*, 2013, 13, 59-71.
- M. Akram, K. Milkowski, J. Gibbins, M. Pourkashanian, Comparative energy and
 environmental performance of 40 % and 30 % monoethanolamine at PACT pilot plant, *Int. J. Greenh. Gas Con.*, 2020, **95**, ARTN 102946, doi:
 102910.101016/j.ijggc.102019.102946.
- L. Zhu, G. W. Schade, C. J. Nielsen, Real-time monitoring of emissions from
 monoethanolamine-based industrial scale carbon capture facilities, *Environ. Sci. Technol.*, 2013, 47, 14306-14314.
- 452 8. J. Mertens, J. Knudsen, M. L. Thielens, J. Andersen, On-line monitoring and controlling
 453 emissions in amine post combustion carbon capture: A field test, *Int. J. Greenh. Gas*454 *Con.*, 2012, 6, 2-11.
- J. Mertens, H. Lepaumier, D. Desagher, M. L. Thielens, Understanding ethanolamine
 (MEA) and ammonia emissions from amine based post combustion carbon capture:
 Lessons learned from field tests, *Int. J. Greenh. Gas Con.*, 2013, 13, 72-77.
- E. F. da Silva, H. Kolderup, E. Goetheer, K. W. Hjarbo, A. Huizinga, P. Khakharia, I.
 Tuinman, T. Mejdell, K. Zahlsen, K. Vernstad, A. Hyldbakk, T. Holten, H. M. Kvamsdal,
 P. van Os, A. Einbu, Emission studies from a CO₂ capture pilot plant, *Energy Procedia*,
 2013, 37, 778-783.
- J. Fagerlund, R. Zevenhoven, J. Thomassen, M. Tednes, F. Abdollahi, L. Thomas, C. J.
 Nielsen, T. Mikoviny, A. Wisthaler, L. Zhu, C. Biliyok, A. Zhurkin, Performance of an amine-based CO₂ capture pilot plant at the Fortum Oslo Varme Waste to Energy plant in Oslo, Norway, *Int. J. Greenh. Gas Con.*, 2021, **106**, Art No 103242.
- M. X. Fang, N. T. Yi, W. T. Di, T. Wang, Q. H. Wang, Emission and control of flue gas pollutants in CO₂ chemical absorption system A review, *Int. J. Greenh. Gas Con.*, 2020, 93, Art No 102904.
- 469 13. A. K. Morken, B. Nenseter, S. Pedersen, M. Chhaganlal, J. K. Feste, R. B. Tyborgnes, O.
 470 Ullestad, H. Ulvatn, L. Zhu, T. Mikoviny, A. Wisthaler, T. Cents, O. M. Bade, J.
- 471 Knudsen, G. de Koeijer, O. Falk-Pedersen, E. S. Hamborg, Emission results of amine
- 472 plant operations from MEA testing at the CO₂ Technology Centre Mongstad, *Energy* 473 *Procedia*, 2014, **63**, 6023-6038.

474	14.	K. Veltman, B. Singh, E. G. Hertwich, Human and environmental impact assessment of
475		postcombustion CO ₂ capture focusing on emissions from amine-based scrubbing solvents
476		to air, Environ. Sci. Technol., 2010, 44, 1496-1502.
477	15.	M. M. Fiume, B. A. Heldreth, W. F. Bergfeld, D. V. Belsito, R. A. Hill, C. D. Klaassen,
478		D. C. Liebler, J. G. Marks, R. C. Shank, T. J. Slaga, P. W. Snyder, F. A. Andersen, Safety
479		assessment of ethanolamine and ethanolamine salts as used in cosmetics, Int. J. Toxicol.,
480		2015, 34 , 84s-98s.
481	16.	B. C. McDonald, J. A. de Gouw, J. B. Gilman, S. H. Jathar, A. Akherati, C. D. Cappa, J.
482		L. Jimenez, J. Lee-Taylor, P. L. Hayes, S. A. McKeen, Y. Y. Cui, S. W. Kim, D. R.
483		Gentner, G. Isaacman-VanWertz, A. H. Goldstein, R. A. Harley, G. J. Frost, J. M.
484		Roberts, T. B. Ryerson, M. Trainer, Volatile chemical products emerging as largest
485		petrochemical source of urban organic emissions, Science, 2018, 359, 760-764.
486	17.	J. Wooley, W. W. Nazaroff, A. T. Hodgson, Release of ethanol to the atmosphere during
487		use of consumer cleaning products, J. Air Waste Manage. Assoc., 1990, 40, 1114-1120.
488	18.	X. Ge, A. S. Wexler, S. L. Clegg, Atmospheric amines - Part I. A review, Atmos.
489		<i>Environ.</i> , 2011, 45 , 524-546.
490	19.	JA. Seo, IH. Bae, WH. Jang, JH. Kim, SY. Bak, SH. Han, YH. Park, KM.
491		Lim, Hydrogen peroxide and monoethanolamine are the key causative ingredients for
492		hair dye-related dermatitis and hair loss, J. Dermatol. Sci., 2012, 66, 12-19.
493	20.	N. Borduas, J. P. D. Abbatt, J. G. Murphy, Gas phase oxidation of monoethanolamine
494		(MEA) with OH radical and ozone: kinetics, products, and particles, Environ. Sci.
495		Technol., 2013, 47, 6377-6383.
496	21.	M. Karl, C. Dye, N. Schmidbauer, A. Wisthaler, T. Mikoviny, B. D'Anna, M. Muller, E.
497		Borras, E. Clemente, A. Munoz, R. Porras, M. Rodenas, M. Vazquez, T. Brauers, Study
498		of OH-initiated degradation of 2-aminoethanol, Atmos. Chem. Phys., 2012, 12, 1881-
499		1901.
500	22.	L. Onel, M. A. Blitz, P. W. Seakins, Direct determination of the rate coefficient for the
501		reaction of OH radicals with monoethanol amine (MEA) from 296 to 510 K, J. Phys.
502		<i>Chem. Lett.</i> , 2012, 3 , 853-856.
503	23.	C. J. Nielsen, B. D'Anna, C. Dye, M. Graus, M. Karl, S. King, M. M. Maguto, M. Muller,
504		N. Schmidbauer, Y. Stenstrom, A. Wisthaler, S. Pedersen, Atmospheric chemistry of 2-
505		aminoethanol (MEA). Energy Procedia, 2011, 4, 2245-2252.
506	24.	S. M. Murphy, A. Sorooshian, J. H. Kroll, N. L. Ng, P. Chhabra, C. Tong, J. D. Surratt,
507		E. Knipping, R. C. Flagan, J. H. Seinfeld, Secondary aerosol formation from atmospheric
508		reactions of aliphatic amines, Atmos. Chem. Phys., 2007, 7, 2313-2337.
509	25.	X. M. Tian, Y. X. Chu, C. K. Chan, Reactive uptake of monoethanolamine by sulfuric
510		acid particles and hygroscopicity of monoethanolaminium salts, <i>Environ. Sci. Tech. Let.</i>
511		2022, 9, 16-21.
512	26.	H. B. Xie, J. Elm, R. Halonen, N. Myllys, T. Kurten, M. Kulmala, H. Vehkamaki,
513		Atmospheric fate of monoethanolamine: enhancing new particle formation of sulfuric
514		acid as an important removal process, Environ. Sci. Technol., 2017, 51, 8422-8431.
515	27.	B. Rosati, S. Christiansen, R. W. de Jonge, P. Roldin, M. M. Jensen, K. Wang, S. P.
516		Moosakutty, D. Thomsen, C. Salomonsen, N. Hyttinen, J. Elm, A. Feilberg, M. Glasius,

