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1 **Applying Air Pollution Modelling within a Multi-Criteria Decision Analysis Framework to**  
2 **Evaluate UK Air Quality Policies**

3

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23

24 **Abstract**

25 A decision support system for evaluating UK air quality policies is presented. It combines the  
26 output from a chemistry transport model, a health impact model and other impact models  
27 within a multi-criteria decision analysis (MCDA) framework. As a proof-of-concept, the  
28 MCDA framework is used to evaluate and compare idealised emission reduction policies in  
29 four sectors (combustion in energy and transformation industries, non-industrial  
30 combustion plants, road transport and agriculture) and across six outcomes or criteria  
31 (mortality, health inequality, greenhouse gas emissions, biodiversity, crop yield and air  
32 quality legal compliance). To illustrate a realistic use of the MCDA framework, the relative  
33 importance of the criteria were elicited from a number of stakeholders acting as proxy  
34 policy makers. In the prototype decision problem, we show that reducing emissions from  
35 industrial combustion (followed very closely by road transport and agriculture) is more  
36 advantageous than equivalent reductions from the other sectors when all the criteria are  
37 taken into account. Extensions of the MCDA framework to support policy makers in practice  
38 are discussed.

39

40 **Key words**

41 Air quality policies; Air pollution modelling; Decision analysis; Health impacts

42 **Highlights**

- 43 • A modelling framework for evaluating UK air quality policies has been developed
- 44 • The framework combines decision analysis, air pollution and impact modelling
- 45 • Multi-criteria decision analysis is used for comparative evaluation of policies
- 46 • The framework is used to evaluate idealized UK air quality policies

47

## 48 **1. Introduction**

49 Atmospheric chemistry-transport models have been used in various ways to evaluate air  
50 quality policies. They have been used mainly as either stand-alone simulation models  
51 (Chemel et al 2014) or embedded within comprehensive integrated assessment tools (Lim et  
52 al 2005, Amann et al 2011, Thunis et al 2012, Carnevale et al 2012a, Carnevale et al 2012b,  
53 Oxley et al 2013). However, if air pollution modelling is to be used in practice to help policy  
54 makers choose amongst potentially competing policies, appropriate methods for  
55 comparative evaluation of such policies are needed (Browne and Ryan 2011). Such methods  
56 include cost-effectiveness analysis (CEA), cost-benefit analysis (CBA) and multi-criteria  
57 decision analysis (MCDA).

58 CEA is mainly used when the policies are assessed against two criteria: monetary (e.g. cost  
59 of the policy) and non-monetary (e.g. effectiveness or benefit of the policy such as health  
60 gain). A cost-effectiveness ratio (cost per unit gain) is calculated for each policy and is used  
61 as the metric for comparative evaluation; the policy with the lowest ratio is deemed to be  
62 the most cost-effective. CBA is similar to CEA except that the non-monetary criterion is  
63 monetised and the ratio of cost to benefit becomes dimensionless, which eases comparison.  
64 CBA can cater for more than two criteria because all the non-monetary criteria are  
65 monetised. MCDA is different from CEA and CBA in one important aspect: the comparative  
66 evaluation between policies is carried out across several criteria without the need to  
67 monetise the criteria i.e., the criteria are maintained in their natural units. Browne and Ryan  
68 (2011) and Scricciu et al (2014) discuss the advantages and disadvantages of the different  
69 methods.

70 The use of MCDA to support environmental decision making has solid foundation (Kiker et al  
71 2005, Zhou et al 2006). It has been recommended for this purpose by some UK Government  
72 Departments (DCLG, 2009). Huang et al (2011) provide a review of the applications of MCDA  
73 in environmental sciences. The applications of MCDA of relevance to this study include  
74 evaluation of flood risk management policy options in Scotland (Kenyon 2007), air quality  
75 policies in the UK (Philips and Stock 2003, Fisher 2006), and climate change mitigation and  
76 adaptation policies (Konidari and Mavrakis 2007, Scricciu et al 2014, Chalabi and Kovats  
77 2014). Apart from the flood risk management MCDA study, the abovementioned studies  
78 describe MCDA frameworks rather than evaluate specific polices.

79 The aim of this study is to demonstrate the use of an air pollution model alongside impact  
80 models within a MCDA framework to evaluate and compare relatively simple UK air quality  
81 policies across several criteria which include health and health inequality. We used the  
82 EMEP4UK chemical transport model (Vieno et al 2010, Vieno et al 2014) to simulate air  
83 pollution over the UK for 2010. Results from an earlier version of the model have been used  
84 for health impact estimation (Doherty et al 2009, Vardoulakis and Heaviside 2012, Heal et al  
85 2013).

86 The paper is structured as follows. Section 2 describes the methods used in this study.  
87 Section 3 gives the results of the MCDA analysis. Section 4 highlights the main findings and  
88 discusses the merits and challenges of this approach in theory and practice, and the final  
89 section concludes. The paper is supported by five technical appendices.

90

91

## 92 **2. Methods**

### 93 **2.1 Multi-Criteria Decision Analysis (MCDA)**

94 Several MCDA methods with varying degrees of complexity could be used to carry out  
95 comparative evaluation of air quality policies. Exposition of MCDA methods are given by  
96 Belton et al (2002) and Figueira et al (2005). The method we used in this study belongs to  
97 the family of Simple Multi-Attribute Rating Techniques (SMART) and is also known as the  
98 weighted-sum method (Cunich et al 2011, Dowie et al 2013). We used the SMART software  
99 tool *Annalisa* (©Maldaba Ltd, <http://maldaba.co.uk/products/annalisa>) for implementing  
100 the MCDA. *Annalisa* has been used as a decision support framework for risk prioritisation of  
101 environmental health hazards (Woods et al 2016).

102 The elements of this MCDA method are: (i) a set of policies, (ii) a set of criteria against which  
103 the policies are evaluated and compared, (iii) a set of preference weights which give the  
104 relative importance of each criterion (the weights add up to 1), (iv) a set of models to  
105 determine the impact of each policy on each criterion (each impact is normalised between 0  
106 and 1), and (v) a method for integrating the impacts and the weights to give a total impact  
107 for each policy across all the criteria. The total impacts of all the policies are the metrics  
108 which are used to compare the policies. If the impacts are burdens then the policy with the  
109 lowest total impact is deemed to be the “optimal policy”. Conversely, if the impacts are  
110 benefits then the policy with the highest total impact is the “optimal policy”.

