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1 **Spatial inter-comparison of Top-down emission inventories** 2 **in European urban areas**

3

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23

24 **Abstract**

25

26 This paper presents an inter-comparison of the main Top-down emission inventories
27 currently used for air quality modelling studies at the European level. The comparison is
28 developed for eleven European cities and compares the distribution of emissions of NO_x,
29 SO_x, VOC and PPM_{2.5} from the road transport, residential combustion and industry sectors.

30 The analysis shows that substantial differences in terms of total emissions, sectorial emission
31 shares and spatial distribution exist between the datasets. The possible reasons in terms of
32 downscaling approaches and choice of spatial proxies are analysed and recommendations are
33 provided for each inventory in order to work towards the harmonisation of spatial

34 downscaling and proxy calibration, in particular for policy purposes. The proposed
35 methodology may be useful for the development of consistent and harmonised European-
36 wide inventories with the aim of reducing the uncertainties in air quality modelling activities.

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38 **Keywords: Top-down emission inventories, Urban areas, Emissions spatial distribution,**
39 **Spatial inter-comparison**

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65 **1. Introduction**

66 Emission inventories represent one of the key datasets required for air quality studies, but
67 they are often recognised as the most uncertain input in the modelling chain (Borge et al.,
68 2014; Guevara et al., 2013; Thunis et al., 2016a; Viaene et al., 2013) as their accuracy greatly
69 varies with the type of pollutant, the activity and the level of spatial disaggregation (Davison
70 et al., 2011). In Europe, this is largely due to the fact that regional and local emission
71 inventories are managed and compiled by several different agencies which rely on different
72 standards, methods and categories. This may be understandable given the different
73 background and scope of the inventories, however it may yield to a heterogeneous and
74 inconsistent picture when collating these data for use in modelling at a larger scale
75 (continental and national levels). Furthermore, it is known that, in emission inventories,
76 different measurement methods are applied for the same sectors, e.g. residential combustion
77 which may result in emissions different up to a factor 5 (Denier van der Gon et al. 2015).

78 For this reason, there exist several top-down implementations that compile EU wide
79 inventories by downscaling national emissions data at a finer resolution: EDGAR (Crippa et
80 al., 2016; Janssens-Maenhout et al., in prep.), HTAP_v2 (Janssens-Maenhout et al., 2015),
81 TNO-MACCII and MACCIII (Kuenen et al., 2014; Kuenen et al., 2015), E-PRTR (Theloke
82 et al., 2009, Theloke et al., 2012), JRC07 (Trombetti et al., 2017). These inventories are all
83 comparable in spatial (i.e. between $\sim 10\text{km} \times \sim 10\text{km}$ and $\sim 7 \text{ km} \times \sim 7\text{km}$) and temporal terms
84 (i.e. annual), geographical extent (i.e. European continent) and thematic resolution (sectors
85 and macro-sectors aggregation) but differences remain in terms of national total emission
86 estimates and/or spatial gridding methodologies. The first type of difference can be caused by
87 model settings, reporting of emission sources, gap filling approaches, assumptions or
88 arbitrary choices and has already been discussed for some inventories (Kuenen et al., 2014;
89 Granier et al, 2011).

90 For the second difference, spatial discrepancies mostly depend on methodological
91 assumptions, proxies' availability and choice of the weighting methodology. The fact that all
92 these inventories are developed at a high spatial resolution (~ 7 to $\sim 10 \text{ km} \times \text{km}$) reinforces
93 this factor. As shown by Zheng et al. (2017), the spatial mismatch between gridded

94 inventories developed from different spatial proxies is largely diminished at coarse
95 resolutions (i.e. 25km x 25km) but tends to increase as grid size decreases (i.e. 4km x 4km).
96 These differences have often been overlooked and only studied for regional (i.e. sub-national)
97 inventories (Winiwarter et al., 2003; Vedrenne et al., 2016) while only a few cases at fine
98 scale have been published (Ferreira et al., 2013). These studies clearly stressed the
99 importance of the assumptions behind the underlying proxies, their level of detail and their
100 accuracy, to explain the very low spatial correlations found between target inventories. It is
101 important to note that these spatial variations have a strong impact on air quality modelling
102 results (Geng et al, 2017; Zhou et al., 2017), especially when the results are considered for
103 policy making and planning options. Top-down emission inventories are often being used as
104 input data for modelling activities at urban scale (Lopez-Aparicio et al., 2017); therefore,
105 particular attention should be given before choosing a specific dataset for this kind of
106 modelling activities.

107 To our knowledge, our study is the only existing spatial inter-comparison between emissions
108 inventories currently used at the European scale. Its novelty lies on defining the possible
109 uncertainties in the spatial proxies behind the disaggregation and allocations of emissions in
110 urban areas and, consequently, on reducing the propagation of errors to air quality models
111 and their applications.

112 This study assesses how a set of six EU wide emission inventories (i.e. EDGAR,
113 TNO_MACCII, TNO_MACCIII, INERISinv, EMEP, JRC07) behave in selected European
114 urban areas in terms of sectorial shares and regional allocation also through the application of
115 a novel approach, namely the diamond analysis (Thunis et al., 2016b), in order to estimate
116 systematically the spatial variability between them. This approach aims to contribute to
117 increasing the reliability of emission inventories. We first describe the methodology and the
118 emission datasets used, before identifying the main differences for the selected urban areas.
119 Finally, recommendations to improve credibility for air quality modelling applications and
120 reduce the level of uncertainty are provided for each inventory.

121

122 **2. Methodology**

123

124 We focus our analysis on the way emissions of NO_x, SO₂, VOC and PPM_{2.5} are spatially
 125 distributed by different European scale top-down inventories. For this reason, the comparison
 126 is not made in terms of absolute, but rather in terms of normalised emission values. The
 127 values attributed to each grid cell of coordinates i and j for the variable $E_{p,s}^*$ represent the
 128 percentage of the total national emission for each emission pollutant “p” and sector “s”, i.e.:
 129

$$\forall s, \forall p: E_{s,p}^*(i, j) = \frac{E_{s,p}(i, j)}{E_{s,p}^{tot}}$$

130 where $E_{s,p}^{tot}$ represents the country total emission for a given sector and pollutant. With this
 131 normalisation, observed differences between inventories at a given grid cell do not depend on
 132 the original national emission value, but instead depend on the downscaling methodology and
 133 ancillary data used (Hiller et al., 2014).

134 The spatial analysis is performed for specific urban areas and for the main emission macro
 135 sectors: non-industrial combustion (SNAP02), industrial activities (SNAP03 and SNAP04,
 136 which are kept together in order to facilitate the comparison within inventories: SNAP34) and
 137 road transport (SNAP07). See the Supplementary Information (SI) for a description of the
 138 SNAP Macro Sectors (Table 1, SI).

139 The SNAP02 macro-sector consists of i) commercial / institutional stationary combustion; ii)
 140 residential combustion; iii) stationary combustion associated with agriculture, forestry or
 141 fishing; iv) other stationary. Given that the sector “ii) residential combustion” is the dominant
 142 one, the discussion in this paper focuses only on this subsector, hereafter referred to as
 143 ‘Residential’.

144 Eleven cities (Barcelona, Bucharest, Budapest, Katowice, London, Madrid, Milano, Paris,
 145 Sofia, Utrecht and Warsaw) were selected across Europe to represent the diversity of
 146 environmental and anthropogenic factors (i.e. meteorology, economic activities, energy
 147 system, population density and land use) over the continental domain; in particular, the
 148 differences in Land Use cover reported in Table 2, SI, will affect the sectorial shares of
 149 emissions in each study site. For each city, the study area covers approximately 35 x 35 km²,
 150 including only whole grid cells without having to split or resample them. With the exception
 151 of EDGAR, all inventories have similar spatial resolution and grid alignment, so it was
 152 possible to define common study areas. The EDGAR inventory has a different spatial

153 resolution and so an alternative definition of the study areas was created resembling the
154 original one, while preserving the integrity of the selected grid pixels. The standard study site
155 and the adjusted EDGAR one for each urban area are shown in the SI with the considered
156 land use pattern (Figure 1, SI).

157 The assessment is supported by the analysis performed by means of the diamond approach
158 (Thunis et al., 2016b), a novel method which, by using total emission ratios, allows the
159 comparison of emission inventories and the identification of the likely cause (activity level or
160 activity share) of differences between them. Given the normalisation by the country totals,
161 the differences seen among inventories in terms of activity levels and share can be directly
162 attributed to the spatial disaggregation methodology.

163 **2.1. Downscaled inventories**

164 We consider six European scale top-down inventories, with 2010 as reference year, unless
165 mentioned otherwise. The selected emission inventories cover a wide and important range of
166 applications, including regulatory purposes (e.g. EMEP), monitoring services (e.g. TNO-
167 MACC, EDGAR) and integrated assessment (e.g. INERISinv, JRC07).

168

- 169 • EDGAR version v4.3.1, January 2016 (European Commission, 2016a; Crippa et al.,
170 2016), hereafter referred to as EDGAR. This inventory provides global emissions for
171 gaseous and particulate air pollutants (BC, CO, NH₃, NMVOC, NO_x, OC, PPM₁₀, PPM_{2.5},
172 SO₂) per IPCC sector (Intergovernmental Panel on Climate Change) covering the whole
173 time-series 1970-2010 at the global scale. Emissions are provided in tons of substance at
174 0.1 x 0.1 degree resolution. A highly detailed re-mapping of the sectors from the IPCC to
175 the SNAP nomenclature has been made to allow comparing with the other databases. The
176 simplified version of the mapping scheme from IPCC to SNAP codes is included in the SI
177 (Table 3) together with the detailed reclassification for a representative SNAP
178 MacroSector (SNAP04, Production Processes, Table 4, SI).