517		M. Bilde, New particle formation and growth from dimethyl sulfide oxidation by
518		hydroxyl radicals, ACS Earth Space Chem., 2021, 5, 801-811.
519	28.	P. Van Rooy, R. Drover, T. Cress, C. Michael, K. L. Purvis-Roberts, P. J. Silva, M. J.
520		Nee, D. Cocker, Methanesulfonic acid and sulfuric acid aerosol formed through oxidation
521		of reduced sulfur compounds in a humid environment, Atmos. Environ., 2021, 261, Art.
522		N0 118504.
523	29.	E. H. Hoffmann, A. Tilgner, R. Schrodner, P. Brauera, R. Wolke, H. Herrmann, An
524		advanced modeling study on the impacts and atmospheric implications of multiphase
525		dimethyl sulfide chemistry <i>Proc. Natl. Acad. Sci. USA</i> , 2016 113 11776-11781
526	30	I Barnes J Hiorth N Mihalopoulos Dimethyl sulfide and dimethyl sulfoxide and their
527		oxidation in the atmosphere <i>Chem Rev</i> 2006 106 940-975
528	31	R Wollesen de Jonge I Elm B Rosati S Christiansen N Hyttinen D Ludemann M
529	51.	Rilde P Roldin Secondary aerosol formation from dimethyl sulfide - improved
530		mechanistic understanding based on smog chamber experiments and modelling <i>Atmos</i>
531		Chem Phys 2021 21 9955–9976
532	32	P R Veres I A Neuman T H Bertram E Assaf G M Wolfe C I Williamson B
532	52.	Weinzierl S Tilmes C R Thompson A B Thames I C Schroder A Saiz-Lonez A
534		W Rollins I M Roberts D Price I Peischl B A Nault K H Moller D O Miller S
535		Meinardi O Y Li I F Lamarque A Kunc H G Kiaergaard D Kinnison I L
536		limenez C M Jernigan R S Hornbrook A Hills M Dollner D A Day C A
537		Cuevas P Campuzano-Jost I Burkholder T P Bui W H Brune S S Brown C A
538		Brock I Bourgeois D R Blake E C Apel T B Rverson Global airborne sampling
539		reveals a previously unobserved dimethyl sulfide oxidation mechanism in the marine
540		atmosphere Proc Natl Acad Sci USA 2020 117 4505-4510
541	33	S F Watts The mass budgets of carbonyl sulfide dimethyl sulfide carbon disulfide and
542	001	hydrogen sulfide. <i>Atmos. Environ.</i> , 2000, 34 , 761-779.
543	34	R J Charlson J E Lovelock M O Andreae S G Warren Oceanic phytoplankton
544	-	atmospheric sulfur, cloud albedo and climate. <i>Nature</i> , 1987, 326 , 655-661.
545	35.	T. S. Bates, B. K. Lamb, A. Guenther, J. Dignon, R. E. Stoiber, Sulfur emissions to the
546		atmosphere from natural sources, J. Atmos. Chem., 1992, 14, 315-337.
547	36.	A. Lana, T. G. Bell, R. Simo, S. M. Vallina, J. Ballabrera-Pov, A. J. Kettle, J. Dachs, L.
548		Bopp, E. S. Saltzman, J. Stefels, J. E. Johnson, P. S. Liss, An updated climatology of
549		surface dimethlysulfide concentrations and emission fluxes in the global ocean. <i>Global</i>
550		Biogeochem, Cycles, 2011, 25, doi: 10.1029/2010gb003850.
551	37.	M. O. Andreae, P. J. Crutzen, Atmospheric aerosols: Biogeochemical sources and role in
552		atmospheric chemistry. Science, 1997, 276, 1052-1058.
553	38.	V. P. Aneia. Natural sulfur emissions into the atmosphere. J. Air Waste Manage. Assoc.
554		1990. 40 . 469-476.
555	39.	K. Jardine, A. M. Yanez-Serrano, J. Williams, N. Kunert, A. Jardine, T. Taylor, L.
556		Abrell, P. Artaxo, A. Guenther, C. N. Hewitt, E. House, A. P. Florentino, A. Manzi, N.
557		Higuchi, J. Kesselmeier, T. Behrendt, P. R. Veres, B. Derstroff, J. D. Fuentes, S. T.
558		Martin, M. O. Andreae, Dimethyl sulfide in the Amazon rain forest. <i>Global Biogeochem</i>
559		<i>Cvcles</i> , 2015, 29 , 19-32.

560 40. P. J. Crutzen, J. Williams, U. Poschl, P. Hoor, H. Fischer, C. Warneke, R. Holzinger, A. 561 Hansel, W. Lindinger, B. Scheeren, J. Lelieveld, High spatial and temporal resolution measurements of primary organics and their oxidation products over the tropical forests 562 of Surinam, Atmos. Environ., 2000, 34, 1161-1165. 563 564 41. J. Kesselmeier, F. X. Meixner, U. Hofmann, A. L. Ajavon, S. Leimbach, M. O. Andreae, 565 Reduced sulfur compound exchange between the atmosphere and tropical tree species in 566 Southern Cameroon, Biogeochemistry, 1993, 23, 23-45. 567 42. S. Meinardi, I. J. Simpson, N. J. Blake, D. R. Blake, F. S. Rowland, Dimethyl disulfide 568 (DMDS) and dimethyl sulfide (DMS) emissions from biomass burning in Australia, 569 Geosphys. Res. Lett., 2003, 30, Art. N0 1454 (1451-1454). 570 43. C. E. Stockwell, P. R. Veres, J. Williams, R. J. Yokelson, Characterization of biomass 571 burning emissions from cooking fires, peat, crop residue, and other fuels with high-572 resolution proton-transfer-reaction time-of-flight mass spectrometry, Atmos. Chem. Phys., 573 2015, 15, 845-865. 574 44. V. Perraud, S. Meinardi, D. R. Blake, B. J. Finlayson-Pitts, Challenges associated with 575 the sampling and analysis of organosulfur compounds in air using real-time PTR-ToF-576 MS and offline GC-FID, Atmos. Meas. Tech., 2016, 9, 1325-1340. 577 S. Trabue, K. Scoggin, F. Mitloehner, H. Li, R. Burns, H. Xin, Field sampling method for 45. 578 quantifying volatile sulfur compounds from animal feeding operations, Atmos. Environ., 579 2008, 42, 3332-3341. 580 46. P. Hobbs, T. Mottram, New Directions: Significant contributions of dimethyl sulphide 581 from livestock to the atmosphere, Atmos. Environ., 2000, 34, 3649-3650. 582 47. J. S. Vandergheynst, D. J. Cogan, P. J. Defelice, J. M. Gossett, L. P. Walker, Effect of 583 process management on the emission of organosulfur compounds and gaseous 584 antecedents from composting processes, Environ. Sci. Technol., 1998, 32, 3713-3718. 585 48. Z. G. Yi, X. M. Wang, G. Y. Sheng, H. M. Fu, Exchange of carbonyl sulfide (OCS) and 586 dimethyl sulfide (DMS) between rice paddy fields and the atmosphere in subtropical 587 China, Agr Ecosyst Environ, 2008, 123, 116-124. 588 49. P. D. Goldan, W. C. Kuster, D. L. Albritton, F. C. Fehsenfeld, The measurement of 589 natural sulfur emissions from soils and vegetation - 3 Sites in the Eastern-United-States 590 revisited, J. Atmos. Chem., 1987, 5, 439-467. 591 J. Williams, N. Y. Wang, R. J. Cicerone, K. Yagi, M. Kurihara, F. Terada, Atmospheric 50. 592 methyl halides and dimethyl sulfide from cattle, *Global Biogeochem*. Cycles, 1999, 13, 593 485-491. 594 51. K. C. Li, D. Shooter, Analysis of sulfur-containing compounds in ambient air using solid-595 phase microextraction and gas chromatography with pulsed flame photometric detection, 596 Int. J. Environ. Anal. Chem., 2004, 84, 749-760. 597 52. M. R. Ras, F. Borrull, R. M. Marce, Determination of volatile organic sulfur compounds 598 in the air at sewage management areas by thermal desorption and gas chromatography-599 mass spectrometry, Talanta, 2008, 74, 562-569. 600 53. E. Smet, H. Van Langenhove, Abatement of volatile organic sulfur compounds in 601 odorous emissions from the bio-industry, *Biodegradation*, 1998, 9, 273-284. R. Raiswell, S. H. Bottrel, The disposal of flue gas desulphurisation waste: sulphur gas 602 54. 603 emissions and their control, Environ. Geochem. Health, 1991, 13, 119-126.

