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2015 Environ. Res. Lett. 10 034001

(http://iopscience.iop.org/1748-9326/10/3/034001)

Environmental Research Letters

LETTER

Global, regional and local health impacts of civil aviation emissions

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Keywords: aviation, air quality, public health

Supplementary material for this article is available online

OPEN ACCESS

RECEIVED
26 November 2014

REVISED
16 January 2015

ACCEPTED FOR PUBLICATION
16 January 2015

PUBLISHED
26 February 2015

Abstract

Aviation emissions impact surface air quality at multiple scales—from near-airport pollution peaks associated with airport landing and take off (LTO) emissions, to intercontinental pollution attributable to aircraft cruise emissions. Previous studies have quantified aviation's air quality impacts around a specific airport, in a specific region, or at the global scale. However, no study has assessed the air quality and human health impacts of aviation, capturing effects on all aforementioned scales. This study uses a multi-scale modeling approach to quantify and monetize the air quality impact of civil aviation emissions, approximating effects of aircraft plume dynamics-related local dispersion (~1 km), near-airport dispersion (~10 km), regional (~1000 km) and global (~10 000 km) scale chemistry and transport. We use concentration-response functions to estimate premature deaths due to population exposure to aviation-attributable $PM_{2.5}$ and ozone, finding that aviation emissions cause ~16 000 (90% CI: 8300–24 000) premature deaths per year. Of these, LTO emissions contribute a quarter. Our estimate shows that premature deaths due to long-term exposure to aviation-attributable $PM_{2.5}$ and O_3 lead to costs of ~$21 bn per year. We compare these costs to other societal costs of aviation and find that they are on the same order of magnitude as global aviation-attributable climate costs, and one order of magnitude larger than aviation-attributable accident and noise costs.

1. Introduction

The International Civil Aviation Organization (ICAO) estimates that the number of air passengers will more than double in two decades, from 3.0 billion in 2012 to 6.4 billion in 2030 (ICAO 2013). Aviation emissions cause an increase in the concentration of pollutants including fine particulate matter ($PM_{2.5}$) and ozone (O_3). Epidemiological studies have reported that exposure to $PM_{2.5}$ and O_3 is associated with an increase in risk of premature death (Pope III *et al* 2002, Ostro 2004, Laden *et al* 2006, Pope III *et al* 2006, US EPA 2006, Lewtas 2007). Aviation emissions impact local, regional and global air quality (Segal and Yamartino 1981, Yu *et al* 2004, Unal *et al* 2005, Carslaw *et al* 2006, Farias and ApSimon 2006, Peace *et al* 2006, Schürmann *et al* 2007, Westerdahl *et al* 2008, Carslaw and Taylor 2009, Dodson *et al* 2009, Hu *et al* 2009, Barrett *et al* 2010, Arunachalam *et al* 2011, Woody *et al* 2011, Zhu *et al* 2011, Carslaw *et al* 2012, Diez *et al* 2012, Hsu *et al* 2012, Carslaw and Beevers 2013, Hsu *et al* 2013, Lee *et al* 2013, Rissman *et al* 2013, Yim *et al* 2013, Wolfe *et al* 2014). Previous studies have partially quantified aviation's air quality and resulting health impacts from $PM_{2.5}$ or O_3 formation around specific airports, for a specific region, or on a global level, respectively. However no attempt has previously been made to quantify the global health burden associated with aviation emissions, accounting for near-airport, regional and global-scale effects.

$PM_{2.5}$ has been linked to increased rates of lung cancer as well as both cardiovascular and respiratory (cardiopulmonary) disease (Pope III *et al* 2002, Laden *et al* 2006, Pope III *et al* 2006). Epidemiological cohort studies such as the Harvard Six Cities and American Cancer Society studies have demonstrated a statistical link between $PM_{2.5}$ exposure and mortality, while clinical and laboratory studies have explored the physiological and molecular mechanisms that might be involved. A review by the American Heart Association

found that air pollutants are linked to a variety of physiological responses which increase the likelihood of fatal cardiovascular or respiratory incidents (Brook *et al* 2010). A follow up to the American Cancer Society cohort study found that O_3, which has been shown in animal laboratory studies to cause oxidative damage when inhaled, is also linked to respiratory disease (Jerrett *et al* 2009).

Barrett *et al* (2010) applied GEOS-Chem (at a global scale) to estimate the concentration of $PM_{2.5}$ due to global aviation emissions. They reported that global aircraft emissions cause ~10 000 premature deaths per year globally, with 80% due to cruise emissions. Lee *et al* (2013) applied CAM-chem to study the impact of aviation emissions on atmospheric O_3, NO_y and $PM_{2.5}$ concentrations confirming the dominant role of cruise emissions in aviation-attributable surface air quality impacts. Woody *et al* (2011) quantified aviation-attributable fine particulate matter emissions from landing and takeoff (LTO) operations (i.e. below 3000 ft above ground level) in the United States at 99 US airports in 2005 and in 2025. Using results from Woody *et al* (2011) and Levy *et al* (2012a) calculated the resulting human health impact from these LTO emissions to be 75 premature deaths in 2005 and 460 in 2025.

