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Marginal climate and air quality costs of aviation emissions

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Abstract

Aviation emissions have been found to cause 5% of global anthropogenic radiative forcing and ~16 000 premature deaths annually due to impaired air quality. When aiming to reduce these impacts, decision makers often face trade-offs between different emission species or impacts in different times and locations. To inform rational decision-making, this study computes aviation's marginal climate and air quality impacts per tonne of species emitted and accounts for the altitude, location, and chemical composition of emissions. Climate impacts are calculated using a reduced-order climate model, and air quality-related health impacts are quantified using marginal atmospheric sensitivities to emissions from the adjoint of the global chemistry-transport model GEOS-Chem in combination with concentration response functions and the value of statistical life. The results indicate that 90% of the global impacts per unit of fuel burn are attributable to cruise emissions, and that 64% of all damages are the result of air quality impacts. Furthermore, nitrogen oxides (NO_x), carbon dioxide (CO_2), and contrails are collectively responsible for 97% of the total impact. Applying our result metrics to an example, we find that a 20% NO_x stringency scenario for new aircraft would reduce the net atmospheric impacts by 700 m USD during the first year of operation, even if the NO_x emission reductions cause a small increase in CO_2 emissions of 2%. In such a way, the damage metrics can be used to rapidly evaluate the atmospheric impacts of market growth as well as emissions trade-offs of aviation-related policies or technology improvements.

1. Introduction

Commercial civil aviation emissions are an increasingly significant contributor to anthropogenic climate change. Aviation attributable carbon dioxide (CO_2) emissions have increased 2.6% per year on average over the past 25 years (International Energy Agency 2017), and it is estimated that commercial aviation already accounts for about 5% of global anthropogenic radiative forcing (Lee *et al* 2009). In addition, aviation emissions adversely affect human health (Masiol and Harrison 2014) and have been associated with ~16 000 premature mortalities annually (Yim *et al* 2015, Eastham and Barrett 2016).

Efforts to mitigate the climate and air quality impacts of aviation emissions have historically focused on technological and operational approaches to improve fuel efficiency (Hileman *et al* 2008, Marais *et al* 2013, ICAO 2017), emissions standards (ICAO 2016a 2008), market-based measures to reduce CO_2 emissions (ICAO 2018, World Bank Group 2018), or the use of alternative aviation fuels (Staples *et al* 2018). However, reductions of one emissions species can come at the cost of increasing emissions of another species, either in absolute terms or by limiting the potential reductions offered by new technologies. For instance, NO_x emissions could be decreased by designing engines with lower combustor temperatures, but

this would result in lower thermodynamic efficiencies, leading to higher CO_2 emissions (Lefebvre 1983, Lieuwen and Yang 2013). In order to evaluate the costs and benefits of different policy and technology scenarios, these trade-offs must be quantified (Mahashabde *et al* 2011). The unique characteristics of each emission species as well as temporal and spatial variation in their impact patterns further complicate evaluating such efforts.

Previous impact trade-off studies have focused on various emissions reductions scenarios for non-aviation sources, primarily at ground-level (Berk *et al* 2006, Driscoll *et al* 2015, Shindell *et al* 2016, Dedoussi *et al* 2019). However, since 91% of all aviation fuel burn occurs during cruise, applying these results to aviation emissions would lead to incorrect population exposure patterns due to differences in transport, chemical interactions, and deposition (Barrett *et al* 2010, Yim *et al* 2015, Cameron *et al* 2017). In addition, high altitude emissions have different climate impacts due to the potential for contrail formation, changes in aerosol radiative impacts (Ban-Weiss *et al* 2012), and extended aerosol lifetimes for emissions above the cloud deck (Lund *et al* 2017b).

This gap is partially filled by existing studies which focused on either the climate or the air quality impacts of aviation. Air quality impacts were calculated for both near ground emissions (Unal *et al* 2005, Levy *et al* 2012b, Yim *et al* 2013, Brunelle-Yeung *et al* 2014, Masiol and Harrison 2014) and for cruise or full flight emissions (Barrett *et al* 2010, Yim *et al* 2015, Eastham and Barrett 2016, Cameron *et al* 2017). Climate impact studies typically estimated the total speciated radiative forcing from one year of aviation emissions (Penner *et al* 1999, Sausen *et al* 2005, Lee *et al* 2009, Brasseur *et al* 2016), or focused on studying one climate forcer, including specifically contrails (Burkhardt and Kärcher 2011, Chen and Gettelman 2013, Schumann and Graf 2013, Chen and Gettelman 2016, Bock and Burkhardt 2019) and aviation NO_x emissions (Wild *et al* 2001, Stevenson *et al* 2004, Köhler *et al* 2008, Hoor *et al* 2009, Holmes *et al* 2011, Søvde *et al* 2014, Skowron *et al* 2015). In addition, Lund *et al* (2017a) presented regionalized aviation impact climate metrics. Freeman *et al* (2018) aimed to identify optimal aviation climate policy considering the trade-offs between NO_x emissions and CO_2 emissions, disregarding the air quality impacts from NO_x emissions.

While the aforementioned studies are valuable for understanding the emissions-to-impact mechanisms, they do not enable consistent comparative assessments of aviation emissions trade-offs considering both climate and air quality impacts. A limited number of studies are available which focused on evaluating these trade-offs. Mahashabde *et al* (2011) evaluated climate, air quality, and noise trade-offs for a set of specific aviation NO_x emissions control scenarios. Dorbian *et al* (2011) presented metrics to evaluate both the climate and air quality impacts of aviation.

These climate metrics are computed per unit of full flight fuel burn, and the air quality costs are quantified for emissions in the landing and take-off flight phase where only ∼10% of fuel burn occurs. As such, Dorbian *et al* (2011) provided a foundation for the climate and air quality assessments of fuel burn reduction, but disregarded air quality impacts from cruise emissions and did not provide insights into the emissions trade-offs of different species.

This paper presents the first set of speciated emissions cost metrics for both climate and air quality (ground-level population exposure to fine particulate matter ($PM_{2.5}$) and tropospheric ozone) per unit of aviation emissions. To evaluate impacts for different flight phases, we present metrics for cruise, and landing and take-off (LTO) emissions, defined as emissions above and below 3000 feet, respectively. In addition, air quality metrics are calculated by world region, so that the variation of societal costs with local and regional operations, atmospheric conditions, and population distribution are captured.

As examples of the utility of these cost metrics, we apply them to evaluate the effects of a global expansion in aviation, consistent in magnitude with current annual growth in aviation. We use this as a benchmark for three scenarios. First, we consider a growth scenario with fuel efficiency increases and reductions in NO_x emissions factors consistent with 10 year technology improvements goals (ICAO 2007, Lee *et al* 2009). Second, we build on the work of Freeman *et al* (2018) and quantify the trade-offs between the climate and air quality impacts of NO_x emission reductions, which are also associated with climate impacts due to increasing CO_2 emissions. Finally, we re-assess the climate and air quality trade-offs of jet fuel desulfurization (Barrett *et al* 2012).