111 The theoretical details of the MCDA method are provided in Supplementary Material A to E.  
112 In summary, Supplementary Material A describes the stakeholder survey used to rank the  
113 criteria (described in Section 2.4: mortality, health inequality, greenhouse gas emissions, air  
114 quality legal compliance, biodiversity, crop yield) in order of their importance.

115 Supplementary Material B describes the method of converting the ranks obtained from the  
116 stakeholders to a set of aggregated weights for the criteria. Supplementary Material C  
117 shows the method of normalising the impacts across the criteria to make them  
118 dimensionless. Supplementary Material D provides details on the measurement of pollution  
119 exceedance. Finally, Supplementary Material E describes the MCDA calculation.

## 120 **2.2 Air pollution modelling**

121 For the purposes of this study, pollutant concentrations of nitrogen dioxide (NO<sub>2</sub>), ozone  
122 (O<sub>3</sub>) and particulate matter with aerodynamic diameter of less than 2.5 µm (PM<sub>2.5</sub>) were  
123 simulated by the EMEP4UK atmospheric chemistry transport model. EMEP4UK is a nested  
124 regional application of the main European Monitoring and Evaluation Programme (EMEP)  
125 MSC-W chemical transport model (Simpson et al, 2012) targeted specifically at air quality in  
126 the UK. EMEP4UK uses one way nesting to scale down from 50 x 50 km horizontal resolution  
127 in the EMEP greater European domain to 5 x 5 km resolution in a nested inner domain  
128 located over the British Isles. Model outputs include surface concentrations of gaseous  
129 pollutants and particulate matter (both primary and secondary) along with their rates of wet  
130 and dry deposition. The driving meteorology for EMEP4UK was taken from the Weather  
131 Research and Forecasting (WRF) model including data assimilation of 6-hourly  
132 meteorological reanalyses from the US National Center for Environmental Prediction (NCEP)  
133 global forecast system. Continuously constraining the WRF fields to observations ensures  
134 that the meteorology supplied to the chemistry-transport model is closely representative of  
135 the real weather conditions prevailing throughout the simulations. Full details of the WRF-  
136 EMEP4UK coupled model are described elsewhere (Vieno et al 2010, Vieno et al 2014).

137

138

## 139 **2.3 Policies**

140 In this study we assess relatively simple policies that would reduce UK emissions from  
141 specific sectors by fixed fractions. We use the Selected Nomenclature for Air Pollution  
142 (SNAP) emission sectors, as defined by the EMEP CEIP (Centre on Emissions Inventories and  
143 Projections: [www.ceip.at](http://www.ceip.at)). In particular, we evaluate policies that control emissions from  
144 the following sectors: SNAP 1. 'Combustion in energy and transformation industries'; SNAP  
145 2. 'Non-industrial combustion plants'; SNAP 7. 'Road Transport'; and SNAP 10. 'Agriculture'.

### 146 **2.3.1 Base simulation**

147 The base simulation was for 2010. It used anthropogenic emissions of primary pollutants  
148 and pollutant precursors as reported in official inventories for that year. Annual gridded  
149 emissions of nitrogen oxides ( $\text{NO}_x = \text{NO} + \text{NO}_2$ ), sulphur dioxide ( $\text{SO}_2$ ), ammonia ( $\text{NH}_3$ ),  
150 Volatile Organic Compounds (VOCs), carbon monoxide, and particulate matter ( $\text{PM}_{10}$  and  
151  $\text{PM}_{2.5}$ ) were taken from the National Atmospheric Emissions Inventory (NAEI,  
152 <http://naei.defra.gov.uk>) for the UK and from CEIP for the rest of Europe. The provided  
153 anthropogenic emissions for each species are apportioned across a standard set of ten SNAP  
154 source sectors as defined by EMEP CEIP. Emissions are distributed vertically within the  
155 model according to SNAP sector. Natural emissions (mainly biogenic isoprene) were  
156 calculated interactively by the model. Model outputs of pollutant concentration and  
157 deposition fluxes were utilised for impacts calculations. A detailed evaluation of the base  
158 EMEP4UK simulation against measured pollutant concentrations is given by Lin et al (2016)  
159 (here we use only the year 2010 from the decade long simulation examined in that paper).



160 **2.3.2 Variant simulations**

161 Variant simulations were performed for 2010 to examine the response of atmospheric  
162 concentrations and deposition rates to a change in UK emissions from several individual  
163 SNAP sectors. Emission from specific SNAP sectors were switched off (i.e. 100% reductions)  
164 to assess the maximum influence of reductions in emissions in a given sector on pollutant  
165 concentrations:

- 166 1. 100% reduction in UK emissions from the 'Combustion in energy and transformation  
167 industries sector' (SNAP 1)
- 168 2. 100% reduction in UK emissions from 'Non-industrial combustion plants' (SNAP 2)
- 169 3. 100% reduction in UK emissions from 'Road Transport' (SNAP 7)
- 170 4. 100% reduction in UK emissions from 'Agriculture' (SNAP 10)

171 In these integrations, the UK anthropogenic emissions of all species in the relevant SNAP  
172 sector were set to zero (in both the outer and inner EMEP4UK domains), while UK emissions  
173 in the other SNAP sectors and all anthropogenic emissions outside the UK were left  
174 unchanged. Natural emissions and meteorology were also unchanged. The differences  
175 between these variant simulations or perturbations and the base simulation therefore arise  
176 solely from the removal of UK anthropogenic emissions in that particular SNAP sector.