A number of studies have assessed aviation sector contributions to near-airport (<10 km) air quality degradation. Field and data analyses have demonstrated a correlation between pollutant concentration and aircraft activity at airports in the US (Segal and Yamartino 1981, Westerdahl *et al* 2008, Dodson *et al* 2009, Hu *et al* 2009, Hsu *et al* 2012, Hsu *et al* 2013, Zhu *et al* 2011), in Europe (Carslaw *et al* 2006, Schürmann *et al* 2007, Carslaw and Beevers 2013) and in Asia (Yu *et al* 2004). Local air quality modeling approaches were also applied to quantify the near-field impact of airport emissions (Farias and ApSimon 2006, Peace *et al* 2006, Carslaw and Taylor 2009, Carslaw *et al* 2012). These studies only investigated a limited number of airports and typically focused on primary pollutants (directly emitted from sources), but not secondary pollutants (formed through chemical reactions). One exception in terms of pollutants considered is Arunachalam *et al* (2011), who used the Community Multiscale Air Quality model (CMAQ) to estimate the population exposure to both primary and secondary $PM_{2.5}$ concentrations at three US airports, applying three grid resolutions of 36 km, 12 km and 4 km. Rissman *et al* (2013) used a modified version of CMAQ with a plume-in-grid model to estimate the $PM_{2.5}$ concentration due to emissions at Hartsfield–Jackson Atlanta International Airport. However, Yim *et al* (2013) show that the variation of air quality impacts around airports is finer than the resolutions applied in the two aforementioned studies. To capture the variation of local impacts associated with aviation emissions, Yim *et al* (2013) applied CMAQ and a local dispersion model to quantify the air quality and health impacts due to the LTO emissions of 20 major airports in the United Kingdom. By merging both regional and local air quality models results, Yim *et al* (2013) estimated that the current UK aviation emissions cause ~110 premature deaths per year. Yim *et al* (2013) found that accounting for local scale dispersion at the sub-grid scale increases estimated PM exposure by 25–31%, but this increase was halved when accounting for plume dynamics (Barrett *et al* 2013). Kim *et al* (2012) employed a hybrid modeling approach using CMAQ and the AERMOD dispersion model to combine the spatially-diffuse secondary $PM_{2.5}$ impacts with localized impacts of primary $PM_{2.5}$ pollutants from the Washington Dulles airport. (See section 2.6 of Kim *et al* 2012 for further airport-specific studies). Another important factor is the level of background ammonia. As noted by Barrett *et al* (2010), the majority of aviation-attributable $PM_{2.5}$ at surface level is secondary inorganic $PM_{2.5}$ formed from neutralization of NH_4^+ with either SO_4^{2-} or NO_3^-. Broadly, this is limited by either the available ammonia, from which NH_4^+ is formed, or the total available sulfate and nitrate ions. High levels of background ammonia therefore result in production of $PM_{2.5}$ in the presence of either SO_x or NO_x aerosol precursors.

Most existing studies focused on the impact of aviation on PM, while only a limited number of studies investigated the impact of aviation emissions on surface ozone. Unal *et al* (2005) applied CMAQ to simulate $PM_{2.5}$ and ozone formation due to the emissions of Atlanta Hartsfield–Jackson International Airport at a ten-day period. Köhler *et al* (2013) and Skowron *et al* (2013) investigated long-term ozone concentration due to aviation emissions, but only in relation to climate impacts.

In this paper, we estimate the concentration of both $PM_{2.5}$ and ozone attributable to aviation emissions, by approximating effects of aircraft plume dynamics-related local dispersion (~1 km), near-airport dispersion (~10 km), and regional (~1000 km) and global (~10 000 km) scale chemistry and transport. In this way we capture the impact of cruise emissions, which take effect on a global to regional scale, and the impact of LTO emissions, which have an impact on a local to regional scale. We estimate the resulting health impact in terms of premature mortalities, monetize this impact and quantify uncertainties. To inform understanding of the relative importance of the health impact of aviation-attributable $PM_{2.5}$ and O_3 concentrations, we compare the health costs to other societal costs of aviation. In particular, we consider (i) climate costs, which result from aviation combustion emissions and measure global welfare losses caused by aviation-induced increases in global surface temperature (e.g. higher flooding risks or lower agricultural productivity), (ii) accident costs in terms of the economic value, which is assigned to injuries and mortalities in aviation accidents, and (iii) noise costs as derived from aviation-related losses in property values. This is the first study to assess the

global health impacts of aviation including effects at a near-airport to global scale, and the first to show that the human health costs of aviation are comparable to its climate costs.

2. Methods

We apply a multi-scale approach to resolve the variation of aviation-attributable $PM_{2.5}$ and ozone at different spatial scales. Global and regional air quality impacts are estimated using chemistry-transport models GEOS-Chem and CMAQ, with aviation-attributable $PM_{2.5}$ and ozone computed as the difference between simulations with all emissions and simulations where only non-aviation emissions are included. Airport vicinity impacts of $PM_{2.5}$ (but not ozone, which is regional in nature) are calculated by merging the results of both local dispersion and regional chemistry-transport models. Premature deaths due to long-term exposure to $PM_{2.5}$ and ozone attributable to aviation emissions are calculated using population density data to compute exposure, and then mapping exposure to risk of early death using concentration-response functions (CRFs). Country-specific values of statistical life (VSL) are calculated to monetize the resultant premature deaths. We use a Monte–Carlo approach to quantify the uncertainties in our calculations.

2.1. Aviation emissions

Aviation emissions for 2006 are from AEDT (Wilkerson *et al* 2010) and include civil aviation emissions of NO_x, hydrocarbons, and fuel burn. Emissions of SO_x, black carbon (BC) and organic carbon (OC) are scaled from fuel burn per Barrett *et al* (2012). Specifically, we assume a scaling factor of 30 mg kg^{-1} fuel for BC and OC, which is comparable with results from the Aircraft Particle Emissions Experiment measurement campaigns which have found a range of 37–83 mg kg^{-1} fuel for OC and 21–98 mg kg^{-1} fuel for BC (Kinsey 2009). Total global fuel burn is estimated to be 188 Tg, of which 36% occurs in North America, 25% in Europe and 20% in Asia. The remaining 19% occurs in other regions (including Africa and Oceania). Emissions are gridded spatially and temporally for air quality modeling.