604	55.	P. R. Mulay, P. Cavicchia, S. M. Watkins, A. Tovar-Aguilar, M. Wiese, G. M. Calvert,
605		Acute illness associated with exposure to a new soil fumigant containing dimethyl
606		disulfide - Hillsborough County, Florida, 2014, J Agromedicine, 2016, 21, 373-379.
607	56.	D. D. Yan, A. C. Cao, Q. X. Wang, Y. Li, C. B. Ouyang, M. X. Guo, X. Q. Guo,
608		Dimethyl disulfide (DMDS) as an effective soil fumigant against nematodes in China,
609		Plos One, 2019, 14.
610	57.	J. Fritsch, T. Fouillet, P. Charles, P. Fargier-Puech, C. Ramponi-Bur, S. Descamps, G. Du
611		Fretay, A. Myrta, French experiences with dimethyl disulfide (DMDS) as a nematicide in
612		vegetable crops, Acta Hortic, 2014, 1044 , 427-433.
613	58.	F. L. Suarez, J. K. Furne, J. Springfield, M. D. Levitt, Morning breath odor: Influence of
614		treatments on sulfur gases, J. Dent. Res., 2000, 79, 1773-1777.
615	59.	S. Meinardi, K. B. Jin, B. Barletta, D. R. Blake, N. D. Vaziri, Exhaled breath and fecal
616		volatile organic biomarkers of chronic kidney disease, Biochim. Biophys. Acta, Gen.
617		Subi., 2013, 1830 , 2531-2537.
618	60.	H. Berresheim, M. Adam, C. Monahan, C. O'Dowd, J. M. C. Plane, B. Bohn, F. Rohrer,
619		Missing SO ₂ oxidant in the coastal atmosphere? - Observations from high-resolution
620		measurements of OH and atmospheric sulfur compounds, Atmos. Chem. Phys., 2014, 14,
621		12209-12223.
622	61.	H. Berresheim, T. Elste, H. G. Tremmel, A. G. Allen, H. C. Hansson, K. Rosman, M. Dal
623		Maso, J. M. Makela, M. Kulmala, C. D. O'Dowd, Gas-aerosol relationships of H ₂ SO ₄ ,
624		MSA, and OH: Observations in the coastal marine boundary layer at Mace Head, Ireland,
625		J. Geophys. Res., 2002, 107, Art N0 D19, 8100 (PAR 8105-8101 - PAR 8105-8112).
626	62.	F. L. Eisele, D. J. Tanner, Measurement of the gas-phase concentration of H ₂ SO ₄ and
627		methane sulfonic acid and estimates of H_2SO_4 production and loss in the atmosphere., J.
628		Geophys. Res., 1993, 98 , 9001-9010.
629	63.	A. Jefferson, D. J. Tanner, F. L. Eisele, D. D. Davis, G. Chen, J. Crawford, J. W. Huey,
630		A. L. Torres, H. Berresheim, OH photochemistry and methane sulfonic acid formation in
631		the coastal Antarctic boundary layer, J. Geophys. Res., 1998, 103, 1647-1656.
632	64.	C. D. O'Dowd, K. Hameri, J. M. Makela, L. Pirjola, M. Kulmala, S. G. Jennings, H.
633		Berresheim, H. C. Hansson, G. de Leeuw, G. J. Kunz, A. G. Allen, C. N. Hewitt, A.
634		Jackson, Y. Viisanen, T. Hoffmann, A dedicated study of New Particle Formation and
635		Fate in the Coastal Environment (PARFORCE): Overview of objectives and
636		achievements, J. Geophys. Res., 2002, 107, Art No 8108.
637	65.	H. Bardouki, H. Berresheim, M. Vrekoussis, J. Sciare, G. Kouvarakis, K. Oikonomou, J.
638		Schneider, N. Mihalopoulos, Gaseous (DMS, MSA, SO ₂ , H ₂ SO ₄ and DMSO) and
639		particulate (sulfate and methanesulfonate) sulfur species over the northeastern coast of
640		Crete, Atmos. Chem. Phys., 2003, 3, 1871-1886.
641	66.	L. J. Beck, N. Sarnela, H. Junninen, C. J. M. Hoppe, O. Garmash, F. Bianchi, M. Riva, C.
642		Rose, O. Perakyla, D. Wimmer, O. Kausiala, T. Jokinen, L. Ahonen, J. Mikkila, J.
643		Hakala, X. C. He, J. Kontkanen, K. K. E. Wolf, D. Cappelletti, M. Mazzola, R. Traversi,
644		C. Petroselli, A. P. Viola, V. Vitale, R. Lange, A. Massling, J. K. Nojgaard, R. Krejci, L.
645		Karlsson, P. Zieger, S. Jang, K. Lee, V. Vakkari, J. Lampilahti, R. C. Thakur, K. Leino, J.
646		Kangasluoma, E. M. Duplissy, E. Siivola, M. Marbouti, Y. J. Tham, A. Saiz-Lopez, T.
647		Petaja, M. Ehn, D. R. Worsnop, H. Skov, M. Kulmala, V. M. Kerminen, M. Sipila,
		26