177 **2.4 Criteria**

178 There is no one ideal or perfect set of criteria to use as basis for comparing the expected  
179 performance of the above air quality policies. The selection of the criteria is a subjective  
180 matter. Ideally from a decision-analytical perspective, the criteria should be independent of  
181 each other. However in practice this independence can rarely be achieved. Informed by a

182 stakeholder workshop, the following six criteria were chosen: mortality, health inequality,  
183 greenhouse gas emissions, air quality legal compliance, biodiversity and crop yield. The  
184 workshop participants came from academia, government departments and environmental  
185 consultancies. The selected criteria represent a spectrum of higher level criteria which span  
186 a range of environmental policy concerns: human health (mortality), social (health  
187 inequality), climate (greenhouse gas emissions), legal compliance (pollution exceedance),  
188 natural ecosystem health (biodiversity) and agricultural ecosystem health (crop yield). The  
189 impacts on all the criteria are presented as burdens. We provide below a brief description  
190 of each criterion and the quantitative metric that is used to model the impact of each policy  
191 on the criterion.

192 *Mortality:* We calculated the mortality impact of long-term PM<sub>2.5</sub> exposure for the base  
193 simulation and each SNAP sector variant simulation using a life table model (Miller and  
194 Hurley 2003) and following the health impact assessment method of COMEAP (2010). The  
195 main output of the life table model used as a metric in the MCDA analysis is the Years of Life  
196 Lost (YLL).

197 *Health inequality:* We reconstructed a socioeconomic deprivation index based on the  
198 Income and Employment domains of the English Index of Multiple Deprivation (IMD) 2010.  
199 IMD is the composite measure of deprivation constructed from a number of deprivation  
200 indicators (such as income, employment, education skills and training) using appropriate  
201 weights to produce a single overall index of multiple deprivation for small geographical  
202 areas known as Lower Super Output Areas (LSOAs). Each LSOA has about 1,500 inhabitants.  
203 The IMD is grouped into 10 deciles with 1 representing the least deprived 10% of the  
204 population and 10 the most deprived 10%. Based on separate life tables created for each

205 decile of IMD (to reflect differences in underlying mortality risk), we used the change in  
206 years of life gained per 5<sup>th</sup> to 9<sup>th</sup> decile of IMD as the measure of health inequality.

207 *Greenhouse gas emissions:* We calculated the CO<sub>2</sub>-equivalent emissions reductions  
208 associated with each policy, based on the impacts on the Kyoto protocol gases (UNFCCC,  
209 2008). Other species that influence climate, such as ozone (O<sub>3</sub>) and aerosols are not  
210 included.

211 *Pollution exceedance:* We used the European Commission's air quality standards to define  
212 the standards for the relevant air pollutants: PM<sub>2.5</sub> and O<sub>3</sub> (Table 1)

213 **Table 1.** EC air quality standards for PM<sub>2.5</sub> and O<sub>3</sub> (EC, 2015)

Pollutant	Concentration	Averaging period	Legal time entered into force	Permitted exceedance each year
PM <sub>2.5</sub>	25 µg m <sup>-3</sup>	1 year	1 Jan 2015	N/A
O <sub>3</sub>	120 µg m <sup>-3</sup>	Max daily 8 h mean	1 Jan 2010	25 day averaged over 3 years

214  
215 NO<sub>2</sub> is also an important pollutant in terms of legal compliance, but due to its short lifetime,  
216 its concentrations show steep gradients away from its sources such as major roads. As the  
217 monitoring sites for which NO<sub>2</sub> exceedances are typically reported (e.g. in 2010 in the UK)  
218 are situated at roadside locations, simulating NO<sub>2</sub> levels comparable with these reported  
219 occurrences, would require road emissions to be modelled explicitly, which is not possible in  
220 the gridded chemistry transport model despite its fairly high horizontal resolution of 5 km  
221 by 5 km. Hence for the purpose of legal compliance only PM<sub>2.5</sub> and O<sub>3</sub>, which have lifetimes  
222 sufficiently long to undergo regional transport, and are hence suitable to be simulated in a 5  
223 km by 5km model, are considered.

224 There is no unique way of quantifying multi-level pollutant exceedance over the whole of  
225 the UK. Supplementary Material D gives the details of the quantitative measures we used. In

226 summary we used as a proxy for legal compliance the total number of surface level 5×5 km<sup>2</sup>  
227 model grids cells in which each pollutant standard is exceeded.

228 *Biodiversity*: Nitrogen-deposition flux (kg-N m<sup>-2</sup> y<sup>-1</sup>) is a quantitative measure of the degree  
229 of loss of biodiversity (e.g., Stevens et al., 2004). Many ecosystems are sensitive to inputs of  
230 reactive nitrogen (i.e. oxidised and reduced forms of nitrogen, such as nitrogen dioxide  
231 (NO<sub>2</sub>), nitric acid (HNO<sub>3</sub>), nitrate (NO<sub>3</sub><sup>-</sup>) aerosol, ammonia (NH<sub>3</sub>) and ammonium (NH<sub>4</sub><sup>+</sup>)  
232 aerosol) by dry and wet deposition. There is a background level of nitrogen deposition from  
233 natural sources that is enhanced by anthropogenic emissions of NO<sub>x</sub> (e.g. from combustion  
234 processes) and ammonia (e.g. from intensive agriculture). Enhanced nitrogen deposition  
235 tends to increase the exposure of ecosystems to acidity (depending upon the local  
236 neutralising capacity of the soil) and also tends to reduce biodiversity (fertilisation favours  
237 generalist species at the expense of specialists). Low levels of reactive nitrogen input are  
238 seen as a measure of a pristine natural environment. Nitrogen deposition was chosen as an  
239 indicator of loss of biodiversity although it is noted that sulphur deposition can also be used  
240 to give a fuller indication of acidity or pH levels.

241 *Crop yield*: Ozone deposition flux (kg-O<sub>3</sub> m<sup>-2</sup> y<sup>-1</sup>) is used to measure the impact of a policy on  
242 crop yield. A major route of ozone removal from the atmosphere is dry deposition to  
243 vegetation. About half of this flux is into plants' stomata, from where ozone directly enters  
244 the plant's vascular system. Because ozone is a strong oxidant, it can cause significant  
245 damage to some plants, including major UK crops such as wheat, and reduce yields.  
246 Irrigated crops are particularly susceptible, as they are more likely to have open stomata.  
247 Current baseline ozone levels in air entering the UK can reduce yields of staples crop such as  
248 wheat and potato by up to 15% (Pleijel et al., 2007; Mills et al., 2011; RoTAP, 2012). This has

249 significant economic and food security implications. Locally produced ozone from precursor  
250 emissions from within the UK itself can further affect crop yields.

