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A systems approach to evaluating the air quality co-benefits of US carbon policies

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Because human activities emit greenhouse gases (GHGs) and conventional air pollutants from common sources, policy designed to reduce GHGs can have co-benefits for air quality that may offset some or all of the near-term costs of GHG mitigation. We present a systems approach to quantify air quality co-benefits of US policies to reduce GHG (carbon) emissions. We assess health-related benefits from reduced ozone and particulate matter ($PM_{2.5}$) by linking three advanced models, representing the full pathway from policy to pollutant damages. We also examine the sensitivity of co-benefits to key policy-relevant sources of uncertainty and variability. We find that monetized human health benefits associated with air quality improvements can offset 26–1,050% of the cost of US carbon policies. More flexible policies that minimize costs, such as cap-and-trade standards, have larger net co-benefits than policies that target specific sectors (electricity and transportation). Although air quality co-benefits can be comparable to policy costs for present-day air quality and near-term US carbon policies, potential co-benefits rapidly diminish as carbon policies become more stringent.

Climate change and regional air quality are major sustainability challenges. Ground level ozone (O_3) and particulate matter ($PM_{2.5}$, particulate matter with diameter $\leq 2.5\,\mu m$) are linked to respiratory diseases and premature death[1,2]. Despite regulatory efforts, 232 and 118 US counties exceeded national O_3 and $PM_{2.5}$ standards, respectively, in 2011 (refs 3,4). Concurrently, changing climate is becoming a global health issue, as increasing temperatures and changing weather patterns threaten human well-being[5].

The Intergovernmental Panel on Climate Change (IPCC) noted that GHG emissions controls can have near-term health co-benefits from reduced air pollution, which may offset a substantial fraction of mitigation costs[6]. Nemet *et al.*[7] summarized 37 peer-reviewed co-benefits estimates, finding a range from US\$2-196/tCO_2 and mean US\$47/tCO_2, with highest values in developing countries. This range reflects co-benefit variability across different study methods, technologies, spatial scales and societies.

Air quality co-benefits estimates are additional to climate benefits from reduced CO_2 emission. In assessing the Social Cost of Carbon (SCC), the US Interagency Working Group estimated marginal damages of CO_2 emitted in 2020 at 43 US\$/tonne (2007 US\$ using 3% discounting; refs 8,9). These monetized impacts of CO_2 emissions include, but are not limited to, reduced agricultural yields, coastal flooding, and increased frequency and severity of weather events[10].

Air pollution and climate change are elements of a coupled social and technical system. Comprehensively assessing potential co-benefits of climate policies to air pollution and associated human impacts, considering variability and uncertainty, requires combining approaches from several disciplines tracing the entire pathway from policies to impacts. First, climate policies influence economic activities and associated emissions of both GHGs and conventional air pollutants. Unlike for GHGs, spatial distribution of air pollutant emissions matters. O_3 and $PM_{2.5}$ formation is nonlinear, and pollutant distribution also impacts population exposure; predicting these requires advanced atmospheric modelling. Atmospheric concentrations must then be linked to human health outcomes through exposure-response calculations. Costs are then derived from economic analyses.

Previous literature has addressed aspects of this system in both physical and societal dimensions, using models to simulate complexities and interactions[11]. In atmospheric chemistry, most analyses using comprehensive chemical models have treated policies exogenously, with associated fixed costs[12,13]. Economic studies[14–19] have focused extensively on drivers of cost variation. However, these studies often use simplified methods linking emissions to concentrations and impacts, neglecting full atmospheric complexity.

It has been noted that the co-benefits literature has had little policy traction[7,11]; one reason given is lack of comprehensive analysis from the full set of disciplines underlying both cost and benefit analysis. With information on how assumptions and uncertainties from various fields combine to influence benefits-per-ton estimates, decision-makers can identify the robustness of policies to variation in drivers of both cost and benefit. As full quantitative uncertainty analysis of all factors is computationally impossible, methods are needed to selectively address the most policy-relevant uncertainties.

