Environmental impact assessment of commercial aircraft operations in the United States

by

Stephen P. Lukachko

S.M., Aeronautics and Astronautics, Massachusetts Institute of Technology, 1997
S.M., Technology and Policy, Massachusetts Institute of Technology, 1997
B.S.E., Aerospace Engineering, University of Michigan, 1992
B.S.E., Mechanical Engineering, University of Michigan, 1992

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Signature of Author: ______________________
Department of Aeronautics and Astronautics
August 31, 2009

Certified by: ______________________
Ian A. Waitz
Jerome C. Hunsaker Professor of Aeronautics and Astronautics, Department Head
Thesis Supervisor

Certified by: ______________________
Edward M. Greitzer
H.N. Slater Professor of Aeronautics and Astronautics
Thesis Committee

Certified by: ______________________
Henry D. Jacoby
Professor of Management
Thesis Committee

Certified by: ______________________
David H. Marks
Morton '42 and Claire Goulder Family Professor of Engineering Systems and Civil and Environmental Engineering
Thesis Committee

Accepted by: ______________________
David L. Darmofal
Associate Professor of Aeronautics and Astronautics, Associate Department Head
Chair, Committee on Graduate Students

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Abstract

The objective of this thesis was to evaluate the environmental trade-offs inherent in multi-criteria objectives of an integrated environmental policy. A probabilistic multi-attribute impact pathway analysis (MAIPA) was formulated to assess the environmental damages of US commercial aircraft operations from 1991–2003. The initial contribution of this work was demonstrating the feasibility of, and identifying requirements for, the FAA Aviation-environmental Portfolio Management Tool (APMT), an integrated assessment capability for US regulatory decision-making.

Non-aircraft sources have been found to dictate marginal emissions costs. The implication is that aviation emissions reductions influence neither the magnitudes nor trends in per-unit marginal damages. In contrast, noise mitigation is the dominant influence on the value of per-unit marginal damages. Trends in sum damages were found to depend on the growth rates of air transport relative to other source emissions. Growth in air transport emissions outpaced non-aircraft sources from 1991–2003. Because growth in marginal costs is nonlinear over this period, aviation emissions damages grow faster than inventories. Applying methods similar to MAIPA to estimate damages for future scenarios suggests that stemming climate impacts is fast becoming the priority.

A reassessment of the environmental benefits derived from mandated phase-outs of noisy aircraft during the 1990's has been carried out. Previous studies estimated a ~80% reduction in population exposure. In contrast, the reassessment estimates a ~2% reduction, providing benefits 17–20 times lower than published estimates of abatement costs. The primary environmental benefit of the noise phase-outs was found to be related to reductions in particulate matter inventories. One way to avoid trade-off inefficiencies is to identify options that bundle benefits. This action provides such an example, where the phase-outs led to reductions in both noise and air quality emissions.

Other contributions in the thesis include the following: a treatment of air transport particulate matter emissions, environmental fate, and health impacts of particulate matter; identification that the major source of reducible uncertainty in emissions damages stems from the assumed extent of ozone and particulate matter production in the engine exhaust plume; and quantification of the environmental trade-offs in decisions specifying aircraft performance for the technology in the US commercial fleet from 1991–2003.

Thesis Supervisor: Professor Ian A. Waitz
Title: Jerome C. Hunsaker Professor of Aeronautics and Astronautics and Department Head

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Dedicated to Sarah.

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Preface

Environmental impact assessments are contentious and this dissertation is not immune to the push and pull of critiques that frame the process—precautionary principle versus economic efficiency in decision-making, moral versus negotiated guidance of environmental goals, individual versus societal rights in policy design, or the overarching philosophical question of whether our technological constructions are part of the natural world. This thesis does not argue the correctness of any of these perspectives. It concerns the comparative benefits of pursuing mitigation of one agent of environmental impact versus another; it is an assessment of benefits, not a benefit-cost analysis. The assessment approach described in the following chapters is scientific, but its economic interpretation for decision-makers carries values to a larger extent. For some, the implied values are anathema to fair environmental policy.

This preface is here to recognize the importance of these discussions and to highlight the urgent need to communicate the science of environmental change, its uncertainties, and a measure of societal preferences for reducing impacts among the many vectors that stem from air transport. The only agenda in this research is to encourage a well-informed environmental practice, expanding opportunities to find solutions that help sustain air transport as a positive part the social fabric. Industry works to provide well-being to society (most directly to its customers and shareholders) primarily through an economic construct, so this effort interprets the social consequences of environmental change with preferences gleaned from observations of economic activity. This is an imperfect but rich medium through which to enhance the use of scientific understanding to set environmental goals in the context of the technological capabilities.[1] The discussions that follow adhere to the theoretical foundation of this approach, giving attention to both the advantages and limitations in the scientific understanding and economic interpretation of environmental change.

Over the final months of this thesis effort, the United States economy entered a recession amidst a series of global economic disruptions, prompting an intense and public reconsideration of our energy,

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[1] Crucially, this integration connects the objectively incommensurate metrics currently used to determine benefits, e.g. reducing noise footprints or emissions per-unit of fuel used, to the mainstream language of environmental impact assessment, establishing a heretofore absent medium for communicating environmental objectives across society. This facility places aviation environmental impacts and those of other anthropogenic activities in the same contextual reference; it also provides an improved capacity to incorporate social health and well-being as factors similarly innate to design and operational decisions as safety and security.

environmental, and infrastructure priorities. The actions taken to mitigate the downturn entail a profound repositioning of both public and private investments likely to reach all sectors of the economy. Therein lies a hazard and an opportunity; it is essential that we ameliorate our ability to thoughtfully evaluate investments in the air transport system for their efficacy and resilience to uncertainty. This thesis applies a pragmatic and theoretically supportable technique to identify the uncertain environmental import of our options and better inform decision-making.

Yet, there is no intention to be dogmatic. Economic assessments are incomplete reflections of how societies make decisions about what is right or fair for two reasons. First, the welfare lens perceives these values through the way people and organizations participate in economies and second, our ability to interpret values from economic decisions is uncertain. Taking from an example described in the introduction, the economic perspective suggests that deciding to reduce an aircraft noise signature relieves an environmental burden from airport-local communities, but adds a burden on other communities in exacerbating air quality and climate impacts, with the effect of reducing their wealth. These distributional observations are valuable to decision-makers, but there is no intrinsic cultural or historical context in these observations that allows us to decide whether such effect is socially acceptable.[2] In short, wealth is an insufficient gauge of societal concerns.

In this sense, relying exclusively on an economic interpretation of environmental risks to the exclusion of other approaches, such as those rooted in justice or moral concerns, limits opportunities to reduce environmental stress. This thesis cannot attain the scope needed to address other lenses and it is particular to air transport in the United States. This does not imply lesser importance of these alternative perspectives, rather that the methods developed here offer a clear and immediately practicable means to make a much needed connection to the extensive and salient knowledge base of environmental science. At the same time, it is important not to interpret the findings discussed presently as the outline for a comprehensive resolution the attendant environmental impacts of air transport. The hazard in presenting

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[2] Environmental damages are an inescapable consequence of providing mobility through air transport and are borne jointly by local airport communities, airlines, manufacturers, the flying public, and taxpayers, as well as global populations far away from airports or flight corridors. Committing to particular technological or operational changes requires some assurance of their effectiveness as remedies. Misguidance wastes valuable resources.

metrics in units germane to aircraft performance, as in this document, is the potential for a narrower discussion, perhaps one that is exclusively technical but presumptively comprehensive of the issues.

With these notes in mind, the thesis follows in seven chapters and seventeen (14) technical appendices.

Notation

Intervals, sets, and logic

$[x, y]$	interval between x and y including x and y
$]x, y[$	interval between x and y not including x and y
$\{x_1, \ldots, x_n\}$	set of values x_1 to x_n
$\exists$	there exists
$\forall$	for all
$\in$	element of
$\cup$	union
$\wedge$	logical AND
$\vee$	logical OR

Change, difference, and ratio

δx	marginal change in a *physical* parameter x usage: marginal economic parameters are denoted differently, see *Parameters*
Δx	change (non-marginal) in a physical *or* economic parameter x
$\Delta x\langle ref\rangle$	difference in parameter x against benchmark computations or data ref with the intent to demonstrate (or refute) consistency with the estimate of x where: ref = see abbreviations near the end of this chapter for ref identifications
$\varepsilon_x\langle ref\rangle$	error in parameter x in comparison to data or computations *ref* with the intent to demonstrate accuracy
$\varphi\langle x_1, x_2\rangle$	ratio of parameter x_1 to parameter x_2

Functions

P(*type*; *x1* ... *xn*)	probability function for variable x
F(*type*; *x1* ... *xn*)	distribution function for variable x where: *type* is the form of the distribution *x1* ... *xn* are the parameters characterizing the distribution define: using P but specification is equivalent for F
P(norm; μ, σ)	normal with mean μ and standard deviation σ
P(logn; μ, σ)	lognormal with geometric mean μ and standard deviation σ
P(gmm; $\mu1$, $\sigma1$, $\mu2$, $\sigma2$)	Gaussian Mixed Model composed of normal distributions defined by the parameters $\mu1$, $\sigma1$ and $\mu2$, $\sigma2$
P(expr; *source*)	experimental function using data from *source*
P(unif; x_1, x_2)	uniform with limits x1 and x2
P(trig; x_1, x_m x_2)	triangular with limits x1 and x2, and central value at x_m
$g(x)$	impulse response or Green's function of parameter x
$f(x)$	forcing function on the parameter x
PV(x)	present value

Statistics

$\hat{x}$	parameter estimator, denoted by a caret
$\underline{\theta}\langle x,w\rangle$	an averaged statistic θ of x using the weighting variable w
$\mu\langle \mathrm{x}\rangle$	mean of x
$\sigma\langle \mathrm{x}\rangle$	standard deviation of x
$\mathrm{CV}\langle x\rangle$	coefficient of variation of x $= \sigma\langle x\rangle/\mu\langle x\rangle$
$\tilde{\mu}\langle x\rangle$	median of x, denoted by the tilde
pct-R$\langle y:[x_1,x_2]\rangle$	range of y from percentile x1 to x2 where x1 is the lower limit (e.g. 0.40 = 40%) and x2 the upper limit (e.g. 0.60 = 60%)
$\mathrm{IQR}\langle y\rangle$	interquartile range = $R\text{-}pct\langle y:[0.25,0.75]\rangle$
$\widetilde{\mathrm{CV}}\langle x\rangle = \mathrm{IQR}\langle x\rangle/\tilde{\mu}\langle x\rangle$	the coefficient of variation defined relative to the median
$\mathrm{var}\langle y \mid x_i\rangle$	variance contribution of xi to parameter y
$\Delta\mu\langle y\vert x_i\rangle$	mean-shift of y with change in variable x
$\rho\langle y\vert x_i\rangle$	correlation of y with xi
SE	standard error of the mean
SE-r	relative standard error of the mean = SE/CV
CI	confidence interval, presented as in (95% CI)

Parameters

n_x number of a given quantity x

where $x = \{\text{ops, seats, psgr, ...}\}$

define: ops ≡ operations or flights (i.e. one landing + one takeoff)

seats ≡ seats

psgr ≡ passengers

i species index where $n_i = 8$

where: $i = \{CO_2, H_2O, NO_x, SO_x, PM_{nv}, PM_v, HC, CO\}$

define:

CO_2 = carbon dioxide

H_2O = water

NO_x = nitrogen oxides = $NO + NO_2$ (nitric oxide + nitrogen dioxide)

SO_x = sulfur oxides = $SO_2 + SO_3 + H_2SO_4$ (sulfur dioxide + sulfur trioxide + sulfuric acid)

usage note: parameters that reference sulfur emissions are typically computed in reference to S (sulfur) to emphasize their fuel origin

PM_{nv} = nonvolatile particulate matter

PM_v = volatile particulate matter

HC = hydrocarbons

usage note: HC transformed in the atmosphere subsequent to emission is referred to as VOC or volatile organic compounds; reactive organic gases or ROG refer to all reactive hydrocarbons in the atmosphere (e.g. VOC + biogenic organics + etc.)

