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Benefit Analysis of the 1st Spanish Air Pollution Control Programme on Health Impacts and Associated Externalities

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Abstract: This paper aims to provide scientific support for decision-making in the field of improving air quality by evaluating pollution reduction measures included in the current Spanish policy framework of the 1st National Air Pollution Control Programme (NAPCP). First, the health impacts of air quality are estimated by using the concentrations estimated by multiscale air quality modeling and the recommended concentration–response functions (CRF), specifically as a result of exposure to particulate matter (PM), nitrogen dioxide (NO_2), and ozone (O_3). Second, the associated external costs are calculated by monetization techniques. Two scenarios are analyzed: a package including existing measures (WM2030) and a package with additional measures (WAM2030). Compared with the baseline scenario, an improvement was found in the health effects of NO_2, PM_{10}, and $PM_{2.5}$, while for O_3 there was a slight worsening, mainly due to the increase in the O_3 metric used (SOMO35), which increases over some urban areas. Despite this, the monetary valuation of the total effects on health as a whole shows external benefits due to the adoption of measures (WM2030), compared with the reference scenario (no measures) of more than € 17.5 billion and, when considering the additional measures (WAM2030), benefits of about € 58.1 billion.

Keywords: health impact assessment; external costs; impact pathway; air pollution

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1. Introduction

Air pollution is considered the largest environmental health risk in Europe. The European Environment Agency (EEA) considers air pollution as the principal environmental factor driving disease, with around 400,000 premature deaths attributed to ambient air pollution annually in the European Union [1], and the World Health Organization (WHO) estimates that one out of every nine deaths is the result of air pollution-related conditions [2].

Air pollution has significant impacts on the health of the population, particularly in urban areas, by increasing mortality and morbidity [2,3]. Epidemiological studies have shown that the exposure to pollutants such as fine particulate matter ($PM_{2.5}$), ozone (O_3), and nitrogen dioxide (NO_2) is associated with cardiovascular and respiratory diseases, leading to increased sickness, hospital admissions, and premature deaths, as well as having an impact on activity and productivity [4,5]. Organizations and institutions, such as the WHO and EEA, undertake regular comprehensive assessments of the impacts of pollution on health at the global, regional, and country level and provide the methodological frameworks for such assessments [2,3,6,7]. The scientific literature is focused on health impact assessments (HIA) in cities or local studies, which allow a highly detailed exposure calculation as well as population and demographic characterization. There are several examples of HIA studies carried out in cities or urban ecosystems (exposure is limited to a city) across the world and Europe [8–10], including Spain, over the last decades [11–15]. There is also research and literature for sectorial quantification [16,17].

Reliable estimates of exposure to air pollutants and related impacts on health are key for better informing policy-makers, providing essential information for implementing, monitoring, and evaluating policies that help to tackle air pollution while also protecting health [2]. Those adverse effects on human health translate into higher welfare costs for the community [18,19]. Globally, welfare losses from ambient air pollution increased by nearly 300 percent between 1990 and 2013, growing from $2.18 trillion to $3.55 trillion, according to the World Bank [20]. Air pollution is an externality in the sense that it imposes unintended and uncompensated costs to people who are external to the transaction of a polluting product (whether this is the production processes, heat, and electricity provision activities or transport activities producing the emissions). It is also an externality that affects a public good, such as clean air quality [21]. In order to support the development of optimal internalization policies, air pollution external costs need to be estimated and taken into account in the design and decision process. Cost–benefit analysis (CBA) is an analytical tool for judging the economic advantages or disadvantages of investment decisions, such as policy measures implemented to improve air quality [22]. In this sense, assessing the external costs and benefits of the measures (or packages of measures) makes it possible to evaluate the change attributable to their implementation.

The European Union establishes a regulation framework for managing air quality, including ambient air quality limit values, pollutant emission legislation, and national emission ceilings [23]. Consequently, in 2019, the 1st National Programme for Air Pollution Control (NAPCP) [24] was approved in Spain, including several packages of measures in order to reduce emissions and improve air quality. The NAPCP includes 50 measures within eight sectorial packages. The Programme also provides details of five other packages with seven further additional measures that are not included in the current legislation. Hence, two scenarios can be assessed: a scenario with measures (WM) and a second, more ambitious, scenario with the additional measures (WAM). The Programme is an integral part of policy planning for framing the climate and energy strategies. Air pollution is a clear marker for sustainable development, as sources of air pollution also release greenhouse gases and other compounds affecting climate [2]. The scenarios of implementation of measures are described in the Programme [24], as well as in the scientific work on modeling developed by Vivanco et al. [25]. There are synergies between air pollution control and climate change mitigation as they share common sources and, to a large extent, solutions; while the majority of air pollutants also impact the climate to some degree, their interaction can also have negative impacts. For example, greenhouse gases, such as methane, contribute to the formation of ground-level ozone, and levels of ground-level ozone increase with rising temperatures. These rising temperatures increase the frequency of wildfires, which in turn increase levels of particulate air pollution [26]. Additionally, the present COVID-19 pandemic has served to highlight the interactions between air pollution and health. Comorbidities feature prominently in the mortality of COVID-19 patients, and among the most important of these are those associated with air pollution [16].

The framework for modeling health impacts and the costs or benefits of implementation of policies for reducing air pollutant emissions uses the impact pathway approach [27], which describes a logical pathway from the activity initially emitting air pollutants, through the quantification of health impacts, to a monetized estimate of pollutant damage for each endpoint–pollutant combination. Figure 1 illustrates a scheme of the CBA for the health impacts of air pollution. In a first step, health impacts are quantified, based on the concentration of pollutants provided by the air quality modeling and the recommended concentration–response functions. Second, health-related impacts are valued in monetary terms using the value of the welfare loss produced by each health endpoint, which reflects the associated external cost. Finally, avoided external costs of each policy scenario are computed and compared with the costs of implementing air quality measures in each scenario.

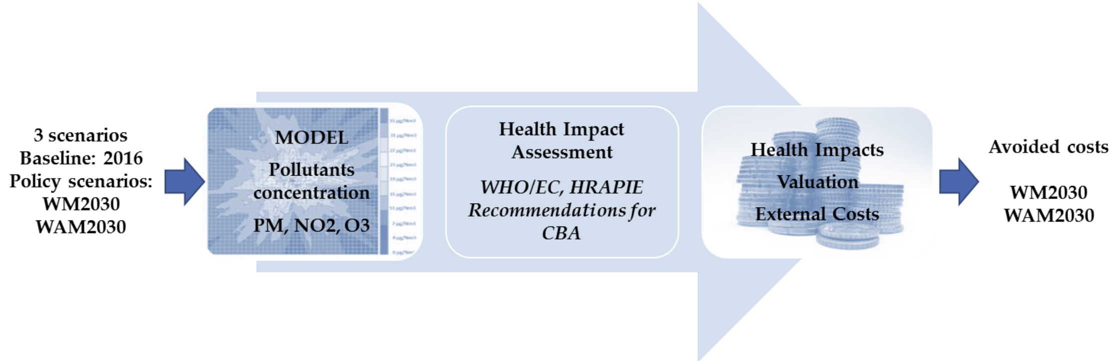


Figure 1. Scheme of the impact pathway approach for external cost calculations associated with the health impacts in the frame of cost–benefit analysis (CBA) of the policy scenarios (with measures: WM2030; and with additional measures: WAM2030). EC: European Commission; WHO: World Health Organization; HRAPIE: Health Risks of Air Pollution in Europe Project.