Here, we illustrate a systems-level approach to analysing how climate policies influence air quality, focusing on US emissions of O_3 and $PM_{2.5}$ precursors to 2030. We assess costs and air-quality-related benefits of three potential national-scale climate policies. We examine the entire pathway linking climate policies, economic sector responses, emissions, regional air quality, human health and related economic impacts, using advanced models at every stage. We first simulate climate policies in the United States Regional Energy Policy (USREP) model. Resulting economic constraints lead to

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economic output changes that vary by policy, economic sector and US region[20]. Changed economic output is used to scale emissions inventories, and the Comprehensive Air Quality Model with Extensions (CAMx; ref. 21) projects resulting ambient pollutant levels. Finally, the Environmental Benefits Mapping and Analysis (BenMAP) program calculates changes in population exposure to pollution, resulting changes to human mortality and morbidity, and corresponding monetized benefits[22]. Using these coupled models, we capture important economic and atmospheric complexities and nonlinearities. We also conduct a policies-to-impacts sensitivity analysis to quantify policy-relevant uncertainties and variabilities: economic growth, technology costs, baseline emissions assumption and representation of health responses.

Results

We first present results for economic costs and emissions, O_3 and $PM_{2.5}$ concentrations, and health and economic benefits for three carbon reduction policies for a base case. We then show our policies-to-impacts sensitivity analysis, examining variation of base case results with uncertainty and variability in key assumptions. Costs and benefits for all scenarios described below are reported in Table 1 and presented graphically in Supplementary Fig. 3.

Economy and emissions. The three carbon reduction policies examined are: Clean Energy Standard (CES), addressing the electricity generation sector; Transportation (TRN), targeting on-road light duty (passenger) and heavy duty (truck) vehicles; and Cap-and-Trade (CAT), with caps applied economy-wide. Each policy is applied in USREP to constrain total carbon emissions from the corresponding sector(s) of the economy, and reduces nationwide CO_2 by 500 million tonnes (or 10%) in 2030 relative to 2006 (USREP base year). We compare each policy to a Business-As-Usual (BAU) case with no carbon emissions constraints. We calculate policy cost (in 2030) as the difference in simulated economic welfare, which includes macroeconomic consumption (capturing market-based activities), and the monetary value of non-working time (leisure; ref. 23).

Changes in economic output under each policy are archived for each USREP sector and region in 2030. Emissions in our economic base year, 2006, are represented in air quality modelling by a nationwide, year-long, spatially and temporally detailed emissions inventory (representing 2005). Each individual emissions source is matched to one of the 17 economic sectors and 12 regions within USREP, and emissions are scaled using the 2030/2006 ratio of output from that sector/region. Thus, spatial and temporal detail for each source is maintained. Moreover, the sectoral structure of USREP is based on detailed input–output data, thereby capturing the empirically observed inter-sectoral linkages within each regional economy. Using this method, we thus capture the full life-cycle and supply chain impacts. For example, if electricity generation from natural gas increases, emissions associated with its production, transportation and use also increase.

Emissions of O_3 and $PM_{2.5}$ precursors decline relative to BAU under all scenarios. Under CES, largest declines are for SO_2 and NO_x, mostly the result of a shift from carbon-intensive coal-fired power plants. Largest reductions under TRN are for CO and NO_x from private and commercial on-road vehicles. In TRN, NH_3 also declines from the agricultural sector, as sectoral output falls owing to increased transportation costs. CAT, which applies to all sectors, reduces less SO_2 and NO_x than CES, and less NO_x and CO than TRN, relative to BAU. Household heating and power generation experience the largest emission reductions under CAT. Emissions projections are detailed in Supplementary Table 1 and Fig. 2.

O_3 and $PM_{2.5}$ Concentrations. Adjusted year-long emissions inventories representing each scenario in 2030 are input to CAMx, which simulates resulting ambient air pollution concentrations. Figure 1 shows the projected difference in O_3 (O_3 season average, top row) and $PM_{2.5}$ (annual average, bottom row) between each climate policy and BAU in 2030. Both O_3 and $PM_{2.5}$ decline overall relative to BAU under all policies. Population-weighted concentration reductions are largest under TRN, but of similar magnitude across policies (0.21–0.99 ppb for O_3 and 0.56–1.16 μg m^{-3} for $PM_{2.5}$). O_3 increases under TRN in some urban centres with large NO_x emissions (often due to heavy traffic); excess NO_x reacts with O_3, thus NO_x reductions there increase O_3. See Supplementary Methods for discussion of NO_x titration.