CO = carbon monoxide

j representative aircraft type

where: $n_j = 19$

define: designations are given in table 3.1

k flight profile segment index where $n_k = 9$

where: $k = \{\text{it, to, cl, ci, ca, cr, da, di, ap}\}$

define:

it = idle/taxi to take-off

to = roll and takeoff

ci = climb to intermediate altitude

ca = climb to flight altitude

cr = cruise at flight altitude

da = decent from flight altitude

di = decent from intermediate alt

ap = approach and landing

il = idle/taxi from landing

see below for altitude definitions at h^y

l airport index or index of an airport-resident U.S. county

where: $n_l = 96$

define: see section 3.3 and appendix 4 for discussion of airport specifications

t time

$T[t_1, t_2]$ period of the interval from time t_1 to t_2

otherwise T = temperature as contextually appropriate, see below at T_x^y

d distance, always flight distance unless otherwise specified

r radius (otherwise economic rates of change as defined below at r et seq.)

h^y altitude of reference height y

where: $y = \{\text{mx, tr, in, fl}\}$

define: mx = mixing height, the edge of the atmospheric boundary layer

tr transition altitude, the point of takeoff power cutback for transition into the flight performance specifications of the first climb mode (i.e. k = to → ci) and similarly for descent (k = di → ap)

in intermediate altitude, the point of performance transition from the first to the second climb mode (i.e. k = ca → ci) and similarly for the descent (k = da → di)

fl flight altitude

T_x^y temperature at condition x at the reference location y

where: $x = \{k, t, f\}$

$y = \{a, h, s, 3, 4\}$

define: k and h are as defined above

t = a thermodynamic reference state, the total or stagnation condition

f = a thermodynamic reference state, the adiabatic flame temperature

a = ambient, usually specifically located by h_y or k as defined above

s = surface, always the surface of the Earth

3 = denotes an engine gas path location, the inlet of the combustor

4 = denotes an engine gas path location, the exit of the combustor

P_x^y pressure at condition x at the reference location y
where: x and y are specified as for T_x^y above

ρ_x^y density at condition x at the reference location y
where: x and y are specified as for T_x^y above

F thrust

W weight

u speed

q_x^y per-unit quantity production (or destruction) of parameter x, sometimes further
specified with location, time, or activity reference y
usage: the per-unit specification for q is per-flight unless otherwise specified

$\dot{q}_x^y$ rate of quantity production (or destruction) indicated by over-dot

Q_x^y inventory of parameter x , sometimes further specified with location, time, or activity reference y
define: for $\{q,Q\}$: x = $\{f,i,n\}$ as indicated above at c_x
y = $\{>h_{mx},<h_{mx},k\}$
where: $>h_{mx} \equiv$ the free troposphere
$>h_{mx} \equiv$ the lower troposphere
k as defined above at k

η efficiency

EI$i\langle y\rangle$ emissions index of species i with reference to y
where: y can be a location or time
y can be an activity
y can be a sourced reference

e environmental quality

p environmental pollution (p = -e)

DNL day-night noise level

LAmax is the maximum sound level over the duration of the noise event LA(t)

RF radiative forcing

RFi instantaneous forcings

X_i concentration of species i

ϱ^l population density of airport-resident county l

w welfare

I_x incidence of condition x

c_x marginal damage cost of parameter x

where: x = $\{f, i, n\}$

define: $f \equiv$ fuel consumption

$i \equiv$ species as described above at i

$n \equiv$ noise

usage: 'marginal damage cost' sometimes abbreviated 'marginal damage' or referred to as 'marginal willingness-to-pay' or MWTP

c_a marginal abatement cost

c_s marginal social cost related to the sum of private and external costs

C total damage cost

usage: 'total damage cost' sometimes abbreviated 'damage' or referred to as in 'willingness-to-pay' or WTP)

r discount rate or the real rate of capital return

r_g rate of consumption growth

r_t time discount rate

θ marginal utility of consumption

$r_g\langle x, [t_1, t_2]\rangle$ compound annual growth rate or CAGR of parameter x over period $T = [t_1, t_2]$

where: T is as defined above

1. Introduction

Air transport development relies on the effectiveness of technology investments to stem the environmental impacts of aircraft operations (NSTC 1995; DfT 1998; EC 1999; NSTC 1999; EC 2001; DfT 2003; FAA 2004d, 2004e; ICAO 2004; JPDO 2004; PARTNER 2004; SASC 2005). Progress in understanding these impacts suggests a number of potentially important environmental objectives. However, their prioritization is uncertain and current resources do not offer a way to make the necessary comparisons.

1.1. Motivation

The potential risks for environmental investments is significant, particularly because the scale of technological change required to reduce impacts may encompass the entire air transportation system. Uncertainty in prioritization is one component of these risks. For decision-makers, this means not knowing the environmental consequences of choosing one mitigation option over another, or how to determine which among several performance specifications provides the most desirable reduction in impacts. The goal of this thesis is to establish assessment capabilities that direct decision-makers toward positive outcomes.

1.2. Objective

Specifically, we want to define the metrics and a means for their evaluation to quantify the environmental trade-offs inherent in design and policy choices.

The objectives of this thesis are to evaluate: (1) metrics to compare the influence of aircraft performance characteristics on environmental change; and (2) methods to incorporate integrated analysis in regulatory assessment tools.

1.3. Approach

To accomplish these objectives, this work uses the environmental costs, or damages, of emissions and noise to compare impacts of changes in climate, air quality, and community noise. To estimate damages, we develop a probabilistic multi-attribute impact pathway analysis (MAIPA) to model the environmental costs of emissions and noise due to changes in climate, air quality, and community noise. These costs represent the economic choices people may make to avoid risks to their health and well-being from environmental change. This approach is methodologically consistent with regulatory norms established in the United States, Canada, and Europe.

1.3.1. Retrospective assessment of US commercial aircraft operations from 1991–2003

MAIPA is used to provide an historical assessment of the environmental impacts of commercial aircraft operations in the United States between 1991-2003.[1] These impacts are characterized by inventory, environmental, risk, and economic metrics, and, ultimately, estimates of environmental damages in terms of aircraft performance parameters. The resolution of results is national and yearly, with the exception of inventories, where results are quarterly.

1.4. Context

Two recent examples illustrate how environmental choices arise in aircraft design and operation and how the results of MAIPA-type assessments clarify decision consequences.

1.4.1. Airline orders reduction in aircraft noise at expense of fuel efficiency

The first Airbus A380, a new very-large commercial transport aircraft with primary application to long, heavily-traveled routes, entered service early in 2007. From the start, environmental performance was a key criterion for the new design, but one of several including operability, safety, security, efficiency, and market applicability. The launch customer, Singapore Airlines (SA) in this case, had an important role in setting requirements.

For SA, operation through London's Heathrow Airport was vital to the success of the A380 in their markets. Heathrow is one of several airports around the world that restrict aircraft operations based on their noise characteristics. These noise rules are in place to minimize impacts on communities close to the airport. SA requested that the A380 design meet restrictions typically applicable to any large commercial aircraft currently in-service, such as the A340 or Boeing B777.

[1] Other aviation environmental impacts exist: production and disposal of aircraft, flight services (food provision, etc.), and infrastructure (such as airside operations). Production impacts are expected to be small ref RCEP, but the author is not aware of any examination concerning disposal issues. Airside operations not associated with flight operations (e.g. fueling, baggage delivery, etc.) and ground-side activities have been assessed elsewhere.

When this request was made, the design process had progressed to a point where Airbus had limited options to respond. Its best option, given resource constraints, was to trade a 1–2% decrease in fuel economy to achieve a 1–2 dB noise reduction and thus a comfortable margin of compliance.[2] An SA A380, the second placed in service, made its first landing at Heathrow on March 18th of 2007.

1.4.2. Increased noise exposure halts airline tests of fuel reduction procedures

The A380 design choice reflects a competition among environmental goals that is replicated in operational choices. Pressure from local communities to mitigate airport impacts, particularly noise, has been an historically significant influence on operations. This is most directly felt by airlines, but as they respond to regional- and global-scale environmental issues, operational decisions that favor reductions in fuel and emissions will be increasingly commonplace.

Consider this recent news thread reported by the Cairns Post. Qantas is an Australian airline that serves domestic and global destinations. In a trial flight path adjustment to lower fuel consumption for southerly arrivals and departures, aircraft were diverted over the city of Cairns in Queensland, increasing noise for residents of the Esplanade district and the Mulgrave River Valley to the southeast (Irby 2007a, 2007b; Koch 2007; Murtagh 2007).

Local protest ended the experiment, returning the status quo. Is the status quo a more favorable balance among noise and emissions impacts? The Cairn's community is trading the hidden costs of noise for those of air quality. The results reported in chapters 5 and 6 of this thesis suggest that their position makes sense from a household perspective. For people exposed to aircraft noise, noise impacts from commercial aircraft operations in the United States between 1991–2003 is estimated to be ~80 times higher than costs resulting from their exposure to emissions impacts.

From a societal perspective, including all regional and global emissions impacts, this equation changes. Again based on results shown later in this thesis, the median annual noise cost for the United States between 1991-2003 accounts for ~1/8th of the estimated sum environmental costs of commercial aircraft operations. In the absence of a way to communicate a comprehensive account of health and welfare risks, there is no voice for the populace outside the region.

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[2] Personal communication. 2004. A. Joselzon. Head of Engineering Environmental Strategy, Acoustics and Environment Department. Airbus France SAS.

1.5. Contributions

There are six primary contributions from this work. These contributions fall into three categories: expanding the capabilities and scope of air transport environmental impact assessment, understanding how environmental impacts are related physically and technologically, and identifying potential mitigation approaches. Additional contributions are stated in the introductions to chapters 2–6. The conclusion chapter 7 reviews these contributions with further discussion of implications.

1.5.1. Integrated approach to air transport environmental impact assessment

Conducted pathfinding research for development of the FAA Aviation-environmental Portfolio Management Tool (APMT).

An important contribution of this work was in demonstrating the feasibility of, and identifying requirements for, the Aviation-environmental Portfolio Management Tool (APMT). APMT is currently under development for application to regulatory decision-making in the United States.

1.5.2. Particulate matter impacts of aircraft emissions

Introduced a treatment of air transport particulate matter emissions, environmental fate, and health impacts of particulate matter.

This thesis extends the scope of impact assessment with a comprehensive treatment of particulate matter emissions. Estimated particulate emissions indices were applied to evaluate the first mass-based PM inventories specific to the operational performance of US commercial aircraft. Through these models, this work identifies precursor emissions as primary sources of environmental damages. Treatment of PM was a necessary basis for the contributions stated in sections 1.5.3, 1.5.5, and 1.5.6 below.

1.5.3. Uncertainties in impact assessments

Identified that the major source of reducible uncertainty in emissions damages stems from the assumed extent of ozone and particulate matter production in the engine exhaust plume.

This thesis evaluates the comparative importance of parametric, structural, and scenario uncertainties in estimated damages. Of these uncertainties, the role of the engine exhaust plume chemistry and microphysics is a primary uncertainty in estimating the change in ambient ozone and particulate matter concentrations due to aircraft operations. With the assumptions made in this thesis, estimated air quality damages are approximately 60% higher than when using the assumptions of large-scale complex air quality models such as EPA CMAQ.

1.5.4. Environmental damages

Identified that the most important factor determining changes in damages over time is the dependence of emissions damages on the background environmental sensitivity.

Emissions impacts of US commercial aircraft are dictated by the progress in controlling emissions from other sources. The attribution of trends to parametric inputs shows that air transport emissions impacts are predominantly determined by the background environmental sensitivity, indicating that the growth of air transport emissions relative to other sources is the key factor that determines damage costs.

1.5.5. Integrated approach to impact mitigation

Reassessed the environmental benefits of the aircraft retirements mandated by the 1990 Aircraft Noise and Capacity Act.