648		Differing mechanisms of new particle formation at two Arctic sites, Geosphys. Res. Lett.,
649		2021, 48 , Art. No e2020GL091334.
650 651	67.	M. J. Lawler, M. P. Rissanen, M. Ehn, R. L. Mauldin, N. Sarnela, M. Sipilä, J. N. Smith, Evidence for diverse biogeochemical drivers of boreal forest new particle formation,
652		Geosphys. Res. Lett., 2018, 45, 2038-2046.
653	68.	R. L. Mauldin, C. A. Cantrell, M. Zondlo, E. Kosciuch, F. L. Eisele, G. Chen, D. Davis,
654		R. Weber, J. Crawford, D. Blake, A. Bandy, D. Thornton, Highlights of OH, H ₂ SO ₄ , and
655		methane sulfonic acid measurements made aboard the NASA P-3B during Transport and
656		Chemical Evolution over the Pacific, J. Geophys. Res., 2003, 108, Art. No 8796.
657	69.	L. L. J. Quelever, L. Dada, E. Asmi, J. Lampilahti, T. Chan, J. E. Ferrara, G. E. Copes, G.
658		Perez-Fogwill, L. Barreira, M. Aurela, D. R. Worsnop, T. Jokinen, M. Sipila,
659		Investigation of new particle formation mechanisms and aerosol processes at Marambio
660		Station, Antarctic Peninsula, Atmos. Chem. Phys., 2022, 22, 8417-8437.
661	70.	D. Davis, G. Chen, P. Kasibhatla, A. Jefferson, D. Tanner, F. Eisele, D. Lenschow, W.
662		Neff, H. Berresheim, DMS oxidation in the Antarctic marine boundary layer:
663		Comparison of model simulations and field observations of DMS, DMSO, DMSO ₂ ,
664		H ₂ SO ₄ (g), MSA(g), and MSA(p), J. Geophys. Res., 1998, 103, 1657-1678.
665	71.	E. S. Saltzman, D. L. Savoie, R. G. Zika, J. M. Prospero, Methane sulfonic acid in the
666		marine atmosphere, J. Geophys. Res-Oceans, 1983, 88, 897-902.
667	72.	S. Huang, L. Poulain, D. van Pinxteren, M. van Pinxteren, Z. J. Wu, H. Herrmann, A.
668		Wiedensohler, Latitudinal and seasonal distribution of particulate MSA over the Atlantic
669		using a validated quantification method with HR-ToF-AMS, Environ. Sci. Technol.,
670		2017, 51 , 418-426.
671	73.	L. Phinney, W. R. Leaitch, U. Lohmann, H. Boudries, D. R. Worsnop, J. T. Jayne, D.
672		Toom-Sauntry, M. Wadleigh, S. Sharma, N. Shantz, Characterization of the aerosol over
673		the sub-arctic north east Pacific Ocean, Deep-Sea Res Pt II, 2006, 53, 2410-2433.
674	74.	A. Sorooshian, L. T. Padro, A. Nenes, G. Feingold, A. McComiskey, S. P. Hersey, H.
675		Gates, H. H. Jonsson, S. D. Miller, G. L. Stephens, R. C. Flagan, J. H. Seinfeld, On the
676		link between ocean biota emissions, aerosol, and maritime clouds: Airborne, ground, and
677		satellite measurements off the coast of California, Global Biogeochem. Cycles, 2009, 23,
678		1-15.
679	75.	D. D. Huang, Y. J. Li, B. P. Lee, C. K. Chan, Analysis of organic sulfur compounds in
680		atmospheric aerosols at the HKUST supersite in Hong Kong using HR-ToF-AMS.
681		<i>Environ. Sci. Technol.</i> , 2015, 49 , 3672-3679.
682	76.	M. D. Willis, J. Burkart, J. L. Thomas, F. Kollner, J. Schneider, H. Bozem, P. M. Hoor,
683		A. A. Aliabadi, H. Schulz, A. B. Herber, W. R. Leaitch, J. P. D. Abbatt, Growth of
684		nucleation mode particles in the summertime Arctic: a case study. <i>Atmos. Chem. Phys.</i> .
685		2016. 16 . 7663-7679.
686	77.	L. C. Maudlin, Z. Wang, H. H. Jonsson, A. Sorooshian, Impact of wildfires on size-
687		resolved aerosol composition at a coastal California site. <i>Atmos. Environ.</i> , 2015, 119 , 59-
688		68.
689	78.	A. Sorooshian, E. Crosbie, L. C. Maudlin, J. S. Youn, Z. Wang, T. Shingler, A. M.
690		Ortega, S. Hersey, R. K. Woods, Surface and airborne measurements of organosulfur and

691		methanesulfonate over the western United States and coastal areas <i>I Geophys Res</i>
692		2015 120 8535-8548
693	79	C Stahl M T Cruz P A Banaga G Betito R A Braun M A Aghdam M O
694	12.	Cambaliza G R Lorenzo A B MacDonald M R A Hilario P C Pabroa J R Yee
695		J B Simpas A Sorooshian Sources and characteristics of size-resolved particulate
696		organic acids and methanesulfonate in a coastal megacity: Manila, Philippines, Atmos.
697		<i>Chem. Phys.</i> , 2020, 20 , 15907-15935.
698	80.	C. J. Gaston, K. A. Pratt, X. Y. Oin, K. A. Prather, Real-time detection and mixing state
699		of methanesulfonate in single particles at an inland urban location during a phytoplankton
700		bloom, Environ. Sci. Technol., 2010, 44, 1566-1572.
701	81.	H. Yuan, Y. Wang, G. S. Zhuang, MSA in Beijing aerosol, Chin. Sci. Bull., 2004, 49,
702		1020-1025.
703	82.	V. M. Kerminen, M. Aurela, R. E. Hillamo, A. Virkkula, Formation of particulate MSA:
704		Deductions from size distribution measurements in the Finnish Arctic, <i>Tellus B</i> , 1997, 49,
705		159-171.
706	83.	M. J. Lawler, E. S. Saltzman, L. Karlsson, P. Zieger, M. Salter, A. Baccarini, J. Schmale,
707		C. Leck, New insights into the composition and origins of ultrafine aerosol in the
708		summertime high Arctic, Geosphys. Res. Lett., 2021, 48, Art No e2021GL094395.
709	84.	J. Burkart, M. D. Willis, H. Bozem, J. L. Thomas, K. Law, P. Hoor, A. A. Aliabadi, F.
710		Kollner, J. Schneider, A. B. Herber, J. D. Abbatt, W. R. Leaitch, Summertime
711		observations of elevated levels of ultrafine particles in the high Arctic marine boundary
712		layer, Atmos. Chem. Phys., 2017, 17, 5515-5535.
713	85.	K. T. Park, S. Jang, K. Lee, Y. J. Yoon, M. S. Kim, K. Park, H. J. Cho, J. H. Kang, R.
714		Udisti, B. Y. Lee, K. H. Shin, Observational evidence for the formation of DMS-derived
715		aerosols during Arctic phytoplankton blooms, Atmos. Chem. Phys., 2017, 17, 9665-9675.
716	86.	X. X. Li, Y. Y. Li, M. J. Lawler, J. M. Hao, J. N. Smith, J. K. Jiang, Composition of
717		ultrafine particles in urban Beijing: Measurement using a thermal desorption chemical
718		ionization mass spectrometer, Environ. Sci. Technol., 2021, 55, 2859-2868.
719	87.	T. A. Pakkanen, V. M. Kerminen, C. H. Korhonen, R. E. Hillamo, P. Aarnio, T.
720		Koskentalo, W. Maenhaut, Urban and rural ultrafine ($PM(0.1)$) particles in the Helsinki
721		area, Atmos. Environ., 2001, 35 , 4593-4607.
722	88.	K. N. Fossum, J. Ovadnevaite, D. Ceburnis, M. Dall'Osto, S. Marullo, M. Bellacicco, R.
723		Simo, D. T. Liu, M. Flynn, A. Zuend, C. O'Dowd, Summertime primary and secondary
724		contributions to Southern ocean cloud condensation nuclei, <i>Sci. Rep.</i> , 2018, 8, Art No
725	0.0	
726	89.	J. M. Makela, S. Yli-Koivisto, V. Hiltunen, W. Seidl, E. Swietlicki, K. Teinila, M.
121		Sillanpaa, I. K. Koponen, J. Paatero, K. Rosman, K. Hameri, Chemical composition of
128	00	aerosol during particle formation events in boreal forest, <i>Tellus B</i> , 2001, 53 , 380-393.
129	90.	M. Dall'Usto, D. C. S. Beddows, P. Tunved, K. Krejci, J. Strom, H. C. Hansson, Y. J.
/30		1 0011, K. 1. Park, S. Becagii, K. Uuisu, 1. Unasch, C. D. U'Dowd, K. Simo, K. M.
131		namson, Arctic sea ice men leads to atmospheric new particle formation, <i>Sci. Rep.</i> ,
132		2017, 7, 1-10.