Table 1 | Total costs and benefits for all scenarios (billion year 2006 US$, undiscounted).

Policy scenario	Cost	Air pollution health benefit (95% confidence interval)
Clean Energy Standard (CES)	$208	$247 ($19–841)
High-Base CES	$295	$334 ($26–1,136)
Low-Cost CES	$145	$134 ($10–457)
High-Cost CES	$248	$271 ($21–924)
2012 Emissions CES	$208	$207 ($16–707)
Transportation (TRN)	$1,028	$287 ($22–981)
High-Base TRN	$1,157	$337 ($26–1,150)
Low-Cost TRN	$813	$210 ($16–716)
High-Cost TRN	$1,249	$340 ($27–1,159)
2012 Emissions TRN	$1,028	$218 ($17–745)
Cap-and-Trade (CAT)	$14	$139 ($11–473)
High-Base CAT	$87	$375 ($29–1,276)
Low-Cost CAT	$26	$123 ($10–419)
High-Cost CAT	$15	$156 ($12–533)
2012 Emissions CAT	$14	$114 ($9–390)
CAT High	$124	$400 ($31–1,360)

The three USREP scenarios used for sensitivity analysis vary baseline economic growth and CO_2 emissions, cost of renewables, and fuel efficiency improvements. High-Base has 14% greater CO_2 emissions under business-as-usual in 2030 (7,100 mmt). Low-Cost reduces the cost of wind technologies by 15% compared to the base case, and improves fuel efficiency by increasing flexibility in the model to substitute fuel inputs with powertrain capital in private vehicles. The High-Cost scenario increases wind technology cost by 15% compared to the base case and decreases the flexibility in the model to substitute fuel inputs with powertrain capital in private vehicles.

Health and economic benefits. Benefits include human mortality and morbidity impacts associated with changes in O_3 and $PM_{2.5}$. Benefits are undiscounted, with a 95% confidence interval (CI) calculated using random/fixed effects pooling for selected concentration-response functions plus uncertainty associated with benefits valuation. Uncertainty associated with individual response functions and valuation functions are presented in Supplementary Information (Supplementary Figs 4 and 5). Morbidity calculations follow recent US regulatory analysis methods[24]. Median co-benefits ranged across policies from US$140 to 290 billion.

Policies-to-Impacts Sensitivity Analysis

We show results of sensitivity to policy stringency, baseline precursor emissions, and economic assumptions and parameters. Although additional uncertainties and variability exist along the policies-to-impacts pathway, as discussed further below, we select these as major influences on policy-relevant variation. Costs per tonne CO_2 are presented in Supplementary Table 2.

Sensitivity to stringency of policy. We tested a cap-and-trade policy (CAT-High) that achieves approximately twice the CO_2 reduction of our base case. Figure 2 shows the percentage of policy