These scenarios demonstrate how decision makers can use the results from this paper to estimate climate or air quality impacts of aviation policies, operational procedures, and technologies. As such, our results aim to enable decision making for the aviation sector. We therefore treat all aviation emissions as the marginal perturbation beyond emission from all other sectors so that aviation emissions are assumed to be the only controllable source of emissions. We subsequently refer to our results as marginal impacts.

2. Methods

2.1. Aviation emissions

The marginal impacts of aviation emissions are calculated using emissions inventories obtained from the US Federal Aviation Administration's (FAA) Aviation Environmental Design Tool (AEDT) (Wilkerson *et al* 2010). AEDT provides fuel burn and emission rates for NO_x, hydrocarbons (HC), and primary particulate matter, for individual flight segments in space and time, for all annual commercial civil flights

globally. For the year 2006, the AEDT inventory contains flights with total fuel burn at 188 Tg, which increases to 240 Tg for the year 2015. AEDT has been validated against other aircraft emissions inventories (Olsen *et al* 2013, Simone *et al* 2013) and its results are found to be consistent with other inventories including AERO2K and REACT4C[4]. The AEDT emissions constitute inputs to the air quality and climate modeling approaches presented below.

2.2. Climate impact

Aviation's contribution to climate change is quantified using the Aviation environmental Portfolio Management Tool - Impacts Climate (APMT-IC) (Marais *et al* 2008, Mahashabde *et al* 2011, Wolfe 2012, 2015). APMT-IC computes probabilistic estimates of aviation's climate impacts under multiple economic and policy scenarios, using a quasi-Monte Carlo method with 100 000 members. Additional simulations are performed to quantify the contribution of uncertainty in each variable to overall uncertainty in the output (i.e. contributions to variance) (Saltelli *et al* 2008).

To determine aviation-attributable climate impacts, APMT-IC first calculates the radiative forcing (RF) associated with both CO_2 and non-CO_2 emissions. APMT-IC follows other studies (Tanaka *et al* 2012, 2018, Fuglestvedt *et al* 2014, Ricke and Caldeira 2014, Zhang *et al* 2014, 2016, Lacey *et al* 2017, de Jong *et al* 2018) by using an impulse response function to estimate how CO_2 concentrations will change in response to a change in CO_2 emissions. The impulse response function models the fraction of a CO_2 emissions pulse remaining in the atmosphere as a function of time (Hasselmann *et al* 1997, Fuglestvedt *et al* 2010, Joos *et al* 2013). To capture the sensitivity of these functions to baseline (all-source) CO_2 concentrations (Moss *et al* 2010), the impulse response functions are derived using the Model for the Assessment of Greenhouse-gas Induced Climate Change (MAGICC6) (Meinshausen *et al* 2011)[5]. The resulting aviation CO_2 RF is computed using the radiative transfer function included in the Fifth Assessment Report of the IPCC (Myhre *et al* 1998, 2013). This approach captures the climate-carbon feedbacks for aviation CO_2 emissions, but does not capture the climate-carbon feedbacks of non-CO_2 climate forcers, which likely results in an underestimate of the relative importance of short-lived climate forcers (Gasser *et al* 2017).

RF due to non-CO_2 emissions (sulfates, black carbon (BC), water vapor and NO_x) are calculated by

tracking their respective direct and indirect effects. These include: a direct black carbon warming impact; a direct high-altitude water vapor warming impact; a semi-direct fuel sulfur cooling impact; a multi-scale indirect NO_x impact of mixed sign; and contrail and contrail-cirrus pathways. The short-term indirect NO_x impacts cover the short-term formation of nitrate aerosol (cooling) and production of tropospheric ozone (warming), both of which last less than one year after emission. Furthermore, NO_x increases OH radical concentrations, and thus reduces methane concentrations, which subsequently reduces tropospheric ozone and stratospheric water vapor impacts. These methane-related impacts are cooling, and develop over the perturbation lifetime of methane (~11 years) (Wild *et al* 2001). Contrails, an indirect impact of emitted black carbon and water vapor, form when water vapor condenses on particles under sufficiently cold and humid conditions. Longer-lasting contrails diffuse and take on water vapor from the ambient environment, leading to large, diffuse contrail-cirrus clouds. Although the exact magnitude of the contrail-cirrus impact remains uncertain, it has been quantified as a warming impact comparable to the magnitude of aviation-attributable CO_2 RF (Lee *et al* 2009, Dorbian *et al* 2011, Kärcher 2018).

We base our non-CO_2 RF estimates for these pathways on the results from FAA's Aviation Climate Change Research Initiative (ACCRI) Phase II report (Brasseur *et al* 2016) which compiled RF estimates from multiple research groups using different climate or chemistry-transport models and satellite observations for contrail estimates.

We scale the BC, H_2O, contrails, nitrate, and sulfate aerosol RFs calculated in ACCRI to each of their respective precursor emissions[6]. We estimate the RF associated with short-term and longer-lived ozone and methane perturbations due to NO_x emissions using the absolute global warming potentials (AGWPs) and atmospheric lifetimes for each of these three forcing pathways individually (Wild *et al* 2001, Stevenson *et al* 2004, Hoor *et al* 2009). These indirect NO_x forcing pathways, along with the nitrate aerosol response, cause a net-NO_x RF response resulting from a cancelation of multiple signals at a given time. On net, initially this NO_x RF response is warming, and later switches to cooling.

Other RFs attributable to aviation are not included. RF due to other non-CO_2 aviation emissions, including non-methane volatile organic compounds (NMVOC), carbon monoxide (CO), and organic carbon (OC) have been shown in prior studies to be negligible (Brasseur *et al* 2016). The indirect radiative impacts of aviation emissions on cloud formation are too uncertain to justify inclusion (Lund *et al* 2017a). Similarly, the impact of aviation-attributable BC on snow albedo is not included here, as it remains highly

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[4] The REACT4C emissions inventory includes an annual fuel burn total of 178.3 Tg for 2006 flight operations (Søvde et al 2014). Aero2K, which is included in the EDGAR emissions database and used in the sixth Coupled Model Intercomparison Project (CMIP6), finds an annual fuel burn total of 154 Tg for the year 2002 (Olsen et al 2013, Crippa et al 2018, Hoesly et al 2018).

[5] A detailed discussion of the derivation of the IRFs is presented in section SI.1.2.2 is available online at stacks.iop.org/ERL/14/114031/mmedia.