733 734	91.	P. K. Quinn, T. L. Miller, T. S. Bates, J. A. Ogren, E. Andrews, G. E. Shaw, A 3-year record of simultaneously measured aerosol chemical and ontical properties at Barrow
735		Alaska L Geophys Res 2002 107 Art No 4130
736	02	W R Legitch S Sharma I Huang D Toom-Sauntry A Chivulescu A M
730	12.	Macdonald K von Salzen I R Pierce A K Bertram I C Schroder N C Shantz R
738		V-W Chang A J. Norman Dimethyl sulfide control of the clean summertime Arctic
730		aerosol and cloud <i>Elementa Sci. Anthron.</i> 2013. 1 , 1, 12
739	03	M Dall'Osto P Simo P M Harrison D C S Beddows A Saiz Lonez P Lange H
740	95.	M. Dali Osto, R. Sillio, K. M. Hallisoli, D. C. S. Beudows, A. Salz-Lopez, R. Lange, H.
741		skov, J. K. Nojgaald, I. E. Mielsell, A. Massing, Abiotic and blotte sources influencing
742		spring new particle formation in North East Oreemand, Atmos. Environ., 2018, 190, 120-
745	04	154. M.D. Willig W. D. Logitch, I. D. D. Abbett Processor controlling the composition and
744	94.	M. D. Willis, W. K. Lealten, J. P. D. Abbau, Processes controlling the composition and
745	05	abundance of Arcuc aerosol, <i>Rev. Geophys.</i> , 2018, 50 , 021-071.
/40	95.	Q. Chen, T. Sherwen, M. Evans, B. Alexander, DIVIS oxidation and sulfur aerosol
/4/		iormation in the marine troposphere: a locus on reactive halogen and multiphase
748	0(chemistry, Atmos. Chem. Phys., 2018, 18, 1301/-1303/.
749	96.	M. Gali, E. Devred, M. Babin, M. Levasseur, Decadal increase in Arctic dimethylsuinde
/50	07	emission, Proc. Natl. Acaa. Sci. USA, 2019, 116, 19311-19317.
/51	97.	S. Sharma, E. Chan, M. Ishizawa, D. Toom-Sauntry, S. L. Gong, S. M. Li, D. W.
152		Tarasick, W. R. Leaitch, A. Norman, P. K. Quinn, T. S. Bates, M. Levasseur, L. A.
/53		Barrie, W. Maennaut, influence of transport and ocean ice extent on biogenic aerosol
/54	00	sultur in the Arctic atmosphere, J. Geophys. Res., 2012, 117, Art No. D12209.
/55	98.	S. Becagli, L. Lazzara, C. Marchese, U. Dayan, S. E. Ascanius, M. Cacciani, L. Caiazzo,
756		C. Di Biagio, T. Di Iorio, A. di Sarra, P. Eriksen, F. Fani, F. Giardi, D. Meloni, G.
/5/		Muscari, G. Pace, M. Severi, R. Iraversi, R. Udisti, Relationships linking primary
758		production, sea ice melting, and biogenic aerosol in the Arctic, Atmos. Environ., 2016,
759	0.0	136 , 1-15.
760	99.	K. D. Arquero, R. B. Gerber, B. J. Finlayson-Pitts, The role of oxalic acid in new particle
761		formation from methanesulfonic acid, methylamine, and water, <i>Environ. Sci. Technol.</i> ,
762		2017, 51 , 2124-2130.
763	100.	K. D. Arquero, J. Xu, R. B. Gerber, B. J. Finlayson-Pitts, Particle formation and growth
764		from oxalic acid, methanesulfonic acid, trimethylamine and water: a combined
765		experimental and theoretical study, <i>Phys. Chem. Chem. Phys.</i> , 2017, 19 , 28286-28301.
766	101.	H. Chen, M. J. Ezell, K. D. Arquero, M. E. Varner, M. L. Dawson, R. B. Gerber, B. J.
767		Finlayson-Pitts, New particle formation and growth from methanesulfonic acid,
768		trimethylamine and water, Phys. Chem. Chem. Phys., 2015, 17, 13699-13709.
769	102.	H. Chen, M. E. Varner, R. B. Gerber, B. J. Finlayson-Pitts, Reactions of methanesulfonic
770		acid with amines and ammonia as a source of new particles in air, J. Phys. Chem. B,
771		2016, 120 , 1526-1536.
772	103.	H. H. Chen, B. J. Finlayson-Pitts, New particle formation from methanesulfonic acid and
773		amines/ammonia as a function of temperature, Environ. Sci. Technol., 2017, 51, 243-252.
774	104.	M. L. Dawson, M. E. Varner, V. Perraud, M. J. Ezell, R. B. Gerber, B. J. Finlayson-Pitts,
775		Simplified mechanism for new particle formation from methanesulfonic acid, amines,

776 777		and water via experiments and ab initio calculations, <i>Proc. Natl. Acad. Sci. USA</i> , 2012, 100, 18710, 18724
779	105	V Derroud V V Li I K Jiang P I Finlawson Ditts I N Smith Size resolved
770	105.	v. Ferraud, A. A. Li, J. K. Jiang, D. J. Finalyson-Fitts, J. N. Shiftin, Size-resolved
790		methylemine and emmonie ACS Fauth Space Chem. 2020. 4, 1192, 1104
701	106	N Dermand L Vu D. D. Center D. J. Einlessen Ditts. Integrated superimental and
/01	100.	V. Perfaud, J. Au, K. B. Gerber, B. J. Filliayson-Pills, integrated experimental and
/82		theoretical approach to probe the synergistic effect of ammonia in methanesultonic actu
/83	107	reactions with small alkylamines, <i>Environ. Sci. Processes Impacts</i> , 2020, 22, 305-328.
/84	107.	J. S. Johnson, C. N. Jen, Role of methanesultonic acid in sulfuric acid-amine and
/85	100	ammonia new particle formation, ACS Earth Space Chem., 2023, 7, 653-660.
/86	108.	J. Shen, HB. Xie, J. Elm, F. Ma, J. Chen, H. Vehkamaki, Methanesultonic acid-driven
787		new particle formation enhanced by monoethanolamine: A computational study,
788	100	Environ. Sci. Technol., 2019, 53 , 14387-14397.
789	109.	J. W. Shen, J. Elm, H. B. Xie, J. W. Chen, J. F. Niu, H. Vehkamaki, Structural effects of
790		amines in enhancing methanesulfonic acid-driven new particle formation, <i>Environ. Sci.</i>
791		<i>Technol.</i> , 2020, 54 , 13498-13508.
792	110.	H. L. Zhao, X. T. Jiang, L. Du, Contribution of methane sulfonic acid to new particle
793		formation in the atmosphere, <i>Chemosphere</i> , 2017, 174 , 689-699.
794	111.	F. R. Rasmussen, J. Kubecka, J. Elm, Contribution of methansulfonic acid to the
795		formation of molecular clusters in the marine atmosphere, J. Phys. Chem. A, 2022, 126,
796		7127-7136.
797	112.	J. Elm, Clusteromics II: Methansulfonic acid-base cluster formation, ACS Omega, 2021,
798		6 , 17035-17044.
799	113.	R. Zhang, J. Shen, HB. Xie, J. Chen, J. Elm, The role of organic acids in new particle
800		formation from methanesulfonic acid and methylamine, Atmos. Chem. Phys., 2022, 22,
801		2639-2650.
802	114.	S. Chee, K. Barsanti, J. N. Smith, N. Myllys, A predictive model for salt nanoparticle
803		formation using heterodimer stability calculations, Atmos. Chem. Phys., 2021, 21, 11637-
804		11654.
805	115.	D. P. Chen, D. F. Li, C. W. Wang, F. Y. Liu, W. L. Wang, Formation mechanism of
806		methanesulfonic acid and ammonia clusters: A kinetics simulation study, Atmos.
807		<i>Environ.</i> , 2020, 222 .
808	116.	M. van Pinxteren, B. Fiedler, D. van Pinxteren, Y. Iinuma, A. Kortzinger, H. Herrmann,
809		Chemical characterization of sub-micrometer aerosol particles in the tropical Atlantic
810		Ocean: marine and biomass burning influences, J. Atmos. Chem., 2015, 72, 105-125.
811	117.	C. Muller, Y. Iinuma, J. Karstensen, D. van Pinxteren, S. Lehmann, T. Gnauk, H.
812		Herrmann, Seasonal variation of aliphatic amines in marine sub-micrometer particles at
813		the Cape Verde islands, Atmos. Chem. Phys., 2009, 9, 9587-9597.
814	118.	M. C. Facchini, S. Decesari, M. Rinaldi, C. Carbone, E. Finessi, M. Mircea, S. Fuzzi, F.
815		Moretti, E. Tagliavini, D. Ceburnis, C. D. O'Dowd, Important source of marine
816		secondary organic aerosol from biogenic amines, Environ. Sci. Technol., 2008, 42, 9116-
817		9121.