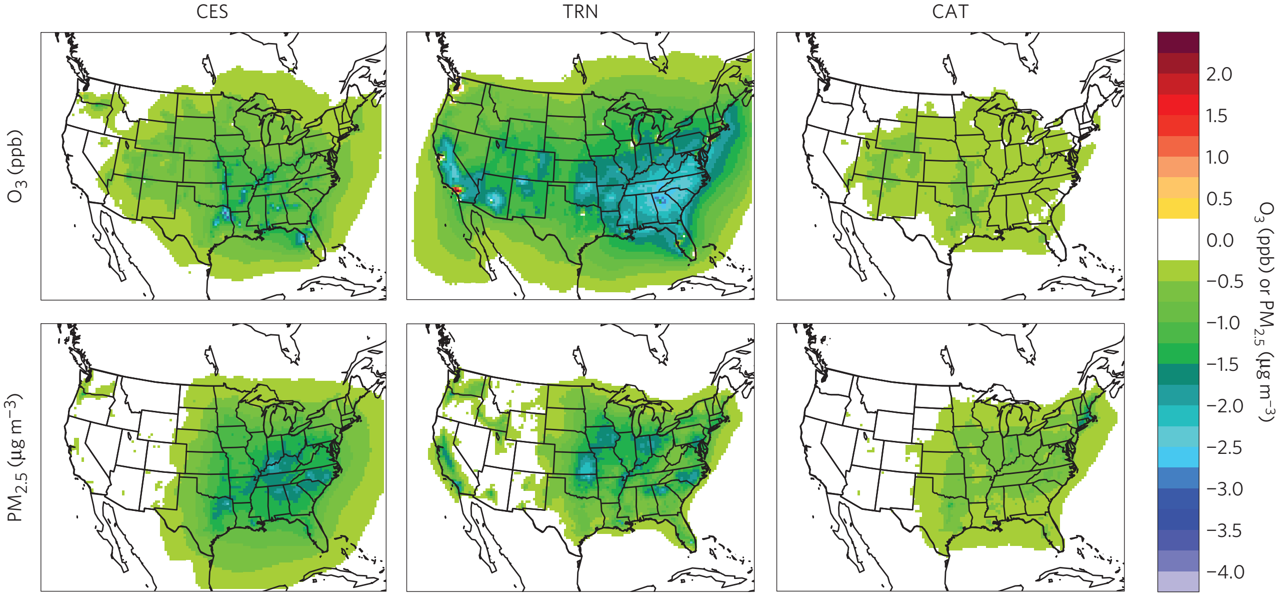

Figure 1 | Spatial maps show the difference in pollution concentration between each base case policy option and BAU. Change in O_3 (O_3 season average daily maximum 8-h O_3, ppb, top row) and $PM_{2.5}$ (annual average, μg m^{-3}, bottom row) in 2030 for carbon policies (CES, TRN, CAT).

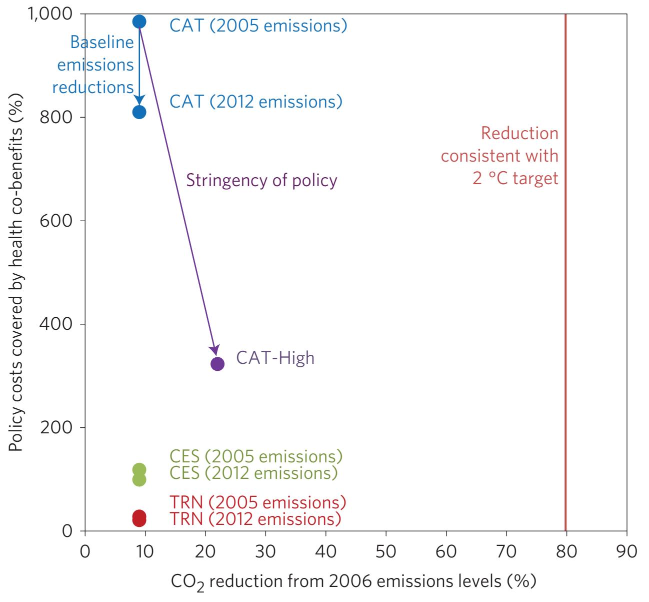

Figure 2 | Percentage of policy costs covered by the median value of policy benefits versus percentage CO_2 reduction relative to 2006. Policy cases (CAT, CES and TRN) and sensitivity scenarios shown for: 2005 and 2012 criteria pollutant emissions inventory and CAT-High sensitivity run. Red vertical line shows the approximate reduction target consistent with a 2 °C temperature increase limit[39].

costs covered by co-benefits versus total carbon reduction relative to 2006 for the CAT base and CAT-High scenarios. The benefit-cost ratio of CAT-High decreases with increasing policy stringency as cheaper controls are exhausted.

Sensitivity to emissions baseline. We use two year-long emissions inventories developed by the US EPA to test the sensitivity of our result to unrelated air pollution controls. Specifically, we recalculate emissions changes for BAU, CAT, CES and TRN scenarios in 2030 with emission inventories representing 2012, and compare estimated co-benefits with the 2005 inventory results. The US EPA estimates that national emissions of NO_x, SO_2, and CO decreased by 20–30% from 2005 to 2012. Using 2012 baseline emissions, calculated human health co-benefits declined by 16–24%, with largest changes for TRN. Figure 2 shows the change in percentage of costs covered by benefits using 2005 and 2012 emissions baselines for each policy scenario.