[6] Contrail impacts are scaled by fuel burn as described in section 2.4.

uncertain for this emissions regime (Fuglestvedt *et al* 2010).

Once RFs have been calculated, APMT-IC converts these to global temperature change using a probabilistic two-box ocean model (Berntsen and Fuglestvedt 2008) in combination with the Roe and Baker (2007) equilibrium climate sensitivity (ECS) distribution. The ECS has a mean of 3.5 °C for a doubling of CO_2 (US Government 2016), which differs less than 4% from the IPCC Coupled Model Intercomparison Project 5 (CMIP5) ECS mean of 3.37 °C and falls within the 1.5 °C–4.5 °C range of the IPCC AR5 (Flato *et al* 2013). Future background global temperature change over time for each RCP scenario is estimated using MAGICC6, under different climate sensitivity assumptions, with results remaining within the temperature distributions of CMIP5 (Collins *et al* 2013).

Finally, APMT-IC uses the calculated global temperature change to estimate the health, welfare, and ecological costs of anthropogenic climate change using (i) the damage function of the Dynamic Integrated Climate Economy (DICE) model (Nordhaus 2017); and (ii) projections of future economic output from the OECD Shared Socio-Economic Pathways (SSP) (Dellink *et al* 2017). To determine the marginal impact of aviation emissions, damages are computed as the difference between damages in a baseline emission scenario, and a scenario where these aviation emissions are included. In this study, marginal speciated aviation climate impacts are derived from a one kilo-tonne pulse of aviation fuel burn occurring in 2015. Future damages from this emissions pulse are discounted using a set of discount rates between 2% and 7%, consistent with widely-used policy guidance (e.g. OMB 2003). To ensure damages are captured for all discount rates, a time horizon of 800 years is used.

A more detailed description of APMT-IC, as well as how impacts are broken down by flight phase, is presented in section SI.1.2.

2.3. Air quality impact

We quantify air quality impacts attributable to a marginal increase in existing emissions in terms of the costs of premature mortalities resulting from population exposure to fine particulate matter ($PM_{2.5}$) and tropospheric ozone (O_3). We use the adjoint of the GEOS-Chem chemistry-transport model (Henze *et al* 2007) to calculate the sensitivity of global costs due to emissions at any location. The GEOS-Chem adjoint has been widely adopted to compute the impacts from (i) combustion emissions in general (Dedoussi and Barrett 2014, Barrett *et al* 2015, Lee *et al* 2015, Turner *et al* 2015); and (ii) aviation emissions on a global and regional level (Gilmore *et al* 2013, Koo *et al* 2013, Ashok *et al* 2014). Impacts are calculated using the sensitivities and AEDT emissions of NO_x, SO_x, HC, CO, BC, and OC for flight operations in 2015. Results are divided by emissions to produce the cost per unit

of mass emitted. Direct air quality impacts of CO_2, contrail-cirrus, and water vapor emissions are considered negligible and not quantified here.

Adjoint simulations are performed on a GEOS-Chem global 4° × 5° model resolution (latitude × longitude) and 47 vertical hybrid sigma-eta pressure levels extending from the surface to 0.01 hPa, resulting in a ~550 m grid height at cruise altitude. The model uses meteorological data from NASA Global Modeling and Assimilation Office, produced using the Goddard Earth Observation System (GEOS-5.2.0) for the year 2009. The EDGAR 4.3.1 and NEI 2011 emissions inventories are used for all anthropogenic sources of non-aviation emissions (US EPA 2015, Crippa *et al* 2016). NO_x emissions from lightning are calculated based on Murray *et al* (2012).

We compute population exposure using the Land-Scan population density product, defined at approximately 1 km (30″ × 30″) spatial resolution globally (Oak Ridge National Laboratory 2015). Premature mortality impacts are estimated for $PM_{2.5}$ and ozone by applying concentration response functions (CRFs) from the epidemiological literature. For $PM_{2.5}$, we estimate changes in cardiovascular disease mortality using the concentration response data from Hoek *et al* (2013). For ozone, we calculate changes in respiratory disease mortality using concentration response data from Jerrett *et al* (2009), consistent with the World Health Organization Global Burden of Disease calculations (GBD 2016 Risk Factors Collaborators 2017). These CRFs are applied for population exceeding 30 years of age and considering the 2015 baseline incidence data from the World Health Organization (WHO 2018).

Finally, following Barrett *et al* (2012), the societal impacts associated with premature mortalities are monetized using a country-specific value of statistical Life (VSL) approach. We conduct income-based country adjustment to the 1990 US VSL (US EPA 2014) by applying an income elasticity of 0.7 (US EPA 2016) on the basis of the Worldbank GDP in PPP per capita for 2015. Using this adjustment, the US VSL in 2015 is USD 10.2 million. An additional set of results are calculated using a global population-weighted average VSL of USD 3.81 million.

As per EPA recommendations (US EPA 2004), we include a cessation lag between time of exposure and mortality. 30% of mortalities are assumed to occur in the first year after emission, 50% are uniformly distributed between 2 and 5 years after emission, and the remaining 20% are uniformly distributed 6–20 years after emission. Future damages are discounted using a set of discount rates between 2% and 7%.

We quantify four sources of uncertainty in monetized air quality impacts using quasi-Monte Carlo simulations with 100 000 members. These uncertainties include uncertainties attributable to (i) atmospheric modeling in GEOS-Chem, (ii) the CRFs,

(iii) VSL estimates in 1990, and (iv) income elasticity of VSL.

Firstly, uncertainty in GEOS-Chem ground-level concentration changes is bounded by comparisons to other models for this regime and to *in situ* measurements. The uncertainty in the response of ground-level ozone concentration to aviation emissions is derived from an inter-model comparison of aviation's impacts on air quality (Cameron *et al* 2017). Due to the large stochastic variability included in the outputs of coupled Climate Response Models (CRMs), we only include the output from the Chemical Transport Models (CTM) and uncoupled runs of the CRM reported in Cameron *et al* (2017). The GEOS-Chem ozone response (0.43 ppbv) differed by less than 5% from the multi-model mean of 0.41 ppbv, while the standard deviation between the model outputs was 20% of the mean value. Using this result as guidance, we add a multiplicative uncertainty to ozone concentration using a triangular distribution with a central value of one and a standard deviation of 0.2. The upper bound of uncertainty (2.0) associated with the changes in $PM_{2.5}$ concentration at ground level is also derived from Cameron *et al* (2017), where the GEOS-Chem average ground-level $PM_{2.5}$ concentration due to aviation emissions is half of that reported by the other two CTM models. The lower bound of uncertainty (0.36) is set by comparisons between *in situ* concentration measurements and GEOS-Chem output for all-source emissions, where studies have found GEOS-Chem overestimates the annual average nitrate $PM_{2.5}$ by up to 2.8 times over most of the US (Heald *et al* 2012, Walker *et al* 2012). Using these two results, we add a multiplicative uncertainty to the $PM_{2.5}$ concentration with a triangular distribution with a minimum value of 0.36, an upper bound of 2.0, and a mean value of 1.0.