From December 31, 1994 through December 31, 1999, FAA mandated a scheduled phase-out of portions of the commercial fleet identified by their failure to meet a limit on noise levels (14CFR91.801-877 Subpart I: Operating noise limits). Analysis of noise trends during these phase-outs finds reductions in noise exposure provided benefits approximately 10 times lower than estimates of the associated phase-out costs published during and after the final compliance date. Combining results from emissions and noise, it is further shown that, by a 2:1 margin, more of the benefit from the Stage 2 noise phase-out rule came through reductions in VOC and PMnv emissions than from noise reduction.

1.5.6. Environmental trade-offs in aircraft technology

Quantified the environmental trade-offs in decisions specifying aircraft performance for the technology in the US commercial fleet from 1991–2003.

The traditional objectives of design toward regulatory standards or the market are marked by minimum NOx, minimum fuel consumption or minimum noise. A more comprehensive perspective recognizes that different sets of environmental performance characteristics can provide equivalent levels of welfare. Trade-offs are estimated using environmental damages and presented as elasticities of performance, or percent change in one performance parameter equivalent to a percent change in another. A damage function describes the sensitivities to performance changes in the US commercial fleet from 1991–2003.

1.6. Chapters

The contents of chapters 2–7 are summarized below. Each incorporates uncertainty analyses into the discussion. Relevant appendices are referenced in each chapter where they appear as background to the material.

- Chapter 2 formulates the multi-attribute impact pathway analysis or MAIPA.
- Chapter 3 evaluates the historical environmental performance of the US commercial aircraft fleet, specifying the fuel, emissions, and noise inventory inputs for the 1991–2003 retrospective assessment.
- Chapter 4 formulates methodology to estimate impacts on the global climate that distinguishes the value of reducing CO2 emissions versus non-CO2 emissions, and estimates marginal and sum climate damages.
- Chapter 5 formulates a methodology to estimate impacts on air quality that distinguishes the value of reducing NOx, SOx, HC, PMnv, and CO emissions that accounts for the formation of ozone and particulate matter. Estimates of marginal and sum air quality damages are reported. In addition, an analysis of marginal emissions costs, including fuel consumption, is presented.
- Chapter 6 formulates a model to estimate trends in noise exposure as a function of a cumulative noise metric. Correlations among trends in air transport noise and emissions damages are discussed and a reassessment of the noise phase-outs mandated by the 1990 Airport Noise and Control Act is presented. In addition, chapter 6 provides a comparative analysis of sum climate, air quality, and noise damages.
- Chapter 7 summarizes the primary contributions of this thesis, including estimates for the environmental trade-offs in policy or design decisions made to specify aircraft performance for the technology in the US commercial fleet from 1991-2003. Chapter 7 also discusses implications and suggests next steps.

2. Integrated air transport environmental impact assessment

The first objective of this thesis is to develop metrics and a means for their evaluation to assess the comparative influence of aircraft performance characteristics on environmental change. This chapter formulates a probabilistic multi-attribute impact pathway analysis (MAIPA) for this purpose. Its goal is to improve the quality and content of decision information available through assessment tools and in doing so, to promote better decision-making for effective mitigation policy.

Contribution 2.1. Conducted pathfinding research for development of the FAA Aviation-environmental Portfolio Management Tool (APMT)

MAIPA is unique as an integrated approach to air transport environmental impact assessment.[1] An important contribution of this work was in demonstrating the feasibility of, and identifying requirements for the Aviation-environmental Portfolio Management Tool (APMT). APMT is currently under development for application to regulatory decision-making in the United States.

APMT represents a fundamental change in aviation-environmental assessment, moving from air transport environmental impact assessment based on quantity-based impact metrics, effect-by-effect analyses, and cost effectiveness decision-making, to one based on a comparative evaluation of multiple impacts—more directly responsive to risks to health and well-being, mindful of uncertainties, inclusive of a broader scientific knowledge base, and prepared to evaluate the synergies or incompatibilities among regulatory options.[2]

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[1] These systems are inherently complex. Commercial air transport consists of interdependent systems driven historically by safety and security requirements as well as economic motivation. The enormous intellectual and capital resources invested to establish this system compose a large technological inertia; change requires a similarly large impulse, carrying an expense and persistence that necessitates careful consideration of investment objectives. In turn, the environmental processes perturbed by air transport weave sources, transformations, and impacts into a latticework of physical and social change that resists decomposition. The geographic scales and persistence of these changes alter the livelihoods of individual communities through accumulative and acute impacts that last hours or days, as well as generations living centuries hence that will experience the lagging consequences of industrial activity today.

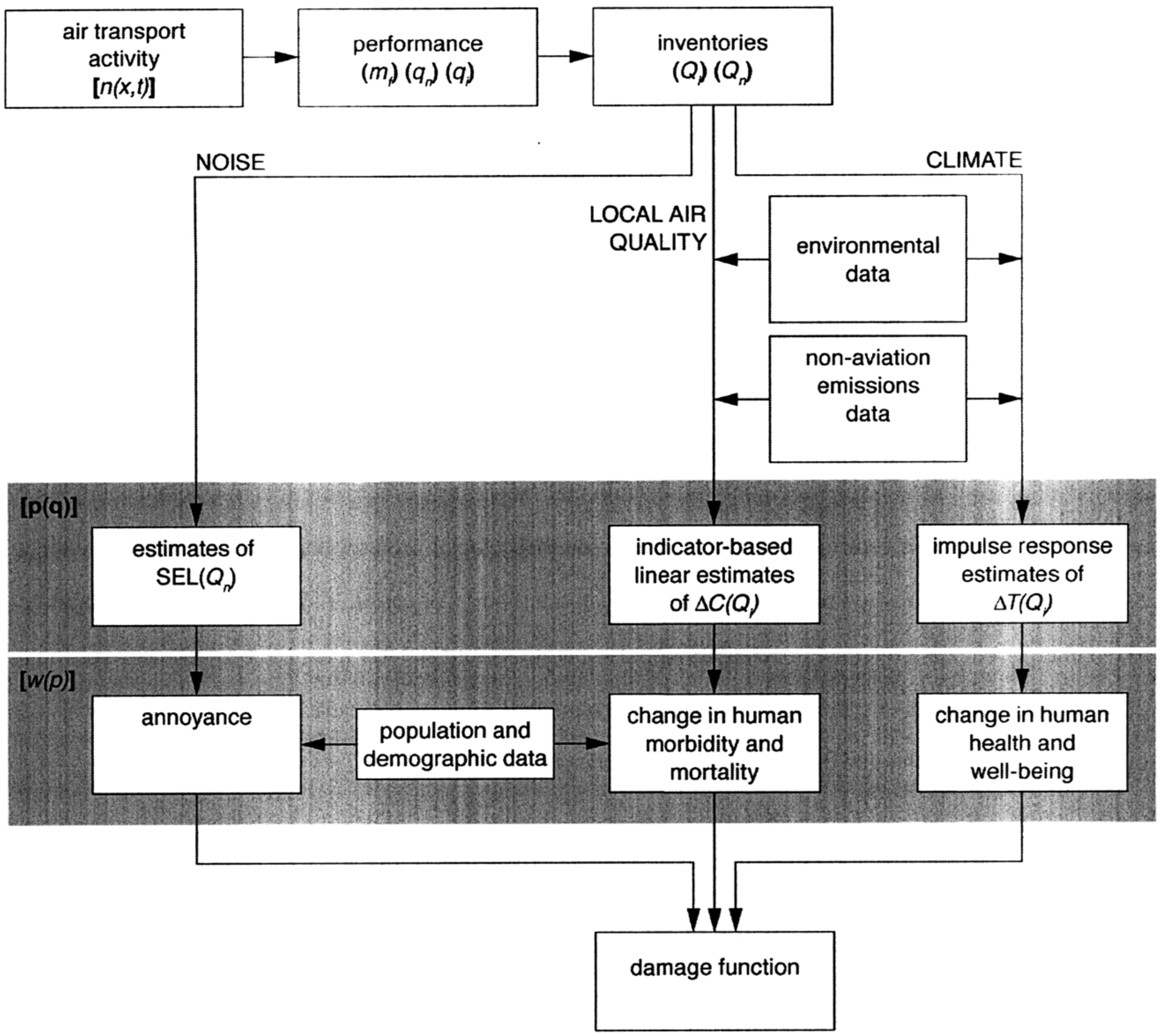
[2] The resolution of trade-off issues lies arguably within the scope of national or international policy; local situations are informative, but not a surrogate for decisions directing investment towards system-wide change. In the air transport context, formulating policies with this broad intent suffers from a limited and uncoordinated flow of scientific information useful for assessment, a situation which places local conflicts at the fore of decision-making. This thesis speaks to this broad perspective by clarifying the relationship between air transportation and the desires of society for environmental health as a part of their well-being. What is needed is a productive means to communicate a rich understanding of the scientific and technical knowledge that defines environmental in a form assimilable into present-day decision mechanisms.

Sections 2.1-2.4 describe MAIPA procedure, with supplemental discussion in Appendix A1. Section 2.5 describes the evaluation of trade-offs using information contained in the damage function. Appendix A2 supplements section 2.5.

2.1. Multi-attribute impact pathway analysis

The evaluation of an impact vector is commonly called an impact pathway analysis; this document uses MAIPA or multi-attribute impact pathway analysis to refer to the unified assessment of many impact vectors. MAIPA mechanistically traces multiple impact vectors that result in climate, air quality, and noise impacts. This is a bottom-up approach, following a chain of environmental effects initiated by the introduction of emissions and noise to their ultimate impact on people. Figure 2.1 sketches a simplified representation of MAIPA. The pathways are grouped by broad categories of impact with vectors pointing left to right.

Figure 2.1. Schematic of the multi-attribute impact pathway analysis



The viability of an assessment is both a technical problem and an issue of public accessibility.[3] The common approach to impact pathway analysis is to treat each chain separately; this picture of environmental impacts, which follows the historically step-by-step discovery of their mechanisms, is the primary reason why regulatory structures are similarly divided in the United States. These are false boundaries, physically and socially; the integrated analysis here examines how impact vectors interact.

2.1.1. Developmental requirements

Assessment metrics need to directly tie actionable technological characteristics with socioeconomic characteristics. Requiring connectable metrics limits the assessment to a subset of theoretical effects and, further, those where the literature on these effects is sufficient to support their inclusion in policy analyses. For example, we use pecuniary environmental metrics, where an intermediary, such as property (e.g. noise) or health (e.g. air quality), as proxies to describe social preferences for environmental quality. A complete impact pathway model can thus be constructed only for certain risks as limited by our ability to evaluate effects in a manner germane to aircraft operations and specify uncertainties (cf. next section 2.2). Early economic assessments of aviation environmental impacts typically applied assessments of other sources to estimate the damages due to air transport.[4]

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[3] *Note on metric characteristics:* There is no reference approach to this task; in lieu, a set of guidelines were developed as part of the thesis effort. The guidelines identify desirable assessment metrics as those that are ordinal, quantitative, connectable, and transparent; these characteristics are common-sensical, but often ignored in making analyses accessible to decision-making.

(1) *Ordinal*: Metrics and related estimation methods are capable of ordering and specifying preferences for environmental quality as the basis for integrative decision-making. With technical metrics, such as total carbon emissions, this characteristic is typically innate, but not necessarily for socioeconomic metrics where their definition often includes assumptions about what is a preferable social order. Ordinality allows prioritization of environmental objectives and explicates mechanisms through which trade-offs among objectives are made.

(2) *Quantitative*: Uses cardinal metrics and methods where magnitude is meaningful, but not necessarily exclusionary; that is we expect uncertainty. Ordinality does not state the extent to which one objective is preferred to another. For this, we need an account of magnitude. Thus, we employ when possible cardinal metrics and methods where magnitude is comparatively meaningful. There exists a set of mitigation options $X = \{x1, \dots, xn\}$ where we can define a probability function where the set contains ΔC specified for each option x; ΔCx is the difference between a reference or baseline condition and one perturbed by implemented mitigations. This study uses a social welfare metric for comparative purposes to meet this guideline, but does not mean an assessment metric should be monetary.