818	119.	X. F. Huang, C. R. Deng, G. S. Zhuang, J. Lin, M. X. Xiao, Quantitative analysis of
819		aliphatic amines in urban aerosols based on online derivatization and high performance
820	1.0.0	liquid chromatography, Environ Sci-Proc Imp, 2016, 18, 796-801.
821	120.	H. Feng, X. N. Ye, Y. X. Liu, Z. K. Wang, T. X. Gao, A. Y. Cheng, X. F. Wang, J. M.
822		Chen, Simultaneous determination of nine atmospheric amines and six inorganic ions by
823		non-suppressed ion chromatography using acetonitrile and 18-crown-6 as eluent additive,
824		<i>J. Chromatogr. A</i> , 2020, 1624 , Art No 461234.
825	121.	M. Wang, Q. Y. Wang, S. S. H. Ho, H. Li, R. J. Zhang, W. K. Ran, L. L. Qu, S. C. Lee, J.
826		J. Cao, Chemical characteristics and sources of nitrogen-containing organic compounds
827		at a regional site in the North China Plain during the transition period of autumn and
828		winter, Sci. Total Environ., 2022, 812, Art No 151451.
829	122.	A. P. Sullivan, K. B. Benedict, C. M. Carrico, M. K. Dubey, B. A. Schichtel, J. C.
830		Collett, A quantitative method to measure and speciate amines in ambient aerosol
831		samples, Atmosphere, 2020, 11, ART No 808.
832	123.	Q. Zhang, C. Anastasio, Free and combined amino compounds in atmospheric fine
833		particles (PM2.5) and fog waters from Northern California, Atmos. Environ., 2003, 37,
834		2247-2258.
835	124.	Z. Y. Liu, M. Li, X. F. Wang, Y. H. Liang, Y. R. Jiang, J. Chen, J. S. Mu, Y. J. Zhu, H.
836		Meng, L. X. Yang, K. Y. Hou, Y. F. Wang, L. K. Xue, Large contributions of
837		anthropogenic sources to amines in fine particles at a coastal area in northern China in
838		winter, Sci. Total Environ., 2022, 839, ARTN 156281.
839	125.	B. K. Place, A. T. Quilty, R. A. Di Lorenzo, S. E. Ziegler, T. C. VandenBoer,
840		Quantitation of 11 alkylamines in atmospheric samples: separating structural isomers by
841		ion chromatography, Atmos. Meas. Tech., 2017, 10, 1061-1078.
842	126.	K. Gorzelska, J. N. Galloway, Amine nitrogen in the atmospheric environment over the
843		north Atlantic ocean, Global Biogeochem. Cycles, 1990, 4, 309-333.
844	127.	V. M. Kerminen, X. Chen, V. Vakkari, T. Petaja, M. Kulmala, F. Bianchi, Atmospheric
845		new particle formation and growth: review of field observations, Environ. Res. Lett.,
846		2018, 13 , Art No 103003 (103001-103038 p).
847	128.	U. Poschl, Atmospheric aerosols: Composition, transformation, climate and health
848		effects, Angew Chem Int Edit, 2005, 44, 7520-7540.
849	129.	IPCC (2021) Climate Change 2021: The Physical Science Basis. Contribution of
850		Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on
851		Climate Change. eds V. Masson-Delmotte et al. (United Kindom and New York, NY,
852		USA), p 2391 pp.
853	130.	A. L. Hodshire, P. Campuzano-Jost, J. K. Kodros, B. Croft, B. A. Nault, J. C. Schroder, J.
854		L. Jimenez, J. R. Pierce, The potential role of methanesulfonic acid (MSA) in aerosol
855		formation and growth and the associated radiative forcings, Atmos. Chem. Phys., 2019,
856		19 , 3137-3160.
857	131.	C. A. Pope, D. W. Dockery, Health effects of fine particulate air pollution: Lines that
858		connect, J. Air Waste Manage. Assoc., 2006, 56, 709-742.
859	132.	M. R. Heal, P. Kumar, R. M. Harrison, Particles, air quality, policy and health, Chem.
860		Soc. Rev., 2012, 41, 6606-6630.
861	133.	J. Lelieveld, Clean air in the Anthropocene, Faraday Discuss., 2017, 200, 693-703.

134.