Secondly, uncertainty in the concentration response is modeled by applying a triangular distribution to the slope of the CRF, based on the central value and 95% confidence intervals reported in the epidemiological literature (Jerrett *et al* 2009, Hoek *et al* 2013). We do not consider alternative CRFs in our uncertainty bounds. In particular, the CRF from Turner *et al* (2016) could lead to larger ozone-related air quality impacts, because it considers annual average concentrations, and not only summertime concentrations (Jerrett *et al* 2009). Since aviation's impacts on ozone peaks during winter (Eastham and Barrett 2016, Cameron *et al* 2017), this could increase the estimated air quality impacts of aviation.

Thirdly, uncertainty in the 1990 US VSL is modeled using a Weibull distribution, based on the 1990 US EPA estimate (US EPA 2014). Finally, we model uncertainty in income elasticity by applying bounds of 0 and 1.4 on a triangular distribution (Robinson and Hammitt 2015, US EPA 2016). These sources of uncertainty are discussed in detail in section SI.1.3.

We do not quantify the error due to model resolution or uncertainty in relative toxicity of the $PM_{2.5}$ components. Although the $4° \times 5°$ model resolution does not allow us to capture localized emissions peaks in highly populated regions near airports (Barrett *et al* 2010, Arunachalam *et al* 2011, Thompson *et al* 2014, Li *et al* 2016, Fenech *et al* 2018), this is likely to affect only LTO emissions and is difficult to correct for without higher-resolution simulations. Regarding species toxicity we follow EPA practice and assume equal toxicity between the $PM_{2.5}$ species, although there is evidence for BC toxicity to be up to ∼10 times higher than for other $PM_{2.5}$ species (Levy *et al* 2012a, Hoek *et al* 2013).

2.4. Normalization of impacts

Our results are presented on a per mass of emissions basis, in order to facilitate their use in quantifying emissions trade-offs.

Since contrail formation is driven by multiple characteristics of aircraft emissions, no clear normalization approach is evident. Previous literature normalized these impacts by unit of fuel burn or CO_2 emitted (Fuglestvedt *et al* 2010, Dorbian *et al* 2011, Lund *et al* 2017a), or by total flight distance (Fuglestvedt *et al* 2010, Lund *et al* 2017a). However, neither method captures the (i) role of soot; (ii) dependence on the water vapor emissions factor through changes in fuel type; (iii) strong spatial and temporal dependence resulting from relative humidity patterns, cloud cover, and time of day; (iv) increase in contrail formation likelihood with increased engine efficiency; or (v) dependence on size of the aircraft (Paoli and Shariff 2016, Lund *et al* 2017a). Since no other method has been proposed, we present our results using the established normalization methods. This is with the explicit caveat that these results, as well as the other short-lived emission results, are unlikely to apply for emissions patterns dissimilar to the present day, and for contrails in case of significant changes in engine efficiency or technology. A more detailed discussion of the challenges associated with scaling contrail impacts is presented in section SI.1.2.7.

3. Results and discussion

3.1. Global results

Table 1 presents the globally averaged marginal air quality and climate costs of emissions based on global full flight emissions. These values can be used for analyzing the climate and air quality impacts associated with a spatially and temporally homogenous change in global emissions. Costs are in 2015 USD and mass is reported in metric tonnes. The climate and air quality results are presented for a discount rate of 3% and results for discount rates of 2%, 2.5%, 5%, and 7% are provided in the SI. The air quality results are found

Table 1. Global aggregate climate and air quality metrics considering a 3% discount rate [$ per metric tonne emission (2015 USD)]. 5th and 95th percentile results are presented in brackets.

		Climate	Air quality		Total[c]
			Country specific VSL	Globally averaged VSL	
CO_2	[$/tonne CO_2]	45 (6.7–120)	N/A	N/A	45 (6.7, 120)
NO_x[a]	[$/tonne NO_x as NO_2]	−910 (−2500 , −120)	23 000 (3500, 72 000)	22 000 (3400, 71 000)	22 000 (2500, 71 000)
Contrail-Cirrus[b]	[$/tonne Fuel Burn]	82 (10, 230)	N/A	N/A	82 (10, 230)
Contrail-Cirrus[b]	[$/Flight km]	0.23 (0.028, 0.64)	N/A	N/A	0.23 (0.028, 0.64)
Fuel Sulfur	[$/tonne S]	−18 000 (−47 000, −2400)	30 000 (4700, 100 000)	31 000 (4800, 110 000)	13 000 (−25 000, 88 000)
BC	[$/tonne BC]	47 000 (6800, 130 000)	14 000 (1800, 44 000)	12 000 (1600, 41 000)	61 000 (17 000, 150 000)
H_2O	[$/tonne H_2O]	2.5 (0.36, 6.7)	N/A	N/A	2.5 (0.36, 6.7)
NMVOC	[$/tonne NMVOC]	N/A	7700 (1100, 21 000)	5200 (830, 17 000)	7700 (1100, 21 000)
CO	[$/tonne CO]	N/A	290 (43, 860)	230 (36, 770)	290 (43, 860)
OC	[$/tonne OC]	N/A	11 000 (1500, 37 000)	9800 (1400, 34 000)	11 000 (1500, 37 000)
Total Cost	[$/tonne Fuel Burn]	200 (30–530)	360 (56–1200)	350 (55–1100)	560 (180–1400)

Note. Climate impacts of NMVOC, CO, and OC were not quantified. Similarly, air quality impacts of CO_2, contrail-cirrus, and H_2O were not quantified. These are indicated as not available (N/A) in the table.

[a] Net-NO_x climate results are calculated as the sum of the impact of four indirect NO_x pathways. These individual pathways are tabulated in table SI.13.

[b] Note that either normalization (per tonne of fuel burn or per flight km) must be used exclusively. For engine technology and fuel properties sufficiently different from current patterns, contrail-cirrus impact scaling remains highly uncertain (see section 2.4).

[c] Total calculated for Country Specific VSL.

to change by $+3\%$ to -11% for discount rates of 2% and 7%, respectively.

Table 2 presents cost metrics broken down by flight phase for a 3% discount rate. Air quality results are presented for country specific VSL with metrics for globally averaged VSL presented in the SI. The results indicate that the largest climate impacts per unit of emission occur in cruise, most likely due to increased atmospheric residence time of emissions at altitude. In contrast, the largest air quality impacts per unit of emission for each species are identified during the LTO phase, due to the co-location of airports and population centers. However, because $\sim$90% of fuel burn occurs in cruise, cruise emissions still dominate the air quality impacts (table 3).