179

- 180 • TNO-MACCII (Denier van der Gon et al., 2010; Kuenen et al., 2011; Kuenen et al.,
181 2014), hereafter referred to as MACCII. The TNO emission inventory was developed for
182 Europe by TNO for the years 2003-2009. It has a 1/8° longitude x 1/16° latitude

183 resolution and covers NO_x, SO₂, NMVOC, NH₃, CO, PPM₁₀, PPM_{2.5} and CH₄. This
184 dataset is not available for 2010, consequently the 2009 dataset has been used instead.

185 • TNO-MACCI_{III} (Kuenen et al., 2014; Kuenen et al., 2015; MACC-III Final Report,
186 2016), hereafter referred to as MACCI_{III}. It is the updated version of the TNO-MACCI_{II}
187 product, which extended the time-series from year 2000 to year 2011. All years were
188 revisited and the spatial distribution proxies updated and improved, often based on user
189 comments.

190

191 • INERIS_{inv} (hereafter referred to as INERIS): The INERIS inventory is based the work by
192 Bessagnet et al., 2016 with the following changes for the Macro Sectors analyzed in this
193 work.

194 MS34: The E-PRTR database is used for Large Point Sources of emissions (Mailler et al.,
195 2017)

196 MS07: Road transport emissions of all considered countries are distributed using a proxy
197 based the combination of several databases and the French bottom-up emission inventory
198 (Mailler et al., 2017)

199 MS02: Residential combustion emissions are distributed based on population, land use
200 and the French bottom-up emission inventory. For all compounds, except PPM, emissions
201 are redistributed according to the population distribution. For PPM_{2.5} the method
202 proposed by Terrenoire et al. (2015) using a logarithmic regression as a function of
203 population density was improved by fitting several parameters to the new proxy and the
204 landuse to treat differently urban areas from non-urban areas.

205 For all sectors and for all compounds, the French and British bottom-up emission
206 inventory at 1km resolution is used to redistribute emissions of France and United
207 Kingdom.

208 The inventory covers NO_x, SO₂, NMVOC, NH₃, CO, PPM₁₀ and PPM_{2.5} as reported by
209 each country in EMEP. It is distributed at a 1/8° longitude x 1/16° latitude resolution and
210 it covers the year 2010.

211

- 212 • EMEP: The EMEP emission inventory is based on emission data reported by the 51
213 Parties belonging to the LRTAP convention, complemented by expert estimates
214 (www.emep.int). The EMEP emission product is normally distributed at $50 \times 50 \text{ km}^2$
215 resolution and calculated by using sectoral emissions as reported by countries and gap
216 filled with data from different models where no or incomplete data are reported by
217 countries (EMEP, 2015). At the 36th session of the EMEP Steering Body, the EMEP
218 Centers suggested to increase spatial resolution of reported emissions from $50 \times 50 \text{ km}^2$ to
219 $0.1^\circ \times 0.1^\circ$ (<http://www.ceip.at>). Nevertheless, the official reporting of gridded emissions
220 in this new resolution is requested from 2017 onwards and currently about half of the
221 EU28 countries have submitted their own gridded data. The higher resolution version
222 used here has instead been rescaled at $1/8^\circ$ longitude x $1/16^\circ$ latitude resolution based on
223 the TNO-MACCI3 emission data and it covers year 2013.
- 224
- 225 • JRC07 (Trombetti et al., 2017). The JRC07 is an inventory recently developed for use in
226 Integrated Assessment Modelling strategies (IAM) in the fields of regional air-quality
227 (Clappier et al., 2015; Carnevale et al., 2012) and land use and territorial modelling
228 (<https://ec.europa.eu/jrc/en/luisa>; Lavallo et al., 2011; Lavallo et al., 2013). The inventory
229 is based on country total emission data from the Greenhouse Gas and Air Pollution
230 Interactions and Synergies Model (GAINS, Amann et al., 2011), as used for the
231 implementation of the EU20-20-20 targets under the assumptions of the 2013 Air Quality
232 package review. It models emissions from 2010 up to 2030 and it currently covers NO_x ,
233 SO_2 , VOC, PPM_{10} , $\text{PPM}_{2.5}$ and NH_3 . It is distributed with different spatial resolutions and
234 here it is being used in its version at $1/8^\circ \times 1/8^\circ$ degrees.
- 235 An overview of the spatial proxies and ancillary data used for the spatial distribution of
236 emissions from the considered sectors is shown in Table 1.

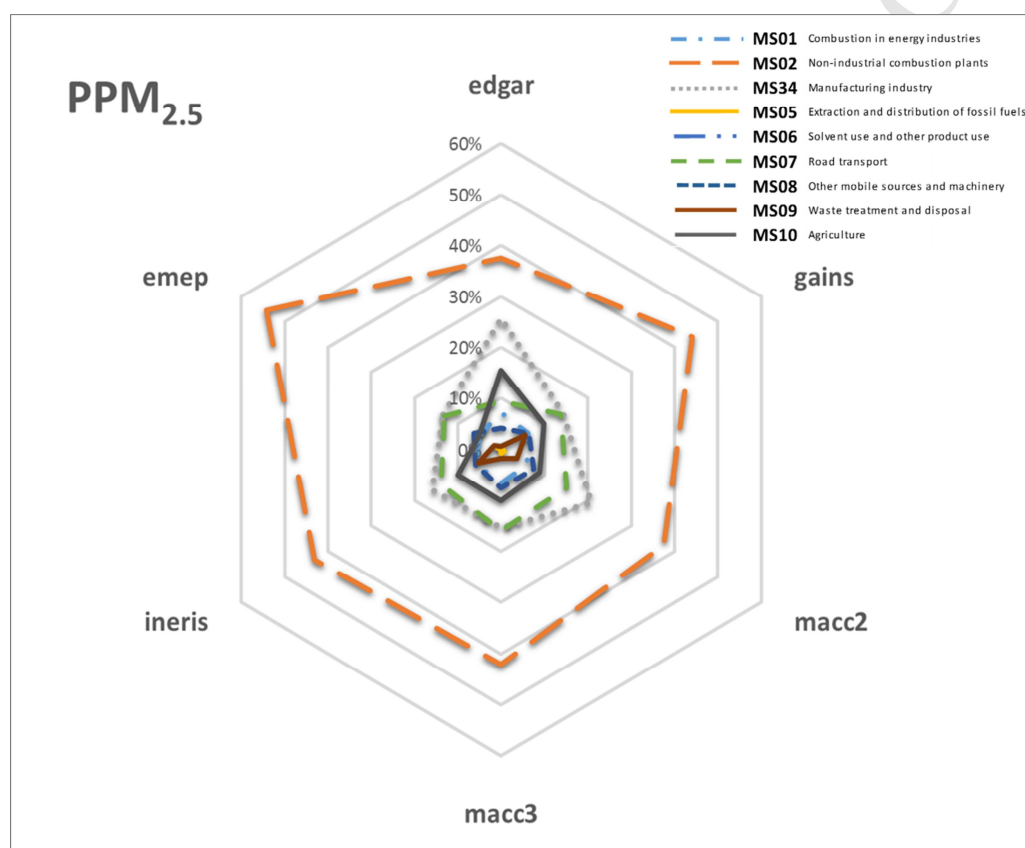
	TNO-MACCII	TNO-MACCIII	INERISinv	JRC07	EDGAR	EMEP
<i>Reference</i>	Kuenen et al., 2014	Kuenen et al. 2015, Kuenen et al. 2016, pers. Comm.	Mailler et al., 2017	Trombetti et al., 2017	Crippa et al., 2016	Mareckova et al., 2016
<i>Temporal Coverage</i>	2003 - 2009	2000 - 2011	2010	2010-2030	1970-2010	2013
<i>Input Emission Data</i>	Official reported emissions to CLRTAP, gapfilled with other data (e.g. GAINS, EDGAR, TNO estimates)	Official reported emissions to CLRTAP, gapfilled with other data (e.g. GAINS, EDGAR, TNO estimates)	EMEP	GAINS	EDGAR	Official reported emissions to CLRTAP, gap-filled by CEIP
<i>SNAP02 (Residential)</i>	Total population (CIESIN and GRUMP http://sedac.ciesin.columbia.edu) and wood use map (based on population and wood availability)	Total population (CIESIN and GRUMP http://sedac.ciesin.columbia.edu) and improved wood use map (based on population and wood availability)	Population (GRUMP http://sedac.ciesin.columbia.edu/gpw) Land Use (USGS)	Population, Industrial and Agriculture Land Use (LUISA, Lavalle et al., 2013) Degree of Urbanization (EUROSTAT, http://ec.europa.eu/eurostat/web/degree-of-urbanisation/overview)	In-house proxy based on rural and urban population (http://sedac.ciesin.columbia.edu/)	50x50 km ₂ : CEIP country reported grids 0.0625x0.125 degrees: TNO-MACCIII emission data
<i>SNAP03 and SNAP04 (Industry)</i>	LPS: E-PRTR, TNO point source database Diffuse: Population	LPS: E-PRTR, TNO point source database Diffuse: Industrial land cover (CORINE (EEA, 2017))	LPS: E-PRTR Diffuse: No Diffuse	LPS: E-PRTR v8 Diffuse: Manufacturing sector employment data (EUROSTAT, 2008), Industrial Land Use (LUISA, Lavalle et al., 2013)	LPS: In-house proxy based on USGS (http://mrdata.usgs.gov/mineral-operations/), E-PRTR v4.2, v6.1, v7 CEC (http://takingstock.cec.org/), CIESIN (http://sedac.ciesin.columbia.edu), Global Energy Observatory (http://globalenergyobservatory.org/), NGDC (https://www.ngdc.noaa.gov/eog/viirs.html); https://www.ngdc.noaa.gov/eog/viirs.html), World Port Index (PUB 150) (http://msi.nga.mil/MSISiteContent/StaticFiles/NAV_PUBS/WPI/Pub150bk.pdf) Diffuse: No Diffuse Population when no LPS is available	50x50 km ₂ : CEIP country reported grids 0.0625x0.125 degrees: TNO-MACCIII emission data
<i>SNAP07 (Road Transport)</i>	TRANSTOOLS network (European Commission, 2005) and Total population	TRANSTOOLS network (European Commission, 2005) and Total population	Proxy based on the correlation between the French bottom-up emission inventory and different spatial databases (CORINE, EEA; ETISplus, http://www.etisplus.eu)	Open Street Map Network (OSM contributors, 2015) Population (LUISA, Lavalle et al., 2013) REMOVE shares of traffic (De Ceuster et al., 2006) AADT UNECE (UNECE, 2005)	In-house EDGAR proxy based on OpenStreetMap (OSM contributors, 2015) and weighted on road type and vehicle category	50x50 km ₂ : CEIP country reported grids 0.0625x0.125 degrees: TNO-MACCIII emission data

