



*Supplement of*

## **Global impacts of aviation on air quality evaluated at high resolution**

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## Versions of GCHP used in this study

Two versions of the GEOS-Chem High Performance (GCHP) model are used in this work. The first, GCHP 12.6.2 (DOI: <https://doi.org/10.5281/zenodo.3543702>) is used without additional modifications. This is the basis of the core simulations performed at C24, C90, and C180 for which mortality outcomes are calculated and for which the effect of in- versus out-of-domain emissions is evaluated. The second, GCHP 14.2.0 (DOI: <https://doi.org/10.5281/zenodo.8411829>) is used but includes a modification to resolve an integration error which occurs at high horizontal resolutions. The error is described at <https://github.com/geoschem/geos-chem/issues/1982>, with its solution (as implemented in this work) described at <https://github.com/geoschem/geos-chem/pull/2006>. This second version of GCHP is used to perform all other sensitivity simulations.

## Listing of simulations

Table S 1. Full listing of simulations performed. LTO: Landing and take-off emissions, below 3,000 feet. Species: chemical species included in the aircraft exhaust.

Set	Name	Cell size (km)	Aviation emissions settings				Species	GCHP model version
			Non-US	US	LTO	Cruise		
Core	AVGLOBAL	400						
		100	Y	Y				
	AVNOUS	50						
		400						
		100	Y	N	Y	Y	All	12.6.2
		50						
AVOFF	400							
	100	N	N					
Sensitivity	AVGLOBAL	50						
		100	Y	Y				
	AVNOUS	100	Y	N	Y		All	
		100	N	N		Y		
	AVNOLTO	100	Y	Y	N			14.2.2
	AVNONOx	100	Y	Y	Y		NOx only	
TRANSPORT	100							
		400	N/A (Simulation included radioactive tracers only)					

## Seasonal patterns of exposure and effect of exposure response function

The 74,300 mortalities attributed to aviation emissions in this work is greater than previous estimates, which have ranged from 10,000 (Barrett et al., 2010) to 58,000 (Quadros et al., 2020). As discussed in the main text, this is in part due to our use of the same Turner et al. (2015) epidemiological data for ozone exposure as was used in Quadros et al. (2020).

Compared to the Jerrett et al. (2009) data used by previous assessments (e.g. Eastham and Barrett (2016), Yim et al. (2015)), Turner et al. (2015) relates mortality to the annual mean 8-hour maximum daily average (MDA8) ozone exposure rather than the ozone season 1-hour maximum daily average (MDA1). MDA1 is by definition always equal to or greater than MDA8, but the differences over the US are typically 10-20% (Seltzer et al., 2020). Turner et al. find that respiratory mortality increases by 12% for each 10 ppb increase in annual mean MDA8 ozone, compared to Jerrett et al. who find a 4% increase in respiratory mortality for each 10 ppb increase in ozone season MDA1 ozone. Since ozone levels are higher during ozone season, the distinction between 4 and 12% does not necessarily indicate greater mortality. Malley et al. (2017) found that shifting from the Turner et al. risk estimate to that of Jerrett et al. increased global ozone-attributable mortality by a factor of 2.2 to 2.6, but we find an increase of 3.2 times (assuming the same diseases are affected). This greater fraction is because the peak in ozone exposure resulting from aviation emissions occurs outside of ozone season, as shown in Figure S1. This effect is also illustrated in a video which shows the diurnal variation in surface  $PM_{2.5}$  attributable to aviation emissions, on a 0 – 0.6  $\mu\text{g}/\text{m}^3$  scale and starting with no spin up:

<https://www.dropbox.com/s/onfkrwliltv6r1w/Surface%20AQ.mp4?dl=0>.