Sensitivity to economic parameters. Three additional scenarios assessed the effect of economic assumptions previously shown to influence economic welfare[25]: baseline growth assumptions; the cost of renewables; and fuel efficiency improvement in private transportation. A first scenario (High-Base) applies high economic growth and thus high CO_2 emissions growth in BAU. A second scenario, an economically optimistic scenario (Low-Cost), assumes lower renewable energy cost and enhanced fuel efficiency improvement. A final scenario (High-Cost) is economically pessimistic, assuming high renewable energy costs and lower fuel efficiency improvement in private vehicles.

The High-Base scenario increases costs and benefits for all policies, because higher CO_2 means that a larger amount must be reduced in 2030 to achieve the 10% reduction target. Low-Cost has smaller benefits, and High-Cost has larger benefits relative to the base case. Because renewables are cheaper and cars cleaner under Low-Cost, its BAU case emits less CO_2 in 2030 relative to the baseline BAU. Therefore, less CO_2 must be reduced to achieve the 10% reduction target, leading to a smaller reduction in co-emitted air pollutants. In the High-Cost scenario, the opposite is true: less renewable energy is produced, and more CO_2 is emitted from vehicles in 2030 in its BAU. Therefore, relatively larger CO_2 cuts are required. Supplementary Table 3 presents co-benefits per tonne CO_2 for all sensitivity simulations.

Comparison of sensitivities for net benefits. Figure 3 presents the range of health and cost responses of each policy scenario to the economic sensitivity runs. Each vertical line represents an economic sensitivity run. Table 2 shows net co-benefits (co-benefit minus cost) in billion US$, and the percentage of cost covered by benefits for differences in policy scenario, economic model choice, and 95% CI for health benefits estimates. Variation across rows reflects policy choice and economic assumptions, while along columns it reflects uncertainty in concentration-response functions and valuations.

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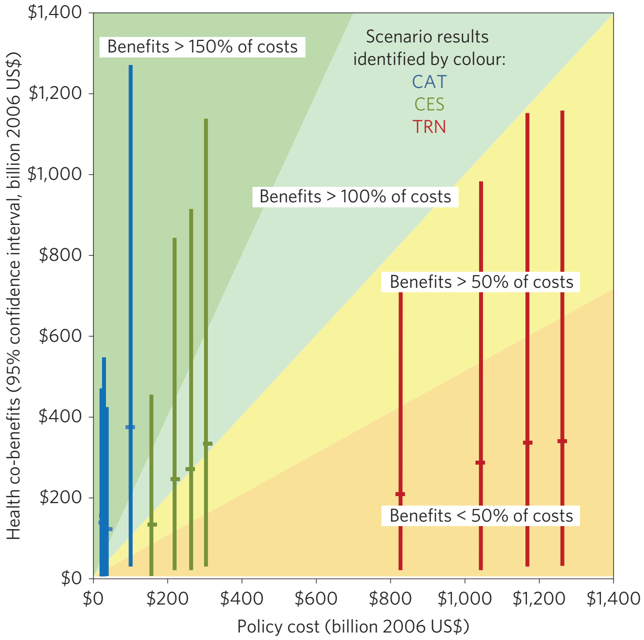

Figure 3 | Human health co-benefits versus policy costs (billion 2006 US$, undiscounted). Vertical lines represent discrete economic sensitivity simulations (blue = CAT, green = CES and red = TRN). Human health benefits are shown as median (rectangle) and 95% CI including health response and valuation (vertical line extent). Values presented in Table 1.

Table 2 | Net benefits (total air quality co-benefits − total cost) of each policy sensitivity case (billion 2006 US$, undiscounted).

Policy & cost assumptions	Benefit assumptions (billions)		
	Median	High	Low
CES			
Standard	$39 (119%)	$633 (404%)	$−189 (9%)
High-Base	$39 (113%)	$842 (386%)	$−269 (9%)
Low-Cost	$−11 (92%)	$312 (315%)	$−135 (7%)
High-Cost	$23 (109%)	$676 (373%)	$−227 (9%)
TRN			
Standard	$−741 (28%)	$−48 (95%)	$−1,006 (2%)
High-Base	$−820 (29%)	$−7 (99%)	$−1,131 (2%)
Low-Cost	$−603 (26%)	$−97 (88%)	$−796 (2%)
High-Cost	$−909 (27%)	$−90 (93%)	$−1,222 (2%)
CAT			
Standard	$125 (985%)	$459 (3,360%)	$−3 (77%)
High-Base	$289 (433%)	$1,190 (1,474%)	$−57 (34%)
Low-Cost	$97 (470%)	$393 (1,604%)	$−17 (37%)
High-Cost	$141 (1,051%)	$518 (3,584%)	$−3 (82%)

Percentage of costs covered by benefits in parentheses. Columns (Median, High, Low) represent 95% CI of the human health benefits (uncertainty associated with concentration response function and valuation function).