862 J. Lelieveld, J. S. Evans, M. Fnais, D. Giannadaki, A. Pozzer, The contribution of 863 outdoor air pollution sources to premature mortality on a global scale, *Nature*, 2015, 525, 864 367-+. A. L. Moreno-Ríos, L. P. Tejeda-Benítez, C. F. Bustillo-Lecompte, Sources, 865 135. 866 characteristics, toxicity, and control of ultrafine particles: An overview, Geosci Front, 867 2022, 13. 868 S. W. Sang, C. Chu, T. C. Zhang, H. Chen, X. R. Yang, The global burden of disease 136. 869 attributable to ambient fine particulate matter in 204 countries and territories, 1990-2019: 870 A systematic analysis of the Global Burden of Disease Study 2019, Ecotox. Environ. 871 Safe., 2022, 238, Art. N0 113588 (113581-113511). 872 137. X. J. Deng, X. X. Tie, D. Wu, X. J. Zhou, X. Y. Bi, H. B. Tan, F. Li, C. L. Hang, Long-873 term trend of visibility and its characterizations in the Pearl River Delta (PRD) region, 874 China, Atmos. Environ., 2008, 42, 1424-1435. 875 D. Chang, Y. Song, B. Liu, Visibility trends in six megacities in China 1973-2007, 138. 876 Atmos. Res., 2009, 94, 161-167. 877 A. Singh, W. J. Bloss, F. D. Pope, 60 years of UK visibility measurements: impact of 139. 878 meteorology and atmospheric pollutants on visibility, Atmos. Chem. Phys., 2017, 17, 879 2085-2101. 880 J. G. Watson, Visibility: science and regulation, J. Air Waste Manage. Assoc., 2002, 52, 140. 881 628-713. 882 141. A. L. Moreno-Rios, L. P. Tejeda-Benitez, C. F. Bustillo-Lecompte, Sources, 883 characteristics, toxicity, and control of ultrafine particles: An overview, Geosci Front, 884 2022, 13, Art No 101147. M. J. Ezell, H. Chen, K. D. Arguero, B. J. Finlayson-Pitts, Aerosol fast flow reactor for 885 142. 886 laboratory studies of new particle formation, J. Aerosol Sci., 2014, 78, 30-40. 887 J. N. Smith, K. F. Moore, P. H. McMurry, F. L. Eisele, Atmospheric measurements of 143. 888 sub-20 nm diameter particle chemical composition by thermal desorption chemical 889 ionization mass spectrometry, Aerosol Sci. Technol., 2004, 38, 100-110. 890 D. Voisin, J. N. Smith, H. Sakurai, P. H. McMurry, F. L. Eisele, Thermal desorption 144. 891 chemical ionization mass spectrometer for ultrafine particle chemical composition, 892 Aerosol Sci. Technol., 2003, 37, 471-475. 893 M. J. Lawler, P. M. Winkler, J. Kim, L. Ahlm, J. Trostl, A. P. Praplan, S. Schobesberger, 145. 894 A. Kuerten, J. Kirkby, F. Bianchi, J. Duplissy, A. Hansel, T. Jokinen, H. Keskinen, K. 895 Lehtipalo, M. Leiminger, T. Petaja, M. Rissanen, L. Rondo, M. Simon, M. Sipila, C. 896 Williamson, D. Wimmer, I. Riipinen, A. Virtanen, J. N. Smith, Unexpectedly acidic 897 nanoparticles formed in dimethylamine-ammonia-sulfuric-acid nucleation experiments at 898 CLOUD, Atmos. Chem. Phys., 2016, 16, 13601-13618. 899 H. Chen, S. Chee, M. J. Lawler, K. C. Barsanti, B. M. Wong, J. N. Smith, Size resolved 146. 900 chemical composition of nanoparticles from reactions of sulfuric acid with ammonia and 901 dimethylamine, Aerosol Sci. Technol., 2018, 52, 1120-1133. 902 147. D.-R. Chen, D. Y. Pui, A high efficiency, high throughput unipolar aerosol charger for 903 nanoparticles, J. Nanopart. Res., 1999, 1, 115-126.

904	148.	P. H. McMurry, A. Ghimire, HK. Ahn, H. Sakurai, K. Moore, M. Stolzenburg, J. N.
905		Smith, Sampling nanoparticles for chemical analysis by low resolution electrical mobility
906		classification, Environ. Sci. Technol., 2009, 43, 4653-4658.
907	149.	D. V. Davis, R. G. Cooks, Site of protonation and bifunctional group-interactions in a,w-
908		hydroxyalkylamines, Org. Mass Spectrom., 1981, 16, 176-179.
909	150.	G. Bouchoux, N. Choret, F. Berruyer-Penaud, R. Flammang, Thermochemistry and
910 011		unimolecular reactivity of protonated alpha,omega-aminoalcohols in the gas phase, <i>Int. J.</i> Mass Spactrom 2002 217 195 230
911 01 2	151	E. D. L. Hunter, S. C. Ling, Evaluated and phase baginities and proton affinities of
912	131.	E. F. L. Humer, S. O. Lias, Evaluated gas phase basicities and proton animities of molecules: An undete, <i>J. Phys. Chem. Bef. Data</i> , 1008, 27 , 412, 656
913	152	S Vantaina K Slowik S D Varaykin A Haintz Vanar prossures and vanarization
914 015	132.	S. Kaptelnia, K. Słowik, S. P. Velevkin, A. Heiniz, Vapor pressures and vaporization anthalping of a series of otherologing. <i>J. Chem. Eng. Data</i> 2005 50 , 208, 402
913	152	L N. Tang, H. D. Munivaluation Determination of variant management from dramlat avanantion.
910	155.	I. N. Tang, H. K. Munkelwitz, Determination of vapor-pressure from droplet evaporation
91/	154	Kinetics, J. Collola Interface Sci., 1991, 141, 109-118.
918	154.	SL. von der weiden, F. Drewnick, S. Borrmann, Particle loss calculator - A new
919		software tool for the assessment of the performace of aerosol inlet systems, <i>Atmos. Meds.</i>
920	165	<i>Iecn.</i> , 2009, 2 , 409-494.
921	155.	M. Kulmala, H. Venkamaki, T. Petaja, M. Dal Maso, A. Lauri, V. M. Kerminen, W.
922		Birmili, P. H. McMurry, Formation and growth rates of ultrafine atmospheric particles: a
923	150	review of observations, J. Aerosol Sci., 2004, 35 , 143-176.
924	156.	1. L. Greaves, A. weerawardena, C. Fong, I. Krodklewska, C. J. Drummond, Protic ionic
925		Cham B 2006 110 22470 22487
920	157	Chem. B, 2000, 110, 224/9-2248/.
927	137.	C. Cal, S. Yang, Q. Zhou, L. Liu, J. Au, S. Zhang, Physicochemical properties of various
920		2-involoxyetitylaminoinum sunonate-based protectome inquids and then potential application in hydrodooxygonotion. Executions in Chamistry, 2010. 7, art. No. 106 (101
929		application in hydrodeoxygenation, <i>Frontiers in Chemistry</i> , 2019, 7, art. No 190 (191-
930	158	112). Y Lu I M Vincent Luna S Calero M Roldan Ruiz R Jimenez M I Ferrer M C
032	156.	Gutierrez, E. del Monte, Aqueous co solvent in zwitterionic based protic ionic liquids as
932		electrolytes in 2.0 V supercapacitors. <i>Cham Sus Cham</i> , 2020, 13 , 5083-5005
937	150	S I Brown D Valcin S Pnadiancherri T C Le I O'rhan K Hearn O Han C I
035	157.	Drummond T. J. Greaves Characterising a protic ionic liquid library with applied
936		machine learning algorithms. I Mol Lig. 2022 367 Art No 120453 (120451-120417)
937	160	R McGraw A Laaksonen Scaling properties of the critical nucleus in classical and
938	100.	molecular-based theories of vanor-liquid nucleation <i>Phys. Rev. Lett.</i> 1996 76 2754-
939		2757
940	161	O Kunjajnen-Maatta T Olenius H Korhonen I Malilal M Dal Maso K Lehtinen H
941	101.	Vehkamaki Critical cluster size cannot in pracetice be determined by slope analysis in
942		atmospherically relevant applications <i>L</i> Aerosol Sci 2014 77 127-144
943	162	Y Liu H B Xie F F Ma I W Chen I Elm Amine-enhanced methanesulfonic acid
944	102.	driven nucleation. Predictive model and cluster formation mechanism <i>Environ Sci</i>
945		<i>Technol.</i> , 2022, 56 , 7751-7760.