237 **Table 1: Description of the features of the inventories (literature reference and temporal coverage), spatial proxies and ancillary data used for the downscaling of**
238 **emissions from the considered sectors**

239 3. Analysis

240 3.1. Comparison at country scale

241 The selected inventories are first analysed and compared in terms of input data, looking at
 242 their macro-sectorial shares of emissions aggregated at the EU28 level (PPM_{2.5} is represented
 243 in Figure 1; see figure 2 in the Supplementary Information for the other pollutants). It is
 244 important to underline that TNO-MACCII, TNO-MACCCIII, INERIS and EMEP are based on
 245 the officially reported emissions by the countries to the CLRTAP, while JRC07 is based on
 246 GAINS. Although the reporting year to CLRTAP might not be the same for all inventories,
 247 this may explain the observed similitudes among some emission inventories.



248

249 **Figure 1 Comparison of the selected inventories for PPM_{2.5} in terms of macro-sectors shares at the**
 250 **country scale. The numbers from 1 to 10 refer to the SNAP sectors, where SNAP34 is the result of**
 251 **merging SNAP03 and SNAP04**

252

253 As expected, there is a good agreement among the inventories based on official reporting to
 254 CLRTAP on the shares of emissions for the targeted macro sectors. The EDGAR emission
 255 inventory shows the largest differences for all pollutants with the exception of SO₂.

256 For NO_x, the share for road transport varies from 37% (EDGAR) to 43% (INERIS), while
257 differences are between 1% and 2% for the residential combustion and the industrial sectors.
258 The most noticeable difference for NO_x is in the EDGAR emission inventory, as it assigns
259 more emissions (~ 7%) to SNAP01 (Combustion in energy and transformation industries),
260 which is compensated by a lower share of emissions in SNAP08 (Non-Road transport).

261 For PPM_{2.5}, the share of emissions from SNAP02 (Residential) ranges from 38% (EDGAR)
262 to 48% (GAINS, on which JRC07 is based) with the exception of EMEP, which assigns
263 much more importance to this sector (54%). This difference between EMEP and the other
264 CLRTAP-based inventories for PPM_{2.5} can be explained by different emission reporting in
265 different years. The EMEP inventory is based on reporting in 2016 while e.g. TNO-MACCI3
266 is based on reporting in 2013. Overall EU28 reported primary PPM_{2.5} emissions from
267 SNAP02 in 2016 are more than 20% higher than in 2013.

268 For SNAP07 (Road Transport) and SNAP34 (Industry), we find the same pattern reported
269 for NO_x, with EDGAR assigning to industry 5% higher emissions than MACC2 and ~10%
270 higher than the other inventories, while reporting a ~6% lower share of the road transport
271 sector. Similar variations are also seen for the agricultural sector (SNAP10).

272 In the case of SO₂, no major difference is observed between the inventories, although it has to
273 be noted that the road transport sector is of negligible importance. Emissions of SO₂ have
274 decreased by 88% between 1990 and 2014 in EU28 mainly as a result of fuel-switching from
275 high-sulphur solid and liquid fuels to low-sulphur fuels (EEA, 2016) but also as a result of
276 the increase in abatement on large plants. Currently, emissions from this pollutant mainly
277 come from point sources linked to the public energy production sector (i.e. coal-fired power
278 plants) that are usually continuously monitored and hence well characterised by all emission
279 inventories. This might not apply though for some other countries where, with the above
280 mentioned increased abatement and fuel switch, the share of emissions of the national total
281 from Large Point Sources has significantly decreased and a higher share of emissions come
282 from Medium Combustion Plants (MCPs) and possibly even from small-scale combustion.

283 Looking at the target sectors for VOC, while there is a good agreement for SNAP02 and
284 SNAP07, EDGAR has higher emissions for the industrial sector for EDGAR. This is most
285 likely an allocation issue, since this difference is partially compensated by an underestimation
286 in SNAP06 (Solvents and other Products use). This compensation between sectors may

287 indicate a potential inconsistency in the mapping of industrial activities related to the use of
288 solvents (e.g. pharmaceutical products, paint manufacturing). This inconsistency highlights
289 that differences in the original mapping and linking tables used in each inventory to match
290 specific pollutant activities to an official reporting format (e.g. NFR to SNAP) may have a
291 large impact when re-mapping activities from one reporting nomenclature to another.

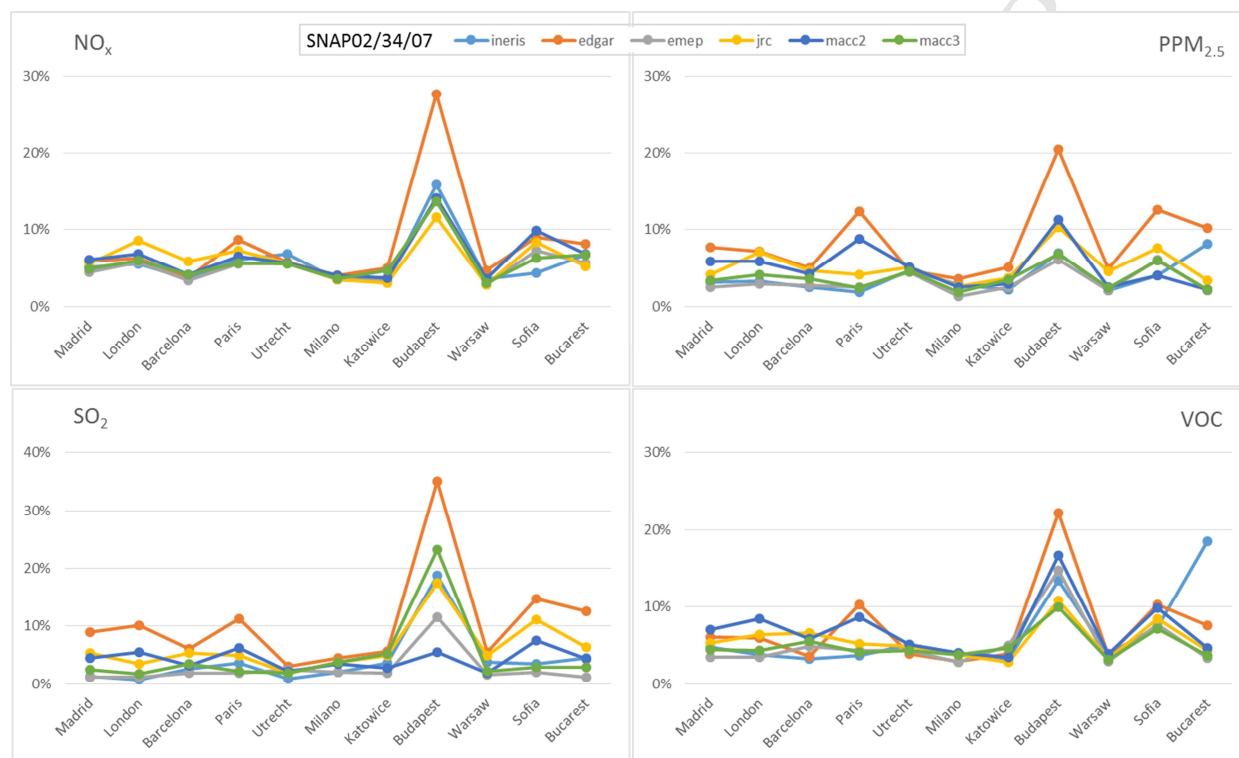
292

293 **3.2. Comparison at regional/urban scale**

294 **3.2.1. Emission totals**

295 We focus here on the regional allocation of emissions, i.e. on the fraction of the sum of
296 national emissions from SNAP02, SNAP34 and SNAP07 which is assigned to a particular
297 city (Figure 2; as in the following figures, the cities are ordered on the x axis by degree of
298 longitude, West to East). All inventories perform similarly for NO_x with an exception in
299 Budapest to which EDGAR assigns almost 30% of the national totals, almost twice the
300 percentage assigned by the other inventories. Budapest consistently shows the largest
301 differences between the inventories for all compounds. Large differences are also observed
302 for Paris, for all pollutants and especially for EDGAR and MACCII, and in Bucharest and
303 Sofia, for SO_2 and $\text{PPM}_{2.5}$. The higher emission share in Paris according to MACCII could be
304 explained by an over-allocation of industrial emissions (SNAP34) to urban areas. Emissions
305 from the industrial sectors that cannot be linked to a specific point source are merged in
306 MACCII and gridded based on total population (Table 1). This approach resulted in an over-
307 allocation of industrial emissions in urban areas (Guevara et al., 2014) and has been corrected
308 in MACCIII where diffuse industrial emissions are allocated to industrial areas according to
309 the CORINE land cover classification 2016 (EEA, 2017b). A good agreement between the
310 inventories is observed in Barcelona, Milano, Warsaw and Utrecht. From an inventory point
311 of view, EDGAR tends to allocate a larger fraction of the national totals to urban areas than
312 the other inventories, in particular for $\text{PPM}_{2.5}$ and SO_2 . The higher estimation ranges between
313 factors 1.5 and 2. The behaviour of INERIS in Bucharest NO_x and SO_2 follows the average
314 trend while, for VOC and $\text{PPM}_{2.5}$, it is outlying. As it appears from the analysis at sectorial
315 level in the next chapters, these higher values are likely due to higher emissions from the

316 industrial sector which represents the ~70% of the total emissions, a share much higher than
 317 the ones reported for the other inventories (~5% - ~40%).
 318 In general, it is clear that the spatial disaggregation methods applied in each inventory work
 319 differently in terms of urban areas and pollutants.