The (near-ground) social cost of emission results from Shindell (2015) fall within the 5th–95th percentile uncertainty bounds of the LTO results presented here, with the exception of LTO NO_x climate results. Shindell's (2015) climate NO_x results are between ten to twenty times smaller than our estimate. This difference has a small impact on the overall results since the climate NO_x impact is at least two orders of magnitude less than the air quality NO_x impact. The difference is likely due to the cancelling of warming and cooling NO_x radiative pathways (see section 2.2), leading to small net NO_x climate costs, and subsequently large percentage differences of net radiative impacts between different sources (Fuglestvedt *et al* 2010, Myhre *et al* 2013). Results from Shindell (2015) are tabulated and discussed further in section SI.2.2.2.

We further compare our full-flight climate results to Dorbian *et al* (2011), and find that for all forcers, uncertainty bounds between the two studies overlap, with their central estimate for cirrus and total fuel burn metrics falling within our uncertainty bounds (see SI.2.3). However, the absolute value of their NO_x, sulfur, BC, and stratospheric water vapor results exceed our uncertainty bounds. This can be attributed to updated RF assumptions for the short-lived climate forcers and the inclusion of a nitrate cooling pathway as a NO_x-related impact (Brasseur *et al* 2016) (see SI.2.3).

The reduced-order climate metrics presented here are calculated for 2015 background atmospheric composition and surface temperature. Under the RCP 4.5 and SSP 1 scenarios, future background temperature change and global GDP are both projected to increase, leading to increased marginal damages in the nonlinear DICE climate damage function. Therefore, when used for future emission years, the climate cost estimates increase by 2% per year for CO_2 which has a long lifetime, and by 4.7% per year for short-lived forcers (assuming a 3% discount rate). A full overview of these 'adjustment rates' for future emission years is presented in the SI. Similarly, we expect the VSL to increase by 2.5% per year, assuming an income elasticity of 0.7 and average year on year growth in GDP as in the SSP 1 scenario (Dellink *et al* 2017).

The air quality costs presented in table 1 are presented for both country-specific and globally averaged VSL while results in table 2 are derived based on country-specific VSL values only. When uniformly applying the global average VSL value, we find less than 10% difference for the cruise impacts, whereas the estimates for the LTO phase decrease by 30% to 50% (see tables SI.14 and SI.15). This difference between LTO and cruise is likely due to the more localized nature of LTO emissions and their impacts (Yim *et al* 2015).

For quickly analyzing scenarios in which fuel burn totals change but emissions composition and distribution remain approximately constant (e.g. operational improvements, sector growth, market-based measures reducing aviation operations), we present the climate and air quality cost per unit of fuel burn. Following Dedoussi *et al* (2019) we refer to these costs as the Climate and Air Quality Social Cost (CAQSC) per unit of fuel burn. These are calculated from the speciated cost metrics presented above.

Table 3 presents CAQSC for each flight phase, while figure 1 presents the breakdown of full flight CAQSC by flight phase. The results indicate that $\sim$90% of the CAQSC results from the cruise emissions. NO_x, CO_2, and contrails are collectively responsible for 58%, 25%, and 14% of the overall cost, respectively, totaling 97%. Air quality impacts account for 64% of total impacts, which is highly sensitive to the discount rate given the long-term nature of climate impacts as compared to the short time scale for air quality impacts (driven by 20 year cessation lag). As such, a 2% discount rate reduces the contribution of air quality impacts to 50%, and a discount rate of 7% increases the contribution of air quality impacts to 80%. Furthermore, 63% of the air quality portion of the full flight CAQSC is caused by the $PM_{2.5}$ impact pathway with the remainder caused by the ozone pathways. This result is consistent with Eastham and Barrett (2016) who found that 58% of the premature mortalities attributable to aviation are due to $PM_{2.5}$ exposure, with the remainder from ozone.

Both the speciated costs and the CAQSC are derived using a marginal impacts assessment (see section 1). Due to nonlinearities in climate and air quality responses, the marginal costs differ from the average cost of a unit of emission. The latter would be derived by apportioning the global all-sector damages to the emissions in question and would be used for determining aviation's fractional contribution to global anthropogenic damages. As discussed in section SI.2.5, the marginal costs of the aviation-attributable impacts are approximately double the average costs.

3.2. Results for regional emissions

The global metrics presented in section 3.1. do not capture regional differences in the climate and air quality sensitivities to a unit of aviation emissions. In turn, the results can only be used to analyze homogenous global

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Table 2. Cost metrics broken down by flight phase. \$/tonne of LTO and cruise emission (2015 USD). 5th and 95th percentile results are presented in brackets.

		Landing and take-off		Cruise	
		Climate	Air quality	Climate	Air quality
CO_2	[\$/tonne CO_2]	45 (6.7, 120)	N/A	45 (6.7, 120)	N/A
NO_x	[\$/tonne NO_x as NO_2]	−590 (−1600, −81)	37 000 (5200, 110 000)	−940 (−2600, −120)	21 000 (3300, 69 000)
Contrail-Cirrus[a]	[\$/tonne Fuel Burn]	N/A	N/A	92 (11, 260)	N/A
Fuel sulfur	[\$/tonne S]	−2600 (−7000, −360)	32 000 (4300, 99 000)	−20 000 (−53 000, −2700)	30 000 (4700, 100 000)
BC	[\$/tonne BC]	18 000 (2000, 52 000)	66 000 (8400, 200 000)	52 000 (7500, 140 000)	7000 (980, 25 000)
H_2O	[\$/tonne H_2O]	N/A	N/A	2.8 (0.41, 7.5)	N/A
NMVOC	[\$/tonne NMVOC]	N/A	19 000 (2700, 52 000)	N/A	2300 (360, 7300)
CO	[\$/tonne CO]	N/A	520 (76, 1500)	N/A	200 (31, 630)
OC	[\$/tonne OC]	N/A	110 000 (13 000, 310 000)	N/A	7000 (980, 25 000)
Total Cost	[\$/tonne Fuel Burn]	130 (20, 340)	590 (84, 1700)	210 (31, 560)	340 (53, 1100)

Note. Results are for a 3% discount rate. For air quality, only results derived using a country specific VSL are presented here.

Climate impacts of NMVOC, CO, and OC were not quantified. Similarly, air quality impacts of CO_2, contrail-cirrus, and H_2O were not quantified. These are indicated as not available (N/A) in the table.