(3) *Connectable* (and *calculable*): Methods and outcome metrics directly tie actionable technological characteristics with socioeconomic characteristics.

(4) *Transparent*: To the extent practicable, methods and outcome metrics are publicly accessible.

[4] Aircraft are mobile, but their origination from airports leads to point source impacts. Most aircraft emissions uniquely occur above the atmospheric boundary layer, and these emissions can be transported back into the surface mixing layer leading to distributed air quality impacts that are atypical of point or mobile sources. Emissions into the upper atmosphere lead to a host of short-lived radiative effects which add to the commonly appreciated effects of carbon dioxide. Noise is higher and not as localized as other mobile sources.

These methods balance the need to minimize the loss of decision-relevant information and maximize its quality. Computations are fully probabilistic, accounting endogenously for uncertainty and variability in input parameters. The first of these implies minimum complexity to the models in order to represent the path from source to damage and still provide useful assessment information. Effects are excluded most often by the availability of observational support, often due to immature measurement capabilities. For this analysis we need technological, operational, environmental, and socioeconomic data to specify required metrics. Admitting methods into assessment approach considers the availability of validation data and the potential for demonstration exercises.

Methods were required to maintain internal consistency in fidelity among the steps in the impact pathway analysis. Each impact pathway evaluated has a resolution-limiting input. Noise valuations, for example, involve meta-analyses that aggregate results from airports spanning the United States, Canada, Europe, and other regions to identify demographic dependencies. Valuing disease incidence is similarly conducted using nationally-averaged heath care costs. For climate, regionalized impact assessments are emerging, but underdeveloped for the assessment practice pursued by this thesis; globally-averaged metrics are the best understood gauges of impacts.

This analysis is specific to environmental impacts directly resulting from flight operations in the United States.[5] Impact assessments are resolved only to the national scale and, where uncertainty allows, the finest time scale is yearly (the exceptions are inventory estimates where the resolution is quarterly). Since metrics are estimated using a retrospective analysis of activity occurring from 1991–2003. Thus, results are intended describe comparative environmental impacts as they exist today, focusing on understanding key uncertainties and identifying relationships that suggest mitigation opportunities.

2.1.2. Scope and resolution limitations

It is critically important to consider the validity of applying the results of the analysis or extrapolations thereof for prospective assessments in the context of the uncertainties and limitations described above. There are limitations on the scope and resolution of MAIPA as applied in this thesis: (1) results are specific to environmental impacts directly resulting from flight operations in the United States; (2) results

[5] Aviation regulatory and design schedules suggest that exploration of decision spaces should be conductible within 3-6 months of initiation in order to allow detailed evaluations of identified options.

are based on a retrospective analysis of activity occurring from 1991–2003; (3) impact assessments are resolved only to the national scale and, where uncertainty allows, the finest time scale is yearly (the exceptions are inventory estimates where the resolution is quarterly); and, (4) include those modes of environmental change calculable along the entire impact pathway.

2.1.3. End-user considerations

An assessment metric needs context. A metric of risk or loss is a poor intermediary if its measure is opaque; this makes it difficult for people to express these preferences, and for decision-makers to obtain useful assessments. For example, the estimation of welfare metrics is theoretically complex, but their monetary expression is comprehensible. But it is helpful if the estimation method is as transparent as possible; methods are preferred that speak directly to stakeholder application (e.g. connecting aircraft performance parameters to risks) and rely upon public information to make transparent the interpretation of results.

Because the state of understanding of particular societal needs will continually change, metrics and methods are further useful if they are amenable to evolving interpretations of causes and effects. In this sense, metrics with invariant interpretations over geographic and historical scales best support the long-term viability of the analysis approach. For impact assessment technique in specific, assessments address air quality impacts at the scale of aggregate industries, particularly power generation and road transport (Small and Kazimi 1995; Mayeres et al. 1996; Levy et al. 1999; McCubbin and Delucchi 1999; Banfi et al. 2000; Matthews and Lave 2000), and assess the national impacts of specific emissions species and noise (Nelson 1978; Oates et al. 1989; Cifuentes and Lave 1993; Delucchi et al. 2002), often for the regulatory assessment of air quality programs (EPA 1997b, 1999c; EC 2003; Nash 2003; Zhang et al. 2004). MAIPA departs from this experience by integrating assessments of emissions and noise impact vectors, using a probabilistic formulation, and developing an assessment practice applicable to air transport.[6]

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[6] Other aviation environmental impacts exist: production and disposal of aircraft, flight services (food provision, etc.), and infrastructure (such as airside operations). Production impacts are expected to be small ref RCEP, but the author is not aware of any examination concerning disposal issues. Airside operations not associated with flight operations (e.g. fueling, baggage delivery, etc.) and ground-side activities have been assessed elsewhere.

2.1.4. Procedure and component models

The impact pathway analysis focuses on common stages of transformation: (1) defining sources and their activity to characterize pollutant inputs; (2) estimating affected changes in environmental quality; and then (3) estimating the decrease in social welfare expected in the presence of these risks. This perspective helps retain focus on consistency in inputs, model fidelity, analysis scope, and geographic and temporal resolution.

Reduced-order source characterization and environmental models were developed for MAIPA (cf. next section at 2.3 and chapter 4), capable of explicating first-order influences, but at a scope and resolution lower than current assessment tools used by FAA and others. Computational burden was also considered; estimation methods are practicable only if they can evaluate statistically significant results in the time frame of an assessment.[7] Constituent models address aircraft performance, emissions chemistry and microphysics, atmospheric chemistry, transport, and radiative processes, disease incidence and mortality, and resource system stresses. Models use common operational data, technological parameterizations, and environmental and socioeconomic conditions. Chapters 3–6 describe the evaluation process in more detail.

Estimation of the pollution input to the environment as emissions and noise through a comprehensive source characterization of the aircraft fleet. Impacts on environmental quality are defined by three metrics: (1) changes in the global climate using surface temperature as the metric of environmental change; (2) changes in air quality measured by changes in atmospheric pollutant concentrations; and (3) changes in environmental noise measured by frequency-weighted sound exposure levels. Only in for air quality analysis are risks to the well-being of exposed populations explicitly determined, measured collectively by the incidence of disease, mortality, and other changes in livelihood. Comparative societal preferences for improvements in environmental quality use observations of societal economic behaviors expressed using the currency metrics conventional to economic analysis.

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[7] Previous studies often apply marginal valuations of impact (e.g. cost per unit emission or noise) estimated for other economic activities (e.g. automotive transport, electricity production, etc.) and assume they are applicable to aviation. However, the spatial distribution of aircraft effects (vertical as well as horizontal), and unique character of impacts (e.g. cloud interactions at altitude) make this benefit transfer practice questionable. This is a common practice, however, and marks most efforts to evaluate the environmental costs. Initial efforts undertaken by the ICAO CAEP to employ cost-benefit analysis in evaluating new proposals have used benefit transfers across industries. To reduce the social and economic distances made in executing such transfers, valuations of change in environmental variables in MAIPA are reserved until after assessing the impacts of commercial aviation.

2.1.5. Evaluation for air transport environmental impact assessment

Subsequent to a comprehensive review under the auspices of the Transportation Research Board (TRB) of the U.S. National Academies of Science and Engineering, the outcome of this three-year process was the definition of requirements, architecture, and a prototype work plan for development of the Aviation-environmental Portfolio Management Tool (Waitz et al. 2006b, 2006c); the initial MAIPA formulation served as the APMT template for these definitions. The decision-making utility of APMT was formally recognized internationally at the 7th Meeting of the Committee on Aviation Environmental Protection (CAEP) of the International Civil Aviation Organization (ICAO) in 2007.

APMT represents a significant and timely change, led by the U.S. Federal Aviation Administration Office of Energy and Environment, in the way national air transport decision-makers approach aviation-environmental issues. The incorporation of the necessary assessment capabilities for economic analysis within APMT has facilitated FAA engagement in shaping global air transport environmental policy. The repertoire of mitigation options that can be addressed using current air transport EIA practice has extended to include market-based approaches and the necessary assessment capabilities have also been built into APMT. These developments are crucially important toward addressing the significant challenge of reducing climate change.[8]

2.2. MAIPA application toward improvement of APMT capabilities

Initially MAIPA was a vehicle to demonstrate the feasibility of benefit estimation in the context of existing practices. MAIPA has also been valuable to investigate potential improvements in the methods and knowledge content of assessment practice, particularly as they inhabit APMT.

MAIPA and APMT today address the benefit estimation problem with different but complementary emphases; table 2.1 compares the methodological approaches and scope in their their current formulations. MAIPA, like APMT, is generally consistent with regulatory guidelines and benefit

[8] APMT is one element in the first complete air transport EIA cost-benefit analysis (CBA) capability for use in national and international decision-making. APMT has been exercised to evaluate tightening NO_x engine emissions limits and the benefits of converting to low sulfur aviation fuels. Currently, APMT assessments include the climate benefits of infrastructure improvements and the value of CO2 engine emissions controls. APMT development continues to expand assessment capabilities. It is important to stress that couching a capability like APMT within the confines of standard-setting for noise and emissions, the historical norm, underestimates the transformative solutions that will be sought through its application. However, the need to pursue alternative sources of energy to power the future aircraft fleet and stem global change does not lessen the consequences of air quality and noise effects.

assessment practices in the United States, Canada, and Europe. Where MAIPA and APMT differ is primarily in resolution and scope; MAIPA trades scope and resolution in order to evaluate methodological choices and trade-offs, and to provide tentative benchmarks for APMT results, while reducing computational requirements.

Table 2.1. MAIPA and APMT approaches to air transport benefit analysis

assessment	MAIPA	APMT	typical aviation-environmental practice
geographic scope	national (U.S.)	global	route (Europe)
activity resolution	representative aircraft types to construct fleet inventories	specific aircraft models to construct location specific or fleet inventories	specific to aircraft type
impact coverage	integrated, activity base consistent across impacts assessed	integrated, activity base consistent across impacts assessed	not integrated; impacts assessed separately with different activity inputs
temporal resolution	quarterly for inventories; yearly for impacts	flight-by-flight for inventories; hourly-daily depending upon impact time-scale	yearly
current application	historical for developmental assessment	prospective for regulatory assessments	prospective for policy development but often for indeterminate point in time
treatment of uncertainty	probabilistic at computational level to assess parametric uncertainty; scenario analyses to assess impact of physical model construction and economic assumptions	probabilistic at computational level to assess parametric uncertainty; scenario analyses to assess impact of physical model construction and economic assumptions	deterministic; no probabilistic computation; individual application specific scenarios
air quality modeling	linearized measurement-based	3-D chemistry transport (CMAQ)	
climate modeling	impulse-response	impulse-response	use physical benefit transfers
noise modeling	probabilistic	INM	

2.2.1. Probabilistic formulation

The importance of communicating uncertainty in environmental analyses is extensively recognized (Iman and Helton 1988; Morgan and Herion 1990), particularly in application to climate change (Manne and Richels 1995; Morgan and Keith 1995; Allen et al. 2000; Webster et al. 2003). We are ultimately interested in a probabilistic approach because the end-use problem is one of managing decision risk.[9] Three aspects of fidelity require attention in the MAIPA: representing uncertainty in model form (structural), considering fidelity in the representation of actual processes; scenario uncertainty, referring to inestimable effect contributions; and specifying uncertainty or variability in input variables (parametric).

The analysis of results focuses on parametric and structural uncertainties. Since this thesis bases damage estimation on a retrospective analysis, scenario uncertainties are assessed only in cases where the lifetimes of physical effects or economic development persist beyond the study time period (i.e. climate change). In the face of uncertainty about the future, it can be attractive to use intermediate metrics, such as inventories, as surrogates for environmental quality. However, intermediate metrics often do not correlate with empirical societal risks (cf. chapters 3-4).

2.2.2. Specification of parametric uncertainties and computation

Practically, the propagation of parametric uncertainties places stricter limits on the resolution of trends and the spatial heterogeneity of impacts.