Variation within each policy choice to economic and policy assumptions shows the robustness of policy design to uncertainties. Percentage-wise, net benefit of CAT varies the most (13–1,685%, a factor of >100) across both economic and concentration-response/valuation assumptions. Although reductions under CAT can come from any economic sector, economic assumptions impact lowest-cost opportunities, resulting in wide cost and benefit variation. The benefit-to-cost ratio for CES and TRN scenarios varies by <25% for all economic sensitivity runs, indicating that benefits roughly scale with costs. As CES and TRN emissions reductions are each limited to a single economic sector, costs and benefits vary less.

Implications for co-benefit assessment

Air quality co-benefits of carbon emissions policies would probably offset much of their economic cost[7]. We estimate that human health benefits associated with air quality improvements offset 26–1,050% of costs depending on the flexibility of the carbon policy. More flexible policies such as CAT are less costly than those that impose reductions from specific sectors (CES or TRN), as the latter fail to equalize marginal abatement costs across economic activities[26]. We find that this flexibility has a relatively larger influence on cost, and a smaller influence on co-benefits.

Using our policies-to-impacts sensitivity analysis, we identify important qualifications to our base case conclusions, going beyond the insights of previous work. Median benefits of CAT aggregated at the national scale exceeded its low costs in all simulated sensitivity scenarios. Although carbon policies that target economic sectors known to contribute substantially to poor air quality (electricity and transportation sectors) have somewhat larger benefits, these policies are also more costly versus CAT. More stringent CAT policy (CAT-High) exceeded the benefits of these sector-specific approaches. However, increasing stringency of CAT policy leads to a smaller fraction of costs offset by co-benefits.

A key finding from our sensitivity analysis is that policy and economic assumptions had a larger impact on policy costs than on median co-benefits both across and within different scenarios. Standard atmospheric science approaches have largely omitted rigorous accounting for cost uncertainty; our analysis suggests this can be the most important policy-relevant uncertain term. This suggests that, for a variety of carbon policy choices, including subsidies that influence the cost of renewables and technologies, net co-benefit is driven by costs rather than benefits.

Large-scale pollutant emissions reductions unrelated to carbon policy will probably decrease human health co-benefits. We saw a 16–24% decrease in human health co-benefits by changing baseline emissions, sublinear relative to emissions decreases. Although our 2012 inventory takes into account present air quality regulations, our 2030 predictions do not account for further regulatory action. In particular, the Mercury and Air Toxics Standards (MATS) are projected to reduce SO_2 emissions from the power sector by >40% by 2016 (ref. 27). As the emissions baseline does not account for MATS SO_2 reductions, co-benefits associated with $PM_{2.5}$ may be biased high. Uncertainty in emissions changes due to future policy is not well understood[28], and although our emissions runs partially address this, future analyses could apply endogenous pollution abatement costs in economic models[29].

Uncertainty in concentration-response functions (crfs) and Value of a Statistical Life (VSL) has a large influence on the magnitude of benefits, but we show that their variation can be comparable to other assumptions along the policies-to-impacts chain (such as economic modelling assumptions). Previous work has found assumptions associated with crfs to be a larger source of uncertainty than assumptions associated with VSL (ref. 30). Uncertainty associated with both the crfs and VSL will be constant across scenarios and sensitivities, so although these assumptions will change the benefit to cost ratio, they do not change how policies compare to each other. In our benefits analysis, we used BenMAP, for consistency with regulatory analyses. Alternatively, representing air quality impacts in a computable general equilibrium model can assess their economy-wide welfare implications (mortality impacts on labour supply, and morbidity impacts on demand for health services), and can capture the response of these impacts to changing prices and policy constraints[31–33].