[a] For engine technology and fuel properties sufficiently different from current patterns, contrail-cirrus impact scaling remains highly uncertain. See section 2.4 for details.
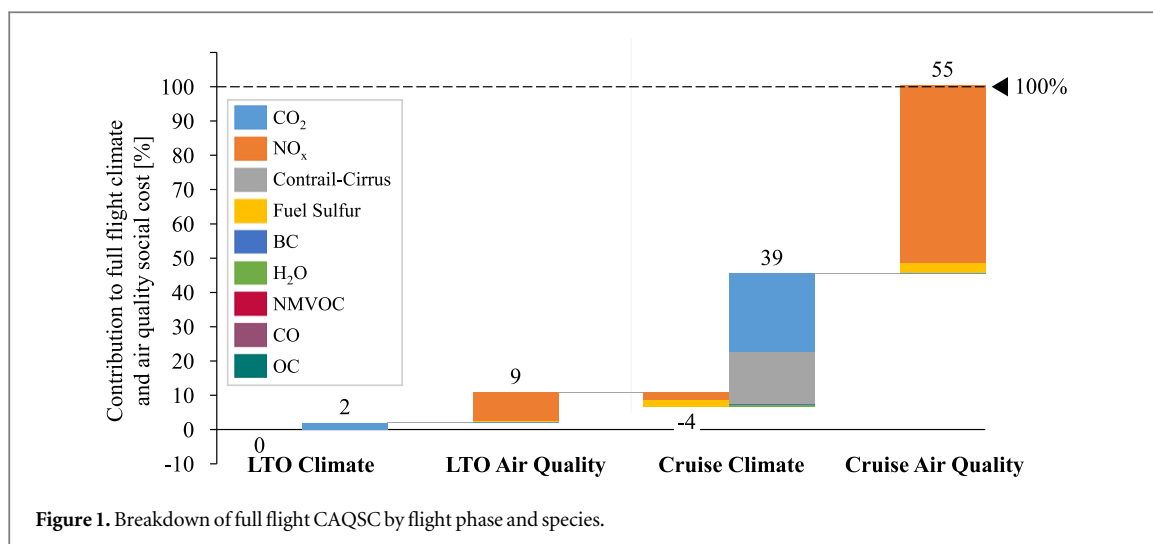
Figure 1. Breakdown of full flight CAQSC by flight phase and species.

Table 3. CAQSC of aviation fuel burn. \$/tonne of fuel burn in flight phase (2015 USD). 5th and 95th percentile results are presented in brackets.

	Landing and take-off	Cruise	Full flight		
			Climate	Air quality	Total
CO_2	140 (21, 370)	140 (21, 360)	140 (21, 360)	N/A	140 (21, 360)
NO_x	550 (70, 1600)	300 (35, 1000)	−14 (−37, −1.8)	340 (52, 1100)	330 (38, 1100)
Contrail-Cirrus[a]	0	92 (11, 260)	82 (10, 230)	N/A	82 (10, 230)
Fuel sulfur	18 (0.92, 58)	6.3 (−18, 52)	−11 (−28, −1.4)	18 (2.8, 62)	7.6 (−15, 53)
BC	4.1 (0.98, 11)	2.1 (0.48, 5.2)	1.7 (0.25l, 4.5)	0.5 (0.067, 1.6)	2.2 (0.6, 5.3)
H_2O	0	3.5 (0.51, 9.3)	3.1 (0.45, 8.2)	N/A	3.1 (0.45, 8.2)
NMVOC	11 (1.6, 31)	0.27 (0.042, 0.85)	N/A	1.2 (0.17, 3.4)	1.2 (0.17, 3.4)
CO	4.3 (0.62, 12)	0.39 (0.06, 1.2)	N/A	0.72 (0.11, 2.1)	0.72 (0.11, 2.1)
OC	0.77 (0.098, 2.2)	0.11 (0.015, 0.37)	N/A	0.16 (0.022, 0.53)	0.16 (0.022, 0.53)
Total Cost	730 (180, 1900)	550 (170, 1400)	200 (30, 530)	360 (56, 1200)	560 (180, 1400)

Note. Results are for a 3% discount rate. Results for alternative discount rates can be found in SI.2.2.2. The air quality results are based on a country specific VSL.

Climate impacts of NMVOC, CO, and OC were not quantified. Similarly, air quality impacts of CO_2, contrail-cirrus, and H_2O were not quantified. These are indicated as not available (N/A) in the table.

[a] For engine technology and fuel properties sufficiently different from current patterns, contrail-cirrus impact scaling remains highly uncertain. See section 2.4 for details.

trends or policies. Regionalized air quality metrics, which quantify global damages due to homogenous changes in emissions in a region, can be used to analyze future aviation scenarios with shifting geographical traffic distributions, policy interventions in selected regions, or heterogeneous adoption of new technologies across the globe[7]. Regionalized climate metrics are not presented. Even though regionalized physical impact metrics exist (Lund *et al* 2017a), very high uncertainty remains regarding the quantification of regionalized damages (Nordhaus 2017).

Figure 2 presents air quality results for regionalized full flight emissions, and table 4 shows results for regionalized emissions metrics by flight phase.

Values for figure 2, alternative results using a globally averaged VSL, and a comparison to results from the literature can be found in SI.2.2.

The results in figure 2 and table 4 show that the highest cost per unit of emissions is for emissions over Europe. For the cruise flight phase, this remains true regardless of whether global or country-specific VSL is used (table SI.17) which likely points to the transport of cruise emissions and their chemical products by prevailing westerly winds from Europe to the populous Asia-Pacific region. For LTO emissions, the magnitude of the impacts varies significantly with the VSL assumption (table SI.18), with costs decreasing by a factor of two in Europe, North America, and the US under a globally averaged VSL assumption. This is because the costs of LTO emissions are more localized and therefore driven by local characteristics such as local VSL and population density.
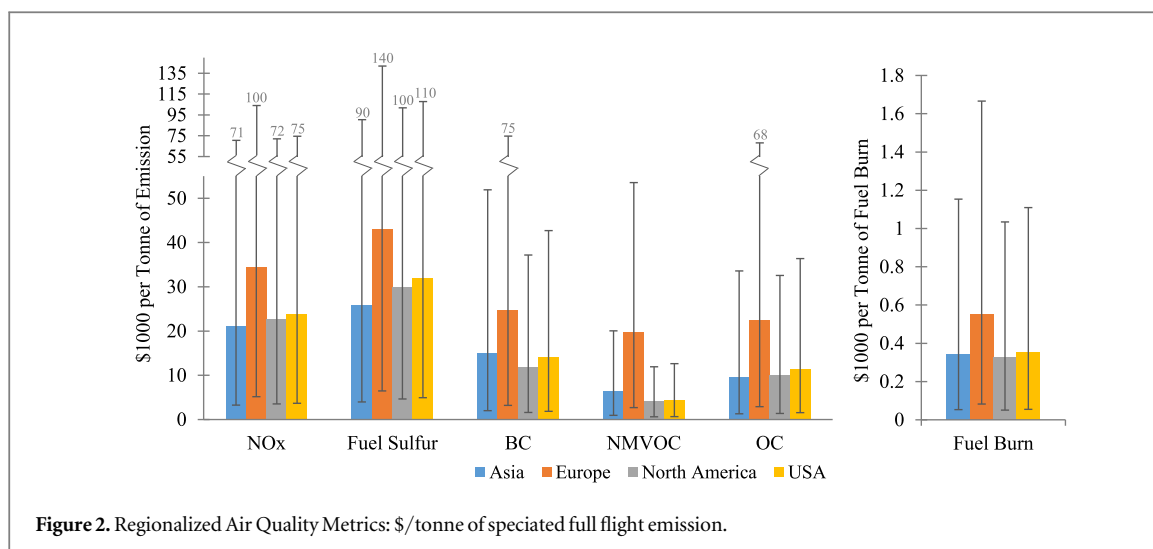
[7] Since the impacts are presented averaged over the region, the metrics must still be used with caution when evaluating highly localized (in space or time) trends (e.g. an individual route).