320

321 **Figure 2: Regional allocation of emissions: fraction of the country emissions from the total of sectors**
 322 **SNAP02 (Residential Combustion), SNAP34 (Industry) and SNAP07 (Road Transport) which is assigned**
 323 **to each city**

324 3.2.2. Sector share

325 In order to better understand why the spatial allocation differs between inventories, we
 326 compare the way each inventory spatially allocates the regional emission in terms of macro-
 327 sectors, more specifically, transport (SNAP07), industry (SNAP34) and residential
 328 combustion (SNAP02). Some uncertainties could be present due to the way different
 329 countries might convert sectors between the different nomenclatures (NFR, SNAP, IPCC).
 330 For each urban area, the contribution of each macro-sector, which will partly depend of the
 331 characteristics of the selected study site (Table 2, SI), it is assessed in terms of percentage of
 332 the total city emission. The regional/city macro-sector percentages (C) are computed as

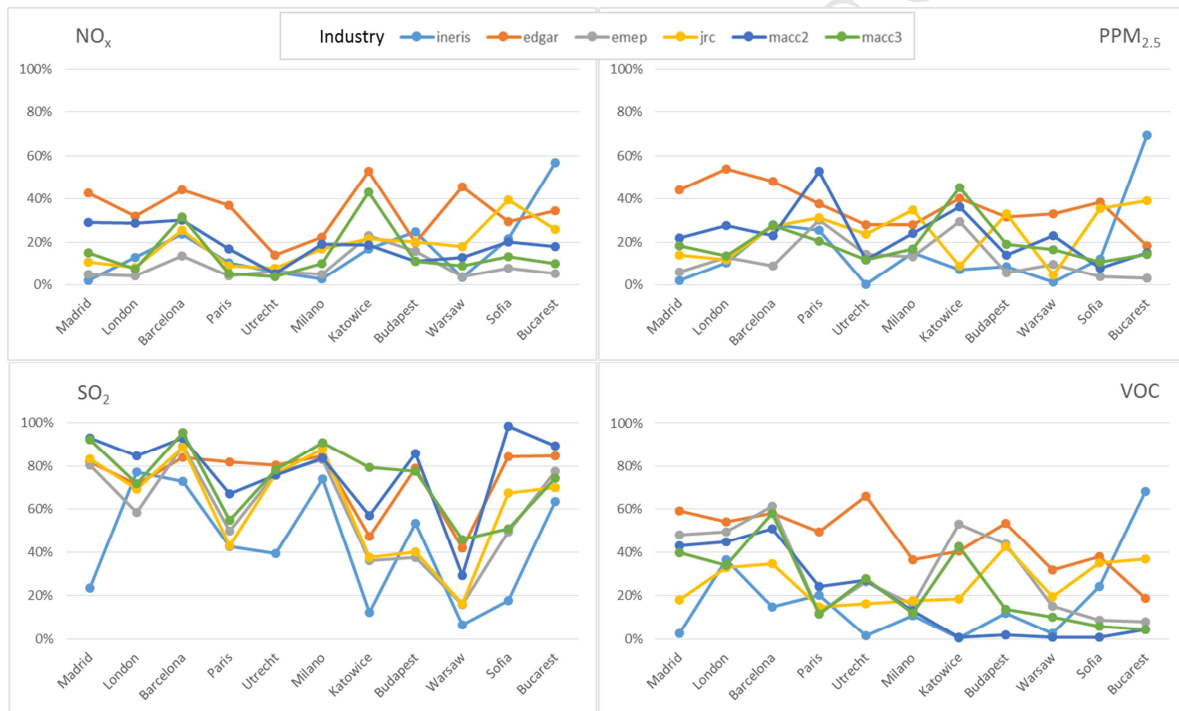
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$$C_m^p = \frac{E_m^p}{\sum_{i=1,M} E_i^p}$$

334

335 where E_m^p represents the total city emission for a pollutant “p” and macro-sector “m” and M
 336 is the total number of sectors (3 in our case).

337 In general, NO_x and SO₂ show the most robust trend among the four pollutants, while it is not
 338 possible to identify a consistent pattern for VOC in terms of cities or in terms of sectors.



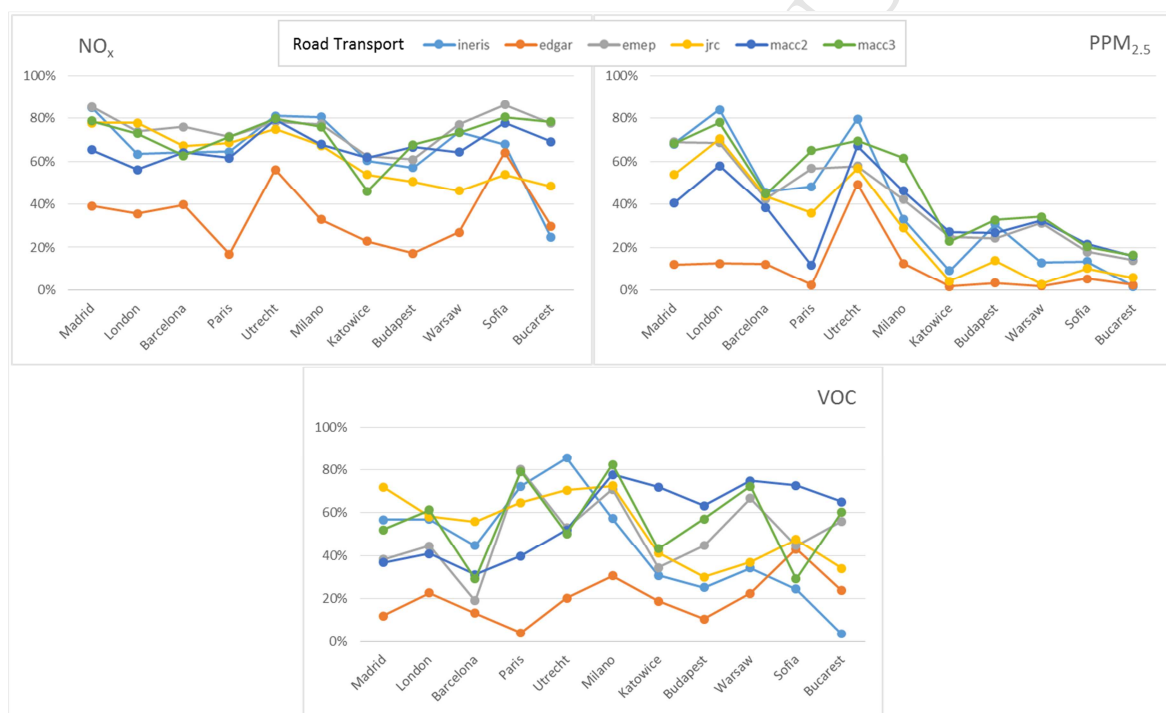
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340 **Figure 3: Sectorial allocation. Share of the total emissions for each city coming from the Industry sector.**

341

342 Among the three macro-sectors, the industrial one is by far the least consistent with large
 343 differences in many cities (values up to 5 times bigger, Figure 3). Similar inconsistency was
 344 highlighted when comparing regional downscaled inventories with bottom-up emission
 345 inventories for the same urban areas (López-Aparicio et al., 2017). While the INERIS
 346 inventory has systematically lower values for SO₂, EDGAR tends to allocate higher industrial
 347 emissions to most of the emission pollutants. It is also noticeable that, as noted
 348 in the previous paragraph, TNO-MACCI3 has reduced the amount of industrial emissions

349 located in urban areas with respect to TNO-MACCII. This also results into a larger relative
 350 contribution from SNAP07 in TNO-MACCIII when compared to the previous version.
 351 The transport sector shows the most similar shares across inventories, with the exception of
 352 VOC (Figure 4; SO₂ not shown due to its low importance for this sector). With the exception
 353 of two cities (i.e. Sofia for NO_x and VOC and Utrecht for PPM_{2,5}), EDGAR systematically
 354 allocates a much lower fraction of transport emissions to urban areas. This is probably due to
 355 the fact that emissions from on-road transport sector are distributed in EDGAR based on road
 356 types and vehicle categories and not considering the population density which is in some way
 357 taken into account in the other inventories.