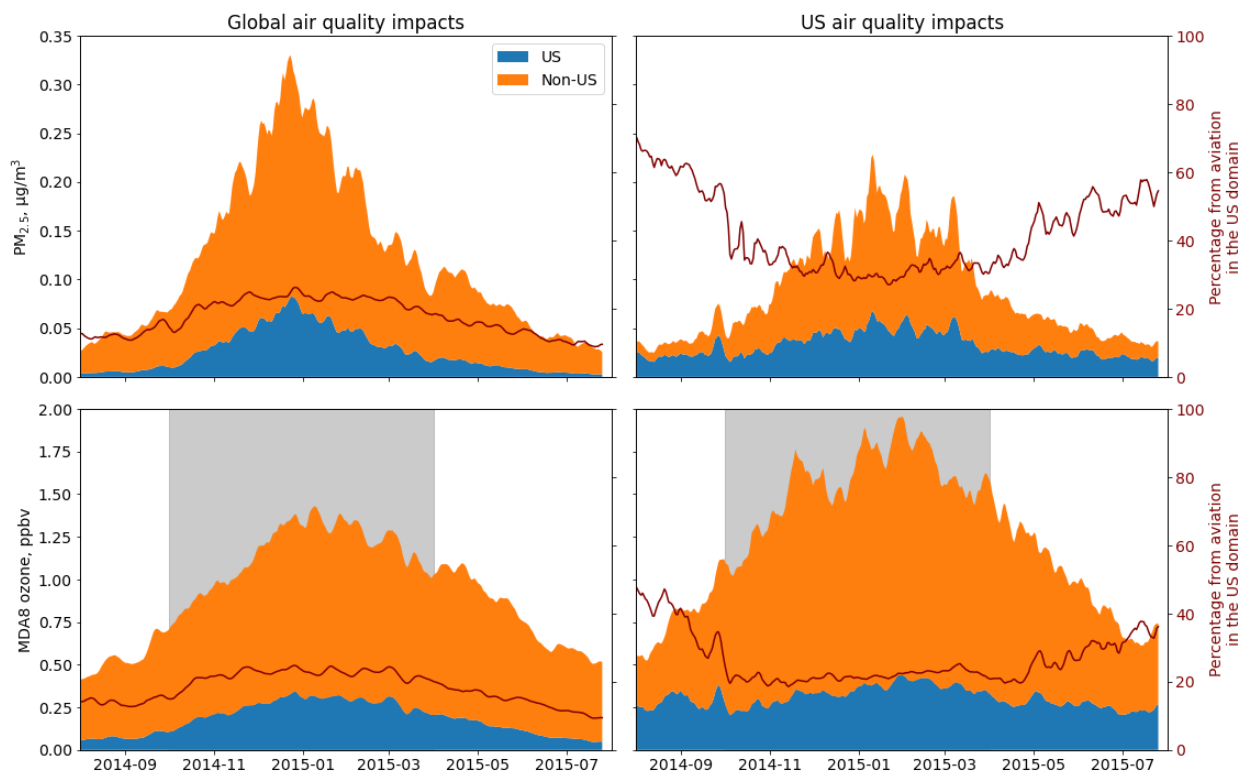


Figure S1. Exposure to air pollutants over the course of one year as a consequence of aviation emissions. Top row: exposure to fine particulate matter ( $PM_{2.5}$ ). Bottom row: exposure to ozone, using the 8-hour maximum daily average (MDA8) metric. Left

*column: global mean exposure. Right column: exposure in the USA only. The blue shaded area shows impacts resulting from emissions in the US domain only, while the orange shaded area shows impacts resulting from emissions outside of the US. The red line shows the percentage of impacts in the target domain (i.e. global on the left, US-only on the right) resulting from US aviation emissions. The grey shaded area indicates the part of the year which would not be included when calculating the US "ozone season", under the definition used by Jerrett et al. (2009) of April through September.*

We also evaluate the effect of thresholding on the total ozone impact. Turner et al. (2015) evaluated the effect of a threshold at 35 ppbv ozone, below which ozone exposure is assumed to have no effect. We find that the application of the threshold reduces ozone impacts by 8.8% in the US and 12% globally. Since Turner et al. (2015) did not include this threshold in their central estimate of ozone impacts, we do not include it in our central assessment.

Finally, in this study we apply the Global Estimate of Mortality Model (GEMM) to estimate mortality attributable to PM<sub>2.5</sub> exposure resulting from aviation emissions. Compared to results generated using five other relative risk data sources (see Table S2), we find that the GEMM-based value is the lowest. Using the Hoek et al. (2013) meta-analysis data employed by Eastham and Barrett (2016) would increase PM<sub>2.5</sub>-related mortality by 16%, whereas using the Krewski et al. (2009) data from the extended analysis of the American Cancer Society cohort – employed, for example, by Dedoussi et al. (2020) in evaluating mortality due to cross-state air pollution – would cause a 68% increase.