## 251 **2.5 Subjective weights**

252 There are various ways of eliciting preference weights on attributes or criteria from  
253 stakeholders. Weernink et al (2014) reviewed preference elicitation methods used in  
254 healthcare decision-making. These methods can be time-consuming because a stakeholder  
255 must follow strict procedures in order to satisfy certain axioms of decision making. We  
256 opted instead for a less time consuming method which has been used in in environmental  
257 health policy (e.g. Kenyon 2007). In this method each stakeholder is asked to rank  
258 (independently from other stakeholders) the criteria in order of their importance as they  
259 perceive it. Supplementary Material A gives the survey questionnaire which we asked the  
260 stakeholders to complete. In this case of six criteria, rank 1 means that the associated  
261 criterion is the most important and rank 6 means that it is the least important. The ranks  
262 should be converted to weights between 0 and 1 such that (i) the weights add up to unity  
263 and (ii) the weights are positioned numerically in the same order as the ranks i.e., for the six  
264 criteria the weight corresponding to rank 1 has the highest numerical value and the weight  
265 corresponding to rank 6 has the lowest numerical value. There are several methods of  
266 achieving transformation between ranks and weights. These methods differ in how steeply  
267 the weights vary with the ranks. We used a method which gives a mildly steep pattern so  
268 that the weights are moderately sensitive to the ranks. Details of the method are given in  
269 Supplementary Material B. In the MCDA calculation the set of weights of each stakeholder  
270 can be used separately, or alternatively, the set of weights aggregated over all stakeholders  
271 can be used. Supplementary Material B also explains the aggregation procedure.

272

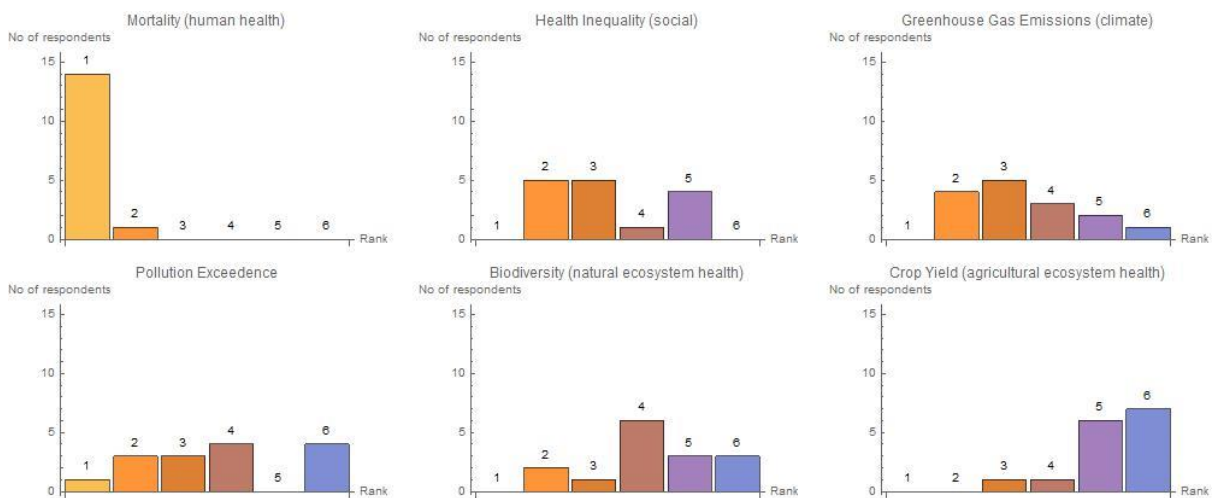
273

### 274 3. Results

275 In this section, the results of the survey questionnaires of ranks and the associated  
276 aggregated weights are presented, followed by the calculated impacts of the air quality  
277 policies on the selected criteria and the MCDA outputs.

#### 278 3.1 Survey questionnaire

279 There were 15 respondents overall, the majority of whom attended the MCDA stakeholder  
280 workshop (approximately 65% response rate). Figure 1 shows the distribution of the  
281 rankings for each criterion. To reiterate, rank 1 means that the criterion was deemed to be  
282 the most important and rank 6 means that the criterion to be the least important. Taking  
283 mortality as an example, fourteen respondents gave it rank 1 and one respondent gave it  
284 rank 2. For Biodiversity, two respondents gave it rank 2, one gave it rank 3, six gave it rank 4,  
285 three gave it rank 5, and 3 gave it rank 6.