Local air quality in the vicinity of a total of 968 airports is explicitly modeled, accounting for 94% of the total fuel burn consumption for aircraft taxi-in and out, takeoff and landing. Of the modeled airports, 26.5%, 22.9%, 19.1% are located in North America, Europe, Asia, respectively. The remaining airports modeled are in other regions. We note that we do not include impacts occurring at many smaller airports, which are also likely to have local-scale impacts. 69% of airports are within the three CMAQ regions, which capture 65% of the global population, 70% of full-flight aviation fuel burn, and 76% of LTO fuel burn. A list of the airports is provided in section 3 in the Electronic Supporting Information (ESI) available at stacks.iop.org/ERL/10/034001/mmedia. Aircraft ground primary particulate matter emissions (BC, OC and primary sulfate) are computed and assigned to runways, terminals and taxiways according to flight modes including taxi-in and out, takeoff and landing. For airports in the United States, the emissions are assigned to terminals and runways according to the AEDT airport database (AEDT 2011). For non-US airports, all operation emissions are assigned to runways, since taxiway and terminal data were not broadly available. (Applying the same approach to the US resulted in a <5% local exposure difference relative to having terminal area information.)

2.2. Air quality modeling

We apply a multi-scale approach to resolve the air quality impacts on three scales: global, regional and local. We use GEOS-Chem (Bey *et al* 2001), a global chemistry-transport model with a spatial resolution of $4° \times 5°$, to simulate global air quality. GEOS-Chem results provide boundary conditions for the regional chemistry-transport model, CMAQ (Byun and Schere 2006), to simulate the air quality in North America, Europe and Asia, with a spatial resolution of 36 km, 40.5 km and 50 km, respectively. For areas outside of these three high resolution regions, GEOS-Chem results are used.

GEOS5 meteorological data from NASA are used to drive GEOS-Chem. We use the Weather Research and Forecast (WRF) (Skamarock and Klemp 2008) to simulate meteorological fields for the CMAQ simulations. Six-hour reanalysis data are used to provide initial and boundary conditions for WRF. For the regions of North America and Asia, the Final Operational Global Analysis (NCEP 1999) data are used. For the European region, the European Center for Medium range Weather Forecasting (Dee *et al* 2011) forecast data are used.

Non-aviation emissions in GEOS-Chem (i.e. as used for the global simulation providing boundary conditions for the three high resolution regions and results for other regions) are described in Bey *et al* (2001), as updated. Non-aviation emissions in CMAQ simulations for North America, Europe and Asia are described in Caiazzo *et al* (2013), in Yim and Barrett (2012), and in the section 2 in the SI, respectively.

We apply the Rapid Dispersion Code (RDC) (Barrett and Britter 2008, 2009) to simulate the local air quality impacts of aircraft ground level emissions. The RDC is based on the approaches described in Barrett and Britter (2008, 2009) that map point-source dispersion calculations to area sources semi-analytically. The RDC reduces the computational time of simulations with multi-area emission sources by 99.5% with a ~5% error in mean concentrations (Lee 2012). The RDC has been used by Lee (2012) and Yim and Barrett

(2012) to evaluate air quality impacts due to airport emissions. The spatial resolution of RDC grid receptors is 400 m with a domain size of 40.4 km × 40.4 km. AERMOD (Cimorelli *et al* 2004) is used to provide the parameterization of a point source dispersion required by RDC for area source computations, as are used to represent runways and terminal areas. The meteorological data (upper air soundings and surface observations) required by AERMOD, are provided by the Integrated Global Radiosonde Archive (IGRA) operated by the National Climatic Data Center (NCDC 2008). The RDC results are compared against results calculated by AERMOD (Cimorelli *et al* 2004) in Lee (2012).

In addition to computing primary PM concentrations in the vicinity of airports, we use an approach described in Lewis and Stevens (1985) to estimate the local concentrations of secondary sulfate $PM_{2.5}$ due to aircraft ground emissions as follows. The secondary sulfate concentration is estimated as

$$\chi_{SO_4^{2-}} = \chi_{SO_x}\left(\varepsilon + kx\left\langle\frac{1}{u}\right\rangle\right),$$

where χ_{SO_x} is the total concentration of sulfur oxides on a common mass basis simulated by RDC to account for dispersion; ε is the percentage of fuel sulfur emitted as sulfate (assumed to be 2%) (Stettler *et al* 2011); k is the average SO_2 transformation rate to SO_4^{2-} (assumed to be 1% h^{-1}) (Lewis and Stevens 1985); x is the distance from the emission source; and $\frac{1}{u}$ is the temporal average of the inverse of wind speed over a year.

We merged the RDC results with the CMAQ results for airports in the three regions considered at a regional scale (North America, Europe and Asia) and with the GEOS-Chem results for other airports. To avoid double counting the impact of aircraft emissions, we apply a mass-conserving approach developed by Isakov *et al* (2007). The PM concentration used for exposure assessment is

$$\chi = \chi_{RDC} + \chi_{CMAQ} - \overline{\chi_{RDC}},$$

where χ is the concentration ($\mu g\,m^{-3}$) of aviation-attributable $PM_{2.5}$ merged from the results of both RDC and CMAQ models ($\mu g\,m^{-3}$), which is referred to as local/regional hybrid results; χ_{RDC} is the $PM_{2.5}$ concentration ($\mu g\,m^{-3}$) calculated by RDC; χ_{CMAQ} is the aviation-attributable $PM_{2.5}$ concentration ($\mu g\,m^{-3}$) simulated by CMAQ; $\overline{\chi_{RDC}}$ is the RDC $PM_{2.5}$ concentration averaged over all the RDC grid cells in a CMAQ grid cell. This method has been applied in other airport studies (Lee 2012, Yim *et al* 2013) and has the effect of conserving ground-level PM mass, but redistributing it to be closer to sources using local dispersion model results.

2.3. Plume correction factor and source modeling

Aircraft taxi, takeoff and landing roll emission sources are modeled as ground level area sources per Barrett *et al* (2013), who found that if the aircraft plume dynamics were not taken into account, the simulated concentration in the near-field from an area source would be over-predicted by a factor of 1.36–2.30. This over-prediction is caused by neglecting the additional mixing due to aircraft exhaust jet mixing and buoyancy. Barrett *et al* (2013) showed that area sources can parameterize the local dispersion of aircraft sources if multiplied by a plume correction factor. We therefore take the plume correction factor into account in our calculations. Elevated sources, which occur in the higher speed winds away from the ground and are spread out due to the speed of airborne aircraft, are captured in CMAQ (and GEOS-Chem) modeling and not local dispersion modeling.