Table S2. Central estimates of mortality from aviation-attributable PM<sub>2.5</sub> and ozone exposure when using different relative risk sources. Differences relative to the source used for the main text (in bold italics) are shown as percentages. MDA1 and MDA8 refer to the maximum daily value using a 1- or 8-hour average respectively. \*NCD + LRI: Non-communicable disease and lower respiratory infections.

Risk data source	Exposure factor	Threshold	Cause of death	Global mortalities estimated
<b><i>GEMM (Burnett et al., 2018)</i></b>	<b><i>Annual mean PM<sub>2.5</sub></i></b>	-	<b><i>NCD + LRI*</i></b>	<b><i>21,200 (-)</i></b>
Hoek et al. (2013) meta-analysis	Annual mean PM <sub>2.5</sub>	-	Cardiovascular	24,700 (+16%)
Vodonos et al. (2018)	Annual mean PM <sub>2.5</sub>	-	All-cause	40,400 (+91%)
Chen and Hoek (2020) meta-analysis	Annual mean PM <sub>2.5</sub>	-	All-cause	45,700 (+120%)
EPA expert judgement (2010)	Annual mean PM <sub>2.5</sub>	-	All-cause	64,500 (+200%)
ACS cohort (Krewski et al., 2009)	Annual mean PM <sub>2.5</sub>	-	All-cause	35,600 (+68%)
<b><i>Turner et al. (2015)</i></b>	<b><i>Annual mean MDA8 ozone</i></b>	-	<b><i>Respiratory disease</i></b>	<b><i>53,100 (-)</i></b>
Turner et al. (2015)	Annual mean MDA8 ozone	35 ppbv	Respiratory disease	46,900 (-12%)
Jerrett et al. (2009)	Ozone season MDA1 ozone	-	Respiratory disease	16,800 (-68%)
Jerrett et al. (2009)	Ozone season MDA1 ozone	56 ppbv	Respiratory disease	9,540 (-84%)
Jerrett et al. (2009)	Ozone season MDA1 ozone	-	COPD + asthma	10,700 (-80%)
Jerrett et al. (2009)	Ozone season MDA1 ozone	56 ppbv	COPD + asthma	6,890 (-87%)

## Non-aviation emissions inventories

In addition to the aviation emissions described in the main text, we simulate a suite of non-aviation emissions in GEOS-Chem. These are listed in Table S3. All listed emissions were identical between simulations. In addition to those listed, the ParaNO<sub>x</sub> extension is used to calculate the effect of in-plume chemical processing for ship emissions (Holmes et al., 2014; Vinken et al., 2011). Where both a regional and global source are available the regional inventory overwrites the global one. For example, both MIX and CEDS provide estimates of electricity generation emissions, but MIX applies to Asia only while CEDS is global. We therefore use MIX in Asia, and CEDS elsewhere (where not otherwise overwritten by other regional inventories).

Table S3. Non-aviation emissions used in GEOS-Chem.

<b>Inventory name</b>	<b>Region</b>	<b>Emissions included</b>
APEI (Meng et al., 2019)	Canada	Multiple anthropogenic sectors
NEI 2011 (US EPA, 2015)	North America	Multiple anthropogenic sectors, including ship emissions
MIX v1.1 (Li et al., 2017)	Asia	Multiple anthropogenic sectors
DICE (Marais and Wiedinmyer, 2016)	Africa	Non-industrial anthropogenic emissions
CEDS (Hoesly et al., 2018)	Global	Multiple anthropogenic sectors, including ship emissions
GEIA NH <sub>3</sub> (Bouwman et al., 1997)	Global	Non-anthropogenic ammonia
POET (Granier et al., 2005; Olivier et al., 2003)	Global	Ethanol
C <sub>2</sub> H <sub>6</sub> 2010 (Tzompa-Sosa et al., 2017)	Global	Ethane from fossil fuel and biofuel
Xiao C <sub>3</sub> H <sub>8</sub> (Xiao et al., 2008)	Global	Propane from fossil fuel and biofuel
Liang Bromocarb. (Liang et al., 2010)	Global	Very short lived bromocarbons (CHBr <sub>3</sub> , CH <sub>2</sub> Br <sub>2</sub> )
Ordonez Iodocarb. (Ordóñez et al., 2012)	Global	Iodocarbons (CH <sub>3</sub> I, CH <sub>2</sub> I <sub>2</sub> , CH <sub>2</sub> ICl, CH <sub>2</sub> IBr)
Decaying plants (Millet et al., 2010)	Global	Volatile organic compounds from decaying plants
AFCID (Philip et al., 2017)	Global	Anthropogenic, fugitive construction and industrial dust
Volcano (Carn et al., 2015)	Global	Sulfur dioxide from volcanic degassing and eruption
GFED (Giglio et al., 2013)	Global	Fire emissions from the GFED v4 inventory
Inorg. Iodine (Carpenter et al., 2013; MacDonald et al., 2014; Sherwen et al., 2016)	Global	Marine emissions of inorganic iodine (HOI, I <sub>2</sub> )
LightNO <sub>x</sub> (Murray et al., 2012)	Global	Lightning NO <sub>x</sub>