See Appendix A1 for additional discussion

- Appendix A1 (Computation and analysis conventions) details the approach to specifying parametric, scenario, and structural uncertainties, the computational implementation of uncertainty propagation, and the approach taken to validate results where measurements may or may not be available.

2.2.3. Identification of structural uncertainties

Structural issues are examined through comparison of model outputs with data and outputs from other models with similar functionality. Only certain elements of the impact pathway can be assessed directly using exogenous data sets (i.e. fuel use and particulate matter emissions measurements). A judicious use (i.e. evaluated against the fidelity guidelines) of measurement data is preferred over employing models to achieve the same output. This limits the extent to which structural issues affect error by confining such

[9] Most people and organizations are risk averse, and there is often the opportunity to combine uncertain assets in such a manner as to maximize a risk-weighted expected return. Nonlinear models generally result in output distributions that are skewed, suggesting that risk may be differentially weighted above or below the mean. Even if outputs are not skewed, costs related to various outcomes may be uneven across the output distribution. Given uncertainty in cost estimates (as in a cost-benefit framework), the expected value of the net surplus may be assumed a measure of welfare (Freeman 1991). Risk preferences are not addressed by the analysis, but there is value in reducing uncertainty for a risk averse populace. In economic terms, for the distribution of environmental costs in particular, certainty equivalents for the benefit stream would be smaller than expected value.

considerations to questions of theoretical plausibility or, more commonly, temporal and geographic resolution.

The latter situation occurs in the context of estimating changes in environmental quality. Chapter 4 introduces these models and reviews the improvements in comparison to APMT results. Consistency between MAIPA outputs and modeling results from equivalent literature studies supplements for the lack of empirical data for comparison. For fuel use and emissions estimates, these comparisons only serve to define the extent to which structural uncertainties may affect output metrics. However, for other situations, comparisons identify structural uncertainties potentially important to the focus of policy efforts, such as the change in air quality impacts stemming from the effect of plume processing on ozone and secondary PM formation (cf. chapter 5).

In contrast, several results from later stages of MAIPA have no equivalents (economic outcomes in particular) and applied methods do not give us the benefit of experimental verification. These results stand on the merits of metric and method choice (thus the need for the guidelines above), specifically the robustness of the underlying theory and community experience with their practice. There are no empirical examples of policy application that might be used to confirm that the fidelity of this analysis is suitable for designing specific decision options, particularly those that are market-based, but this is not a unique problem among environmental assessments. The United States Environmental Protection Agency (EPA) has examined the national experience with economic incentives for other purposes (EPA 2001c), but there is no follow-up work to identify whether the underlying analyses enabled good policy design. Validity (which includes accuracy assessment) is ultimately judged by the success of policies ex post. This thesis does evaluate one previous regulatory effort, the mandated phase-out of noisy aircraft, to establish the beginnings of the type of evaluation that seeks to understand how policy design conforms with its intent (cf. chapters 3 and 6).

2.2.4. Methodological limitations under an economic framework

The economic framework through which trade-offs are identified is perhaps the most restrictive of the assessment steps. The damage function is a means to compare ways of addressing aviation environmental issues and in this sense, monetization bundles different goods so they may be juxtaposed against other bundles. The use of an economic framework is by definition limited to those impacts which can be inferred through observations of transactions or which can be related to a reference transaction such that a given environmental impact has an economic correlate.[10] Valuation methods are categorized broadly by the manner in which they seek to determine preferences for environmental amenities in the absence of a market. They are also differentiated in their ability to evaluate use-related and non-use impacts resulting from a change in environmental quality, and are often limited to evaluations of willingness-to-pay at the margin due to the restrictions of metrics for amenities such as clean air and quiet (i.e. a complete marginal damage curve is difficult to obtain). There are difficulties separating these non-market goods from bundling with other private and public goods.

This analysis relies primarily on the extensive basis of theoretical development and application experience with valuation techniques that use statistical inference or cost data directly to infer preferences for environmental quality. This restricts the detailed valuation of environmental impacts to using costs (i.e. physical damages and their resultant impacts) where surrogates for environmental effects are measured through changes in markets. Alternative methods are available for some impact endpoints to assess the valuation of non-use attributes, a particularly contentious issue in the case of climate change. Regardless, this is a situation where the economic lens may simply be at the limit of its field-of-view; other avenues of decision advice may be better options.

Tables 2.2 and 2.3 overview impacts that are and are not evaluated through MAIPA with an estimated magnitude of effect and quality of research available for decision making noted where possible. The absence of impact vectors biases marginal costs downward with the largest factor likely to be the multiplier effects of environmental damages. Unanticipated swings in future results are an unavoidable consequence of continued research in the relatively young endeavor to quantify the impacts of aviation on the environment.

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[10] Most people and organizations are risk averse, and there is often the opportunity to combine uncertain assets in such a manner as to maximize a risk-weighted expected return. Nonlinear models generally result in output distributions that are skewed, suggesting that risk may be differentially weighted above or below the mean. Even if outputs are not skewed, costs related to various outcomes may be uneven across the output distribution. Given uncertainty in cost estimates (as in a cost-benefit framework), the expected value of the net surplus may be assumed a measure of welfare (Freeman 1991). Risk preferences are not addressed by the analysis, but there is value in reducing uncertainty for a risk averse populace. In economic terms, for the distribution of environmental costs in particular, certainty equivalents for the benefit stream would be smaller than expected value.

Table 2.2. Evaluated impact vectors

restricts the detailed valuation of environmental impacts to use costs (i.e. physical damages and their resultant impacts) where surrogates for environmental effects are measured through changes in markets					
impact	**source**	**agent**	**environmental metric**	**valuation endpoint**	**economic metric**
annoyance	noise	noise	dBA SEL	exposure area	depreciation of owner-occupied housing values in 55+ and 65+ DNL exposure areas
mortality	NOx, SOx, VOC	PMv	PM2.5 in μ/m^3	change in risk of mortality, sudden and chronic	value of a statistical life
	PMnv	PMnv	PM2.5 in μ/m^3		
	NOx	O3	O3 in ppm		
morbidity	NOx, SOx, VOC	PMv	PM2.5 in μ/m^3	change in risk of cardiovascular and respiratory disease and acute onset of similar conditions	out-of-pocket costs for health care
	PMnv	PMnv	PM2.5 in μ/m^3		
	NOx	O3	O3 in ppm		
	NOx	NO2	NO2 in ppm		
	SOx	SO2	SO2 in ppm		
	CO	CO	CO in ppm		
change in human support systems	CO2	CO2	globally-averaged surface temperature Ts in K	• change in agricultural patterns • sea level rise • disease incidence • shifts in ecosystems and human institutions as WTP to preserve associated capital • changes in markets primarily as related to forestry and energy production) • non-market impacts based on time use for leisure	% change in global GNP
	NOx	O3 CH4			
	H2O	clouds/ contrails			
	SOx	PMv			
	PMnv	PMnv			

Table 2.3. Impact vectors not considered in MAIPA applications

impact	agents	vector	state	key barrier to evaluation	magnitude
climate	CO_2 H_2O NO_x SO_x PMnv	morbidity and mortality associated with change in surface temperature	emerging	epidemiological evidence	O(0.01) to O(0.1)
		ecosystem services	emerging	comparative metric (economic or otherwise)	O(1) to O(10)
ozone depletion	NO_x	skin cancer risk from UV radiation	maturing	assessment appropriate method to evaluate aircraft impacts upper tropospheric / lower stratospheric ozone depletion	O(0.01) to O(0.1)
air quality	PMnv VOC metals	cancer risks from hazardous air pollutants	emerging	few data on composition of trace species in engine exhaust	unknown
	PMnv NO_x SO_x	PM-related mutagenesis and neurological effects	nascent	epidemiological evidence	unknown
	CO	mortality associated with change in atmospheric concentrations	emerging	epidemiological evidence	O(0.01) to O(0.1)
	NO_x SO_x	visibility, agricultural productivity, acidification of water bodies, likely apply only to a subset of airports near natural areas, farmland, or confined water bodies	nascent	characterization of regional effects of airport sources	unknown
	CO_2 H_2O NO_x SO_x PMnv	morbidity and mortality associated with of changes in air quality indirectly caused by change in surface temperature	nascent	epidemiological evidence	O(1e-01) to O(1)
	all emissions	macroeconomic impacts of changes in quality	emerging	analysis approach for air transport to determine multiplier effect	O(1) to O(10)
noise	noise	noise health and welfare effects including disaggregation of annoyance metric	maturing	epidemiological evidence	O(1) to O(10)
		macroeconomic impacts of changes in noise	emerging	analysis approach for air transport to determine multiplier effect	unknown
		domestic animal and wildlife response	emerging	comparative metric, economic or otherwise, for evaluation against other impacts	unknown

2.3. Evaluation of the damage function

The quantitative results of this analysis provide a comparative analysis of performance factors that effect changes in environmental impact. Given the relatively high cost of any technological or operational change in the air transportation system, this is essential to identifying investments that have the highest potential for environmental benefits. This section outlines the computational problem associated with estimating $c(w)$ and the damage function.

2.3.1. MAIPA calculation of damage estimates

Referring back to figure 2.1, environmental costs depend upon environmental quality impacts (p) as a function of emissions and noise, $p(q_i,q_n)$. Emissions and noise, in turn, are functions of geography and time, $q_i(x,t)$ and $q_n(x,t)$, as dictated by operational activity and technology. For a design change, consequent changes in qi and qn are not necessarily independent; q_i and q_n are realized in discrete pairs related to the performance of an aircraft type. New aircraft types will offer a different balance of emissions and noise performance than those currently in the fleet. The functions p related to emissions (p_i) and noise (p_n) are typically non-linear and cannot necessarily be assumed independent for a given policy option intended to reduce impacts. As q_i and q_n exhibit dependency through technology, p_i and p_n exhibit dependency through operations.

Equation 2.1 shows a generalized damage function without income effects for a location X (e.g. communities around an airport). The equation integrates marginal damages (c) for a change in emissions and noise occurring over time following implementation of policy k at time $t = 0$. The result is the change in damage cost (C) associated with the policy k. The integration is made for the change in emissions i, ΔQ_i, and noise, ΔQ_n, against a background from all other sources, Q_{ref}. To this is added an error, ε, resulting from sources not quantified or unknown. Here the time horizon is infinite, but with a given discount rate, r, only a portion of this timeline will be of practical consequence and the stream of future changes is uncertain.