2.4. Health impacts

2.4.1. CRF for $PM_{2.5}$

$PM_{2.5}$ exposure is estimated by overlaying the aviation-attributable $PM_{2.5}$ concentrations, pieced together from the GEOS-Chem global simulation, the three CMAQ regional simulations, and the 968 dispersion computations, onto population taken from the Global Rural-Urban Mapping Project (GRUMPv1) with a spatial resolution of 30 arc-seconds (GRUMP 2011). The resultant premature deaths are estimated using CRFs reported by the WHO (WHO 2004). While this CRF is older than alternative CRFs reported in the literature, we select it because it provides for direct comparison to similar studies and exhibits the property of reducing risk at higher exposure and thus may provide a more representative burden of disease estimate for developing countries, where a higher background pollutant concentration is expected (Barrett *et al* 2012). We present results for an alternate CRF and also discuss the impact of other CRF choices in the ESI.

The WHO CRF describes the relationships between annual average $PM_{2.5}$ exposure and the risk of premature death due to lung cancer and cardiopulmonary disease. The CRF takes the form

$$\text{premature deaths} = \sum_k \frac{RR_k - 1}{RR_k} B_k P_k,$$

where the relative risk is $RR_k = \left(\frac{\chi_A + 1}{\chi_B + 1}\right)^{\beta}$, χ_A represents the $PM_{2.5}$ including both aviation and non-aviation emissions, and χ_B represents the concentration where only non-aviation emissions are taken into account, β is a disease specific power coefficient, and B_k is the baseline incidence rate for each disease based on the WHO Global Burden of Disease (GBD) (WHO 2004) database, P_k is population above 30 years of age which is exposed to $PM_{2.5}$, and k is a population exposure grid cell index. Further information on data sources is provided in section 6 in the ESI. We note that the toxicity may be different among $PM_{2.5}$ species. However, since the differential toxicities are uncertain (Levy *et al* 2012b), we assume an equal toxicity for all

$PM_{2.5}$ species in the premature death estimation consistent with EPA practice and previous studies.

2.4.2. CRF for ozone (O_3)

We apply a log-linear CRF to estimate premature deaths due to long-term exposure to aviation-attributable ozone (Jerrett *et al* 2009). The CRF has previously been used in assessments of health impacts due to ozone exposure (US EPA 2011, Fann *et al* 2012). Premature mortality due to aviation-attributable ozone exposure is estimated as

$$\text{premature deaths} = y_0 \cdot \left(1 - \frac{1}{\exp\left(\beta \cdot \Delta O_3\right)}\right),$$

where y_0 represents the baseline incidence rate (deaths due to all respiratory diseases). ΔO_3 is the averaged daily maximum ozone concentration (ppb) due to aviation emissions. We note that while this is strictly applicable during the ozone season, we take the annual average daily maximum ozone perturbation due to aviation emissions because of the relatively small impact of aviation, the variability of the ozone season in different regions, and because application of this approach to the US results in a <10% error. While this is small relative to other sources of uncertainty, we correct for this in our uncertainty quantification approach.

2.5. Valuation

We monetize premature deaths due to aviation emissions. The valuation is based on the VSL distribution reported by the US EPA (2011) with a mean of $7.4 m (in 2006US$). A Weibull distribution is applied to fit the data with a scale parameter of 7.75 and a shape parameter of 1.51 according to US EPA (2012). We estimate the VSLs for other countries based on their gross national income and an income elasticity range of 1–2 (Hammitt and Robinson 2011) as in Barrett *et al* (2012). Our valuation estimation also takes into account a 20-year cessation lag for $PM_{2.5}$ impacts (US EPA 2011) so that 30% of the total premature deaths occur in the first year, 50% occur equally in years two to five and the remaining 20% occur equally in years six to 20. The cessation lag is not applied for O_3 health impacts due to a lack of evidence to support any cessation lag structure. We estimate the net present value of damage at discount rates of 2%, 3% and 7%.

To compare the resulting health costs to other societal costs of aviation, we provide cost estimates of global accident costs, global climate costs and global noise costs of aviation. We derive global noise costs from He *et al* (2014). Climate costs are estimated based on the results in Dorbian *et al* (2011). For accident costs, we conduct our own analysis based on accident, fatality and injury statistics. The methodology employed for calculating consistent cost estimates is described in section 11 in the ESI.

2.6. Uncertainty

The concentration results, which we discuss in the result section, are nominal values from model simulations, while premature deaths are shown as a central estimate with 90% confidence intervals. Similar to Yim *et al* (2013) and Yim and Barrett (2012), we apply a Monte–Carlo approach to assess uncertainty in premature death and valuation estimates. Uncertainty associated with atmospheric modeling and the CRFs are taken into account in the calculations. A triangular distribution—defined by low, nominal and high multipliers—is assumed except where otherwise specified. The uncertainties of simulated $PM_{2.5}$ and O_3 vary for both GEOS-Chem and CMAQ for different regions. The uncertainty distributions for the two models are based on the normalized mean bias obtained from model validation exercises. Uncertainties in AERMOD-computed concentrations are represented by a $T(0.5, 1, 1.5)$, i.e. a ±50% triangularly distributed uncertainty (Chang and Hanna 2004), while the additional uncertainties of RDC are represented by $T(0.9, 1, 1.1)$ based on validation results provided in Lee (2012). The potential reduction in aircraft-attributable concentrations due to aircraft plume mixing and buoyancy are represented by a factor with a distribution $T(0.58, 0.71, 0.88)$ (Barrett *et al* 2013).