SoilNOx (Hudman et al., 2012)	Global	NO <sub>x</sub> from soil and fertilizers
SeaSalt (Jaeglé et al., 2011)	Global	Sea salt picked up from the ocean surface
DustDEAD (Zender, 2003)	Global	Entrained mineral dust from the DEAD model
MEGAN (Guenther et al., 2012)	Global	Biogenic emissions

Emissions of soil NO<sub>x</sub>, mineral dust, sea salt, and biogenic emissions are calculated prior to the simulation at a global resolution of 0.5° latitude by 0.625° longitude, with the calculated fluxes for each hour archived and used during all simulations at all resolutions. This approach ensures consistent emissions between simulations at different resolutions. The only exceptions are lightning NO<sub>x</sub> and emissions of inorganic iodine, which are calculated online based on meteorological fields. Emissions of lightning NO<sub>x</sub> are calculated based on the archived flash density and convective depth fields, with the functional result that they are also resolution-independent. Differences in emissions of inorganic iodine are expected to be a negligible contributor to differences.

## Disaggregating the effect of resolution on exposure

Increasing model resolution affects both the accuracy of the model calculations and the degree to which pollution and population are “collocated” in the data. To isolate this effect, we calculate total population exposure globally in two ways. First, we calculate exposure by using the original model output. We then perform a second calculation in which we degrade all model output to the coarsest resolution (C24, or ~400 km). The subsequent estimated global mean population exposures for each pollutant are shown in Table S4.

*Table S4. Global mean population exposure to each pollutant calculated at the original model resolution (“Raw”) or after degrading the model output to C24 resolution (“Degraded”). Results are given as the percentage of the “Raw” C24 calculation.*

	PM <sub>2.5</sub>		Soot		Ozone (MDA8)	
	Raw	Degraded	Raw	Degraded	Raw	Degraded
C24 (~400 km)	100%	-	100%	-	100%	-
C90 (~100 km)	133%	116%	114%	90.3%	117%	121%
C180 (~50 km)	135%	116%	122%	85.0%	120%	125%

Comparing the “degraded C180” to “raw C24” results isolates the effect of improved resolution of physical phenomena. For PM<sub>2.5</sub> and ozone the “degraded C180” exposure is 16% and 25% greater respectively than the “raw C24” exposure. This is less than the combined effect (comparing “raw C180” to “raw C24”) for PM<sub>2.5</sub> (+35%) but greater than that for ozone (+20%). This suggests that the artificial diffusion associated with coarse-resolution simulation reduces the estimated impacts of PM<sub>2.5</sub> but enhances them for ozone.

For soot, we find that exposure calculated using the “degraded C180” results is 15% lower than the “raw C24” calculation, whereas the “raw C180” results showed greater exposure than the “raw C24” (+22%). This implies that using a higher-resolution model causes faster removal of soot from the atmosphere, but that this effect is exceeded by the increase in exposure associated with collocation of pollution sources and population. Performing the same calculation using the ~100 km (C90) output data shows changes in surface air quality which are consistent with using the ~50 km results for PM<sub>2.5</sub> and ozone.