Figure 2. Regionalized Air Quality Metrics: $/tonne of speciated full flight emission.

Table 4. Regionalized LTO and cruise air quality cost metrics.

		LTO				Cruise			
		Asia-Pacific	Europe	North America	USA	Asia-Pacific	Europe	North America	USA
NO_x	[$/tonne NO_x as NO_2]	44 000	67 000	18 000	20 000	19 000	31 000	23 000	24 000
Sulfates	[$/tonne S]	37 000	52 000	20 000	24 000	25 000	42 000	31 000	33 000
BC	[$/tonne BC]	83 000	120 000	41 000	50 000	5700	11 000	7200	7400
NMVOC	[$/tonne NMVOC]	18 000	56 000	7400	8000	2000	3200	2400	2400
CO	[$/tonne CO]	480	1100	380	400	180	270	220	220
OC	[$/tonne OC]	110 000	190 000	60 000	76 000	5700	11 000	7200	7400
Total	[$/tonne Fuel Burn]	720	1100	280	320	310	480	330	360

Note. Uncertainty and results for globally averaged VSL can be found in tables SI.17 and SI.18.

The region represents emissions region and does not necessarily correspond to where the impact occurs.

3.3. Model sensitivity and uncertainty

The sensitivity of the climate and air quality metrics to each uncertain parameter considered in this study (see section SI.1.2 and SI.1.3) is estimated by deriving total-effect indices. These indices represent the fraction of the total output uncertainty attributable to an uncertain input variable though both its direct (i.e. first-order) contribution to output variance, as well as the higher-order effects due to its interactions with other variables (Saltelli *et al* 2008).

For the climate impacts, we find the uncertainty associated with the equilibrium climate sensitivity and the climate damage function to be the largest contributors to overall outcome uncertainty, with total effect indices of 0.64 and 0.45, respectively. For the air quality impacts, we find the uncertainty associated with the VSL_{1990} to be the largest contributor to uncertainty, with a total-effect index of 0.85. Uncertainty in the GEOS-Chem $PM_{2.5}$ concentration and income elasticity have total-effect indices of 0.12 and 0.13, respectively, while other uncertainties each have indices of 0.07 or less.

Because higher-order effects are included in the total effect indices for each variable, the sum of all the total effect indices may exceed one. For the climate model, the sum over all the total-effect indices was 1.16, while for air quality the sum was 1.15, indicating significant effect interaction. An additional discussion on uncertainty, Monte Carlo convergence, as well as sensitivity to RCP and SSP scenarios is presented in the SI. The Monte Carlo datasets are also available as described in the Data Availability Statement.

3.4. Analysis of aviation growth and mitigation scenarios

The results presented in previous sections can be applied to support decision making about policies, operational procedures, and technologies in the aviation sector. Here we present analyzes of the climate and air quality impacts of global air traffic growth as well as three approaches which could reduce these impacts. These approaches include (i) fleet improvements; (ii) NO_x stringencies with a CO_2 trade-off; and (iii) fuel desulfurization.

3.4.1. Uniform emissions growth of 4.7%

Aviation passenger traffic is projected to grow at 4.7% per year (ICAO 2016b). If emissions grew at the same rate throughout all markets and if there were no improvements in fuel efficiency throughout the global fleet, annual fuel burn would have increased by 11×10^6 tonnes from 2015 to 2016. Further assuming no changes in emission composition, this fuel burn increase would have been attributable to total climate and air quality costs at 6400 million USD (90% confidence interval 2000–16 000).

3.4.2. Growth of 4.7% with improved aircraft entering fleet

Over the history of aviation, new aircraft with system level design improvements have continuously entered the fleet. Assuming (i) new aircraft are delivered to match growth; (ii) these aircraft have NO_x emissions indices 45% below the current fleet average, to meet ICAO targets[8]; and (iii) these aircraft also have a 10% reduction in CO_2 intensity per seat mile below fleet average[9], we could have reduced the additional costs of growth from 2015 to 2016 by 1800 million USD (90% confidence interval 360–5760) to 4600 million USD. In turn, the climate and air quality costs of a year's aviation traffic growth could be offset by replacing ~12% of the fleet average with new aircraft. This percentage could be lower if the oldest, most polluting aircraft are replaced first and if additional climate and air quality impacts of reductions in other emissions species were considered.

3.4.3. NO_x stringencies

Engine level NO_x stringencies have been considered as a means to achieve NO_x reduction targets (ICAO 2016b). However, the benefit of such stringencies is limited in part by a fundamental NO_x–CO_2 trade-off. Higher combustor temperatures result in increased thermodynamic efficiency, thereby reducing fuel burn. However, higher combustor temperatures also lead to increased NO_x formation, as described by the Zeldovich mechanism (Kundu *et al* 1998).

Freeman *et al* (2018) studied the optimal climate policy of this trade-off, and assumed a baseline case where a 20% reduction of aviation NO_x leads to a 2% increase in CO_2 emissions. We build on this work and calculate the climate and air quality impacts of introducing new aircraft into the fleet that achieve these goals relative to the fleet average, neglecting any other changes in emissions which might be associated. Again, we assume new aircraft to be delivered to match growth homogenously throughout all markets. We find the costs associated with the CO_2 increase to be 32 million USD (90% confidence range 4.8–82), and the

benefit of the NO_x decrease to be 730 million USD (90% confidence range 80–2500). This yields a net benefit of 700 million USD (90% confidence range 58–2500) during the first year of operation of the new aircraft. While this result suggests that stricter NO_x regulation is likely net-beneficial from a climate and air quality perspective, a full cost benefit analysis needs to account for additional issues such as feasibility constraints, development costs and the increase in fuel requirements.