(2.1)
$$\Delta C^k = \left(\int_0^{\infty} \int_{(Q^{ref}+\Delta Q^k)}^{Q^{ref}} c(Q)\, e^{-rt}\, dQ\ dt \right) + \varepsilon$$

To estimate total damage costs due to aircraft operations, MAIPA estimates $c(Q)$ as the difference between two states: (1) a baseline where air traffic effects are as they stand presently; and (2) the state where these effects are hypothetically removed (so that ΔQ_i and ΔQ_n are equal to all emissions and noise from air transport). As a result, equation 2.1 can be recast approximately as shown in equation 2.2, where it is assumed that individual effects are additive and thus separable.

$$(2.2) \qquad C \cong \underbrace{\sum_{i=1}^{I}\left(\int_{0}^{\infty}\int_{(Q_i^{ref}+\Delta Q_i)}^{Q_i^{ref}} c_i(Q_i)\, e^{-rt}\, dQ_i\, dt\right)}_{\text{emissions contribution}} + \underbrace{\left(\int_{0}^{\infty}\int_{(Q_n^{ref}+\Delta Q_n)}^{Q_n^{ref}} c_n(Q_n)\, e^{-rt}\, dQ_n\, dt\right)}_{\text{noise contribution}} + \varepsilon$$

In equation 2.2, we assume parameters reference air transport. The equation does not contain the feedbacks that would occur if policies implementing technology or operational change were carried out over a time scale similar to the fleet replacement cycle. For sequential policies, it may be necessary to revise the damage function to account for changes in the states of air transport, background sources, and environmental quality.[11]

A simplified summary of inventory calculations is shown with equations 2.3, for an emitted species, i, at a particular location, X, for the time period, T. A similar formulation is used for noise. Inventories, Q_i, are the product of fleet operations, n, and the rate of emissions production, $\dot{q}_i$, integrated over time. The index, j, refers to a particular aircraft type and q_i is dependent on technological characteristics.

$$(2.3) \qquad Q_i(X,T) = Q_i^{<h_{bl}} + \sum_J n^j \left(\int_{t(\uparrow h_{bl})}^{t(\downarrow h_{bl})} \dot{q}_i^j(t)\, dt\right) \quad \rightarrow \quad Q_i(T) = \sum_X Q_i$$

Typically, inventories are calculated as the sum of regional inventories over airports X, as written in equation 2.3. MAIPA inventories are calculated directly from U.S. national activity statistics, essentially treating the United States as a single airport (cf. chapter 4). We book-keep two inventories, one including only portions of the flight below the atmospheric boundary layer, $< h_{bl}$ (equation 2.4), assumed to impact

[11] In cases where a technological system such as the aircraft is being designed, such interdependencies arise explicitly and a model is required to relate qi and qn directly. This case is not considered in this thesis but the framework is amenable to its inclusion. For example, an exogenous aircraft design model could be specified for this purpose.

air quality, and the other including all flight segments, impacting the entire atmosphere and leading to climate effects.

$$(2.4) \qquad Q_i^{<h_{bl}}(T) = \sum_J n^j \cdot \left[\underbrace{\int_{t=0}^{t(\uparrow h_{bl})} \dot{q}_i^j(t)\ dt}_{\substack{\text{emissions during take-off} \\ \text{up to the mixing height } h_{bl}}} + \underbrace{\int_{t(\downarrow h_{bl})}^{t=t_{end}} \dot{q}_i^j(t)\ dt}_{\substack{\text{emissions during landing} \\ \text{under the mixing height } h_{bl}}} \right]$$

The emissions rate for species i and aircraft type j is the product of the fuel consumption rate ($\dot{q}_f$) and the emissions index (EI), a metric of technological emissions performance as in equation 2.5.

$$(2.5) \qquad \dot{q}_i^j(t) = \dot{q}_f^j(t)\ \mathrm{EI}_i^j(t)$$

MAIPA calculates valuations of environmental quality C for discrete changes in environmental quality over a given time period (e.g. a quarter or year). For air quality, this is done assuming marginal costs equal average costs $c \cong \bar{c}$, implying that a linear change in total damages over Δp is locally a good approximation of $p(Q)$. In this case, we use Q_i from equation 2.5 to obtain an estimate for c as in the equation 2.6. Where $C(p)$ is nonlinear, as in the cases of noise and climate change, $c \neq \bar{c}$ and we instead use an approximation to the derivative as in equation 2.7.

$$(2.6) \qquad c \cong \bar{c} = \Delta C / \Delta Q \qquad \text{for linear} \qquad C = \beta_1 p + \beta_0$$

$$(2.7) \qquad c = dC/dQ \qquad \text{for nonlinear} \quad C = f\left(p(Q)\right)$$

With equations 2.6 and 2.7, equation 2.2 simplifies as in equation 2.8; the summation over aircraft types J is added because this is the base unit of technology constructed for MAIPA. The linear form implies separability of effects in estimating marginal damage costs; the roles of bundling and indivisibility are not investigated in this thesis.

$$(2.8) \qquad \Delta C = \sum_{j=1}^{J} n_j \cdot \left[\sum_{i=1}^{I} c_i \Delta q_i + c_n \Delta q_n \right]_j + \varepsilon$$

This leads us to the form of the damage function estimated in this study shown in equation 2.9, where the summation over J in equation 2.8 for emission i has been pulled into the calculation of ΔQ_i and noise marginal damages are given on a per operation basis such that Δq_n in equation 2.7 is included in the calculation of c_n.

2.3.2. Form of damage function

The coefficients ($\hat{\mathbf{c}}$) are vectors of estimators of marginal damage costs, denoted in this document by the use of carets. These estimators are assumed unbiased (in the statistical sense, not in reference to error), and are uncertain, given as distributions of possible values (cf. next section 2.4). Practically, policies built around technology standards (as is the case in aircraft environmental regulation) should be congruent with aircraft performance. MAIPA accommodates this in selecting the parameter references for marginal damage costs. These are fuel composition (carbon, hydrogen, and sulfur); emission indices for nitrogen oxides (NO_x as NO_2), carbon monoxide (CO), hydrocarbons (HC as VOC), and non-volatile particulate matter (PM_{nv}, i.e. soot); and sound exposure level (dB SEL). Since the SEL is a time-integrated parameter, it makes sense to use a per-operation normalization as is done in equation 2.9. Fuel marginal damages are estimated from the EI-weighted summation of marginal damages for the constituent species.

The form of the damage function is most relevant to supply-side decision-making. It can be alternatively specified to stress consumption-oriented options, such as the evaluation of market-based abatement strategies, but the marginal damage units would more conveniently be presented using mass units (e.g. $/t, cf. section 4 for tabulation of marginal damages in mass units). Assuming a linear form for the damage function implies separability, i.e. the marginal impacts of one species do not depend on another. This is a physical simplification appropriate to small perturbations in environmental quality Equation 2.9 does not tell us directly how much environmental quality we will get from an expenditure; for this, we need the marginal abatement cost estimator.

See Appendix A2 for additional discussion

- A comprehensive cost-benefit analysis (CBA) capability is envisioned as the next step air transport environmental impact assessment. Appendix A2 (Benefit assessment in cost-benefit analysis) looks at the application of benefit assessment to CBA in the context of the discussion in this chapter.

Equation 2.9. Form of the estimated damage function

$$\Delta \mathrm{C} = \underbrace{\boldsymbol{\alpha}\cdot\hat{\mathbf{c}}_{fuel}}_{\text{(1) fuel consumption}} + \underbrace{\boldsymbol{\beta}\cdot\hat{\mathbf{c}}_{comp}}_{\text{(2) fuel composition}} + \underbrace{\Delta\mathbf{EI}_i\cdot\hat{\mathbf{c}}_i}_{\text{(3) emissions via combustion}} + \underbrace{\Delta\mathbf{n}^j\cdot\hat{\mathbf{c}}_n^j}_{\text{(4) noise}} + \underbrace{\varepsilon}_{\text{inestimable / unknown damages}} = \text{damage function}$$

units $\quad \hat{C}\langle\$\rangle \quad \hat{c}_{fuel}\langle\$/\mathrm{t}\rangle \quad \hat{c}_{comp}\langle\$/\mathrm{EI}\rangle \quad \hat{c}_i\langle\$/\mathrm{EI}\rangle \quad \hat{c}_n\langle\$/\mathrm{op}\rangle \quad \mathrm{EI}\langle\mathrm{g/kg\text{-}fuel}\rangle$

where (1)

$$\boldsymbol{\alpha}\cdot\hat{\mathbf{c}}_{fuel} = \begin{bmatrix} \Delta Q_{fuel} & \Delta Q_{fuel}^{<h_{bl}} \end{bmatrix} \begin{bmatrix} \hat{c}_{fuel} \\ \hat{c}_{fuel}^{<h_{bl}} \end{bmatrix} = \begin{bmatrix} \text{climate damage} \\ \text{cost of fuel} \\ \text{consumption} \end{bmatrix} + \begin{bmatrix} \text{air quality damage} \\ \text{cost of fuel} \\ \text{consumption} \end{bmatrix}$$

define

Q_{fuel} = fuel inventory

$Q_{fuel}^{<h_{bl}}$ = fuel inventory below mixing height

$\hat{c}_{fuel}$ = marginal damage cost of fuel

$\hat{c}_{fuel}^{<h_{bl}}$ = marginal damge cost of fuel for flight segments $< h_{bl}$

where (2)

$$\boldsymbol{\beta}\cdot\hat{\mathbf{c}}_{comp} = \begin{bmatrix} \Delta\mathrm{EIC} & 1 & \Delta\mathrm{EIS} \end{bmatrix} \begin{bmatrix} \kappa_1\hat{c}_{\mathrm{EIC}} \\ \kappa_2\hat{c}_{\mathrm{EIC}} \\ \hat{c}_{\mathrm{EIS}} \end{bmatrix} = \begin{bmatrix} \text{damage cost} \\ \text{of fuel carbon} \end{bmatrix} + \begin{bmatrix} \text{damage cost} \\ \text{of fuel hydrogen} \end{bmatrix} + \begin{bmatrix} \text{damage cost} \\ \text{of fuel sulfur} \end{bmatrix}$$

define

$\hat{c}_{\mathrm{EIC}}$ = marginal damage cost of fuel carbon

$\hat{c}_{\mathrm{EIH}}$ = marginal damage cost of fuel hydrogen

$\hat{c}_{\mathrm{EIS}}$ = marginal damage cost of fuel sulfur

$\kappa_1 = (1+\gamma Y_0\lambda) =$ factor change in carbon damage cost with composition change

$\kappa_2 = \dfrac{\gamma Y_0}{\mathrm{EIC}_0}(1-\lambda) =$ factor change in hydrogen damage cost with composition change

specify

$Y = (\mathrm{H/C}) =$ fuel hydrogen-carbon ratio

$\lambda = Y/Y_0 =$ ratio of fuel H/C new to old fuel composition

$\gamma = \phi\cdot\hat{c}_{\mathrm{EIH}}/\hat{c}_{\mathrm{EIC}} =$ factor change in EIH due to contrail abatement ϕ

where (3)

$$\Delta\mathbf{EI}_i\cdot\hat{\mathbf{c}}_i = \underbrace{\begin{bmatrix} \Delta\mathrm{EIPM_{nv}} & \Delta\mathrm{EINO_x} \end{bmatrix} \begin{bmatrix} \hat{c}_{\mathrm{EIPMnv}} \\ \hat{c}_{\mathrm{EINOx}} \end{bmatrix}}_{\text{climate damage cost of emissions}} + \underbrace{\begin{bmatrix} \Delta\mathrm{EIPM_{nv}^{<h_{bl}}} & \Delta\mathrm{EINO_x^{<h_{bl}}} & \Delta\mathrm{EICO^{<h_{bl}}} & \Delta\mathrm{EIVOC^{<h_{bl}}} \end{bmatrix} \begin{bmatrix} \hat{c}_{\mathrm{EIPM_{nv}}}^{<h_{bl}} \\ \hat{c}_{\mathrm{EINO_x}}^{<h_{bl}} \\ \hat{c}_{\mathrm{EICO}}^{<h_{bl}} \\ \hat{c}_{\mathrm{EIHC}}^{<h_{bl}} \end{bmatrix}}_{\text{air quality damage cost of emissions}}$$

where (4)

$$\Delta\mathbf{n}^j\cdot\hat{\mathbf{c}}_n^j = \underbrace{\begin{bmatrix} n^1 \ldots n^j \ldots n^{J=20} \end{bmatrix} \begin{bmatrix} \hat{c}_n^1 \\ \vdots \\ \hat{c}_n^j \\ \vdots \\ \hat{c}_n^{J=20} \end{bmatrix}}_{\text{noise damage cost}}$$

define

$j =$ index of representative aircraft type (REPACT) : $J = 20$

$n =$ number of operations (flights)

NO_x and PM_{nv} marginal damages are further resolved into components relevant to climate and air quality impact vectors, the latter indicated by superscript $<h_{bl}$ indicating that only the portion of a flight within the atmospheric boundary layer is accounted. For this study, all cloud-related climate effects (including

contrails) are attributed to hydrogen emissions since the thermodynamic and microphysical interactions of water vapor emissions are the mechanism for their creation. The parameter (ϕ) in equation 2.9 represents the reduction in cloud formation potential that may be achieved by, for example, route alterations to avoid humid air masses, a function of mesoscale and synoptic meteorology. In this study, $\phi = 1$ without specifying its dependence on these conditions. The damage function accommodates specifications of alternative fuel compositions via the parameters κ_1 and κ_2, the latter of which is zero in the absence of cloud impacts ($\phi = 0$).