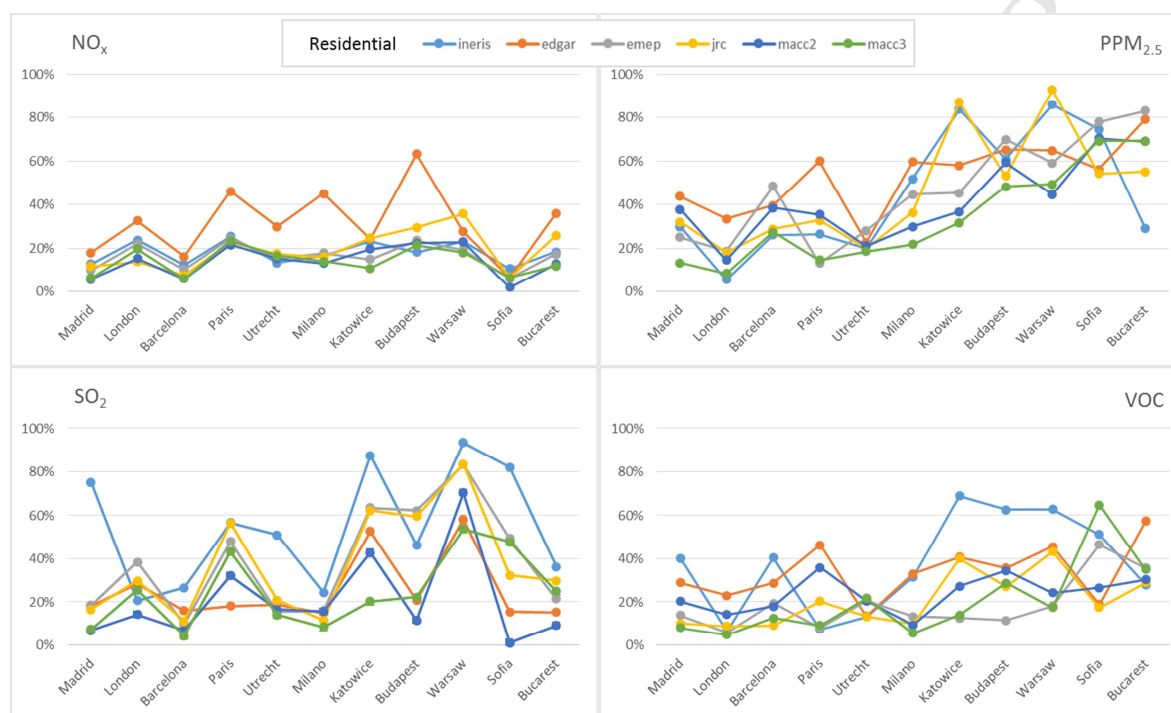


358

359 **Figure 4: Sectorial allocation. Share of the total emissions for each city coming from the SNAP07, road**
 360 **transport sector.**

361 The residential sector (SNAP02) shows good agreement among the inventories for NO_x and,
 362 in particular, there is no difference between TNO-MACCII and TNO-MACCIII (Figure 5). In
 363 the case of PPM_{2,5}, although the trends are quite consistent, there are differences in terms of
 364 percentages, indicating greater variability in emitting sources (Fuelwood, Coal), which are
 365 distributed differently by each inventory (Table 1). This is especially true for Eastern
 366 European cities, such as Bucharest, Katowice and Warsaw.

367 The EDGAR emission inventory do different patterns from all other inventories; in particular,
 368 there are larger emission estimates from the residential sector for NO_x and, to a less degree,
 369 for some cities for $\text{PPM}_{2.5}$ and VOC, which are partially compensated by lower contributions
 370 from the road transport emissions.



371
 372 **Figure 5: Sectorial allocation. Share of the total emissions for each city coming from the SNAP02,**
 373 **residential combustion sector.**

374 3.3. Activity / share analysis: the diamond approach

375 3.3.1. Methodology

376 Thunis et al. (2016b) proposed a methodology to compare emission inventories for different
 377 pollutants (i.e. PPM , NO_x , VOC, SO_2 , etc...) and activity macro-sectors (i.e. transport,
 378 industry, residential, etc...), on the basis of emission ratios between two inventories. In a first
 379 step, the emission for a pollutant p and a macro-sector s ($E_{s,p}$) is expressed as the product of
 380 an emission factor (e) and an activity (A). The emission ratio between two inventories then
 381 equals to the product of an emission factor ratio and an activity ratio:

382

$$383 \quad \hat{E}_{s,p} = \hat{e}_{s,p} \hat{A}_s$$

384

$$385 \quad \text{with} \quad \hat{E}_{s,p} = \hat{E}_{s,p}^{(1)} / \hat{E}_{s,p}^{(2)} ; \hat{e}_{s,p} = \hat{e}_{s,p}^{(1)} / \hat{e}_{s,p}^{(2)} \quad \text{and} \quad \hat{A}_s = \hat{A}_s^{(1)} / \hat{A}_s^{(2)}$$

386

387 in which superscripts 1 and 2 identify the two inventories for a pollutant p and a macro-sector
388 s .

389 The methodology detailed in Thunis et al. (2016b) aims to quantify inconsistencies in terms
390 of emission factors and activity ratios ($\hat{e}_{s,p}$ and \hat{A}_s) from the limited knowledge we have of
391 the total emission ratio ($\hat{E}_{s,p}$). It assumes that one pollutant species (denoted as p^*) can be

392 identified as reference for which the emission factors are equal in the two inventories (i.e.

393 $\hat{e}_{s,p^*} \approx 1$). With this condition, it is then possible to deduce the emission factors and activity

394 ratios from the total emission ratios: $\hat{A}_s \approx \hat{E}_{s,p^*}$ and $\hat{e}_{s,p} \approx \hat{E}_{s,p} / \hat{E}_{s,p^*}$.

395 The need to select a reference pollutant is a disadvantage of this methodology as discussed in

396 Thunis et al. (2016b). However, in this work we follow an alternative approach that does not

397 require a reference pollutant. We assume that the activity and emission factor ratios behave as

398 random variables with probability distributions following a Gaussian law centered around 1.

399 These distributions are then used to estimate the probability that the activity and emission

400 factors ratios take specific values within given intervals, while satisfying the known

401 constraint on total emission ratios. The activity and emission factor ratio are then those

402 characterised by the highest probability. The activity and emission factor ratio are used as X

403 and Y coordinates in the “diamond” diagram, where each sector-pollutant couple is

404 represented by a specific point (Figure 6). As a result of the construction, the diagonals (slope

405 = -1) provide information on the overall under-/over-prediction in terms of total emissions.

406 We can define a diamond shaped area where activity, activity shares and total emissions all

407 remain within given degrees of variation. For example, the red diamond indicates ratios of

408 activity, emission factor and total emissions all within 100% (or a factor 2) differences, while

409 the green diamond indicates ratios within 50% (or a factor 1.5). Colors and symbols are used

410 to identify pollutants and sectors, respectively. These choices are made to facilitate the

411 identification of the different ratios. The size of the symbol is then made proportional to the
412 emission magnitude (i.e. the emission for one sector is compared to the total emitted for one
413 given pollutant). This feature helps identify the biggest contributors and potential sectors to
414 mitigate. It has to be in fact remarked that this kind of analysis does not aim to draw final
415 conclusions but is instead a screening tool to highlight possible sources of inconsistencies
416 between inventories. The reader is referred to Thunis et al. (2016b) for more details.

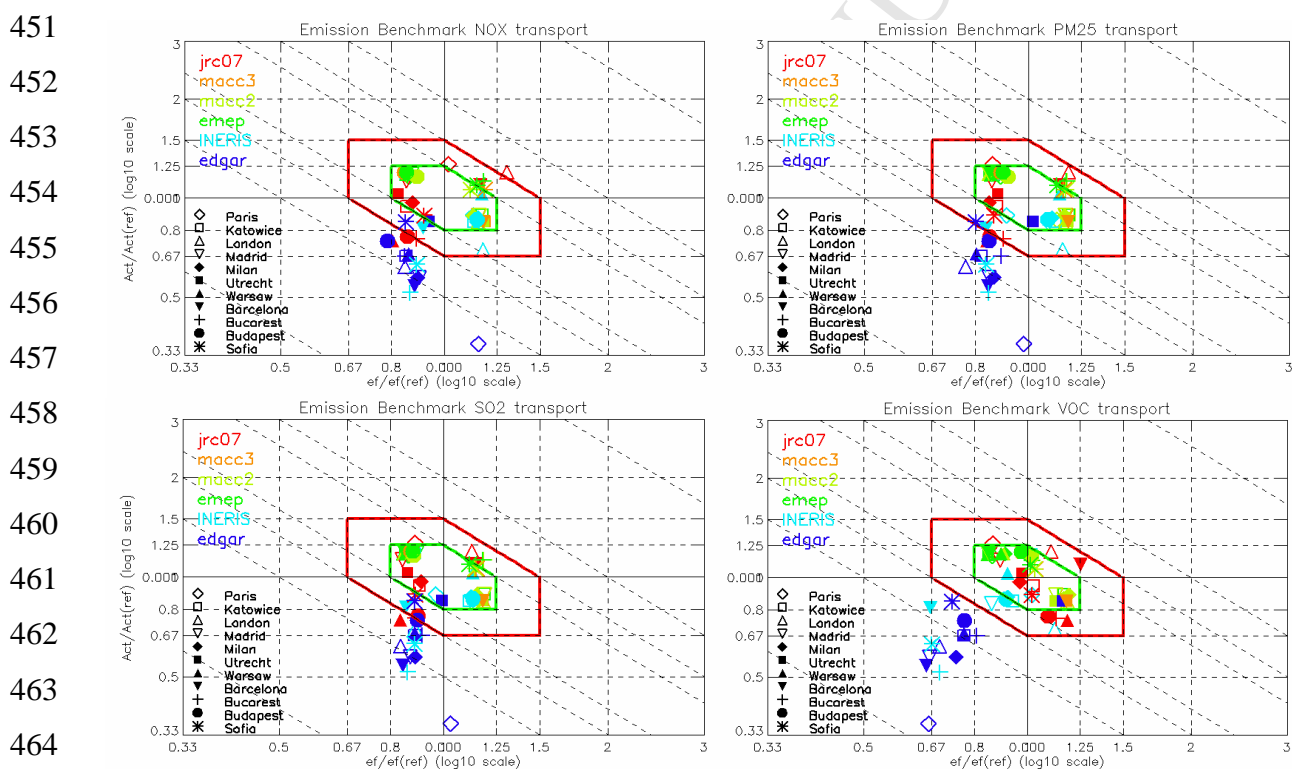
417 This approach allows us to compare the 6 inventories for 4 pollutants and for 11 cities.
418 However, the “diamond” approach only allows relative comparisons because no emission
419 inventory can be considered as the reference inventory. A synthetic inventory was therefore
420 created for the relative comparison and to be used as a reference dataset. The synthetic
421 emission values are computed as the median values of the 6 existing inventories. The results
422 discussed in the next sections are based on the differences and similarities between the six
423 top-down inventories when compared to the synthetic dataset in terms of emission sector
424 share and activity data (i.e. the “data on the magnitude of human activity resulting in
425 emissions or removals taking place during a given period of time”, IPCC, 2006). Even if the
426 median values could be affected by outlying values, the general trends describing the nature
427 of discrepancy between inventories is expected to be anyway meaningful.