Stettler *et al* (2013) reported that the methods which have been widely used to estimate aircraft BC emissions (FOA3 for LTO emissions and fleet average EIs for cruise emissions), may result in an underestimation of BC emissions during LTO cycle and at cruise by a factor of $T(1.99, 3.97, 5.96)$ and $T(2.70, 2.93, 3.28)$, respectively (Stettler *et al* 2013). A sensitivity analysis is performed to estimate the sensitivity of the health impact results to different BC emission calculations, including the current widely-used method, FOA3, and FOX as developed by Stettler *et al* (2013). This is discussed in section 10 of the ESI.

For the WHO-CRF, the uncertainties of cardiopulmonary diseases and lung cancer baseline incidences are represented by $T(0.06, 0.16, 0.25)$ and $T(0.09, 0.23, 0.38)$, respectively (Ostro 2004). For the ozone-CRF, the relative risk is represented by the distribution $T(1.010. 1.040, 1.067)$ as reported by Jerrett *et al* (2009). A factor of 0.9 is taken to represent the over-estimation due to averaging the daily maximum ozone concentration over a year instead of over ozone season.

The uncertainty in the VSL for the US is represented by a Weibull distribution with a mean of $7.4 m (in 2006US$) as described in section 2.5 (US EPA 2011), while the uncertainty of the VSLs for other countries are based on the uncertainty found in literature. A list of the VSLs and their uncertainty ranges is provided in the section 7 in the ESI.

Table 1. The mean ground level concentrations of $PM_{2.5}$ (ng m^{-3}) and O_3 due to full flight (FF) and landing and takeoff (LTO) only emissions. The global values are based on CMAQ results for the three regions and GEOS-Chem results for other regions, and GEOS-Chem replaced CMAQ where available for global results. The percentage of each $PM_{2.5}$ species is also given for full flight emissions.

	FF/LTO $PM_{2.5}$ (ng m^{-3})	FF BC (ng m^{-3}) (%)	FF OC (ng m^{-3}) (%)	FF SO_4 (ng m^{-3}) (%)	FF NO_3^- (ng m^{-3}) (%)	FF NH_4^+ (ng m^{-3}) (%)	FF/LTO O_3 (ppb/ppt)
Global	6.2/0.5	0.6 (1.7)	0.5 (1.2)	37.6 (36.5)	41.9 (41.2)	19.3 (19.3)	0.6/10.7
North America	9.0/1.2	1.3 (5.1)	−0.2 (−0.6)	12.7 (34.7)	65.4 (40.9)	20.9 (20.0)	1.1/25.4
Europe	18.2/4.8	0.3 (1.1)	−0.3 (0.1)	7.0 (12.6)	69.9 (63.4)	23.1 (22.8)	1.0/29.8
Asia	15.1/0.7	0.6 (1.4)	1.0 (0.9)	20.0 (28.1)	57.1 (48.9)	21.3 (20.7)	0.9/12.5
Other	3.8/0.3	0.6 (1.3)	0.4 (2.1)	55.5 (45.4)	28.1 (35.4)	15.4 (15.9)	0.5/8.8

3. Results and discussion

We first describe results for global and regional air quality impacts, then computations of near-airport air quality. Air quality impacts accounting for all scales are then mapped to health impacts and monetized to enable comparison with other societal costs of aviation.

3.1. Global and regional air quality impacts

Table 1 shows the global (GEOS-Chem and CMAQ based) and regional (CMAQ based) surface PM and ozone impacts of aviation, with PM impacts speciated. The impact of LTO emissions (i.e. up to 3000 ft) is also given, as these are the emissions that are currently regulated.

The global average impact of aviation emissions on surface O_3 is 0.6 ppb. This result is consistent with Lee *et al* (2013), which reports that aviation emissions lead to 0.5 ppb increase in O_3 in July, whereas up to several ppb in January. Our estimate show that 2% (10.7 ppt) of the total aviation impact on surface O_3 is attributable to LTO emissions. Compared with the results calculated by only using GEOS-Chem, the global area-weighted ground level O_3 attributable to full flight aviation emissions when including nested CMAQ computations increases by 12%, but the O_3 impact due to LTO emissions decreases by 6%, consistent with increased NO_x emissions decreasing O_3 formation in VOC limited regions (which are captured by the higher resolution CMAQ modeling).

Among the regions, North America experiences the highest aviation impact on surface O_3 (1.1 ppb), of which LTO emissions contribute for 25.4 ppt. Aviation emissions cause a 0.9 ppb increase in annual average O_3 concentration in Asia, which is lower than the impact in Europe and North America. In Europe, the O_3 impact due to LTO emissions is ~2.4 times higher than that in Asia. The aviation-attributable O_3 concentration in other regions is ~0.5 ppb.

We estimate that global aviation emissions result in an average 6.2 ng m^{-3} ground level $PM_{2.5}$ perturbation. Figure 1 depicts the annual ground level $PM_{2.5}$ concentration due to aviation emissions, where GEOS-Chem and CMAQ results have been merged. Compared with the results calculated by only using GEOS-Chem, the global area-weighted ground level aviation-attributable $PM_{2.5}$ decreases by 29%, while the standard deviation of the concentration increases by 22%, representing increased spatial resolution.

Our estimates show two peaks of $PM_{2.5}$ in Northern India (0.47 μg m^{-3}) and Northeastern China (0.35 μg m^{-3}), coincident with peaks in ammonia concentrations (Barrett *et al* 2010), and also peaks in the central Europe and San Francisco, which are associated with major airports. Of the total ground level aviation-attributable $PM_{2.5}$, nitrate (NO_3^-) and sulfate (SO_4^{2-}) account for 42% and 38% by mass, respectively. BC and OC together account for ~1% on average. As seen in table 1, aviation has a negative impact on OC in North America and Europe. Aircraft NO_x emissions have been shown to reduce ambient OC (Ashok *et al* 2013, Woody and Arunachalam 2013), as they deplete radical species in the vicinity of airports and consequently slow the oxidation of organic aerosol precursors (Woody and Arunachalam 2013). Woody and Arunachalam (2013) note, however, that aviation's impact on OC may be sensitive to model grid resolution.