An assessment of the external costs of the measures proposed in the NAPCP is conducted in this study. Specifically, we present the external costs associated with the potential improvements on air quality that would be achieved under different scenarios of measures included in the NAPCP and the additional measures considered. The benefits are assumed as avoided costs, and they are based on the assessment of the reduction of the health impacts of the most relevant pollutants for the population of mainland Spain and the Balearic Islands. In Section 2, the methodology and the sources of information required for the analysis are described, from the modeling description to the statistical data sources, health assessment, and monetary valuation methods explanation. Section 3 includes the results in terms of health outcomes by pollutants, and the associated external costs. In Section 4, we discuss the results, taking into account the results of current studies and related literature, and also the analysis of results in terms of territorial distribution of the avoided impacts. Finally, the conclusions, recommendations, and future lines are presented in Section 5.

2. Methodology

Air quality modeling provides the basis for the health impact assessment (HIA) and cost–benefit analysis (CBA) of the policy scenarios for the reduction of pollutant emissions by 2030 (WM2030 and WAM2030). Air quality estimates were obtained from simulations, performed with the CHIMERE model [28], that were applied to a domain covering the Iberian Peninsula at a spatial resolution of $0.1° \times 0.1°$, nested within a European domain at $0.15° \times 0.15°$. The CHIMERE model has been extensively used and evaluated in Europe [29] and, in particular, in Spain [30–32]. Model performance for estimating atmospheric concentrations has been shown to be comparable to that of other air quality models applied in Europe [33]. To estimate the impacts of the NAPCP relative to the reference year 2016, emissions for that year were provided by the Ministry for the Ecological Transition and Demographic Challenge (METDC). These emissions are based on the National Official Emission Inventory for 2016, calculated at a spatial resolution of $0.1° \times 0.1°$ (EMEP grid). Emissions for the NAPCP scenarios were calculated using emission reductions, relative to the reference year, which were also provided by the METDC. Details of the pollutant emission control measures can be found in [24,25]. Meteorological data were obtained from simulations of the Integrated Forecasting System (IFS) of the European Centre for Medium-Range Weather Forecasts, ECMWF (www.ecmwf.int) for 2016, which were obtained from the Meteorological Archival and Retrieval System (MARS) archive at ECMWF through the access provided for research projects by the Spanish Meteorology Agency (AEMET). The same meteorology was used for the WM2030 and WAM2030 scenarios, in order to evaluate impacts only due to changes in emissions. More information on the model configuration and the emission reductions corresponding to the measures in the NAPCP are included in [25] and references therein. Annual mean concentrations for SO_2, NO_2, PM10, and PM2.5 were calculated, as well as SOMO35 for O_3 (SOMO35: this metric for ozone is defined as the sum of means over 35 ppb (daily maximum 8-hour) [34]. For the use in this study, following the recommendation by [27], the result is converted to a change in annual daily maximum 8-h mean in excess of 35ppm. For that, SOMO35, the annual sum of daily maximum running 8-h average O_3 concentrations above 35 ppb across a whole year is divided by the number of days in the year of the calculation [35]. Since the estimation of impacts and costs strongly depends on the accuracy of the air quality estimates, the model outputs were combined with observations to reduce spatiotemporal biases model resulting from model or input data uncertainty. The methodology for combining the model estimates with the observations is more fully described in [25,36,37]. Briefly, the differences between the modeled concentrations (annual mean for NO_2 and PM and daily maximum 8-hour mean for O_3) and those observed at the air quality stations were interpolated (using kriging) across the modeling domain. These interpolated differences were then subtracted from the model estimates to obtain the corrected concentrations. This was done separately for rural sites and for suburban/urban sites, and then the two datasets were combined, weighted by grid cell population density. This methodology partially takes into account local effects, since urban, suburban, and industrial sites are used. In addition, traffic stations are also used for NO_2 (one of the most traffic-affected pollutants). Furthermore, the same corrections were applied to the reduced-emission scenarios by assuming that the biases (calculated for the 2016 reference year) are proportional to the atmospheric concentrations.

The impacts on health were estimated following the recommendations issued by the Health Risks of Air Pollution in Europe (HRAPIE) project [27,38]. This project proposed concentration–response functions (CRF) for key pollutants to be included in cost–benefit analysis supporting air quality policy, considering the meta-analysis and the findings from the project Review of Evidence on Health Aspects of Air Pollution (REVIHAAP) [5].

2.1. Health Impact Assessment

The methodology allows the estimation of health impacts by using the concentrations simulated by the air quality model and the recommended CRF. The key pollutants are particulate matter (PM), nitrogen dioxide (NO_2), and ozone (O_3). This assessment requires input information on demographic and health data, and provides the quantification of health impacts in terms of a health outcome by applying the following equation to each cell in the domain of the air quality model:

$$\mathrm{I} = \mathrm{C}_i \times \mathrm{P}_a \times \mathrm{P}_r \times \mathrm{R} \times \mathrm{CRF} \quad (1)$$

where I is the impact expressed as number of attributable additional cases (outcome), C_i is the concentration of the pollutant i, P_a is the fraction of the population within the age group considered, P_r is the fraction of population at risk within the age group, R is the incidence ratio, and CRF is the concentration–response function, or change in incidence per unit of concentration. The CRF values are given as relative risks (RR).

Relative risks (RR) have an uncertainty that is expressed as confidence intervals (CIs). These CIs provide the upper and lower boundaries of the 95% CI of the estimate, taking into account only the uncertainty in the relative risks. Table 1 summarizes the CRFs used in this study, with details of the pollutant and type of exposure, the population group at risk, the specific range of concentrations for applying the CRF, and corresponding health outcome, as well as the RR and the original study reference. There are two types of exposure and subsequent health impact: short-term exposure and long-term exposure. Only those effects that contribute to the total effect—additive effects—are considered, and the effects of both Group A* (pollutant–outcome effects for which enough data are available for a reliable quantification) and Group B* (pollutant–outcome effects for which there is more uncertainty) are quantified.

Table 1. Recommended concentration–response functions (CRFs) used in the study [27]. RAD: Restricted activity days, MRAD: minor restricted activity days.

Pollutant and Metric	Range of Concentration	Health Outcome (Impact/Population Group)	Type	RR (95% CI) per 10 µg/m^3	Original Study
NO_2, annual mean	>20 µg/m^3	Long-term Mortality, all (natural), age over 30 years	B*	1.055 (1.031–1.080)	[39]
NO_2, annual mean	All	Prevalence of bronchitis symptoms in asthmatic children aged 5–14 years.	B*	1.021 (0.990–1.60)	[40]
NO_2, annual mean	All	Acute Mortality, all-causes (natural), All ages	A*	1.0027 (1,0016–1.0038)	[41]
NO_2, 24-h mean	All	Hospital admissions respiratory diseases, all ages	A*	1.0180 (1.0115–1.0245)	[42]
O_3, SOMO35	>35 ppb (>70 µg/m^3)	Mortality, all (natural) causes, all ages	A*	1.0029 (1.0014–1.0043)	[43,44]
O_3, SOMO35	>35 ppb (>70 µg/m^3)	Hospital admissions, CVD diseases, age over 65 years	A*	1.0089 (1.0050–1.0127)	[43]
O_3, SOMO35	>35 ppb (>70 µg/m^3)	Hospital admissions, CVD and respiratory diseases, age 65+ years	A*	1.0044 (1.0007–1.0083)	[43]
O_3, SOMO35	>35 ppb (>70 µg/m^3)	MRADs, all ages	B*	1.0154 (1.0060–1.0249)	[45]
$PM_{2.5}$, annual mean	All	Mortality, all cause (natural), age over 30 years	A*	1.062 (1.040–1.083)	[39]
PM_{10}, annual mean	All	Post-neonatal infant mortality, (age 1–12 months) all cause, expressed as deaths	B*	1.04 (1.02, 1.07)	[46]
PM_{10}, annual mean	All	Bronchitis in children, age 6–12 (or 6–18) years	B*	1.08 (0.98–1.19)	[47]
PM_{10}, annual mean	All	Chronic bronchitis in adults (age over 27 years)	B*	1.117 (1.040–1.189)	[48–51]
$PM_{2.5}$, daily mean	All	Hospital admissions CVDs, all ages	A*	1.0091 (1.0017–1.0166)	[52]
$PM_{2.5}$, daily mean	All	Hospital admissions, respiratory diseases, all ages	A*	1.0190 (0.9982–1.0402)	[53–57]
$PM_{2.5}$ annual average	All	RADs, all ages	B*	1.047(1.042–1.053)	[58]
$PM_{2.5}$, annual average	All	Work days lost, working age population (age 20–65 years)	B*	1.046 (1.039–1.053)	[58]
PM_{10}, daily mean	All	Asthma symptoms in asthmatic children, aged 5–19 years	B*	1.028 (1.006–1.051)	[59]