428 It is noteworthy to remark that, in this work, urban emission totals are further scaled by their
429 country totals as explained in the methodology. This step is made to ensure that all urban
430 inventories originate from similar country totals and that the observed differences in the
431 diamond approach focus on the differences in terms of spatial allocation of the emissions
432 rather than on country scale biases. In this particular case, the value on the X axis is now an
433 indication of the differences in terms of activity shares rather than in terms of emission
434 factors.

435 **3.3.2. Analysis in terms of sector**

436 *Transport Sector* – There is an overall agreement between the inventories both in
437 terms of activity intensity and sectorial share as indicated by the fact that most points are
438 concentrated within the diamond shape (Figure 6). This is probably explained by the fact that
439 similar proxies are used for the spatial and sectorial disaggregation from the country totals,
440 allowing to allocate similar amounts of emissions to the considered study areas. Indeed, the

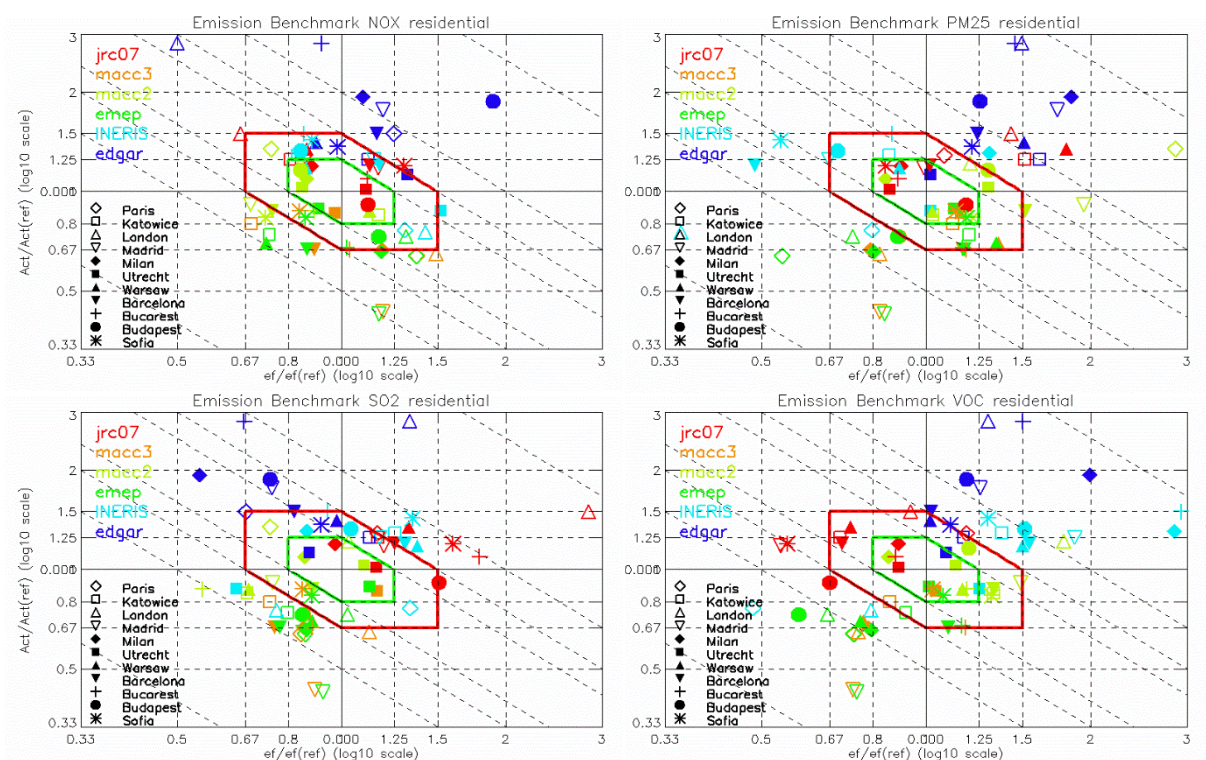
441 spatial information related to the road network (e.g. Open Street Map) is one of the most
 442 precise and shared pieces of information (especially at the spatial resolution considered in
 443 this work). The proxies used to allocate traffic intensity in each inventory are also quite
 444 similar and do not impact the emission distribution significantly. Activity is however lower
 445 according to EDGAR (especially Paris, Barcelona) and INERIS (for the Eastern European
 446 cities). This pattern is especially visible for VOC, probably related to the way the inventories
 447 deal with the evaporative emissions. It is also interesting to note that the diagram does not
 448 show the same consistency if city totals are not scaled to the inventories (not shown),
 449 indicating that most of the differences between inventories tend to originate from differences
 450 in country total estimates rather than from the spatial disaggregation proxies.



465 **Figure 6: Comparison between inventories using the diamond approach for the Road Transport sector**

466
 467 *Residential Sector* – As noted above, the most consistent trends in this sector across
 468 cities appear for NO_x , with the exception of EDGAR (Figure 7) and, for a few cities, of
 469 EMEP and MACCIII. The larger differences observed for PPM2.5 and VOC are mostly due
 470 to a problem in terms of activity share (points spread along the horizontal axis) rather than in

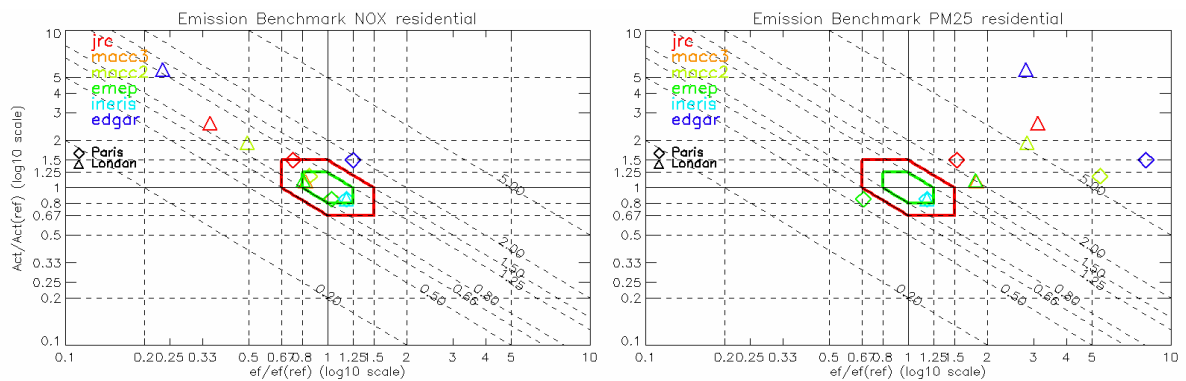
471 terms of activity intensity. Given the fact that NO_x emissions are more consistent than the
472 other pollutants, the difference must be due to activities which are not a significant source of
473 NO_x emissions, such as wood burning. It is interesting also to note the INERIS behaviour; for
474 most cities in this inventory, the points representative of $\text{PPM}_{2.5}$ and VOC are aligned on the
475 same horizontal line, indicating a similar proportional overestimation of the activity (wood
476 and coal burning) in all cities. This similar overestimation probably results from using a
477 parameter proportional to population to scale up wood-burning emissions. In INERIS,
478 emissions from SNAP02 are in fact distributed according to a proxy based on population,
479 land-use and the French bottom-up emission inventory, fitting other parameters in order to
480 differentiate urban and non-urban areas. Differences across the cities are hence mainly due to
481 inconsistencies in terms of shares of activities within the same sector, with a proportion that
482 depends on the importance of wood and coal burning in each city: the higher the importance
483 of wood and coal burning, the higher the uncertainty of emission distribution in this sector.
484 For instance, in countries such as Germany and Spain, emissions from residential heating are
485 lowest, whereas Romania, Poland and France have the highest levels (Terrenoire et al., 2015).
486 This confirms the importance of updating the emission estimates from the residential
487 combustion sector, as stated by Denier van der Gon et al. (2015) and developing a proxy
488 which would allow for a better and common representation of the spatial distribution of wood
489 and coal usage, also taking into account site-specific features such as the proliferation of
490 district heating in many cities which results in a smaller and secondary usage of conventional
491 wood-fired stoves.



492

493 **Figure 7: Comparison between inventories using the diamond approach for the Residential Combustion**
 494 **sector**