286

287 Figure 1. Distribution of ranks for each criterion, as selected by survey correspondents.

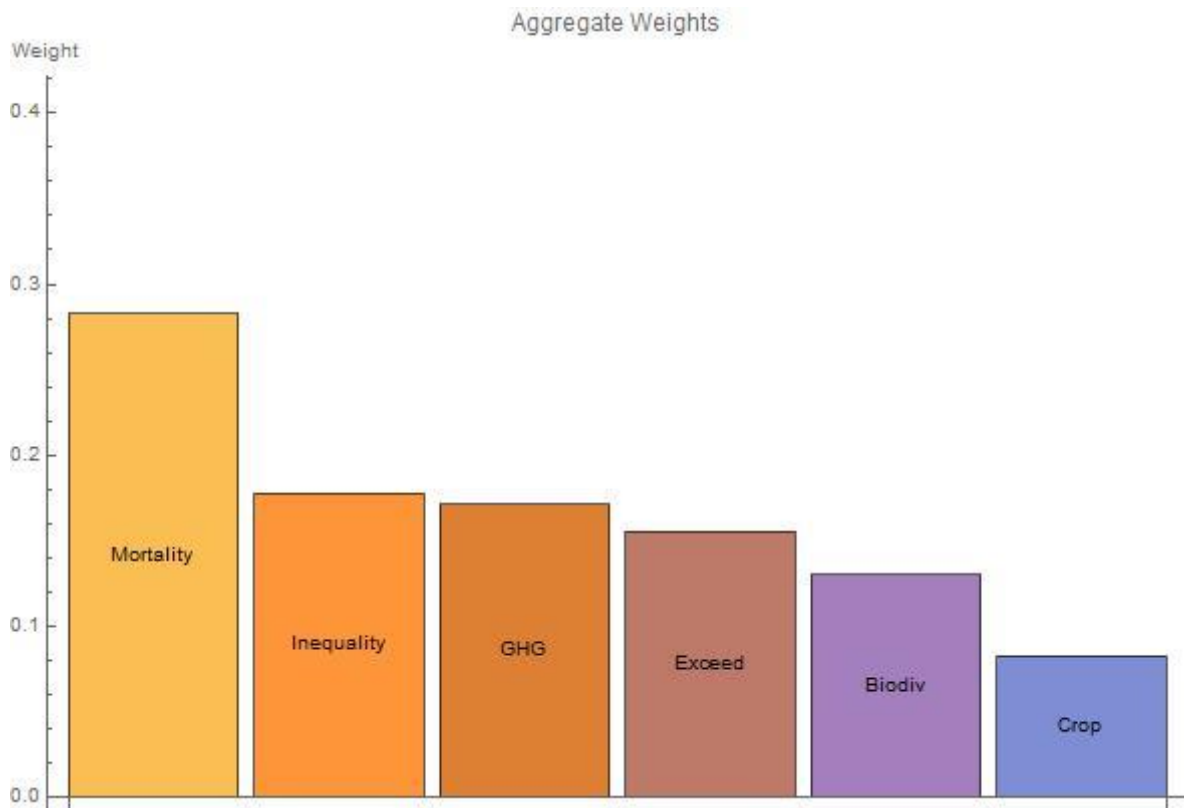
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289 Supplementary Material B describes the method for mapping ranks to weights. As explained  
290 previously, the map is a mathematical transformation which converts the ranks to weights  
291 such that the weights are positive, add up to unity and are in the same numerical order as  
292 the ranks. Applying this transformation gives the following weights: 0.2857 (rank 1), 0.2381  
293 (rank 2), 0.1905 (rank 3), 0.1429 (rank 4), 0.0985 (rank 5) and 0.0476 (rank 6). The ratio of  
294 two weights represents the relative importance between the associated ranks. For example,  
295 rank 1 is deemed to be 1.2 ( $=0.2857/0.2381$ ) times more important than rank 2, and 6.0  
296 ( $=0.2857/0.0476$ ) times more important than rank 6. Individual weights are then aggregated  
297 proportionally to the number of respondents who selected the associated ranks so that the  
298 aggregated weights also add up to unity (Supplementary Material B).

299

300 Figure 2 shows the aggregated weights for the 6 criteria across all 15 respondents.. The  
301 weights can be interpreted as follows. Overall the respondents judged that mortality is the  
302 most important criterion and crop yield is the least important. The ratio of two weights  
303 represents how important one criterion is judged to be relative to the other. For example,  
304 mortality was considered to be 1.6 times more important than health inequality and 3.4  
305 times more important than crop yield. Biodiversity was considered to be 1.6 times more  
306 important than crop yield.

307



308

309 Figure 2. Aggregated weight of each criterion.

310

311 Having established the relative weights to be assigned to each criteria, we now apply the air  
 312 pollution modelling simulation results to calculate the impact of each policy on each of the  
 313 criteria in the sections below.

### 314 **3.2 Mortality**

315 We calculated mortality impacts applying the life table model to the simulated air pollution  
 316 levels for 2010. Table 2 gives the population-weighted annual mean PM<sub>2.5</sub> concentration (µg  
 317 m<sup>-3</sup>) per socio-economic (SE) deprivation decile group along with the YLL (years) associated  
 318 with long-term PM<sub>2.5</sub> exposure summed over the whole population in England.

319  
 320  
 321  
 322  
 323



324 **Table 2.** Annual mean PM<sub>2.5</sub> concentrations on ( $\mu\text{g m}^{-3}$ ) and associated mortality per decile group  
 325 for the baseline and for 100% SNAP emission reduction (perturbation) in each of the four SNAP  
 326 sectors.

SE-deprivation decile groups	Baseline		SNAP 1		SNAP 2		SNAP 7		SNAP 10	
	PM <sub>2.5</sub>	YLL	PM <sub>2.5</sub>	YLL	PM <sub>2.5</sub>	YLL	PM <sub>2.5</sub>	YLL	PM <sub>2.5</sub>	YLL
1 (the least)	9.175	20,667	8.341	18,789	8.690	19,575	8.421	18,969	7.901	17,797
2	9.180	24,373	8.352	22,175	8.706	23,115	8.462	22,467	7.877	20,914
3	9.186	26,261	8.364	23,912	8.721	24,932	8.475	24,229	7.881	22,532
4	9.208	27,492	8.393	25,060	8.752	26,131	8.492	25,356	7.921	23,652
5	9.202	28,691	8.393	26,171	8.749	27,280	8.482	26,449	7.929	24,726
6	9.228	29,621	8.420	27,030	8.772	28,159	8.499	27,283	7.966	25,574
7	9.272	29,671	8.462	27,082	8.816	28,214	8.524	27,280	8.023	25,679
8	9.316	30,697	8.502	28,019	8.857	29,187	8.547	28,167	8.081	26,634
9	9.366	31,554	8.548	28,803	8.907	30,011	8.575	28,894	8.140	27,431
10 (the most)	9.450	34,057	8.634	31,121	8.996	32,423	8.631	31,110	8.244	29,717
Total	N/A	283,084	N/A	258,162	N/A	249,452	N/A	260,204	N/A	244,656
Total relative to baseline		0		-24,922		-33,632		-22,880		-38,426

327  
 328 Table 2 shows that the burden of PM<sub>2.5</sub> pollution in 2010 is about 283,000 YLL with SNAP 1  
 329 (Industrial combustion plants) contributing about 25,000 YLL, SNAP 2 (non-industrial  
 330 combustion plants) 34,000 YLL, SNAP 7 (road transport) 23,000 YLL and SNAP 10  
 331 (Agriculture) 38,000 YLL. Hence changes in PM<sub>2.5</sub> concentrations due to removing UK  
 332 emissions in the agriculture sector have the largest impact on mortality due to the large  
 333 geographical area it covers compared to other sectors. This finding is in agreement with that  
 334 of Vieno et al (2016) who compared the impacts of reductions in individual pollutants and  
 335 reported that reductions in ammonia (NH<sub>3</sub>) – whose emissions occur primarily from  
 336 agriculture – had the greatest effect in area-weighted PM<sub>2.5</sub> concentrations.