Our global and regional models results show that the air quality impact due to aviation emissions varies among the different regions. In North America, 9.0 ng m^{-3} of $PM_{2.5}$ is attributable to aviation emissions. Of the aviation-attributable $PM_{2.5}$ in North America, ~13% (1.2 ng m^{-3}) is due to LTO emissions. In Europe, the annual average $PM_{2.5}$ due to aviation emissions is 18.2 ng m^{-3}, which is the highest among the regions, and is double of that of North America. In Asia and other regions, the average $PM_{2.5}$ concentrations due to aviation emissions are 15.1 ng m^{-3} and 3.8 ng m^{-3}, respectively. As can be seen in figure 1, in the other regions (not modeled at high resolution with CMAQ), aviation contributes to $PM_{2.5}$ in limited regions including the Middle East and western parts of Russia.

3.2. Local air quality impact in different regions

We estimate the near-airport (within 20 km) ground level aviation-attributable $PM_{2.5}$ averaged over all airports in each region, combining our local dispersion calculations with CMAQ results using the mass conserving scheme described. Our results show that

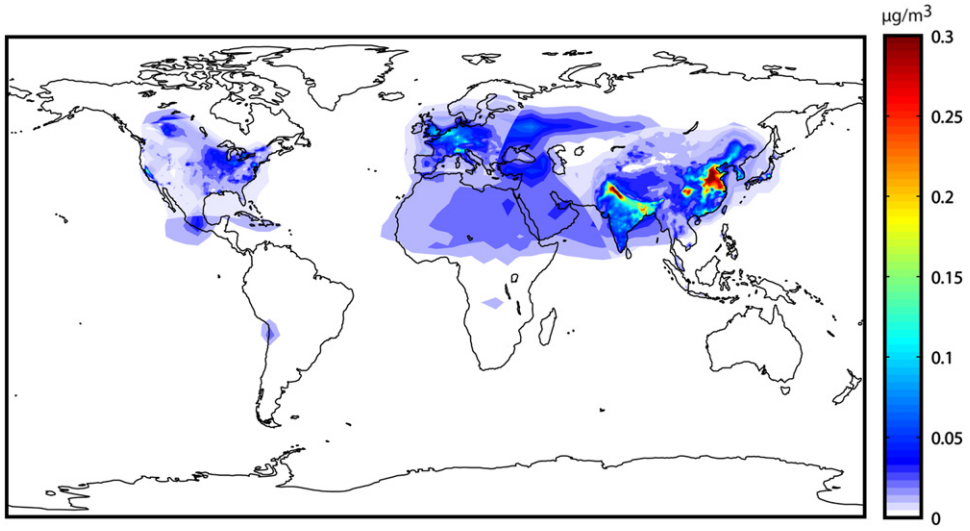

Figure 1. Annual mean $PM_{2.5}$ concentration ($\mu g\ m^{-3}$) due to aviation emissions in 2006. The CMAQ results—simulated by finer resolution grid cells—in North America, Europe and Asia are overlaid onto the GEOS-Chem results.

Table 2. Premature deaths per year in different regions due to the population exposure to aviation-attributable $PM_{2.5}$ and ozone (90% confident intervals) calculated using the WHO-CRF.

	Full flight	LTO	LTO/FF (%)
North America	1500 (850–2300)	650 (290–1300)	43
Europe	3700 (2100–5500)	1800 (1100–2600)	49
Asia	8200 (3700–13 000)	740 (420–1200)	9
Other regions	2700 (1400–4200)	780 (420–1300)	29
Global	16 000 (8300–24 000)	4000 (2400–6200)	25

primary $PM_{2.5}$ due to aviation emissions contributes to 44–61% of total aviation-attributable $PM_{2.5}$ at 2 km distance from airports. However, the percentage decreases as distance from airport increases. At 20 km from airports, the percentage drops to less than 6%.

Our results show that aviation emissions lead to an average $PM_{2.5}$ concentration of 44.2 ng m^{-3} in the 20 km vicinity of all airports globally. For airports in Asia, the mean near-field impact of $PM_{2.5}$ due to aviation emissions is 74.1 ng m^{-3}, the highest among regions and more than double as the $PM_{2.5}$ in North America (29.5 ng m^{-3}). This is consistent with the peak in available ammonia amplifying aviation's $PM_{2.5}$ contribution, particularly the effect of cruise emissions. On average, the mean aviation-attributable $PM_{2.5}$ impacts in the vicinity of airports in Europe and in other regions are 58.5 ng m^{-3} and 26.2 ng m^{-3}, respectively.

The population exposure to aviation-attributable $PM_{2.5}$ in different regions varies due to different regional population densities in the vicinity of airports, the variation in available ammonia and in aviation emissions. Our results show that 23% of airports have near-field population exposure to aircraft-attributable $PM_{2.5}$ higher than the global average exposure, of which 17% are located in North America, 33% and 34% are located in Europe and Asia, respectively, and the remaining 16% are located in other regions. Aviation emissions result in 44.9 people·mg m^{-3} mean $PM_{2.5}$ exposure within 20 km averaged over all airports globally. Among regions, the mean exposure in the vicinity of airports in Asia is the highest (142.6 people·mg m^{-3} per year), a factor of ~3.2 higher than the global average. The relatively high near-field $PM_{2.5}$ exposure in Asia is due to this region having both relatively high aviation-attributable $PM_{2.5}$ concentration (due to the extent of available ammonia) and mean population density in the vicinity of airports. Within 20 km of airports, the average population surrounding all airports in Asia is 1.6 million, 87% higher than the global average. The average aviation-attributable $PM_{2.5}$ exposure within 20 km of airports in Europe, North America and other regions is 42.3, 19.5 and 15.3 people·mg m^{-3}, respectively.