The spatial domain of this study was the Iberian Peninsula and the Balearic Islands. A geographic information system (GIS) was used to calculate the population exposed to each pollutant concentration in each cell of the domain. Each air-quality model grid cell was assigned to projected geographic coordinates corresponding to its centroid. A population grid was created in order to distribute the population within the pollutant concentration grid. This was done by disaggregating the Spanish (Iberian Peninsula and Balearic Islands) population data (inhabitants by settlements according to the official census data [60]). In some cases, cities, villages, or even lower level settlements were split between grid cells. In this case, the population was allocated according to the population density by area. Next, the population data was reaggregated by cell, and linked to the pollutant exposure of that cell.

Population and health data stratified by age and regional location considering two levels of territorial units (NUTS2 and NUTS3) were used in this study. The Nomenclature of Territorial Units for Statistics (NUTS) was drawn up by Eurostat decades ago in order to provide a breakdown of the economic territory of the European Union into territorial units for the production of regional statistics and for targeting political interventions at a regional level. In Spain, the territory is organized by the administrative boundaries corresponding to 17 Autonomous Communities and 2 Autonomous Cities (*Comunidades y ciudades autónomas*) in the level NUTS2, and 57 Provinces (*Provincias*) in the level NUTS3 [61].) Population data by region for 2016 and the projected population for 2030 were obtained from the National Statistics Institute (INE) according to the official census [62].

Official health data on mortality and morbidity for 2016 were used as the baseline incidence of mortality and morbidity rates by cause and age group. Health data were collected as much as possible at NUTS3 level. In some cases, national health statistics were used due to lack of data at regional level. The main source of data was the National Statistics Institute (INE) (www.ine.es). The number of deaths corresponding to total non-accidental causes (International Classification of Diseases, 10th revision (ICD-10), codes A00-R99) and cardiovascular (CVD; ICD-10, codes I00–I99) and respiratory (ICD-10, codes J00-J99) diseases for 2016 were provided by the INE database. The National Health Survey developed by the INE was used as the information source for those impacts related to days of reduced activity or symptoms of asthma [63]. Finally, for the data that was unavailable at regional or national level, such as minor restricted activity days, the United Nations mid-estimates for population were used, as recommended by the referenced literature [27].

2.2. External Costs Calculation

Health outcomes, such as mortality impacts in terms of premature deaths or the reduction of life time expectancy (years of life lost, YOLL), as well as morbidity impacts in terms of attributable cases, can be translated into costs (external costs). Valuation of health effects was calculated by multiplying impacts (e.g., respiratory hospital admissions) by an appropriate estimate of the monetary value of each impact (e.g., the cost of a respiratory hospital admission) [23], as shown in equation 2.

$$E = I \times V \tag{2}$$

where I is the impact assessed using Equation (1) and V is the monetary value of each impact.

As for the valuation of mortality effects from air pollution, there is an ongoing discussion about which of two alternative metrics should preferably be used: loss of lifetime expectancy expressed as total number of years of life lost (YOLL) per year across the population and valued using the metric value of life year (VOLY); or premature deaths expressed as number of deaths per year and valued using the value of statistical life (VSL) [64]. Depending on the metric, the values differ, and several studies have compared both approaches. For example, the impact assessment undertook by the European Commission found that external costs of air pollution in European countries could range between €330 billion, using VOLY, and €940 billion in 2010, using VSL [65]. The VSL is a commonly used economic method of valuing risk to life which is derived from the trade-offs people are willing to make between fatality risk and wealth—it might be alternately phrased “the value of preventing a fatality” [19]. There are several published estimates of VSL in the literature considering different variables (e.g., age [66] or labor market-based [67,68]). Also, VSL is not a natural constant, but varies with individual preferences and opportunities, so there is heterogeneity in the individual VSL levels [69]. In this study, the valuation (V) of the estimated health impacts is done using monetary values of the different health endpoints recommended by the OECD for mortality (VSL) published in the year 2012 [70–73]. Those values were recommended for mortality valuation in the last meeting of the UNECE Task Force on Integrated Assessment Modelling [74], and have been used in recent analysis by the European Chemicals Agency (ECHA) [75] and other recent studies ([23,27]). The monetary values of the external costs per unit (V) of health outcome are shown in Table 2.

Table 2. Monetary values (V) used for the health impact assessment, in $€_{2005}$.

Pollutants	Health Outcome	Value €2005/Unit	Source
NO_2, O_3, PM	Mortality (VSL)	3,060,000	OECD [71]
NO_2, PM	Bronchitis in children	588	[27]
NO_2, O_3, PM	Respiratory hospital admissions	2200	[27]
O_3, $PM_{2.5}$	Cardiovascular hospital admissions	2200	[27]
O_3	MRADs, all ages	42	[27]
PM_{10}	Infant mortality	5,355,000	OECD [71] Average value of High and Low Value
PM_{10}	Asthma symptoms in asthmatic children	42	[27]
PM_{10}	Chronic bronchitis in adults	53,600	[27]
$PM_{2.5}$	RADs	92	[27]
$PM_{2.5}$	Work days lost	130	[27]

External costs have been updated to the present value by a currency conversion from $€_{2005}$ to $€_{2019}$ in order to consider the inflation rate for the year 2019, according to the ex-ante policy implementation (1.26 according to Consumer Prices Index variation provided by INE [76]).

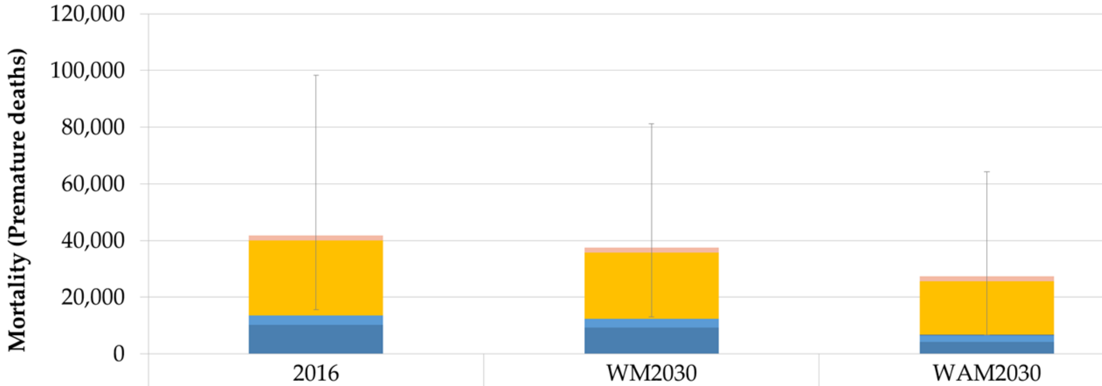
3. Results

Impacts on health of the implementation of the NAPCP have been investigated. Impacts on air quality of these pollutants are discussed with more detail in [25]. Overall, significant improvements are found for the WAM2030 scenario for all the pollutants. For ozone, and in particular for SOMO35, the model, in spite of a general improvement of air quality, estimates an increase of concentration over some areas in Galicia, Northern Spain (Asturias), Barcelona, and Madrid for the scenario WAM2030. This behavior, related to the reduction of NOx emissions and the chemistry of ozone, is analyzed more deeply in [25].