Near-airport surface air quality evaluated at different resolutions

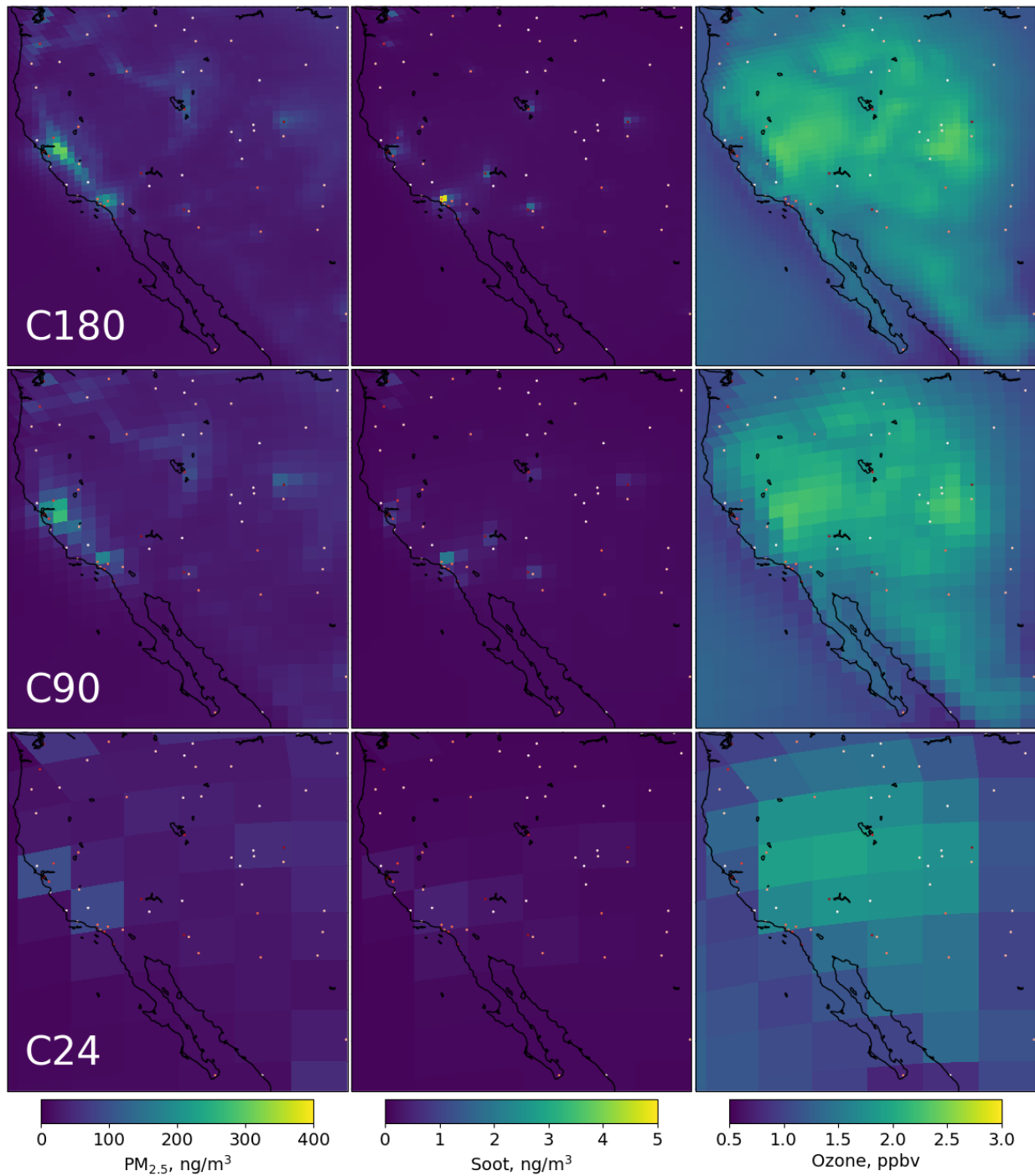


Figure S 2. Changes in annual mean PM<sub>2.5</sub> (left), soot (center), and MDA8 ozone (right) over the US west coast due to aviation simulated at C180 (top), C90 (middle), and C24 (bottom) resolution. Red dots indicate airports which served at least 100,000 passengers in 2015 according to the US Bureau of Transportation Statistics from T-100 segment data.