3.3. Health impacts

Table 2 shows estimated premature mortalities due to aviation emissions. Global aviation emissions cause 16 000 (90% CI: 8300–24 000) premature deaths per year due to population exposure to aviation-attributable $PM_{2.5}$ and ozone. Of the total premature deaths, 87% and 13% are due to $PM_{2.5}$ and ozone, respectively, while 25% is attributable to the LTO portion of emissions. Comparing with the approach of only using

Table 3. Aviation fuel burn (FB) in different regions and the resultant costs (2006US$bn) for a 2% discount rate due to the health impact of aviation full flight and LTO emissions. This table also includes the ratio of health costs to aviation fuel burn occurring in each region.

	Full flight emissions			LTO emissions		
	FB (Tg)	Cost ($bn)	Cost/FB ($/tonne)	FB (Tg)	Cost ($bn)	Cost/FB ($/tonne)
North America	55.8	7.08	127	7.0	3.07	439
Europe	34.7	10.02	289	4.5	4.09	909
Asia	40.7	2.25	55	5.4	0.41	76
Other regions	56.9	1.48	26	5.3	0.48	91
Global	188.1	21.16	112	22.1	8.19	371

GEOS-Chem (a global model), our multi-scale approach estimates 7% and 29% higher global premature deaths due to full flight and LTO emissions, respectively. The lower increase for full flight emissions is consistent with the relatively diffuse impact of the dominant cruise emissions being captured by the lower resolution global model.

Our estimate shows that aviation emissions cause 2100 (90% CI: 1000–3300) ozone-related premature deaths per year worldwide. LTO emissions alone account for 2.6% of the ozone-associated premature deaths due to aviation emissions. This result highlights that the long-term health impact of O_3 due to LTO aviation emissions is marginal, compared to the $PM_{2.5}$ health impact. However, the ozone-exposure due to full flight emissions accounts for 12% of the total premature deaths due to the both aviation-attributable $PM_{2.5}$ and O_3. Of the total ozone health impact due to aviation emissions, 62% occurs in Asia, while 7% and 10% occurs in North America and Europe, respectively. From table 1 it can be seen that the O_3 mixing ratio attributable to aviation in the three regions is 0.9–1.1 ppb, suggesting that population density drives the breakdown of mortalities by region. The remaining 21% occurs in other regions.

In Barrett *et al* (2010), it is found that 80% of health impacts on a global scale are due to non-LTO emissions. In this study with its increased regional resolution combined with dispersion calculations at 968 airports, we capture more of the LTO impacts (partly countered by the inclusion of ozone in this study which is dominated by cruise emissions) and revise this estimate down to 75%. However, as shown in table 2, regions with relatively high concentrations of airport fuel burn have relatively high contributions from LTO emissions. Specifically, in North America and Europe 43% and 49% of early deaths are due to LTO emissions, respectively. On the other hand, 91% of early deaths in Asia are due to non-LTO emissions. Asia accounts for 20% of global civil aviation fuel burn, but over 50% of early deaths due to aviation emissions. This is consistent with Asia incurring a relatively high component of intercontinental air pollution from aviation (Koo *et al* 2013). In other words, the prominence of Asia is due to population density and the amplifying effect of available ammonia on nitrate rather than local LTO emissions (which contribute little to nitrate exposure due to the timescale required for oxidation of NO_x).

While figure 1 would suggest that the majority of aviation's air quality impacts are captured in the three high resolution regions, table 1 shows that ~2700 early deaths occur in the other regions due to aviation emissions each year, greater than the ~1500 in North America. The ratio of total population exposure to $PM_{2.5}$ due to aviation emissions for North America to other regions is 0.53, whereas the ratio of population-weighted average $PM_{2.5}$ concentration due to aviation emissions for North America to other regions is 4.05. This is consistent with the major air quality impacts of aviation being captured in the high resolution regions, while the population in other regions means that the diffuse impacts of aviation still contribute 17% of global early deaths.

We calculate premature deaths due to aviation-attributable $PM_{2.5}$ exposure within 20 km from each airport worldwide. Our results show that aviation-attributable $PM_{2.5}$ causes 5000 (90% CI: 2000–9900) premature deaths within 20 km from ~1000 airports, which account for ~32% of the total premature deaths due to both aviation-attributable $PM_{2.5}$ and O_3. Of the total airport vicinity premature deaths (i.e. those within 20 km of airports), 25% occur in North America; 38% in Europe; 22% in Asia; and the remaining 15% in other regions. We do not detail early deaths in the vicinity of each individual airport because impacts are calculated for aviation in general and not specific airports, so impacts within 20 km of a specific airport cannot be attributed to that airport.

3.4. Valuation and comparison to other societal cost of aviation

We monetize the premature deaths due to aviation emissions (in 2006 US$). The central monetized health impact is $(21.16, 20.58, 18.72)bn per year for a (2, 3, 7)% discount rate choice (ESI table S16). Of the total cost, the damage in North America accounts for $(7.08, 6.89, 6.27)bn per year, the damage in Europe amounts to $(10.02, 9.74, 8.82)bn per year—the highest among the different regions—while the damage in Asia is $(2.25, 2.19, 2.00)bn per year. The damage in other regions accounts for the remaining $(1.48, 1.44, 1.32)bn per year.

Table 3 shows that the resulting regional health cost of aviation emissions is not proportional to