3.1. Impacts on Health

The exposure to air pollutions leads to various health effects that have been quantified, for the conditions existing in 2016 and those in 2030, for the scenarios WM2030 and WAM2030.

Regarding mortality, results show a remarkable decrease in premature deaths due to exposure to high levels of NO_2, PM_{10}, $PM_{2.5}$, and O_3 that goes from 41,713 premature deaths in 2016 to 37,386 premature deaths in 2030 for the WM2030 scenario (10% reduction), and 27,267 premature deaths in the WAM2030 scenario (35% reduction). Figure 2 shows the results of the mortality quantification due to the exposure to the different pollutants. The error bars show the uncertainty range related to the CRF relative risks (max and minimum RR).



	2016	WM2030	WAM2030
O3	1707	1723	1692
PM2.5	26,535	23,302	18,946
PM10	23	21	18
NO2 Acute	3172	3051	2338
NO2	10,299	9310	4291

Figure 2. Mortality attributable to air pollution (NO_2, O_3, PM_{10}, $PM_{2.5}$) in 2016, and scenarios with measures (WM2030) and with additional measures (WAM2030).

As expected, $PM_{2.5}$ is the pollutant with the highest impact in terms of premature deaths, and its impact is reduced by 12% in WM2030 and 29% in WAM2030. NO_2 is also a large contributor, but its impact declines notably in the WAM2030 scenario, especially due to reductions in the long-term exposure to this pollutant.

A recent publication of the EEA [3] quantifies the impacts attributable to exposure to $PM_{2.5}$, NO_2, and O_3 in Europe for 2016 for 41 countries across Europe. For Spain, the report establishes 33,300 premature deaths due to NO_2 (7700 premature deaths), O_3 (1500 premature deaths), and $PM_{2.5}$ (24,100 premature deaths). The Eionet Report from the European Topic Centre for the Air Pollution, Transport, Noise and Industrial Pollution (ETC/ATNI) of the EEA [35] establishes 33,300 premature deaths due to NO_2 (9500 premature deaths), O_3 (1700 premature deaths), and $PM_{2.5}$ (26,000 premature deaths) in Spain in 2017. These results are in general agreement with ours, especially those related to impacts from $PM_{2.5}$ and ozone, while impacts from NO_2 are lower, even considering that the EEA assessments do not include the NO_2 acute mortality or the PM_{10} infant mortality. There could be several reasons for these discrepancies. First, in this study we used an exhaustive allocation of parameters, such as mortality rates at NUTS2 and NUTS3 level, to each cell of the air quality model, which helps to make a fine-tuned characterization of the exposed population, avoiding the assumption of the same ratios of mortality or morbidity for the entire population. The second reason could be differences in the atmospheric concentration estimates used, and the third is that the quantification of the impact for one pollutant could include effects attributable to another with which it is correlated. Individuals are simultaneously exposed to all pollutants [52], and so the approach to estimate the number of avoided cases or deaths associated with each pollutant may lead to overestimations due to double-counting of effects.

The health impacts included in the report assessment of the EU Clean Air Strategy carried out by the International Institute for Applied Systems Analysis (IIASA) [23] used 2005 as reference year, and compared the impacts for the EU-28, considering the projection of population, under different policy scenarios (legislation as 2017 and the National Emission Ceilings Directive (NECD) Emission Reduction Requirements in 2030—ERR legislation, including one, two, or more scenarios, depending on the emission control scenario, PRIMES2016 and PRIMES Climate and Energy Policy). Under 2017 legislation, they estimated an annual mortality of 22,000–23,000 and 264,000–274,000 premature deaths as a consequence of O_3 and $PM_{2.5}$, respectively. Under ERR2030 legislation, they estimated

22,000 premature deaths as a consequence of O_3 exposure, and 245,000–205,000 due to $PM_{2.5}$. The study did not provide individualized results for each European country, and, therefore, cannot be directly compared to our results. However, it confirms the ratio of the impacts of both pollutants found in our study: that ozone impacts are around 8% of those of $PM_{2.5}$.

The CBA for the Clean Air Policy Package (CAPP) conducted by Holland [77] quantified deaths from $PM_{2.5}$ and O_3 exposure by time series from 2010 to 2030 (using reported emissions for 2010 and estimated emissions under the legislation at this moment for future years (year 2014). For Spain, they found a mortality attributable to $PM_{2.5}$ of 23,963 and 21,668 premature deaths in the baseline scenarios for 2010 and 2030, respectively. In our study, results for the projected scenarios in 2030 are 23,302 premature deaths in the less ambitious scenario WM2030, and 18,946 expected premature deaths in WAM2030.

Regarding O_3, while the CBA CAPP found between 1574 in the baseline, and 1366 expected premature deaths, in the best scenario, for 2030, our results do not follow the same decreasing trend under both scenarios. In comparison with the estimates for 2016, we found that the premature deaths would increase in 2030 under the scenario WM, and would decrease under WAM, although these results are not statistically significant. This unexpected result could be explained by the increase in ozone levels in urban areas at night, associated with a reduction in the titration of O_3 by NO. The inclusion of the titration reaction in the chemical balance, as an improved representation of the chemical and photochemical processes, is a key aspect in the interpretation of health impacts from ozone. These findings mean that measures to reduce NO_2 could decrease O_3 destruction during the night (in absence of solar radiation), and, consequently, increase O_3 concentrations in some areas. In the WAM scenario, some of the additional measures partially buffer those effects, and so there is an overall decrease in premature deaths due to $O_{3.}$

The avoided impacts on health due to the implementation of the air quality measures of the WM and WAM scenarios in 2030, in comparison with the situation in 2016, are shown in Figure 3. The results show that the implementation of the measures provided in the NAPCP could avoid a large number of premature deaths and other negative effects, such as cases of hospitalization due to respiratory or CVD diseases, bronchitis and asthma symptoms, and days of reduced activity. However, focusing on the associated impacts of O_3, the changes are small. The sign of the results for avoided impacts in the WM2030 scenario is negative, which means that there are no avoided impacts, and instead, the mortality, respiratory, and CVD hospital admissions and minor restricted activity days due to O_3 would increase (SOMO35 increases only over a few areas, but highly populated ones). In the WAM2030, the impact is positive, but negligible. Although a general improvement of SOMO35 is found, this does not happen in specific urban areas, such as Barcelona or Madrid, with high population density, thus enhancing the negative impacts on health. In the WAM2030 scenario, there are small amounts of avoided impacts due to the implementation of additional measures that partially buffer those effects. Overall, the results of the total avoided impacts of the four pollutants in the scenario WAM2030 are much better than those of WM2030, which implies that the additional measures implemented in WAM are able to compensate for the increasing trend of ozone impacts.