3. Characterization of commercial aircraft sources

This chapter evaluates the historical environmental performance of the US commercial aircraft fleet, specifying the fuel, emissions, and noise inventory inputs to MAIPA. This work, along with early versions of the environmental models discussed in Chapters 4-6, was the basis for defining requirements and a prototype for APMT. The goal of source characterization was to establish a standard description of the technological and operational features of the historical US commercial fleet. In particular, the objectives were: (1) to develop a probabilistic computation through the estimation of fuel, emissions, and noise inventories; (2) to demonstrate an accuracy in the estimated fuel consumption inventories comparable to higher-resolution modeling capabilities accepted by the assessment community as benchmark methods.; and, (3) to construct a mass-based particulate matter inventory.

Contribution 3.1. Treatment of particulate matter emissions

This thesis extends the scope of impact assessment with a comprehensive treatment of particulate matter emissions. Estimated particulate emissions indices were applied to evaluate the first mass-based PM inventories specific to the operational performance of US commercial aircraft. The analysis of PM trends made possible with these inventories reveals a decline in PM emissions from 1991-2003. These results are discussed subsequently in this chapter as well as in chapters 5 and 7.

Contribution 3.2. Historical and probabilistic emissions and noise inventories for US commercial operations 1991-2003

This thesis presents the first detailed historical emissions and noise inventories for US commercial aircraft operations. Quarterly estimates are provided for 10 inventories: fuel consumption; CO_2, H_2O, SO_x, PM (nonvolatile and volatile), NO_x, HC (as VOC), and CO emissions; and noise (as SEL dBA). These inventories are based on performance characterizations of aircraft technology in historical fleet between 1991-2003 using parametric inputs defined by open-source data; a total of 19 representative aircraft types are specified.

Result 3.1. MAIPA estimated fuel consumption metrics compared to benchmark DOT data

The source characterization methods reproduce the historical fuel consumption trends described by airline-reported data compiled by the US Department of Transportation (DOT) for US commercial operations. The following results summarize comparisons evaluated for per-flight and inventory fuel consumption metrics:

- Mean errors in estimated per-flight fuel consumption are in the range -27% to 19% across the set of all 19 representative aircraft types.
- Operations-weighted mean error over all quarterly estimates of fleet fuel consumption from 1991-2003 is -1.6% ± 0.12%.

Result 3.2. MAIPA estimated fuel consumption and emissions metrics compared to benchmark inventory evaluation tools

Estimates of annual national fuel consumption and emissions inventories have accuracy and precision comparable to current higher-fidelity inventory evaluation tools. Comparative per-flight and inventory metrics were obtained from: (1) the NASA/Boeing 1991 inventory developed for the IPCC Special Report on Aviation and the Atmosphere, and (2) inventories estimated for 2000-2002 using the System for Assessing Global Emissions, the tool currently employed to provide inputs for APMT. Comparisons of MAIPA and benchmark fuel consumption inventories show the following results:

model	MAIPA 91	NASA 91	MAIPA 00q3	SAGE Oct 00
error	+4%	-18%	-1.4% +/- 0.1%	-4.4% +/- 0.9%
basis	DOT reports	DOT reports	DOT reports	airline data

Comparisons of estimated $EINO_x$, EIHC, and EICO with EIs estimated using SAGE and NB in suggest consistency among methods for aircraft with a long history of operation. With the exception of aircraft types introduced towards the end of the MAIPA analysis period, SAGE and NB estimates are within a standard deviation of the equivalent MAIPA result.

Result 3.3. Impact of fidelity controls on analysis scope and resolution

The developmental guidelines discussed in chapter 2 were established in part to maintain consistency among data and procedures. These requirements provide for mutually-consistent emissions and noise inventories. Along with an historical focus, this thesis provides statistically-discernible trends in the aviation-environmental impacts of US commercial operations. Exercising fidelity controls congruent with the methodological guidelines outlined in chapter 2 presented a challenging restriction on analysis resolution (i.e. time, geography, technology), reducing the informational content of the assessment. The extent of these restrictions evidenced by inventory results further recommended development of reduced-order correlates to the complex environmental models included in mainline assessment tools. As later chapters demonstrate, assessment results contain useful information for decision-making despite these resolution limitations.

Result 3.4. Challenges to future emissions mitigation efforts

Fuel consumption by regional operations is ~12% of total fuel consumption between 1991–2003; this is smaller than the short-haul component (~80%), but is higher than the long-haul component (~8%). While regional aircraft consume less fuel per flight, their operational frequency is higher than in the long-haul fleet, thus the larger fuel consumption. Only a few of the largest regional aircraft are subject to current technology standards.

Over all representative aircraft types, for NO_x, HC, PM_{nv}, and CO, the ratios of the emissions index averaged over the entire flight to the emissions index averaged only over the portion of the flight within the atmospheric boundary layer are generally less than one. With the influence of free tropospheric emissions on air quality noted by Barrett et al. (2009), results indicate that:

- Emissions controls that attend only to the landing-takeoff cycle, as current regulations are setup to accomplish, have lower efficacy by as much as 85% for CO and HC, and 50% for NO_x than assumed in the current framework.
- Correlations among trends in emissions between 1991-2003 suggest that technology standards have been more successful in limiting emissions within the atmospheric boundary layer than overall emissions, which has been their intention. They also reinforce the limited efficacy of EI controls in stemming emissions growth.

The nonvolatile PM emissions index declines at a rate of -1.56% between 1991-2003. These trends are strong enough to offset increases in fuel use. PM_{nv} emissions are nominally controlled by smoke regulations; however, these regulations were not changed between 1991-2003. Instead:

- Reductions in $EIPM_{nv}$ stem from the retirement of aircraft through the phaseout of Stage 2 aircraft mandated by ANCA. This also underlies a negative correlation between fuel consumption and HC emissions trends. Noise trends suggest that as the oldest aircraft are retired, additional reductions in both noise as well as PM_{nv} and HC emissions may be realized.

Result 3.5. Resolving structural issues in evaluation of risk-based assessment metrics versus refinement of traditional quantity-based metrics

The current assessment analysis suggests damage estimates are influenced more by the structural uncertainties in assessment modeling than the errors in existing source characterization capabilities. For example, in source characterization, the importance of establishing a capability that allows particulate matter to be included in an assessment analysis has more influence on damage estimates than reducing errors in inventory estimates.

Section 3.2 provides a brief overview of the derivation and application of fuel consumption, emissions, and noise metrics in MAIPA. This section is supported with an extensive methodological discussion contained in appendices A1-A9. Section 3.3 reports the evaluation of error, uncertainty, and inter-model consistency of fuel consumption and emissions metrics. The discussion in this section focuses on method evaluation. Section 3.4 reports the an analysis of correlations among trends in fuel consumption, emissions, and noise metrics, highlighting insights made available by enabling comparative analyses

3.1. Evaluation of pollutant metrics

Aircraft technology in-service between 1991-2003 is described by aggregating aircraft models into 19 representative aircraft types. A set of probability functions for 11 estimators characterizes the environmental performance of a representative aircraft type.

$$\text{environmental performance} = f\left(t_x,\ \dot{q}_f,\ EI_i\ \ldots I = 8,\ q_n\right) \tag{3.1}$$

In equation 3.1, $\dot{q}_f$ is the fuel consumption rate, t_x is the time-in-mode, and EI_i is the emissions index of species i, and q_n is a metric of per-flight noise production. These estimators derive from operations, emissions, noise, and fuel consumption data extended to cover all flight conditions of a nominal flight discretized with a 9-segment performance schedule using physics-based models. This section provides a brief overview of the derivation and application of these metrics in MAIPA. Much of the detail is contained in appendices as referenced in the discussion.

3.1.1. National operations data

For MAIPA, two parameters obtained from airline-reported operations data—revenue aircraft-kilometers flown and revenue aircraft departures performed—describe the input historical US air transport activity.[1] These data are collected by the US Department of Transportation (DOT) under Parts 241 and 298 of Title 14 (Aeronautics and Space) of the US Code of Federal Regulations (14 CFR 241 and 14 CFR 298), commonly known as Form 41 (F41) and Form 298C (F98) data respectively. Data availability limited the analysis period to 1991-2003.

The unit of activity for inventory calculations is a nominal aircraft flight; the unit of technology for inventory calculations is a representative aircraft type. A representative aircraft type consists of the aggregated technological and operational characteristics of several specific aircraft models to summarize the performance of a portion of the historical aircraft fleet. Nineteen (19) representative aircraft types form the constituent technological components of the fuel, emissions, and noise inventories presented later in this section.

Probability functions for flight distance for all representative aircraft types are specified by quarterly F41T2 statistics from 1991-2003. Deriving the flight distance probability function from longitudinal data in this way implicitly assumes that market applications of an aircraft type are temporally consistent (e.g. the distribution as applied does not change over the period of the analysis).

See Appendix A3 for additional discussion:

- Appendix A3 (Low complexity models of environmental performance) looks at the application of low complexity parametric models of flight performance in the context of assessment practice. A performance model based on the Breguet equation highlights the importance of flight distance specification to model error.

[1] Air transport activity is a service supplying mobility. In the market, mobility supply decisions are based on forecasts of future demand for air transportation and are ultimately communicated as service schedules (e.g. Official Airline Guide) and equipment choice for city-pair markets. Since these decisions determine departures and kilometers-traveled, they are the primary economic input.

3.1.2. Representative aircraft types

Flight distance is the largest parametric influence on error and variance in estimates for per-flight fuel consumption and emissions; similar market application among representative aircraft types tends to minimize this influence; section 3.4 shows how flight distance factors into inventory error. Representative aircraft types were selected using similarity in nominal flight distance calculated from F41 data.

Like the distribution for flight distance, representative aircraft type performance specifications are also static. These performance specifications determine operations over a canonical flight profile scaled to fit a given flight distance (cf. next section 3.3). Analysis of errors and variance contributions in fuel consumption results suggests that a more-narrowly defined representative aircraft type tends to improve the accuracy of inventory estimates. To minimize spread in technology performance for a given representative aircraft type, further divisions within the candidate aircraft model groupings (initially made based on flight distance) were made to increase resolution of technology performance within the constraints of model fidelity (i.e. the scope and resolution of inputs and computation). Table 3.1 summarizes the set of 19 representative aircraft type aggregates identified using this process.

Table 3.1. Representative aircraft type assignments for DOT aircraft model identifiers

- Assignments categorized into long-haul, short-haul, and regional fleet segments.
- Aircraft models are listed by their DOT Form 41 identifiers.
- Gray-shaded entries indicate out-of-production models.