946	163.	J. Xu, V. Perraud, B. J. Finlayson-Pitts, R. B. Gerber, Uptake of water by an acid-base
947		nanoparticle: theoretical and experimental studies of the methanesulfonic acid-
948		methylamine system, Phys. Chem. Chem. Phys., 2018, 20, 22249-22259.
949	164.	V. Perraud, J. R. Horne, A. S. Martinez, J. Kalinowski, S. Meinardi, M. L. Dawson, L.
950		M. Wingen, D. Dabdub, D. R. Blake, R. B. Gerber, B. J. Finlayson-Pitts, The future of
951		airborne sulfur-containing particles in the absence of fossil fuel sulfur dioxide emissions,
952		Proc. Natl. Acad. Sci. USA, 2015, 112, 13514-13519.
953	165.	J. G. Murphy, P. K. Gregoire, A. G. Tevlin, G. R. Wentworth, R. A. Ellis, M. Z.
954		Markovic, T. C. VandenBoer, Observational constraints on particle acidity using
955		measurements and modelling of particles and gases, Faraday Discuss., 2017, 200, 379-
956		395.
957	166.	Z. Klimont, S. J. Smith, J. Cofala, The last decade of global anthropogenic sulfur dioxide:
958		2000-2011 emissions, Environ. Res. Lett., 2013, 8, 1-6.
959	167.	T. J. Breider, L. J. Mickley, D. J. Jacob, C. Ge, J. Wang, M. P. Sulprizio, B. Croft, D. A.
960		Ridley, J. R. McConnell, S. Sharma, L. Husain, V. A. Dutkiewicz, K. Eleftheriadis, H.
961		Skov, P. K. Hopke, Multidecadal trends in aerosol radiative forcing over the Arctic:
962		Contribution of changes in anthropogenic aerosol to Arctic warming since 1980, J.
963		Geophys. Res., 2017, 122 , 3573-3594.
964	168.	G. M. Hidy, C. L. Blanchard, The changing face of lower tropospheric sulfur oxides in
965		the United States, <i>Elementa-Sci. Anthrop.</i> , 2016, 4, Art. N0 000138 (000131-000120).
966	169.	D. I. Stern, Global sulfur emissions from 1850 to 2000, Chemosphere, 2005, 58, 163-175.
967	170.	M. Amann, Z. Klimont, F. Wagner, Regional and global emissions of air pollutants:
968		Recent trends and future scenarios, Ann. Rev. Environ. Resour., 2013, 38, 31-55.
969	171.	P. K. Quinn, G. Shaw, E. Andrews, E. G. Dutton, T. Ruoho-Airola, S. L. Gong, Arctic
970		haze: current trends and knowledge gaps, <i>Tellus B</i> , 2007, 59 , 99-114.
971	172.	D. Hirdman, J. F. Burkhart, H. Sodemann, S. Eckhardt, A. Jefferson, P. K. Quinn, S.
972		Sharma, J. Strom, A. Stohl, Long-term trends of black carbon and sulphate aerosol in the
973		Arctic: changes in atmospheric transport and source region emissions, Atmos. Chem.
974		<i>Phys.</i> , 2010, 10 , 9351-9368.
975		



[MEA] = 3.3 ppb



Figure 1. Size distributions of particles from MEA (1.5 ppb) reacting with (A) 0.68 ppb MSA, (B) 1.4 ppb MSA and (C) 2.8 ppb MSA, and size distribution from MEA (3.3 ppb) reacting with (D) 1.5 ppb MSA, (E) 3.3 ppb MSA and (F) 6.1 ppb MSA. Measurements were conducted at the same sampling ports distributed equally along the length of the flow tube, but experiments displayed in panel (A-C) were performed with a total flow rate of 23.4 L min⁻¹ (resulting in reaction times between 0.3 and 4.5 s), while experiments displayed in panel (D-F) were performed with a total flow rate of 10.7 L min⁻¹ (resulting in reaction times between 0.5 and 7.7s). All experiments were performed under dry conditions, and size distributions are the

average of 3 to 8 replicates (error bars correspond to one standard deviation) for each reaction time. All size distributions were corrected for particle losses through the sampling lines. Total particle concentrations and geometric mean diameters as a function of reaction times are given in Fig. S12.



Figure 2. Particle formation rate (J_{>2.5 nm}) for the MSA+MEA system under dry conditions as a function of the product of the MSA and MEA mixing ratios in ppb. Each data point represents an average over 3 to 8 individual SMPS scans taken at 2.3-2.4 s reaction time, with error bars representing one standard deviation, and corrected for particle losses through the sampling lines. The red line is a linear fit to the data ([MSA] × [MEA] \leq 2 ppb²) with a slope of (5.3 \pm 0.03) × 10⁶ particles cm⁻³ ppb².



Figure 3. Acid/base molar ratio measured by the TDCIMS for the MSA+MEA system, under dry conditions (red data points) and at 52% RH (blue data points) measured at 4.5 s reaction time. MEA was measured in POS ion mode while MSA was measured in NEG ion mode, and the ion distributions in the MS spectra were similar at all reactant concentrations (Fig. S6). Text S5 provides more detailed information on how the acid/base molar ratios were estimated. All measurements were performed with initial MEA concentration of 1.5 ppb. No significant difference was observed in the measured molar ratio across for experiments performed with [MSA] = 0.68 ppb, [MSA] = 1.4 ppb or [MSA] = 2.8 ppb, in either dry or humid conditions; thus the data points represent average values across the [MSA] concentrations range for each RH condition. For each data point, the error bars represent one standard deviation. The dashed line corresponds to an acid/base molar ratio of unity for reference.



Figure 4. Evolution of the particle size distributions as a function of relative humidity (RH) from the reaction of MSA (0.7 ppb) with MEA (1.4 ppb). Panel (A) represents the evolution as a function of time while panel (B) represents a snapshot of the size distributions at a given RH (each distribution is an average over three SMPS scans with the error bars representing one standard variation). All measurements were performed at 4.5 s reaction time, and particle size distributions were corrected for particle loss through the sampling lines.



Figure 5. Size distributions (A and B) and evolution of the particle total concentrations and geometric mean diameters measured as a function of the reaction time (C and D) in the flow reactor for the MSA+MEA reaction system at 50% RH. All lines in panels C and D are guides to the eye. All data originate from replicate scans (n = 5) and are displayed with one standard deviation. All size distributions were corrected for particle losses through the sampling lines.



Figure 6. Logarithm of the particle formation rate ($J_{>2.5 \text{ nm}}$) for the MSA+MEA system under humid conditions (RH ranging from 8 to 56%) as a function of the log of the water concentration (in molecule cm⁻³). Each data point represents an average over 3 to 6 individual SMPS scans taken at 4.5 s reaction time, with error bars representing one standard deviation. All data points were corrected for particle loss through the sampling line. Red data are for [MSA] = 0.68 ppb and [MEA] = 1.5 ppb while the blue data are for [MSA] = 2.8 ppb and [MEA] = 1.5 ppb. The slopes of the lines are 0.04 for 2.8 ppb MSA and 0.2 for 0.68 ppb MSA.



Figure 7. Representative averaged size distribution (red trace) from the reaction of MSA (1.4 ppb) with MEA (1.5 ppb). For comparison, a size distribution for MSA (1.4 ppb) reacting with MA (4.8 ppb) is also shown (green trace). Both size distributions were taken at ~4 s reaction time. Each size distribution was averaged over five consecutive scans and the shaded area corresponds to one standard deviation uncertainty. The thick line corresponds to a log normal fit to the averaged data. Both size distributions have been corrected for particle losses through the sampling lines.



Figure 8. Comparison between particle formation rate ($J_{>2.5 \text{ nm}}$) for the MSA+MEA and the MSA+MA systems under humid conditions (RH ranging from 8 to 56%) as a function of the product of the MSA and the amine mixing ratios in ppb. Each data point represents an average over 3 to 6 individual SMPS scans taken at 4.5 s reaction time, with error bars representing one standard deviations. The dashed lines are exponential fits to the data and the solid lines are the tangent to the fits with slopes of 5.5×10^6 and 1.2×10^6 for the MSA+MEA and MSA+MA systems respectively. All data points were corrected for particle loss through the sampling lines.