## Zonal mean aviation-attributable ozone evaluated at different resolutions

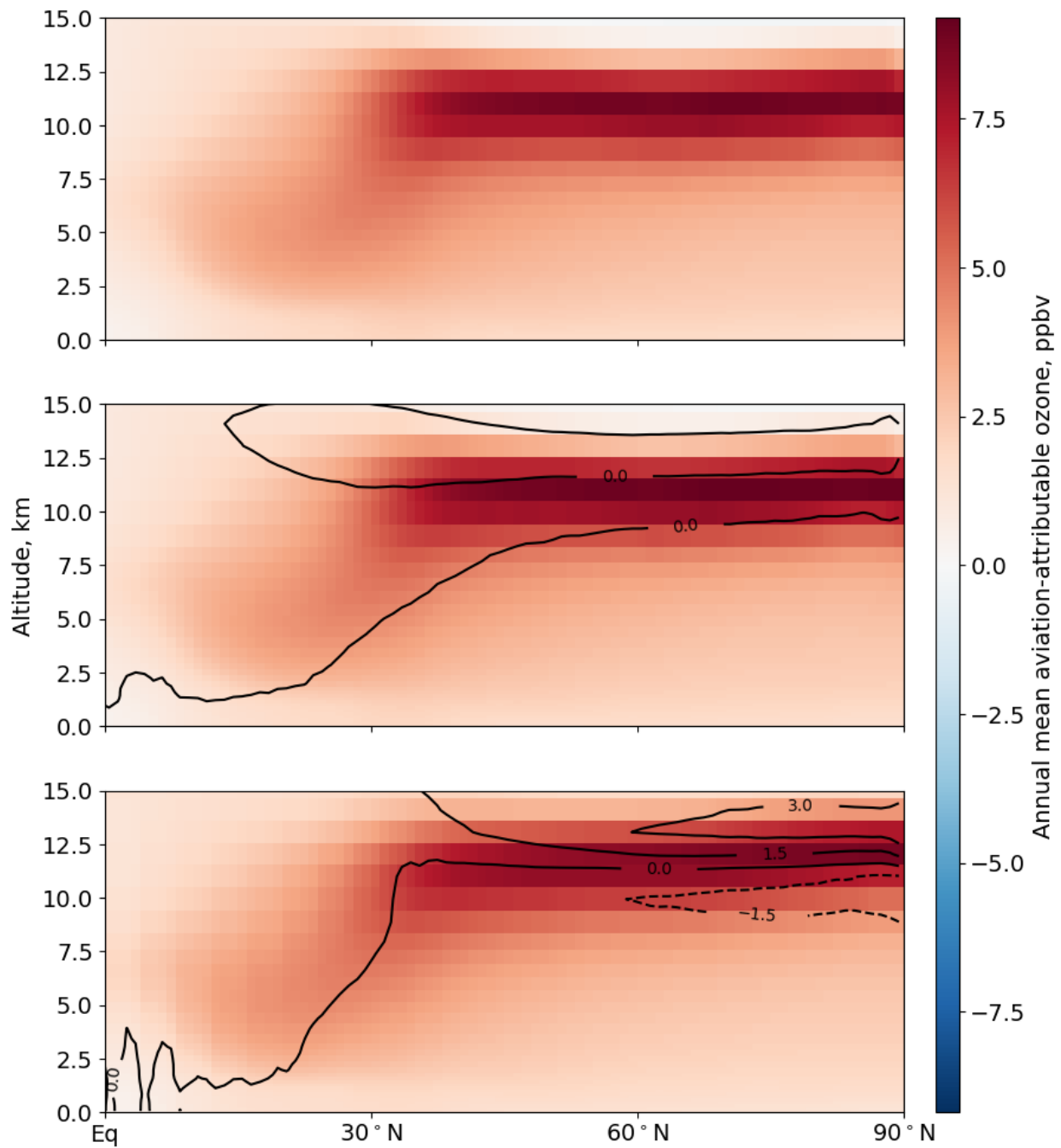


Figure S 3. Annual mean aviation-attributable change in zonal mean ozone at C180 (top), C90 (middle), and C24 (bottom) for the Northern Hemisphere. Contours show the difference between the estimate at that resolution and at C180, with positive values indicating that the simulation at the lower resolution is overestimating the change in ozone relative to the simulation at C180. Each contour corresponds to a 1.5 ppbv difference.

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