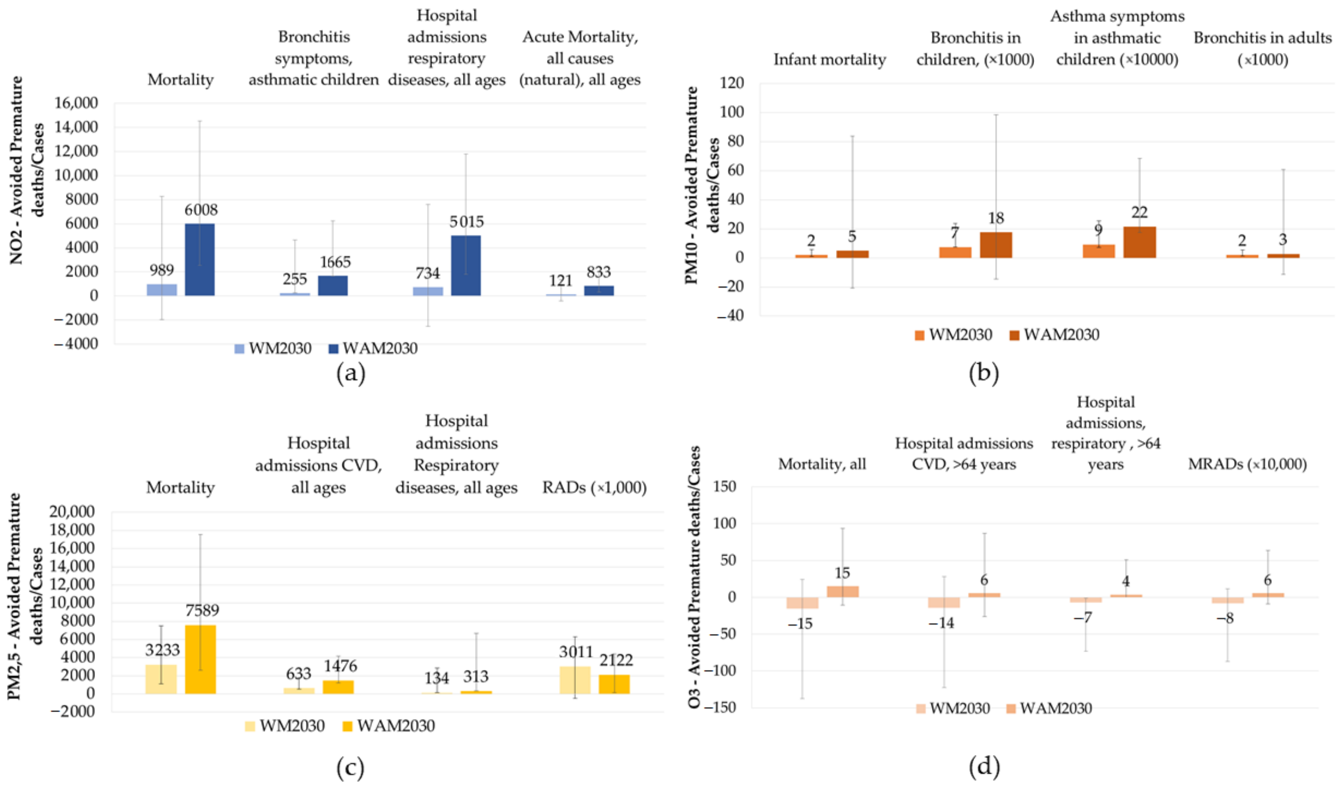


Figure 3. Avoided health impacts by pollutant: (**a**) NO_2, (**b**) PM_{10}, (**c**) $PM_{2.5}$, and (**d**) O_3. The results of the uncertainty analysis (from the CIs of the CRF and relative risk (RR) values) are shown as error bars.

The analysis of the spatial distribution of premature deaths caused by all the studied pollutants (Figure 4) shows that they are concentrated around cities such as Madrid, Barcelona, and Valencia (red and orange points in the figure) (More info about location of the main cities mentioned in the study is available at the National Geographic Institute web [78]). These results are in general agreement with the results on concentrations reduction found by [25]. While in the WM2030 scenario there are some cells where the number of premature deaths is above 1600 in Barcelona or 1400 in Madrid, in WAM2030 there are no cells with more than 1400 premature deaths in Madrid and only one cell in Barcelona. These results indicate the success of the potential implementation of the proposed additional measures included in the Programme. Premature deaths are lower than 10 per grid cell (approximately 100 km^2) for most of the domain. In addition, some cities in the north (in the regions of Asturias and Galicia) and in the south (Seville and Granada) have 600–700 premature deaths/100 km^2 (blue points in Figure 4). However, these results of total premature deaths caused by all the studied pollutants could be overestimated due to double-counting effects. The impacts estimated for each pollutant have been added to determine the total impact attributable to exposure to air pollution. However, concentrations of different pollutants are correlated (sometimes strongly), so this may lead to a double counting. In case of the effects of $PM_{2.5}$ and NO_2, this could be up to 30% [38].

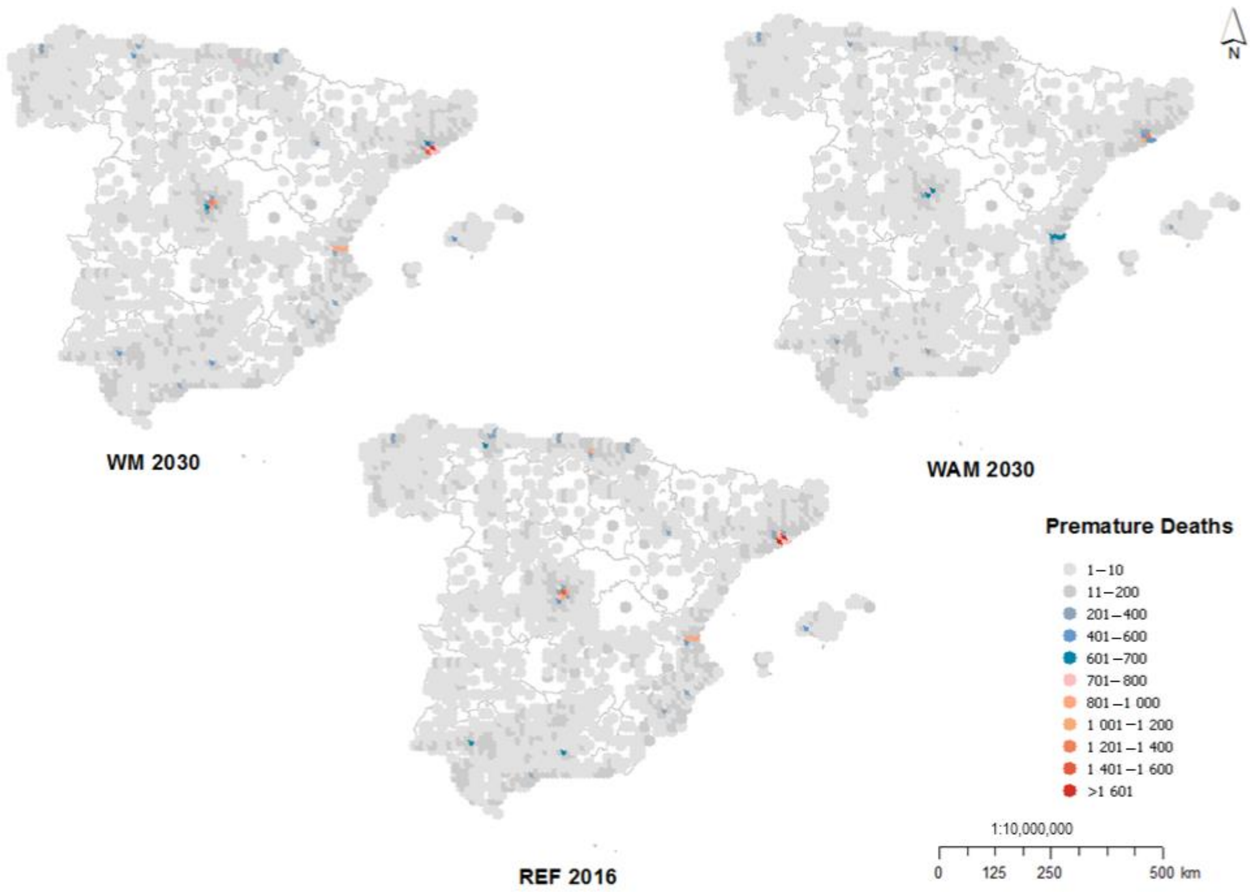


Figure 4. Spatial distribution of all pollutant mortality (premature death) in terms over the reference scenario (REF 2016, year 2016) for the scenarios WM2030 (on the left) and WAM2030 (on the right).