337

### 338 **3.3 Health inequality**

339 As outlined, above health inequality is defined as the change in YLL (associated with long-  
 340 term PM<sub>2.5</sub> exposure) per 5<sup>th</sup> to 9<sup>th</sup> decile of socioeconomic deprivation index in England.  
 341 Table 2 shows that both overall, and for each SNAP sector, the most deprived parts of the  
 342 population are exposed to higher levels of PM<sub>2.5</sub>, and that there is an (almost monotonic)

343 increase in exposure for each sector as deprivation rises. Table 3 gives the change in YLL  
 344 ( $\Delta$ YLL) calculated by subtracting YLL at the 5<sup>th</sup> decile group from that at the 9<sup>th</sup> decile group:

345  
 346 **Table 3.** Change in YLL per 5<sup>th</sup> to 9<sup>th</sup> decile deprivation score for baseline and each SNAP perturbation

	Baseline	SNAP 1	SNAP 2	SNAP 7	SNAP 10
Change in PM <sub>2.5</sub> , $\mu\text{g}/\text{m}^3$	0.164	0.155	0.158	0.093	0.211
Change in YLL in years	2,863	2,632	2,731	2,445	2,705
Relative to baseline	0	-231	-132	-418	-158

347  
 348 Table 3 shows that the reductions in road transport emissions (SNAP 7) have the biggest  
 349 impact in reducing health inequalities ( $\approx$  420 YLLs), followed by industrial combustion plants  
 350 emissions ( $\approx$  230 YLLs), agricultural emissions ( $\approx$ 160 YLLs) and then non-industrial  
 351 combustion plants ( $\approx$ 130 YLLs).

352

### 353 **3.4 Greenhouse gas emissions, biodiversity and crop yield**

354 Table 4 gives CO<sub>2</sub>-equivalent emissions (measure of greenhouse gas emissions), the N-  
 355 deposition flux (measure of impact on biodiversity), O<sub>3</sub>-stomatal conductance flux (measure  
 356 of impact on crop yield) for the baseline and SNAP perturbations for the UK.

357 **Table 4.** CO<sub>2</sub>-eq emissions, N-deposition flux and ozone stomatal deposition flux for baseline and  
 358 each SNAP perturbation

	Baseline	SNAP 1	SNAP 2	SNAP 7	SNAP 10
CO <sub>2</sub> -eq (Gg/yr)	563,341	369,711	457,148	452,612	526,048
Relative to baseline	0	-193,630	-106,193	-110,729	-37,293
N deposition (Gg/yr)	278.925	268.943	277.096	265.646	219.76
Relative to baseline	0	-10.0	-1.8	-13.3	-59.2
O <sub>3</sub> deposition (Gg/yr)	1838	1850.58	1844.98	1872.52	1840.54
Relative to baseline	0	12.6	7.0	34.5	2.5

359

360 It is shown that for CO<sub>2</sub>-eq emissions, SNAP 1 (industrial combustion plants) contributes  
 361 around 34%, followed by SNAP 7 (road transport) 20%, SNAP 2 (non-industrial combustion  
 362 plants) 19%, and SNAP 10 (agriculture) 7%. For N-deposition, agriculture is most important,  
 363 again due to the larger geographical area for emissions in this sector. Reducing UK emissions

364 leads to an increase in O<sub>3</sub> deposition – this is because the ozone titration reaction (O<sub>3</sub> + NO  
 365 →NO<sub>2</sub> +O<sub>2</sub>) is reduced as emissions of NO fall, and hence ozone concentrations are higher.  
 366 Transport emissions (SNAP 7) have the largest effect on ozone deposition change owing to  
 367 their high NO<sub>x</sub> content.

368 **3.5 Pollutant exceedance**

369 Table 5 gives the number of 5km grids for which O<sub>3</sub> and PM<sub>2.5</sub> exceeded the permitted levels  
 370 in 2010 according to the definitions in Table 1. As explained above NO<sub>2</sub> was not considered  
 371 due to insufficient model resolution.

372 Table 5. Pollutant exceedance for O<sub>3</sub> and PM<sub>2.5</sub>.

Country	Baseline	SNAP 1	SNAP 2	SNAP 7	SNAP 10
England					
O <sub>3</sub>	0	0	0	0	0
PM <sub>2.5</sub>	0	0	0	0	0

373  
 374 The above table shows that the EU permitted levels of O<sub>3</sub> and PM<sub>2.5</sub> are never exceeded in  
 375 the simulations. Although non-legislative thresholds could be used (e.g. 95<sup>th</sup> or 97.5<sup>th</sup> centile  
 376 for each pollutant), these levels would be arbitrary and would not represent “legal  
 377 compliance”. This means that the pollutant exceedance criterion ends up playing no part in  
 378 the MCDA analysis. Although pollution exceedance did not impact the MCDA calculation we  
 379 cannot remove it because it was selected by the stakeholders. The stakeholders also ranked  
 380 it in terms of its importance in relation to other criteria. We only found in the impact  
 381 modelling afterwards that it does not affect the MCDA calculation. It would not be  
 382 appropriate to remove it and re-rank the remaining criteria without going back to the  
 383 stakeholders.

384

385 **3.6 Normalised impacts**

386 Because the impacts on the criteria are in different units, the impacts should be normalised  
 387 so that they become dimensionless. Supplementary Material C describes a method for  
 388 normalisation for each criterion which is to divide by the maximum impact across all policy  
 389 options. Other methods could also be used and the Discussion section comments on the  
 390 sensitivity of the results to the normalisation method chosen.