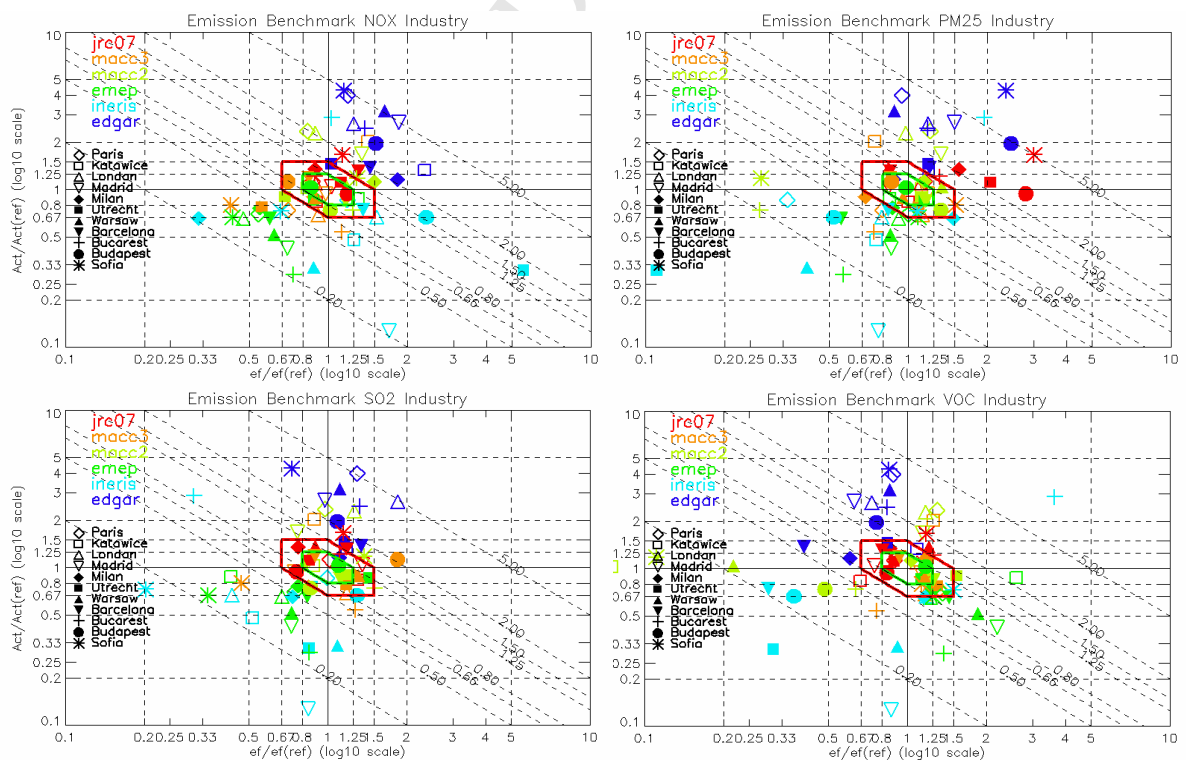
495 *Residential Sector: INERIS as a reference inventory* - INERIS can be considered as a
 496 reference inventory for France and UK since the national emission inventories are directly
 497 introduced as spatial proxies (1km² resolution) in the European emission dataset. In this
 498 section, assuming that the national bottom-up inventories are supposed to be the most
 499 accurate, the terms “over”- or “under-estimation” compared to this new reference can then be
 500 used for London and Paris. Figure 8 provides a rather different picture of the spread of
 501 emissions from the residential sectors for both NO_x and PPM_{2.5} compared to Figure 7. For
 502 NO_x, the differences in terms of activities and share for JRC07 and EDGAR in London are
 503 emphasised and become much bigger, while the MACCIII ones are now more similar to the
 504 reference inventory. For PPM_{2.5}, in Paris the patterns are similar to those shown before, while
 505 the results are more diverse for London. Here, MACCII and JRC show larger differences
 506 from the reference inventory than before with problems of both activity levels and activity
 507 shares; while EDGAR, MACCIII and EMEP are closer to the activity levels of the reference
 508 inventory.



509

510 **Figure 8: Comparison between NO_x and $\text{PPM}_{2.5}$ inventories using the diamond approach for the**
 511 **residential sector with INERIS as a reference**

512 *Industrial Sector* – This sector is the one that needs the biggest efforts and
 513 improvements. In particular, EDGAR consistently has higher emissions from this sector,
 514 while INERIS and EMEP assign lower values (Figure 9). There are in general differences
 515 between all the inventories and for all the pollutants that appear to be due to discrepancies
 516 both in terms of activity levels and shares, as indicated by the wide horizontal and vertical
 517 spreads of the points in Figure 9.



518

519 **Figure 9: Comparison between inventories using the diamond approach for the Industrial sector**

520 Being largely based on Large Point Source (LPS) information, the differences seen in this
521 industrial sector probably result mainly from differences in the choice of the relevant
522 databases, reporting location and ‘weight’ of the facilities: as reported by Wang et al., 2012,
523 the spatial accuracy of the LPS information can significantly affect the accuracy of the
524 associated chemical transport models. The LPS databases differ in terms of spatial accuracy
525 and thematic details (capacity or size of the single emitting facility) which strongly affect the
526 resulting spatial variability, often combined with other inconsistencies with larger
527 consequences (opening and closure of facilities and regular updates of the underlying
528 databases) (Janssen-Maenhout et al., 2015; Ferreira et al., 2013). All the inventories
529 considered rely on different versions of the E-PRTR database. The industrial emissions that
530 cannot be linked to a specific LPS facility (i.e. diffuse fraction) since they are often below the
531 threshold of individual facility reporting (e.g. to E-PRTR, <http://prtr.ec.europa.eu>) are
532 included in this sector. Especially for small countries, the existence of threshold makes the
533 PRTR dataset less valuable and it requires additional data for point sources falling below the
534 threshold. Hence, the diffuse fraction has to be spatially allocated according to different
535 proxies that may greatly contribute to the inconsistencies among inventories (Table 1). As
536 already pointed out in section 3.2.1, in TNO-MACCII for example, emissions from SNAP34
537 are distributed based on the E-PRTR database, their TNO internal LPS database and on
538 population distribution in the case of the diffuse fraction. The TNO-MACCCIII introduces an
539 improvement in the distribution of this part of the industrial sector emissions that cannot be
540 represented by point sources (LPS). This improvement can be observed in Figure 9 when
541 comparing MACCII and MACCCIII and will avoid a likely over-allocation of industrial
542 emissions in urban areas. An alternative choice is the one of the EDGAR inventory which
543 doesn’t define any share of diffuse industrial emission but the whole national total is assigned
544 to point sources.

545 To summarise, of the three sectors considered, road transport is the most robust, with
546 inconsistencies mostly on activities for EDGAR and, to a minor extent, INERIS. This sector
547 has also been reported by Lopez-Aparicio et al. (2017) as the most consistent although, when
548 comparing it with bottom-up approaches, all considered inventories showed underestimation
549 of NO_x and PPM₁₀ emissions. As stated in this same paper, non-exhaust emissions due to
550 resuspension are the main reason of discrepancies for PPM₁₀, whereas the disaggregation of

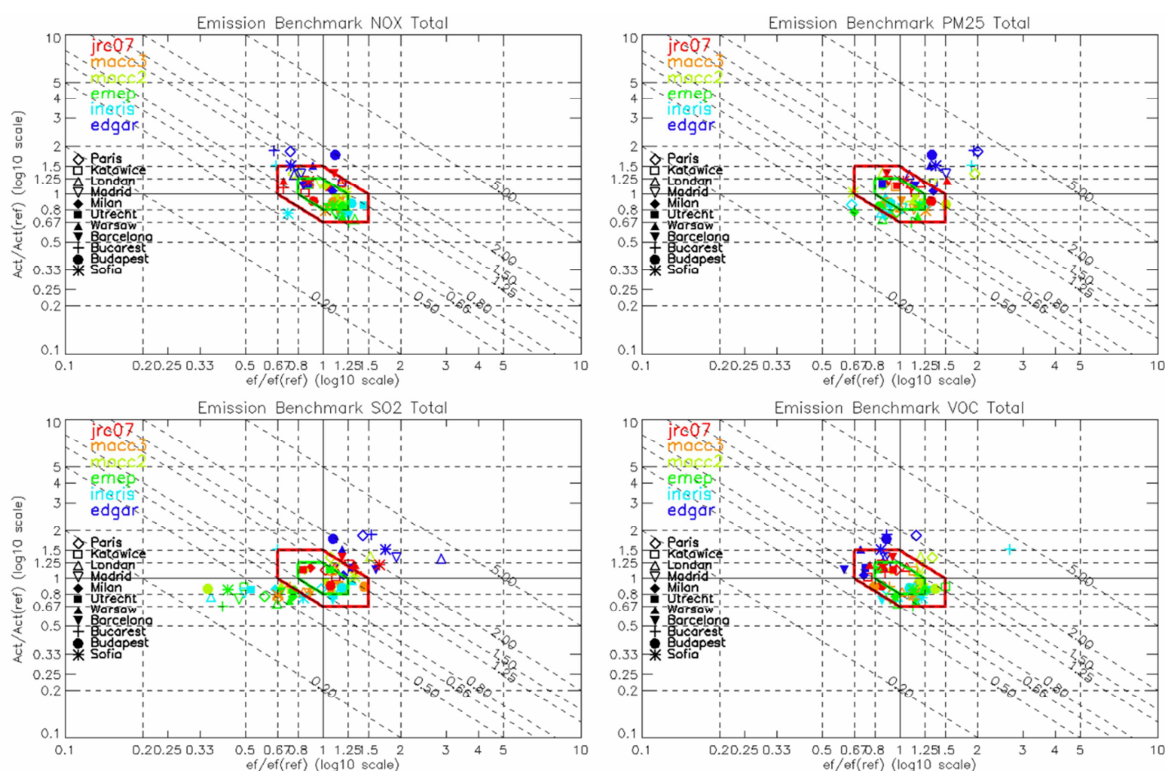
551 traffic emissions in urban areas based on population may entail lower activity and the
552 subsequent underestimation of NO_x emissions.

553 The other two sectors, in particular the industrial sector, highlight problems with both activity
554 levels and activity shares. It is also interesting to note that in general the problems are similar
555 for all cities in each inventory. This might mean that specific parameters of each urban area,
556 such as land use, population density and degree of urbanization, play an important role in
557 emission distribution.

558 **3.3.3. Analysis in terms of pollutants**

559 If we sum-up the emissions from the three sectors and use the diamond approach, we observe
560 greater consistency between the inventories for all pollutants than for single macro sectors
561 (Figure 10). This consistency results from the compensation effects of higher and lower
562 estimations in the individual macro-sectors. This is particularly notable for the EDGAR
563 inventory, where the estimates of traffic emissions, which are lower when compared to the
564 other datasets, are compensated by higher ones from the industrial and residential sectors.

565 The largest consistencies are mostly observed for NO_x and VOC and the lowest for $\text{PPM}_{2.5}$
566 and SO_2 . For SO_2 , the discrepancy mostly lies in the sectorial share as indicated by the large
567 horizontal spread. It is interesting to note the differences for SO_2 between MACCII and
568 MACCIII which are important in cities like Budapest, but small in others like Paris. These
569 differences can be attributed to changes in the proxies used to distribute industrial emissions
570 resulting in differences in terms of share. The proxies for industrial activities were in fact a
571 specific target of the upgrade to MACCIII: as previously indicated, diffusive industrial
572 emissions are allocated based on population in MACCII whereas industrial land use is used in
573 MACCIII.