Figure 5 shows the spatial distribution of the ratio of avoided deaths to the premature deaths of the reference scenario. The decrease of premature deaths in the scenario WM2030 ranges from −1.5% to 22%, in comparison with the reference, with the largest reductions (>21%) in cities such as Madrid (in the middle of the country) and some points in the northwest (Galicia and Asturias). There are negligible increases in premature deaths in some locations along the Mediterranean coast (from 0.0181 premature deaths in 2016 to 0.0185 premature deaths in the scenario WM2030). This result is aligned with the air quality estimates. The coastal Mediterranean model grid cells present a large fraction over sea, with lower dry deposition (and thus higher concentrations), so these results must be carefully considered, due to the uncertainty in model resolution [25]. In most of the Spanish mainland, there would be some reductions on premature deaths with a vast area of the territory in the range from 0 to 10%. Under the WAM2030 scenario, the percentage of avoided deaths reaches more than 60%. The highest value (61% reduction), within the metropolitan area of the city of Madrid, indicates that 239 premature deaths would be avoided in an area where 389 premature deaths were attributable to pollution in 2016. Substantial reductions (>50%) are also predicted for urban areas around other cities, such as Barcelona, Seville, and Granada. Large reductions are also predicted for the northwestern area covering part of Galicia (ES11), Asturias (NUTS2 ES12), and Castille and Leon (ES41).

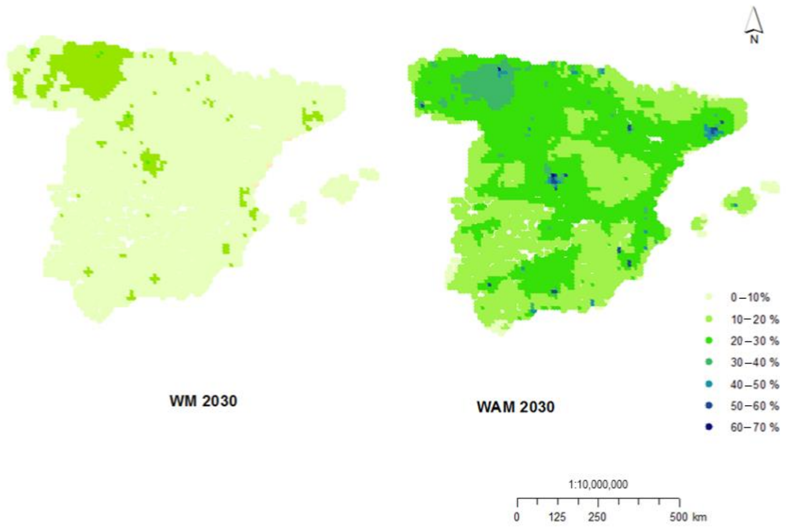


Figure 5. Spatial distribution of reduction of premature deaths in terms of percentage of avoided deaths, with respect to the reference scenario (2016), for the scenarios WM2030 (**left**) and WAM2030 (**right**).

3.2. Costs Associated with the Expected Impacts on Health

These impacts translate into medical costs and loss of wellbeing, known as external costs or externalities. Overall externalities are quantified for the extended set of concentration–response functions A* + B*. These costs amount to a total of 168,900 M €$_{2019}$ in 2016, 156,170 M €$_{2019}$ in 2030 under the scenario WM2030, and 110,785 M €$_{2019}$ under the scenario WAM2030 (Figure 6a). The figure for 2016 is around 15% of annual gross domestic product (GDP), highlighting the severity of these impacts, and suggesting that Spain should take urgent action to reduce air pollution.

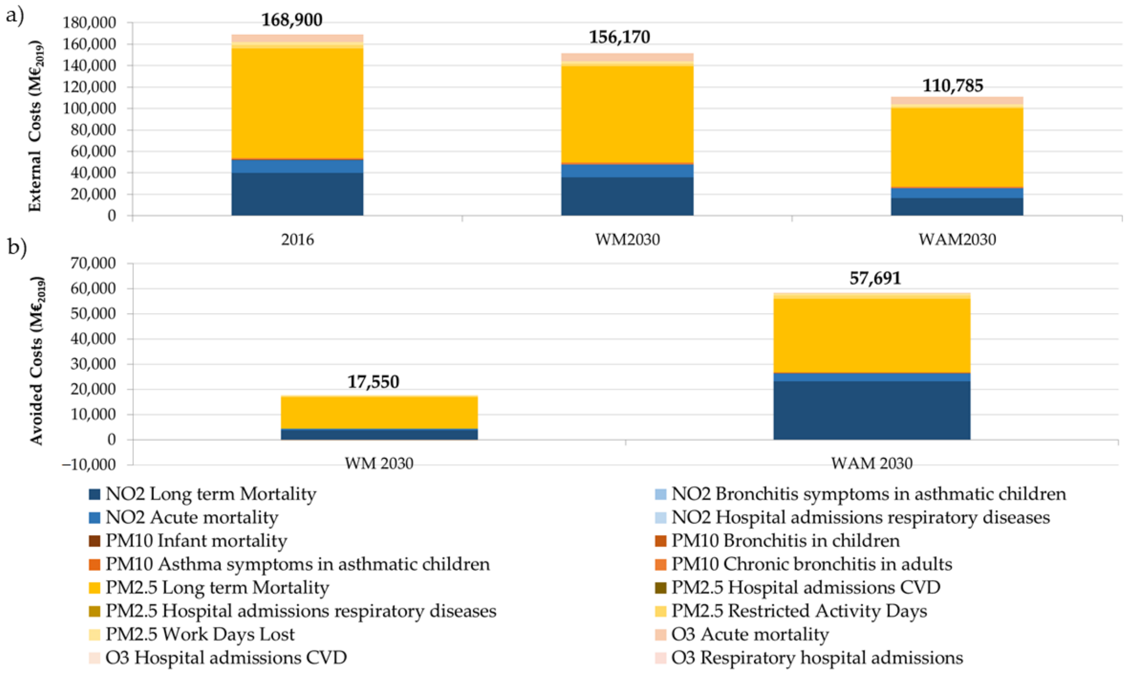


Figure 6. External costs associated with the assessed health impacts by pollutant, in M €$_{2019}$, under the baseline scenario (2016) and the policy scenarios (WM2030 and WAM2030). The figure on the top (**a**) shows the total external costs associated with the health impacts for the pollutants and outcomes considered. The bottom figure (**b**) shows the external costs avoided by the implementation of the measures (WM2030 and WAM2030).