391 Table 6 gives the normalised impacts across all criteria.

392

393 Table 6. Normalised impacts

	Baseline	SNAP 1	SNAP 2	SNAP 7	SNAP 10
Mortality	1.0000	0.9120	0.8812	0.9192	0.8643
Health Ineq.	1.0000	0.9193	0.9539	0.8540	0.9448
GHG emissions	1.0000	0.6563	0.8115	0.8034	0.9338
Exceedance	1.0000	1.0000	1.0000	1.0000	1.0000
Biodiversity	1.0000	0.9642	0.9934	0.9524	0.7879
Crop yield	0.9816	0.9883	0.9853	1.0000	0.9829

394

395 The entries in Table 6 are obtained as follows. The highest mortality impact is 283084 YLLs  
 396 which corresponds to the baseline (Table 2). All other mortality impacts are normalised by  
 397 this value: 258262/283085 (SNAP 1), 249452/283084 (SNAP 2), 260204/283084 (SNAP 7)  
 398 and 244656/283084 (SNAP 10). For health inequality, the largest change in YLL per 5<sup>th</sup>-9<sup>th</sup>  
 399 decile is 2863 YLLs which also corresponds to the baseline. All other health inequality  
 400 impacts are normalised by this value: 2632/2863 (SNAP 1), 2731/2863 (SNAP 2), 2445/2863  
 401 (SNAP 7) and 2705/2863 (SNAP 10). The other entries are derived in the same manner.

402

403 For all criteria, the highest impacts were for the baseline case except for the impact on crop  
 404 yield where it is highest for SNAP 7 (road transport) reductions (section 3.4). This explains  
 405 why the crop yield entry for the baseline is below unity and that of SNAP 7 is unity. All the

406 entries for exceedance are 1 because there are no exceedances and all the impacts are  
 407 equal.

408

409

410 **3.7 MCDA results**

411 The total impacts (burdens in this case) for each policy option are obtained by integrating  
 412 the impacts and the criteria using the calculation method described in Supplementary  
 413 Material E. The results are shown in Figure 3 using the *Annalisa* MCDA template:

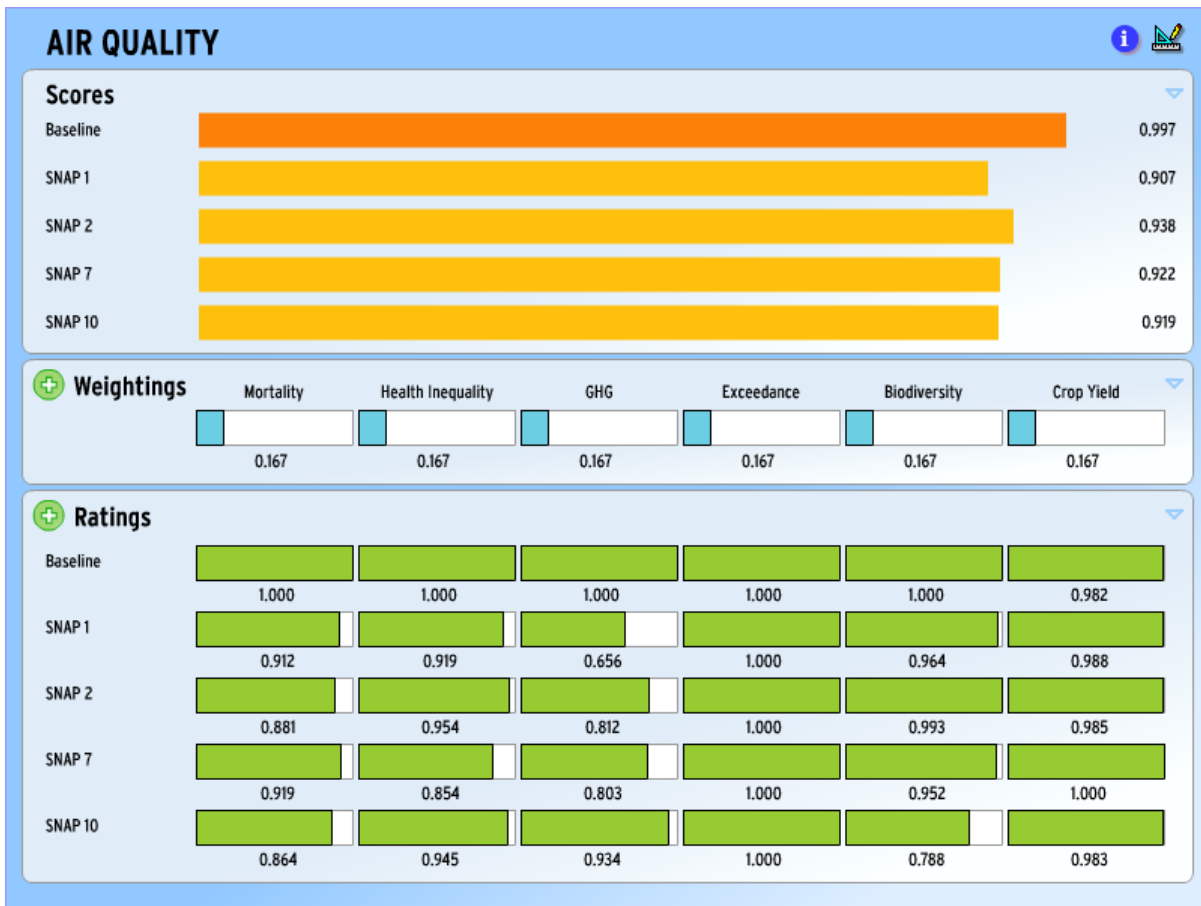


414

415 **Figure 3.** MCDA results.

416

417 The template is divided into three rectangular windows. The middle window (“Weightings”)  
418 gives the group’s aggregated relative weight (importance) of each criterion (Figure 3). The  
419 lower window (“Ratings”) is a 5 by 6 matrix which gives the burden of each option on each  
420 criterion (e.g. column 1 gives the normalised mortality burdens for the four policy options  
421 and the base case, column 3 gives the normalised greenhouse gas emissions burdens for the  
422 four policy options and the base case). The top window (“Scores”) gives the overall burden  
423 of each option across all the criteria. The higher the score the higher is the integrated  
424 burden. The option with the lowest score i.e. SNAP 1 (industrial combustion) represents the  
425 policy with the smallest integrated burden. This is followed very closely by SNAPs 7 (road  
426 transport) and 10 (agriculture). The “scores” are dimensionless numbers and their ratios  
427 can be interpreted as their relative strength; for example 100% perturbation in SNAP 1  
428 yields 0.896 times less burden than the base case. Naturally this outcome depends on the  
429 relative weights and the normalisation constants chosen. Figure 4 shows the counterpart  
430 results if all the criteria were weighted equally.