Although we have illustrated several policy-relevant uncertainties and variabilities along the policies-to-impacts pathway, there are numerous aspects that we have not quantitatively assessed. Year-to-year meteorological variability can change the distribution and formation of $PM_{2.5}$ and O_3. Climate changes can affect air quality: rising temperatures will probably increase O_3 formation on the order of increases predicted here by changing emissions[34,35]. However, health benefits are dominated by $PM_{2.5}$ (refs 36,37), and the influence of climate change on $PM_{2.5}$ distribution and health impacts remains difficult to quantify[38]. Uncertainty quantification in modelling transport and chemistry of pollutant formation is also limited both by model fidelity and scientific knowledge, particularly with respect to the formation of $PM_{2.5}$. Further applications of our approach, however, could incorporate future quantitative estimates of these influences in a policy-relevant way. For example, although meteorological variability may change the absolute level of air pollution co-benefits, it may not affect selected policies differently. In this way, our approach is distinct from traditional uncertainty and sensitivity analysis.

There are additional uncertainties which cannot be captured within this framework. Regional atmospheric models such as CAMx cannot capture sub-grid scale variability. If maximum reductions occur in areas of high population density with large spatial gradients (for example, primary $PM_{2.5}$ from vehicles), the corresponding scenario could have larger benefits. Secondary $PM_{2.5}$, however, is not sensitive to resolution on the scale of most regional modelling[36].

Whereas our model assumes that sectoral reductions will be homogeneous across all sources within each sector, different sources within each sector will react differently. For example, if carbon policy reduces coal-fired power plant output, some individual plants may close while others operate normally. This can affect the spatial distribution of calculated benefits.

Implications for policy

Our approach suggests several insights for decision-makers considering co-benefits of different climate policies. We find co-benefits comparable with policy costs for existing air quality and realistic climate policy goals in the US, suggesting that substantial co-benefits for CO_2 reduction are not limited to developing regions.

A US carbon cap would have a measurable, positive impact on regional air quality relative to BAU, similar in magnitude to policy specifically targeting O_3 and $PM_{2.5}$. Air pollution reductions relative to BAU estimated here are comparable to the 1.4% and 38% reductions of NO_x and SO_2 proposed by the US EPA for 2014 and recently upheld by the US Supreme Court. However, EPA estimates of air quality policy costs (US$800 mil, ref. 24) are an order of magnitude smaller than carbon policy costs. This result should also be interpreted with caution, as our BAU case does not include air quality improvement.

We find diminishing relative co-benefits with both baseline emissions improvements and increasing climate policy stringency. Figure 2 suggests that very stringent climate policies—necessary to meet a 2° global target, estimated as $\geq$80% reductions[39]—may be offset to a much lower degree by air quality co-benefits. Macro-economic costs of CO_2 abatement are increasing more than proportionally to the abatement target. This means that although initial policy actions can be motivated based on air pollution co-benefits, this strategy has important limits. Whereas we conduct national scale analysis, state and regional air quality decision-makers assess policies based on both regional cost-benefit assessment and regulatory attainment status. Benefits in individual regions largely follow $PM_{2.5}$ concentration changes, but policy makers might be more concerned with O_3 from a regulatory standpoint. Costs can vary substantially across regions, reflecting, among other things, regional disparities in energy intensity (energy consumption/GDP) and electricity generation fuel mix[40]. Regional benefit/cost ratios may thus vary, creating winners and losers. Also, carbon policies may be applied at state or regional, rather than national scale. Future work could examine these regional differences.

The co-benefits estimates presented here should not be interpreted as a comprehensive benefit-cost analysis for the considered carbon policies. Co-benefits are additional to (and of larger magnitude than) estimates of the SCC. We focus on O_3 and $PM_{2.5}$; additional air quality improvements such as reductions in mercury and other air toxics could have co-benefits not captured here. Furthermore policies themselves may have additional health and economic benefits: for example, reduced vehicle transportation use may improve health by encouraging walking and bicycling. Our results suggest, however, that cost-benefit analyses of climate policy that omit regional air pollution could greatly underestimate benefits.