The largest external costs are from premature deaths due to exposure to $PM_{2.5}$, followed by exposure to NO_2. Several studies on CBA of air pollution also show that the total external costs due to mortality strongly outweigh those due to morbidity. Other external costs associated with $PM_{2.5}$, due to work days lost and the restricted activity days (RAD), are visible in the total costs in the graph (Figure 6a), but the contribution is negligible, considering the cost of mortality due to $PM_{2.5}$ and NO_2.

The cost reduction in the WAM2030 scenario is very notable, in comparison with WM2030. Figure 6b shows the results in terms of avoided costs, highlighting the potential reduction of external costs due to the reduced mortality associated with long-term exposure to NO_2. The avoided costs associated with each scenario are 17,550 M €$_{2019}$ for WM2030, and more than 58,115 M €$_{2019}$ in the most ambitious scenario, WAM2030. That means that with additional measures (WAM), there is the potential to reduce total external costs by more than 33%, with respect to the baseline scenario. Impacts from ozone decrease the avoided costs slightly, but the effect is negligible.

The analysis of these impacts, in terms of costs and benefits, has been done at NUTS2 level. Table 3 shows the contribution to the avoided costs due to the reduction of air-pollution-related mortality and morbidity per NUTS2 region. In Figure 7, these results are showed in absolute terms. The highest potential to reduce the external costs is estimated for Catalonia and Madrid, both above 3000 M €$_{2019}$ in WM2030 scenario, and 15,000 and 12,000 M €$_{2019}$, respectively, in WAM2030. In all cases, avoided costs are substantially higher in WAM2030 than in the WM2030 scenario. Additionally, in Andalusia, located in the south, there is a large decrease in estimated costs, due to the large area of this region (cumulative effect of a large number of small reductions), as well as the reduction of health impacts in some cities (Seville, Granada).

Table 3. Distribution of the reduction in external costs by NUTS2 region.

Region	NUTS2	Avoided Costs of Morbidity Relative to 2016 (%)		Avoided Costs of Mortality Relative to 2016 (%)	
		WM2030	WAM2030	WM2030	WAM2030
Andalusia	ES61	18.9%	18.6%	13.1%	12.6%
Aragon	ES24	1.8%	2.3%	1.8%	1.7%
Principality of Asturias	ES12	6.7%	5.6%	7.7%	5.5%
Balearic Islands	ES53	1.1%	1.0%	0.8%	1.1%
Cantabria	ES13	0.9%	1.0%	1.1%	1.0%
Castille and Leon	ES41	3.7%	4.1%	4.5%	3.3%
Castille—La Mancha	ES42	2.8%	3.5%	2.3%	2.2%
Catalonia	ES51	21.9%	21.4%	22.6%	26.7%
Valencian Community	ES52	10.2%	10.0%	8.6%	9.1%
Extremadura	ES43	0.7%	10.9%	1.3%	1.0%
Galicia	ES11	4.0%	4.2%	7.9%	5.0%
Community of Madrid	ES30	20.6%	19.7%	20.1%	21.5%
Region of Murcia	ES62	21.7%	1.9%	2.1%	2.6%
Chartered Community of Navarra	ES22	1.0%	1.2%	10.7%	0.6%
Basque Country	ES21	3.3%	4.0%	34.7%	5.5%
La Rioja	ES23	0.7%	0.8%	0.7%	0.6%

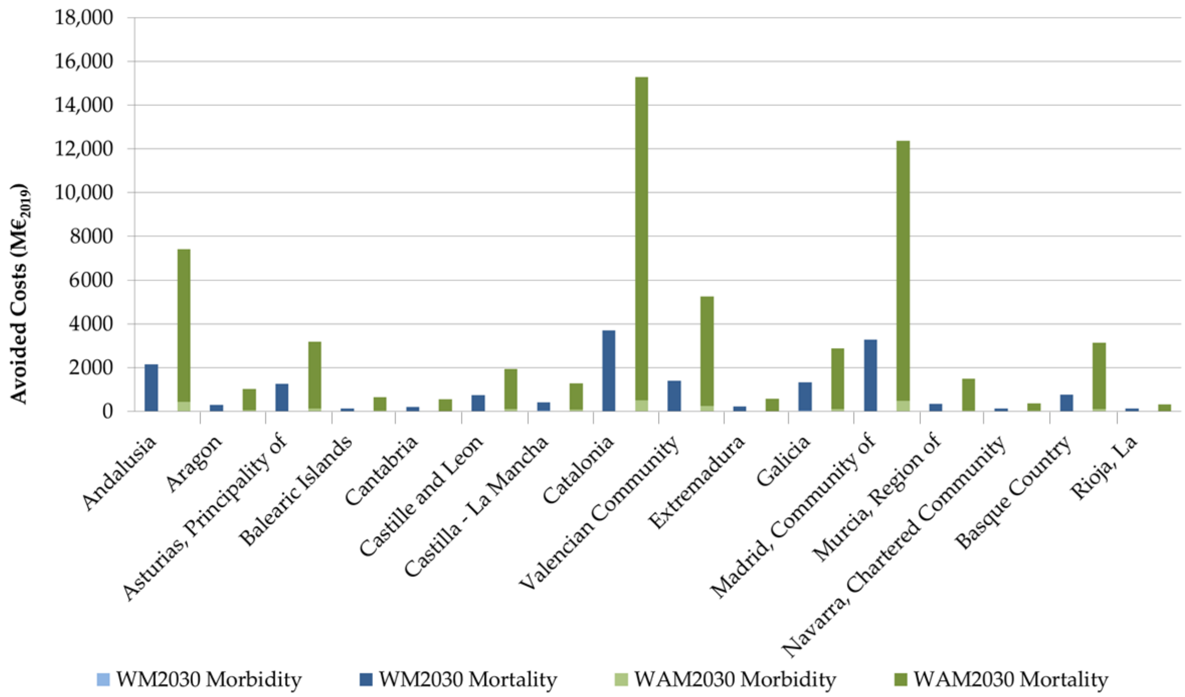


Figure 7. Distribution of the avoided costs per NUTS2.

Note that, in order to complete the cost benefit analysis, the total external costs avoided should be compared with the costs of implementing the measures. For future work, this will be done in order to estimate the net benefit. In this sense, the costs of implementation and the benefits of reducing impacts on health do not affect NUTS regions equally, so the regionalization of the net benefit will be a challenge in the foreseen research work. For instance, the investment for measures implementation could come from local, regional, and national governments, or even from private entities in different NUTS regions.

4. Discussion

This study aims to calculate the external economic costs of premature mortality and morbidity from air pollution, in order to support the Spanish government in its ambition to act decidedly in reducing air pollution. As the World Bank claims, the fact that global welfare losses from fatal illness attributable to air pollution are in the trillions of dollars is a call to action [20]. The effective implementation of the NAPCP would bring about a significant decline in health-related impacts: more than 4000 premature deaths could be avoided in 2030, increasing to more than 14,400 with additional measures.

The measures envisaged in the NAPCP are completely aligned with the aims to integrate health improvement, sustainability criteria, and new models of development joined to economic progress. The analysis performed provides valuable information on the economic and social benefits of improving air quality that is of use to policy-makers and decision-takers, as well as to citizens. Health impact assessments, such as this one, may also enable law and regulation-makers to pay special attention to the most affected regions, thereby preventing inequality in the face of risk and ensuring a fair health–benefit distribution. Valuing the costs of effects on health associated with pollution helps to further highlight the severity of the problem. Governments must confront a broad array of development challenges, and monetizing the costs of pollution can help them decide how to allocate scarce resources to improve the lives of citizens. Our findings provide additional evidence that reducing air pollution in Spain should be among the main priorities in any attempt to improve the welfare of the population. The avoided costs due to the measures implemented to reduce air pollution have been allocated to the population of each NUTS2 region of the territory under study (Table 4), in order to calculate the avoided costs per capita and year. The avoided costs per capita reach appreciable values, although there are significant differences between regions. The inhabitants from Galicia, Asturias, and Castille and Leon would be the main beneficiaries due to joined co-effect of health impact reduction and low population, in comparison with other regions.