431

432 **Figure 4.** MCDA results with equal weightings.

433

434 This shows that reduction in industrial combustion emissions is still the best single policy  
 435 even if equal weights are assigned to all the criteria.

436 **4. Discussion**

437 From a scientific perspective, atmospheric chemistry transport models are very useful in  
 438 contributing to the understanding of the spatio-temporal dynamics of air quality, while  
 439 impact models provide a link to relevant outcomes from a policy perspective. These models  
 440 are also useful because they can be used to evaluate how policies based on reduction of  
 441 emissions in various sectors impact air quality. However in practice policy makers take into  
 442 account multiple criteria when assessing policies in addition to their impact on pollutant

443 exposures. To enable policy makers to make effective use of the pollutant outputs from air  
444 pollution models, we suggest that pollution and impact models are embedded within  
445 decision analytical frameworks which support decision making. The use of an MCDA  
446 framework allows a more transparent assessment of policies where the evidence base for  
447 the impacts of the policies on the criteria (“Ratings”) is shown alongside the importance  
448 assigned to the criteria (“Weightings”) and the overall impacts of the policies (“Scores”). The  
449 main contribution of this paper is to demonstrate as a proof-of-concept the use of a MCDA  
450 framework that employs both air pollution and health and non-health impact models to  
451 evaluate UK air quality policies.

452 For this approach to move forward from a proof-of-concept to a practical decision support  
453 tool further development is required. Firstly, the set of policies and criteria selected for this  
454 study emerged from “informal discussions” in a workshop. There are however formal  
455 facilitator-led procedures such as “decision conferencing” which guide stakeholders (or  
456 policy makers) as a group to reach some consensus on the appropriate policies and criteria  
457 (e.g. Quaddus and Siddique 2001, Mustajoki et al 2007, Phillips and e Costa 2007). These  
458 procedures are however very time-consuming but nevertheless they are necessary in  
459 practice.

460 Secondly, the axioms of MCDA require that all the criteria are independent. If some of the  
461 criteria are dependent, then they are best embedded in a hierarchical decision tree  
462 structure and appropriate methods for eliciting the weights of hierarchical criteria should be  
463 used (Scricciu et al 2014). It can be argued that the criteria used here are nearly  
464 independent although it is debatable whether the criteria of mortality and health inequality  
465 are truly independent.



466

467 Thirdly, no sensitivity or uncertainty analyses were carried out in the MCDA because the  
468 decision problem was illustrative rather than real. In practice sensitivity and uncertainty  
469 analyses should be performed. However what is important in decision analysis is not the  
470 quantification of uncertainty per se but whether the uncertainty in the evidence base  
471 (“ratings”) or variability in the importance of weights attached to the criteria (“weightings”)  
472 will change the rankings of the integrated impacts (“scores”). Simple sensitivity analysis can  
473 be performed using the above interactive decision tool by changing the numbers to reflect  
474 the uncertainty in the “ratings” and variability in the “weightings”. The uncertainties in the  
475 evidence matrix require either carrying out extensive probabilistic simulations of the models  
476 or using experts to define the uncertainty in the central estimates (e.g. Tuomisto et al 2008).  
477 Sensitivity analysis should also be performed to determine sensitivity of the “scores” to the  
478 chosen normalisation method. We have normalised the impact of each policy option by the  
479 maximum impact across all options. Other approaches would normalise by the highest  
480 possible impact (e.g. normalising by worst case scenario) or by presenting the impacts as  
481 percentage changes from the baseline. There is not a preferred method. It depends on the  
482 exact application and the choice of the normalisation method can influence the outcome.

483

484 Fourthly, legal compliance was not an issue in this MCDA but could be in the future. More  
485 thought may be required to differentiate between modelling different types of compliance  
486 for air quality in the MCDA, e.g. in relation to soft law ‘target values’ for some pollutants  
487 and mandatory law ‘limit values’ for others (EC, 2008).

488

489 Finally, the policy analyses were carried out by perturbing via model simulations the  
490 emissions of some of the SNAP sectors by -100%. Clearly this large reduction in emission in  
491 any SNAP sector does not represent a realistic policy option and the question then is  
492 whether more realistic reductions in emissions can be deduced from the -100% perturbation  
493 result via linear scaling. Linearity simulation experiments performed with the air pollution  
494 model (not shown here) suggest that the results are scalable for at least three of the  
495 impacts (CO<sub>2</sub>-eq emissions, N and O<sub>3</sub> deposition fluxes), but further analysis is required to  
496 ascertain the scalability of the results for all outcomes.

## 497 **5. Conclusion**

498 This study demonstrates a proof-of-concept MCDA method which uses an atmospheric  
499 chemistry transport model (WRF-EMEP4UK) for the purpose of evaluating and comparing  
500 country-wide air pollution related policy options. The policy options were formulated in  
501 terms of reductions of 100% in emissions in four sectors: energy and industrial combustion,  
502 non-industrial combustion, road transport and agriculture. Six criteria were used for the  
503 comparative evaluation of the policy options: mortality, health inequality, greenhouse gas  
504 emissions, pollution exceedance, biodiversity and crop yield. The selection of the policy  
505 options and the criteria were informed by a workshop of interested stakeholders. The  
506 MCDA analysis consisted of three main steps: (i) eliciting the relative weights (importance)  
507 of the criteria from the stakeholders (acting as proxy policy makers), (ii) calculating the  
508 impacts of each policy option on each criterion, and (iii) combining the weights with the  
509 modelled impacts to rank the options in terms of their overall impact scores. This ranking  
510 can be used to guide policy makers on how the different policy options compare relatively in  
511 terms of their overall impact across all the criteria. Using the six criteria, it is found that

512 reductions in industrial combustion has the largest overall impacts, followed very closely by  
513 reductions in road transport and agricultural emissions. Reductions in agricultural emissions  
514 are important for mortality and N-deposition.

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523

524

525

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