Methods

The MIT US regional energy policy (USREP) model. USREP is a recursive-dynamic computable general equilibrium (CGE) model of the US economy designed to analyse energy and GHG policies. USREP has been widely used to investigate energy and climate policy, including interactions with tax policy, and effects on economic growth, efficiency and distribution[20,40–42]. USREP is described in detail in Rausch *et al.*[40,42], and further model details are presented in the Supplementary Information. We conduct simulations from 2006 to 2030, with a five-year timestep. CO_2 emissions grow to 6,200 million metric tonnes (mmt) under the Business-As-Usual (BAU) case. Economic output and CO_2 emission under each policy is archived from USREP by region and sector (see Supplementary Information for details).

The three USREP scenarios used for sensitivity analysis vary baseline economic growth and CO_2 emissions, cost of renewables, and fuel efficiency improvements. High-Base has a 14% greater CO_2 emission under business-as-usual in 2030 (7,100 mmt). Low-Cost reduces the cost of wind technologies by 15% compared to the base case, and improves fuel efficiency by increasing flexibility in the model to substitute fuel inputs with powertrain capital in private vehicles. The High-Cost scenario increases wind technology cost by 15% compared to the base case and decreases the flexibility in the model to substitute fuel inputs with powertrain capital in private vehicles.

Future emissions projections. We match USREP economic sectors and regions to individual emissions sources by matching Standard Classification Codes used in US EPA national emission inventories to USREP sector categories. The 2030 scaled emissions inventory is prepared for regional modelling using the Sparse Matrix Operator Kernel Emissions (SMOKE) emission preprocessing program[43]. SMOKE creates gridded and speciated hourly emissions files for input to CAMx. Supplementary Table 4 shows benefits per short ton of SO_2 and NO_x reduced for three base case scenarios. We assume emissions factors (pollutant per economic activity) remain constant from 2005 to 2030, not accounting for technological improvement, to test the potential magnitude of effects of co-benefits alone. Our emissions sensitivity test partly addresses the influence of improved technology between 2005 and 2012 on implied emissions factors.

CAMx. Photochemical modelling simulations are conducted for each BAU case and policy scenario using CAMx, a three-dimensional, Eulerian photochemical model that simulates emission, transport, chemistry and removal of chemical species in the atmosphere[21]. CAMx is approved by the US EPA for air quality analysis[44]. We use modelling inputs, including full year 2005 meteorological inputs and emissions inventories, developed (and evaluated versus measurements) by the US EPA (ref. 45). Meteorological conditions represent present climate and are held constant throughout our study. The CAMx modelling domain used covers the continental US with 36 km by 36 km horizontal resolution, shown previously to be appropriate for national-scale benefits analysis for $PM_{2.5}$ and O_3 (refs 36,46).

For O_3, daily maximum 8-h average concentration is calculated for May through September (the O_3 season) and then averaged. For $PM_{2.5}$, annual average is calculated from hourly sums of individual species concentrations (sulphate, nitrate, ammonia, black carbon, and primary and secondary organics). Simulated average daily maximum 8-h O_3 (in ppb) and annual average $PM_{2.5}$ (in $\mu g\, m^{-3}$) from each model run are compared to BAU, and differences used in health incidence estimation and valuation.

Health incidence estimation and valuation. We follow the methodology of the US EPA's Regulatory Impact Analysis process, using the US EPA's BenMAP program[22], whereby modelled changes in ambient concentrations are related to health incidences through concentration-response functions and projected health and census data[37]. Calculated O_3 and $PM_{2.5}$ changes are overlaid with forecast US census data and county-level mortality data, both representing 2030 (ref. 47), and applied to these concentration response functions. Changes to mortality dominate benefits (over morbidity responses) and are valued using the EPA's estimate for the Value of a Statistical Life (VSL) based on 26 value-of-life studies[22].

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Author contributions

T.M.T., S.R. and N.E.S. designed the modelling framework and the research approach. T.M.T. linked the framework and conducted the atmospheric modelling and human health analysis. S.R. developed the economic modelling tool and conducted the economic model runs. R.K.S. assisted with the human health analysis. All authors contributed to writing the text.

Additional information

Supplementary information is available in the online version of the paper. Reprints and permissions information is available online at www.nature.com/reprints. Correspondence and requests for materials should be addressed to T.M.T.

Competing financial interests

The authors declare no competing financial interests.