574

575 **Figure 10: Comparison between inventories using the diamond approach for the total emissions from all**
 576 **the considered sectors**

577 3.3.4. Uncertainties of emissions at urban scale

578 In this section we quantitatively summarise the results described previously. For this purpose,
 579 an estimate of the standard uncertainty is calculated for each pollutant and each sector for the
 580 6 emission inventories. The relative standard uncertainties (u) for each city are calculated for
 581 each pollutant “p” and sector “s” according to the following formula:

582

$$u(p, s) = \frac{t^{(n-1)} * \sigma}{E_{p,s}^{mean} \sqrt{n}}$$

583 where $E_{p,s}^{mean}$ is the mean of the 6 emission values for a given sector and pollutant, “ $t^{(n-1)}$ ”
 584 represents the Student’s t-test probability value corresponding to a 95% confidence level and
 585 n is the number of available inventories (n=6). The uncertainties apply to the emissions at
 586 urban scale since the starting point of all emission inventories is the national emissions total,
 587 which is identical for all of them. The uncertainties therefore reflect the expected variations

588 resulting from the application of different spatial proxies to allocate the emissions in the
 589 urban environment.



590
 591 **Figure 11: Percentage standard uncertainties for the transport, residential and industry sectors for the 11**
 592 **selected cities. VOC for SNAP07 and NO_x for SNAP02 are not shown here, since, for these pollutants, the**
 593 **contribution from these sectors to the total emission is less than 10% (average value of all considered**
 594 **inventories)**

595 **Figure 11** shows uncertainties up to 100% (and over this threshold for VOC) for the residential
 596 and industrial sectors whereas uncertainties of ~25% are found in the transport sector. These
 597 remarkably small uncertainties for the road transport are due to the consistency among the
 598 inventories observed in the previous paragraphs and explained by the usage of very similar
 599 spatial proxies.

600 It has also to be noticed that the uncertainty is very low and similar for all pollutants,
 601 including PPM_{2.5} and VOC which, differently from NO_x and SO_x, have a significant
 602 contribution from non-exhaust emissions. This entails that even if non-exhaust emissions

603 from resuspension is a major source of uncertainty in national inventories, they are spatially
604 distributed in the same way by the different dataset we analysed.

605 Furthermore, it is also interesting to note the overall good agreement for Utrecht and
606 Barcelona. When looking at the combination of all the considered sectors, uncertainties are
607 generally reduced for all cities and pollutants, due to a compensation effect, although they are
608 still very high for SO₂ and some cities such as Paris, Bucharest, Budapest and Sofia.

609 **4. Conclusions**

610 With the analysis presented in this paper we introduce an innovative approach for the spatial
611 analysis of proxy-based emission inventories gridded at the European scale (i.e. ~7 to ~10 km
612 resolution).

613 Several emission inventories are available for Europe, differing substantially in terms of total
614 emissions, sectorial emission shares and spatial distribution. It should be noted that total
615 emissions per major sector and country can be different in the existing inventories even when
616 based on the country reported data to CLRTAP. Since reporting takes place on an annual
617 basis, emissions are reported annually for every historical year back to 1990. When
618 methodological changes are made in the countries' inventory, these changes are implemented
619 for all historical years and this may lead to significant changes in historical emission
620 estimates. A key example is residential combustion (SNAP02) where country reported
621 PPM_{2.5} emissions for the EU28 have increased by more than 20% between 2013 and 2016
622 reporting. The changes are due to the differences in measurement techniques to quantify PPM
623 emissions from small combustion installations and the lack of a clear definition for the basis
624 on which PPM emissions should be reported.

625 Hence, while for the most important cities, bottom-up inventories often do exist providing
626 more accurate information at a higher spatial resolution, for extensive air quality modelling it
627 is still of utmost importance to be able to rely on consistent and harmonised European-wide
628 inventories. In order to assess the potential impact of the choice of a specific inventory for air
629 quality modelling, we analysed their spatial patterns of behaviour looking at representative
630 urban areas over Europe.

631 A distinctive outcome of the work presented in this paper is the significant difference
632 between regional emission inventories due to the choices made in terms of disaggregation

633 approach and selection of spatial proxies. Moreover, our study underlines those sectors where
634 additional efforts are needed in the framework of regional air quality assessments.

635 For all inventories, it appears necessary to review, compare and develop new methodologies
636 and proxies for the spatial disaggregation of emissions from the industrial sector. The large
637 inconsistencies observed may be in part due to the different methodologies and assumptions
638 used to allocate the diffuse industrial emissions. Emissions from Medium Combustion Plants
639 (MCP, >1 MWth and <50 MWth) are just now starting to be regulated by the EC (Directive
640 2015/2193) and consequently no information on these facilities (e.g. geographic location,
641 emissions) is available. Considering that the number of MCP in the EU is estimated to be
642 around 143,000 (European Union, 2016b) and that large part of them are used for providing
643 electricity and energy for processes, having detailed information of these facilities would
644 improve the allocation of industrial emissions and reduce the observed discrepancies.

645 As it was previously highlighted (e.g. Guevara et al., 2014; López-Aparicio et al., 2017), the
646 use of the population density as proxy to allocate the diffuse fraction of industrial emissions
647 results in an over-allocation of emissions in urban areas (e.g. TNO-MACCII). Even though
648 the distribution of diffuse emission based on land use cover data is an improvement (e.g.
649 TNO-MACCI), this approach still needs further development. The main reason is that the
650 land use classification includes in the 'industry' class areas that are commercial rather than
651 industrial. It is also important that inventories base the distribution of emissions from Large
652 Point Sources emissions on the latest available LPS dataset and that, for the sources below
653 the emission thresholds which can be very important especially in small countries,
654 appropriate complementary dataset are adopted.

655 Particular attention should also be given to the residential sector, if possible comparing
656 bottom-up estimates to better calibrate the spatial patterns of emissions from wood and coal
657 burning, in order to reflect the significant variations between countries. Furthermore, city-
658 specific features such as district heating should be taken into account; in these cases, a much
659 lower share of residential emissions would be expected over the city compared to individual
660 heating sites. At the same time, the traditional proxies used for gridding residential emissions
661 (e.g. population density) would not be any more relevant.

662 Based on the differences highlighted in this analysis, we list the main aspects for each
663 inventory that could be important to review. It has to be noted that these issues have been

664 identified by a comparison between gridded Top-down inventories and, since there is no way
665 to directly verify the results of the disaggregation, they have to be considered as hints for a
666 critical revision of the chosen downscaling methodologies. Considering that none of the
667 analysed inventories can be considered as a true reference, it is also important to emphasise
668 that the consensus found for certain sectors/pollutants (e.g. NO_x traffic emissions) does not
669 necessarily indicate that the uncertainty in the emission inventories is low. A high level of
670 consensus may be due to similar assumptions used in all the inventories or similar sources of
671 uncertainties (e.g. laboratory versus on-road traffic emission factors, Degraeuwe and Weiss,
672 2017):

673

674 *EDGAR*: The importance of residential and road traffic emissions appears to be
675 systematically estimated as lower (SNAP02) and higher (SNAP07) over urban areas (the split
676 of national totals in terms of macro-sectors seem to be in line with the other inventories).

677 This emission inventory is generally the one that presents the largest inconsistencies when
678 compared to the other analysed emission inventories. At the same time, it is the only one that
679 offers a global spatial coverage, hence dealing with a wider range of data sources which need
680 to ensure consistency and representativeness for the different parts of the globe. This fact
681 indicates that when working at different scales, the availability and detail of spatial proxies
682 may change.

683 *INERIS*: The spatial disaggregation of emissions from on-road traffic should be checked for
684 some eastern cities (Bucharest, Sofia) for which much lower values are reported.

685

686 *MACCII-MACCIII*: As expected, a general improvement from MACCII to MACCIII is
687 observed with very large changes for some of the cities. In general, in MACCIII, industrial
688 and residential emissions are now distributed more outside of the city domains and less
689 within the urban areas (Kuenen et al., 2014). For the industrial sector, the area sources which
690 were distributed using population density in MACCII are gridded over the industrial land use
691 area in MACCIII.

692 For the residential sector, MACCIII assigns lower amounts of wood or coal burning to the
693 city centres. The estimates for wood combustion and the spatial distribution have been

694 revised; for Eastern European countries, the emissions from this source have been
695 significantly increased.

696

697 *EMEP* Industrial emissions proxies and methodologies should be checked, since for all
698 pollutants much lower values than the other inventories are reported.

699

700 *JRC* Particular attention should be given to residential emissions in Eastern Europe (in
701 particular Poland); the country inter-variability of urban residential emissions (Wood and
702 Coal burning) has not been properly addressed.

703

704 These first results provide an insight for the identification of the main issues and differences
705 among the emission inventories commonly used at the European scale for air quality
706 modelling and some recommendations are provided with the aim of working towards the
707 harmonisation of spatial downscaling and proxy calibration, in particular for policy purposes.

708 Further work will be needed in order to provide a deeper insight into emission spatial patterns
709 through a comparison at a finer scale with local bottom-up inventories, which rely on massive
710 and detailed spatial information such as point sources, detailed censuses and traffic statistics
711 or, as alternative, with the national grids at 0.1*0.1 degrees resolution recently reported to
712 *EMEP* by many European countries. Such a comparison would help calibrate proxies at a
713 regional/local scale rather than using common ones for such diverse and extended areas.

714 Finally, and considering that one of the main aims of the analysed inventories is to provide
715 emission inputs for air quality modelling, future work should also consider the influence of
716 uncertainties in proxy-based emission inventories when they are used in atmospheric
717 chemistry models.

718

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- Tools for comparison of downscaled emissions at different levels are presented
- Urban Emissions of NO_x, PPM_{2.5}, SO_x and VOC from SNAP02, 34 and 07 are compared