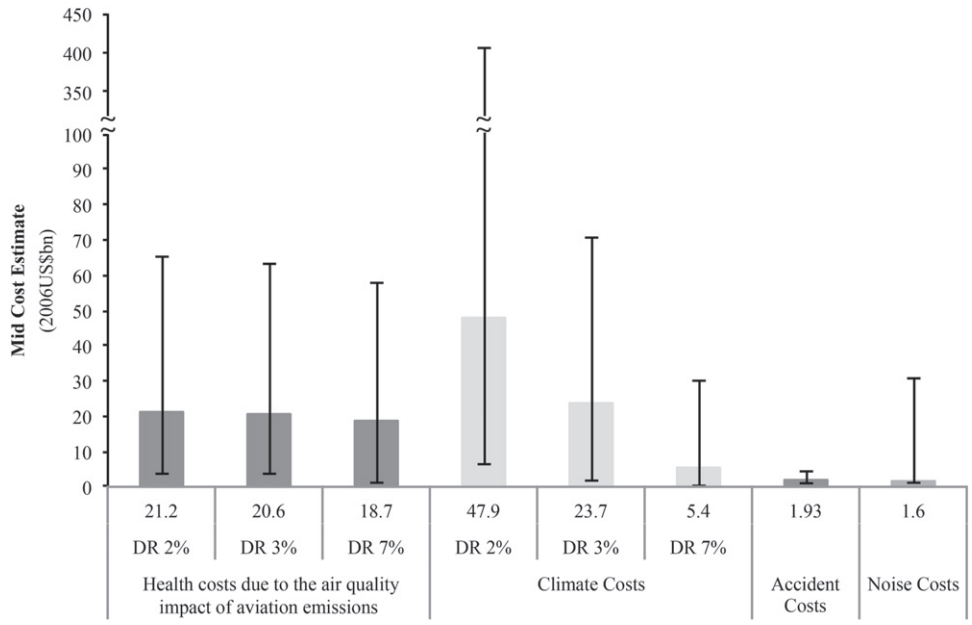

Figure 2. Comparison of the central net present value cost estimate for 2006 aviation emissions-related health costs with mid net present value cost estimates for climate impacts from aviation emissions, mid cost estimates for accident costs of aviation and mean annualized aviation noise costs (2006US$bn), differentiated according to discount rates (DRs), where applicable. The capitalized mean noise costs are added for reference. The error bar for health costs is bounded by the 5% and 95% percentile of monetized impacts. See the section 11 in the ESI for details on error bars for the other costs.

aviation fuel burn occurring in each region. The global average ratio is US$112/tonne for full-flight emissions and the figure for LTO emissions is a factor of 3.31 higher. The corresponding factor for number of early deaths per unit fuel burn is 2.13. LTO operations cause more early deaths per unit emission than at cruise due to the proximity of emissions to the population. However, when monetized this difference is magnified due to the relatively greater importance of LTO emissions in richer regions (with higher VSLs) such as North America. The influence of regional variation in VSL is also evident in noting that while 51% of full-flight aviation emissions-attributable early deaths occur in Asia, only 11% of monetized impacts occur there.

Dorbian *et al* (2011) estimated that the air quality marginal damages per tonne of fuel burn in LTO in the United States is US$230, which is ~50% lower than our result of US$439. This may be because Dorbian *et al* (2011) used a regional air quality model only, which does not resolve local impacts, and FOA3-based BC emissions rather than the higher FOX-based BC emissions.

To understand the relative importance of health costs due to the air quality impact of aviation emissions, we compare them to the estimates of other aviation-induced societal costs, i.e. noise costs, accident costs and climate change costs as shown in figure 2. (See ESI for the monetization approach.) The figure compares the central values for global health costs due to the air quality impact of aviation emissions in 2006 with estimates for climate costs, accident costs and noise costs for the same year and various discount rates, where applicable. Note that the bars shown here are based on mid- or mean estimates and that significant uncertainty exists about actual costs, as indicated by the error bars in this figure.

Our results show that the health costs of aviation emissions are on the same order of magnitude compared to climate costs for discount rates of 2% and 3%. For a consistent discount rate of 7%, climate costs are one order of magnitude smaller than health costs. Comparing the emissions-related health costs to the global accident costs of aviation, the central estimate of the health costs exceed the mid accident costs estimate by one order of magnitude. Aircraft accidents have a high public visibility but are rare occurrences (~0.4 fatal accidents per million hours flown, for a total of ~1050 fatalities in 2006, see ESI section 11), but the societal costs as calculated here are significantly lower than the health costs of aviation emissions. We also find that the mean estimate of the annualized noise costs is one order of magnitude lower than the central values for the health costs due to aviation emissions.

4. Conclusions

We produce the first multi-scale global assessment of the air quality and human health impacts of aviation, accounting for both fine particulate matter and ozone, estimating that aviation emissions result in ~16 000 early deaths each year. We find that $PM_{2.5}$ exposure causes 87% of early deaths. While cruise emissions dominate causing 75% of early deaths due to aviation

emissions, approximately half of early deaths are caused by LTO emissions in North America and Europe—regions with relatively high aviation and airport fuel burn. In contrast, 91% of early deaths are caused by non-LTO emissions in Asia. This suggests that LTO emissions reductions in North America and Europe will provide regional benefits, while the benefits of non-LTO emissions reductions will be diffuse and also felt in Asia.

A global total of ~5000 people who live within 20 km of airports are estimated to die prematurely each year due to aviation emissions, with 38% of airport vicinity early deaths in Europe. Our results suggest, in contrast with previous analyses, that primary $PM_{2.5}$ emissions from aviation are a significant contributor to health risk when airport vicinity exposure is captured. A significant uncertainty in our estimates of the subgrid contribution to $PM_{2.5}$ exposure is the aviation BC emissions inventory.

Finally we show that the monetized health costs of aviation emissions exceed aviation's fatal accident costs and noise costs by an order of magnitude, and is on the same order as aviation's climate costs for discount rates of 2% and 3% (as are appropriate to climate change costing Johnson and Hope 2012). This suggests that environmental benefits of fuel burn reductions are as much in air quality as they are in climate. Furthermore, this implies that when assessing the environmental impacts of aviation biofuels that result in reductions in emissions, the air quality impacts may be in the same order as the climate impacts. For example, paraffinic biofuels would be expected to eliminate SO_x emissions and reduce BC emissions by ~80% (Speth *et al* 2015).

Acknowledgments

This study is based on work from 2006–2014 sponsored in part by the UK Engineering and Physical Sciences Research Council (EPSRC), in part by the FAA Office of Environment and Energy, and in part by MIT (including final assembly of this multi-scale study). Any views or opinions expressed in this work are those of the authors and not the FAA or EPSRC. We also thank the anonymous referees for their comments.

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