3.4.4. Ultra-low fuel sulfur

Another approach for reducing emissions from the aviation sector is sulfur removal from jet fuel (Barrett *et al* 2012). Sulfur, an element naturally occurring in fossil fuels, is currently estimated to be present in jet fuel at a concentration of 550–750 ppm, which exceeds the 15 ppm of standard US highway diesel.

Reducing the fuel sulfur content from the assumed average 600 ppm to 15 ppm would reduce fuel sulfur emissions of one tonne fuel by 0.585 kg. At the same time, Barrett *et al* (2012) estimated fuel sulfur removal to result in a 2% increase in the life cycle CO_2-equivalent (CO_2e) emissions, predominantly due to decreased refinery efficiencies. We find CO_2e emissions increase by 75 kg CO_2e/tonne fuel by assuming a specific energy of 43 MJ kg^{-1}. In sum, we find fuel desulfurization to result in a combined climate and air quality benefit of 10.8 USD/tonne fuel burn (90% confidence interval −11 to 55). Assuming this low fuel sulfur is used in all flights, the total annual benefit would be 1,000 million USD (90% confidence interval −4300 to 12 000). However, the uncertainty bounds show that desulfurization could also lead to a disbenefit, due to the loss of climate benefits of cruise level sulfur emissions.

Given the localized air quality impacts of sulfur emissions during LTO, one can expect the benefits of jet fuel desulfurization to be particularly high for the LTO cycle in Europe (section 3.2). If it were possible to remove sulfur only from fuel used during LTO in Europe, the combined climate and air quality benefits would be 26 USD/tonne fuel burn (90% confidence interval −1.2–82). This benefit is more than double the sulfur removal benefit per unit of global full flight fuel burn, suggesting it could be beneficial to use low sulfur fuel on short flights in the European region. However, a full cost benefit analysis would be required to analyze the net societal benefit of this approach (e.g. Barrett *et al* 2012).

4. Summary and conclusions

This work presents a method for comparing the climate and air quality impacts of aviation emissions, by estimating the social costs per unit of emitted mass by species. The cost metrics are broken down by flight phase and by the region of emission, both per tonne of

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[8] Consistent with the ICAO target set for NO_x reductions for 2016 in production aircraft when compared to 2006 (ICAO 2007).
[9] Consistent with a ~1% per year improvement in CO_2 intensity per seat mile over 10 years (Lee et al 2009).

emission and per tonne of fuel burn (the CAQSC metric).

4.1. Limitations of current approach and future research needs

Our results are applicable for the assessment of marginal changes in aviation emissions inventories, such as short- or medium-term changes in air traffic, or advancements in aircraft technology or operations. However, the results presented here are not applicable to evaluate all emissions scenarios, and exclude the impact of some uncertain factors.

Firstly, the results presented would not be applicable for evaluating certain emission scenarios. These scenarios include highly localized emissions changes, e.g. resulting from in-flight altitude or changes in flight tracks. Additionally, our results are not applicable to evaluate changes in contrail impacts due to changes in engine or fuel technologies. To capture contrail impacts over a wider range of emissions scenarios, development of a more representative scaling method for contrail impacts would be necessary. Given the large impact of contrails (14% of impacts, see table 3), this remains a major research need.

Secondly, our results do not consider the impacts associated with some uncertain physical modeling aspects. Our climate results exclude the impact of climate-carbon feedbacks, the impact of differing temperature responses due to different climate forcers, and the impact of aerosol-cloud interactions. In particular, aerosol-cloud interactions could have a large impact on results, but the scientific literature has yet to agree on the sign of this impact (Lund *et al* 2017a). Moreover, the impacts due to BC are likely underestimated in this work, resulting from (i) the exclusion of BC radiative impact on albedo changes (section 2.2), (ii) not accounting for differential toxicity in air quality impacts, and (iii) the use of a large modeling grid (section 2.3). Further advances in epidemiological, atmospheric modeling research, and computational efficiency are necessary to include these effects.

Third, derived uncertainty bounds for the cost metrics remain large, ranging from 10% to 200% of the mean cost values. Only two physical modeling factors, equilibrium climate sensitivity and contrail RF, contribute significantly to this uncertainty, while monetization of impacts induces significant uncertainty for both the climate and the air quality results. For air quality, this uncertainty is largely associated with the value of statistical life, while for the climate model, the uncertainty results from the damage function. For this study, we apply the DICE damage function and its uncertainty, which is derived from 26 underlying studies (Nordhaus 2017). However, we note an even larger range of values has been reported in literature, with central social cost of carbon estimates ranging from 36 [2007] USD to 417 USD/tonne (US Government 2016, Ricke *et al* 2018, Pindyck 2019). This suggests

further research into these valuation methods is necessary to further reduce uncertainties.

Finally, different economic valuation approaches can have significant impacts on our results. For instance, some regulators use the Value of Life Years (VOLY) lost instead of VSL to quantify the costs associated with air pollution. Since air quality damages disproportionally affect an older segment of population, a VOLY approach will likely lead to lower air quality impacts. For example, Tollefsen *et al* (2009) find the air quality damages of a VOLY approach to be 64%–68% of the VSL impacts.

4.2. Research application

Using the stated assumptions, our results indicate that three components are responsible for 97% of climate and air quality damages per unit fuel burn, with individual contributions of NO_x at 58%, CO_2 at 25%, and contrails at 14%. These species can subsequently be seen as primary targets for future strategies to reduce the atmospheric impacts of aviation emissions.

To reduce the climate impact of aviation, measures aimed at reducing CO_2 emissions and contrails are expected to lead to the greatest net climate benefit. In contrast, we find 94% of air quality impacts (which are 64% of total impacts) to be driven by NO_x. This suggests that measures aimed at reducing NO_x emissions could lead to the greatest net benefits, even if such measures lead to a small but uncertain climate NO_x disbenefit and small increase in CO_2 emissions.

Finally, we find that the air quality impacts of aviation emissions significantly exceed the climate impacts, with air quality impacts being between 1.7 times (full flight) and 4.4 times (LTO) higher than the climate impact per unit of fuel burn. This finding must be contrasted to ground-based industries, where post-combustion emissions control and access to cleaner fuels is wide-spread. For example, Dedoussi *et al* (2019) find the climate and air quality impacts of the US power sector to be of similar magnitude following significant declines in co-pollutant emissions over the past 15 years. This points towards potential political and technological opportunities for reducing the atmospheric impacts of the aviation sector.

Data availability statement

The Monte Carlo uncertainty datasets are openly available (Figshare ID: ERL-106869.R2). Other data that support the findings of this study are available from the corresponding author upon reasonable request. The data are not publicly available and some data cannot be shared for legal and/or ethical reasons.

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