Table 4. Avoided costs associated to reduction of impacts on health per capita for the scenarios WM2030 and WAM2030.

Region NUTS2	Code NUTS2	Avoided Costs Associated to Morbidity Reduction per Inhabitant (€$_{2019}$/Inhabitant·Year)		Avoided Costs Associated to Mortality Reduction per Inhabitant (€$_{2019}$/Inhabitant·Year)	
		WM2030	**WAM2030**	**WM2030**	**WAM2030**
Andalusia	ES61	16	45	217	704
Aragon	ES24	13	45	244	806
Principality of Asturias	ES12	59	138	1,320	3190
Balearic Islands	ES53	7	18	100	447
Cantabria	ES13	13	40	324	933
Castille and Leon	ES41	55	168	1,282	3195
Castille—La Mancha	ES42	13	47	214	679
Catalonia	ES51	20	55	395	1577
Valencian Community	ES52	16	44	257	919
Extremadura	ES43	7	24	225	607
Galicia	ES11	16	48	615	1327
Community of Madrid	ES30	24	64	451	1634
Region of Murcia	ES62	9	29	224	919
Chartered Community of Navarra	ES22	13	40	174	501
Basque Country	ES21	12	40	328	1313
La Rioja	ES23	19	57	371	992
Average		20	56	421	1234

Some of the measures could also be associated with other benefits, in terms of environmental external costs or savings of resources. However, here we have focused on the assessment of the benefits associated with change in health impacts. We are aware that the health benefits from implementing the measures are an underestimation of the total benefits. We did not quantify other health effects (different outcomes not included in HRAPIE and the subsequent scientific knowledge, e.g., low birth-weight, changes in lung function, neurological effects, or cancer rates [79]), effects of other air pollutants (e.g., health effects of secondary volatile organic compounds and aerosols), or the potential long-term global climatic effects of CO_2 release from fossil-fuel sources (e.g., external costs associated with indirect effects on health, biodiversity, and materials as consequences of climate change). Such potential impacts are out of the scope of this work, but concerns over such recognized unquantified impacts should be included in decision-making processes to provide a comprehensive overview of the overall impact of air pollution. This study focuses on the pollutants and health impacts recommended for CBAs of air quality policies in Europe [27]. In this sense, we present a complete assessment within those recommendations. Many studies focusing on the external costs of air pollution have shown that the largest and most damaging cost of pollution is related to premature mortality [20]. Morbidity is frequently left out of the assessment. There were three reasons why we included morbidity. First, they are included in the abovementioned recommendations for CBAs. Second, this approach is considered more coherent for communication purposes in policy-making support, due to the comprehensive inclusion of a range of effects, activities, and populations affected. Finally, citizens have seen how some healthcare resources for the current COVID-19 pandemic, such as beds in hospitals or human resources, have been displaced to respond to the sudden need caused by the virus. Thus, the quantification of how many additional resources would be available (e.g., how many cases of hospitalizations due to air pollution can be avoided) by implementing certain measures would help improve the contingency management in the face of healthcare crises as consequences of natural extreme events or pandemics.

As highlighted above, the additional costs of pollution not assessed here make reducing exposure a priority for achieving the goals of shared, inclusive, and sustainable prosperity. The growing challenge of ambient air pollution is likely to require the most ambitious action plans and measures [20]. As we found in this study, there are important improvements in terms of avoided costs when the additional measures included in the WAM2030 scenario are implemented, in comparison with WM2030. Moreover, when air-pollution-related health risks are added to the rest of the health risks that, unlike air pollution, are typically within the remit of health agencies, it is crucial to emphasize the need for governments to consider this important health burden and to call on all international, national, regional, and local environmental and health organizations, as well as the different polluting economic sectors, to work together to face this challenge.

Future work will assess the effects of implementing measures in highly populated cities, such as Madrid and Barcelona, in order to obtain a more accurate assessment of the avoided health impacts, as well as evaluate the influence of model spatial resolution on the impact estimates. CHIMERE will be applied at a spatial resolution of approximately $1 \times 1\ km^2$ for these cities, applying proxies to distribute emissions at this higher resolution (although this will add its own uncertainty). Additional research on the synergies between NOx and O_3 dynamics is needed in order to reach the most suitable strategies for pollution impacts on health. Moreover, it is important to bear in mind that the methodology presents several sources of uncertainty, starting with uncertainties in air-quality modeling, such as the applications of the reductions, provided at national scales by the level of SNAP category (SNAP is the Selected Nomenclature for Air Pollution, the standard classification for sources of air pollution), fixed meteorology (2016), fixed boundary conditions, and a coarse spatial resolution. Benefits of PM_{10} reductions are underestimated due to the lack of information on the reduction in the $PM_{2.5\text{-}10}$ fraction. Uncertainties associated with the estimation of health impacts and monetary valuation of the impacts is also important and, therefore, results should be considered with caution. Those uncertainties could be reduced by using additional CRFs, avoiding double-counting, as well as functions adapted for Spanish conditions, instead of the WHO and HARPIE CRFs recommended for Europe. Ongoing epidemiological research for Spanish case studies is being developed, but the application at national scope has been not reported in the literature. In addition, values for monetarization could be adapted for the Spanish case.

5. Conclusions

Impacts of the potential implementation of the measures included in the 1st National Programme for Air Pollution Control (NAPCP) have been investigated by applying a cost–benefit analysis. Using air quality modeling, the attributable health impacts associated with air pollution under different scenarios of pollution reduction measures are assessed, as are the external costs associated with these health impacts.

In terms of impacts on health, the envisaged reduction of NOx and $PM_{2.5}$ emissions will reduce health risks from NO_2 and $PM_{2.5}$ exposure, and, at a national scale, from O_3 exposure. Nevertheless, due to the so-called titration effect, the health risks from ozone exposure could increase slightly over cities such as Madrid and Barcelona. However, impacts from ozone exposure are small, compared with those of $PM_{2.5}$ and NO_2. In general, there is an improvement in health impacts for the scenarios WM2030 and WAM2030, with respect to those of 2016. Most significantly, total mortality impacts in 2016 are estimated at 41,700 premature deaths, a figure that could be reduced by 35% if the measures proposed in the Programme are implemented.

Monetary valuation of total health effects shows that air pollution costs can be as high as 15% of the national GDP, and that important benefits can be expected from adopting the additional measures in WAM2030, quantified at around 58,115 M $€_{2019}$. These numbers can give policy-makers an idea of the magnitude of the benefits that the NAPCP will bring to the Spanish economy, and an estimation against which to compare the costs of implementing the measures proposed in this plan.

For future work, in order to complete the CBA, the total external costs avoided will be compared with the implementation costs in future works, in order to estimate the net benefit of the investment on air pollution measures beyond the societal value of reduced mortality and morbidity. Additionally, future research will be focused on cities such as Madrid and Barcelona, in order to obtain a more localized assessment of the avoided health impacts, as well as cost-effective measures designed for those urban ecosystems.

However, it is important to consider that the methodology presents several sources of uncertainty, starting with uncertainties in air-quality modeling, such as the application of reductions at a national level and by SNAP category, fixed meteorology (2016), fixed boundary conditions, and a fairly coarse spatial resolution. Uncertainties associated with the estimation of health impacts and the monetary valuation of these impacts are also important and, therefore, the results should be considered with caution.

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