Long >2400 km F41 codes	Equiv REPACT
816 B747-100	b747o
817 B747-200/300	b747o
822 B747sp	b747o
819 B747-400	b747
820 B747f	b747
626 B767-300/300er	b767
625 B767-200/200er	b767
624 B767-400	b767
730 DC10-10	dc10
732 DC10-30	dc10
733 DC10-40	dc10
760 L1011-1/100/200	l1011
765 L1011-500	l1011
740 MD11	md11
627 B777	b777

Short 720-2400 km F41 codes	Equiv REPACT
715 B727-200/231a	b727
710 B727-100	b727
711 B727-100c/Qc	b727
619 B737-300	b737
616 B737-500	b737
617 B737-400	b737
618 B737-300lr	b737
655 MD80	md80
654 MD87	md80
622 B757-200	b757
623 B757-300	b757
640 DC9-30	dc9
650 DC9-50	dc9
630 DC9-10	dc9
645 DC9-40	dc9
620 B737-100/200	b737o
621 B737-200c	b737o
694 A320-100/200	a320
698 A319	a320
699 A321	a320
614 B737-800	b737n
615 B737-700/700lr	b737n
634 B737-900	b737n
675 B717-200	b717

Regional <720 km F41 codes	Equiv REPACT
608 E145	e145
629 CRJ-200ER	tfan
603 F100	tfan
674 E135	tfan
868 BAE146-100	tfan
628 CRJ-100/100ER	tfan
835 AvroRJ85	tfan
602 F28-4000/6000	tfan
867 BAE146-200	tfan
601 F28-1000	tfan
866 BAE146-300	tfan
456 SF340/B	tprp
461 E120	tprp
441 ATR-42	tprp
416 C208	tprp
483 DHC8-100	tprp
442 ATR-72	tprp
469 BA Jetstream31	tprp
445 CV-660	tprp
405 Beech 1900A/B/C	tprp
467 Swearingen Metro Iii	tprp
471 BA Jetstream 41	tprp
449 Dornier 328	tprp
489 S360	tprp
484 DHC8-300	tprp
408 BAE-ATP	tprp
450 F27/A/B/F/J	tprp
550 L188A/C	tprp
430 CV-580	tprp
435 CV-600	tprp

3.1.3. Airport-specific operational data

MAIPA was designed for a national scale. To represent geographic variability in the assessment analysis of local and regional effects, airport-specific operational and demographic data are aggregated as probability functions.[2] To do this, DOT operations were first attributed to major airports using data assembled for use in the FAA Model for Assessing Global Exposure to the Noise of Transport Aircraft or MAGENTA (CAEP 2001b, 2001a).[3]

The MAGENTA database contains operational frequencies categorized by aircraft model and noise exposure metrics for 96 airports in the US (out of 1724 civil airports worldwide); these airports were selected based on availability of detailed operational and route data. To construct distributions for demographic and environmental metrics, MAGENTA aircraft models were first matched to representative aircraft types using the list of assignments in appendix table A4.7. These assignments apportion operations reported via F41T2 and F98A1 to the 96 MAGENTA airports.

MAGENTA airports are anonymous by government agreement, designated by index; county-specific demographic and environmental data could not be explicitly associated with a MAGENTA airports. A likelihood indicator— the root mean square of population density and number of operations—was used to attach an index to an airport name, and thus county demographics and environmental data. Likely matches were identified by comparing indicator values calculated using MAGENTA operations and population density data with indicator values calculated using equivalent data for airport-counties obtained from the US Census (population density) and the FAA Terminal Area Forecast (operations). County environmental and demographic data for the top three matches to a MAGENTA airport were averaged to specify the distributions of housing statistics (house price, number of units), ambient air quality statistics, population, and non-aviation emissions inventories that are input to the air quality and noise models presented later in chapters 5 and 6.

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[2] While not a topic in this thesis, variance and mean-shift associated with this uncertainty are tied to the necessity for policy regionalization in efforts to improve economic efficiency.

[3] At the time of this analysis, a unified source of airport-specific operations data does not exist for the US that differentiates by aircraft. These data are necessary for the consistent evaluation of local effects, both air quality and noise-related. Total historical airport operations since 1976 are available from issues of the FAA Terminal Area Forecast (TAF), but not at the aircraft type resolution. Since then, a database has been constructed that expands the MAGENTA data and ties it to ETMS.

See Appendix A4 for additional discussion:

- Appendix A4 (Historical operations data and representative aircraft types) describes the activity data available through DOT Form 41 and Form 298C and its use as a source characterization input to the MAIPA. It also details the approach to aggregating certificated aircraft types into representative aircraft type groupings to characterize fleet technology operating in the historical aircraft fleet.

3.1.4. Emissions metrics

MAIPA pollutant inputs consist of emissions inventories estimated for eight (8) species: carbon dioxide (CO_2), water vapor (H_2O), sulfur oxides (SO_x as SO_2); nitrogen oxides (NO_x as NO_2); hydrocarbons (HC as VOC); carbon monoxide (CO); and 2 categories of particulate matter (PM), volatile particulate matter (PM_v) composed of precipitate or volatile mass formed through oxidation of SO_x, NO_x, and VOCs, and nonvolatile particulate matter (PM_{nv}) composed of carbonaceous mass (i.e. soot).[4] These inventories are estimated using methods consistent with current practice and guidance from the US EPA and FAA (EPA 1985, 1992) (FAA 1997, 1998; EPA 2001a, 2003c),[5] and the European Union (EC-ECAC 1998; IPCC 1999; Carlier and Smith 2004; Jalinek et al. 2004).

Emissions inventories (Q_i) are the sum over all representative aircraft types (J=19) of MAIPA per-flight emissions estimates for species i multiplied by DOT-reported number of operations (n^j_{ops}) (equation 3.2). Historical inventories are computed for each quarter from 1991-2003.

(3.2)
$$Q_i = \sum_J n^j_{ops} q^j_i$$

Probability distributions for representative aircraft type per-flight emissions (q_i) are estimated using a 9-segment performance schedule as the discrete sum of products in equation 3.3. Each segment (x) is

[4] The step of transforming emitted hydrocarbons to an equivalent amount of VOCs establishes consistency with current inventory metrics used for other industries. Organic chemicals emitted into the atmosphere are typically described as VOCs (or 'hydrocarbons', (c.f. Code of Federal Regulations, Title 40 part 5/Section 100 for complete definition). HC emissions as measured do not compositionally correspond exactly to the volatile organic compound (VOC) definition used for air quality assessments. EPA (1992) suggests a conversion from HC to VOC for commercial aircraft by multiplying HC by a factor of 1.0947.

[5] MAIPA inventory evaluations are consistent with the methods incorporated within the FAA Emissions Dispersion Modeling System (EDMS), a legacy estimation tool for emissions inventories within the atmospheric boundary layer that has been incorporated into SAGE. EDMS remains the inventory evaluation model required by EPA regulations for inventory development toward reporting and air quality compliance demonstrations.

specified using a set of probability functions for 10 estimators $\{t_x, \dot{q}_f, EI_i \; ... I = 8\}$ where $\dot{q}_f$ is the fuel consumption rate, t_x is the time-in-mode, and EI_i is the emissions index of species i.

(3.3) $$\int_0^{t_{flt}} \dot{q}_i(t)\,dt = \int_0^{t_{flt}} \dot{q}_f \cdot \mathrm{EI}_i(t)\,dt = \sum_X \dot{q}_{f_x} \cdot \mathrm{EI}_i^x \cdot t_x$$

3.1.5. Fuel consumption metrics

Parsing equation 3.3, the discrete summation of the products $\dot{q}_f \cdot t_x$ gives an estimate of per-flight fuel consumption specific to each representative aircraft type (equation 3.4).

(3.4) $$q_f = \int_{t=0}^{t_{flt}} \dot{q}_f(t)\,dt \approx \sum_X \dot{q}_{f_x} \cdot t_x$$

The fuel consumption inventory (Q_f) is the sum over all representative aircraft types (J=19) of per-flight fuel consumption multiplied by the DOT-reported number of operations (n_{ops}^j) (equation 3.5).

(3.5) $$Q_f = \sum_J n_{ops}^j q_{f_j}$$

3.1.6. Flight performance

A set of operational rules was developed to provide a common specification of flight procedures and operational conventions for all representative aircraft types. Flight traffic and safety regulations specify one set of rules which restrict aircraft performance; within these restrictions an optimization objective determines the schedule aircraft inputs for the desired operational point. This flight management problem underlies the standard flight performance models applied in MAIPA to specify times-in-mode. Additional measurement data obtained at engine certification specify fuel consumption rates as a function of the engine thrust. Together, time-in-mode and fuel consumption rate provide the parametric description of flight performance.

The Society of Automotive Engineers (SAE) Aerospace Information Report 1845 (AIR, see SAE 1986) specifies flight performance during departure takeoff and climb, and approach descent and landing, from ground to an intermediate altitude where operational rules become less restrictive. MAIPA employs the

SAE-1845 flight parameterizations specific to individual aircraft models developed for use with the FAA Integrated Noise Model (INM, see Bishop and Mills 1992; FAA 1999). En route profile and performance follow Eurocontrol Base of Aircraft Data (BADA) performance schedules (cf. Eurocontrol 2003).

Estimators of times-in-mode inherit the performance variability among the aircraft models within a representative aircraft type, parametric uncertainties associated with input parameters, and operational variances in specifications based on aircraft condition (e.g. weight) and pilot discretion. The most important of these inputs is the flight distance distribution; differences in flight distance and, to a lesser extent, cruise altitude largely determine representative aircraft type TIM estimates. Section 3.6 examines these influences quantitatively.

See appendices A5 and A6 for additional discussion:

- Appendix A5 (Flight performance model) elaborates the methods used to specify representative aircraft type flight performance, including the parametric implementation of activity data, standard flight procedures, and aircraft performance models to define representative aircraft type flight operations and times-in-modes estimators.
- Appendix A6 (Estimation of fuel consumption rate) details the application of engine measurement data to estimate fuel consumption rates based on representative aircraft type performance parameters.

3.1.7. Gaseous emissions indices

For species emitted in a simple ratio to fuel flow—carbon dioxide (CO_2), water vapor (H_2O), and sulfur oxides (SO_x)—EIs were specified using fuel composition standards with corrections for engine performance, accounting for uncertainties in fuel specifications. $EICO_2$ and EIH_2O derive from typical fuel hydrogen/carbon (H/C) ratios with adjustments for combustion inefficiencies. Estimates of total sulfur emissions, $EISO_x$ as SO_2, are determined from typical fuel sulfur levels.

For the regulated gaseous emissions—nitrogen oxides (NO_x), unburned hydrocarbons (HC), and carbon monoxide (CO)—EIs are estimated from public certification data reported as a function of engine ground power setting. Engine models are assigned to representative aircraft types based on certification data. Boeing Method 2 (BM2), introduced in appendix 7, corrects these data to altitude conditions. Emissions data are specific to an engine model randomly selected for each simulation iteration. The estimation of

EIs accounts for certification measurement uncertainty and uncertainties in interpolating and extrapolating data to flight conditions different from the original emissions measurements.

See Appendix A7 for additional discussion:

- Appendix A7 (Estimation of gaseous emissions indices) reviews the MAIPA specification of gaseous emissions indices (EI) as a function of flight performance.

3.1.8. Particulate matter emissions indices

The results of this thesis indicate significant environmental damages from aircraft particulate matter due to air quality effects and resultant mortality risks. As reported in chapter 5, MAIPA estimates show secondary PM sourced to aircraft exhaust emissions is the largest component of annual damages.

To credibly estimate emissions impacts, it was crucial to address the absence data or methods to construct PM inventories in mass units, the basis for epidemiological correlations with disease and mortality incidence as well as climate model estimates of radiative forcing. To fill this gap, a comprehensive, probabilistic treatment of the initial quantities and subsequent atmospheric evolution of PM emissions was built to enable mass-based inventory estimates, the first particulate emissions inventories relevant to commercial aircraft.

Four types of particulate matter result from aircraft emissions—nonvolatile carbonaceous, and volatile particulate matter components, sulfate, nitrates, and organics. Non-volatile particulate matter (PM_{nv}) and sulfate volatile particulate matter (PM_v) are detectable within the exhaust plume near the engine exit.[6] Aircraft PM_{nv} is established over millisecond time-scales within the combustor (Dakhel et al. 2007), while sulfates are formed in the near-field plume (< 1s downstream of the engine exit). These are referred to as primary PM in this thesis (cf. Lukachko *et al.* 2008).

Secondary PM refers to the sum of volatile PM produced in the atmospheric oxidation of gaseous NO_x to nitrate and related ammonium salts, continuing oxidation of SO_x to sulfate and related ammonium salts, and oxidation of HC to volatile organics. Current sampling programs are now examining volatile organic particulates in the exhaust plume. Measurements indicating their presence were reported several years ago (Yu et al. 1999, Schumann et al. 2002). Recent ground measurements confirm that condensible organics

[6] PM_v composed solely of black carbon soot and volatile organics estimated using methane (CH4) as the EI mass basis

are present in aircraft engine exhaust and that a portion of PM_v is attributable to organics (Wey et al. 2006; Knighton et al. 2007; Lobo et al. 2007; Yelvington et al. 2007). Given the emerging nature of measurements that resolve organic speciation in the gaseous and condensed phases, empirical data does not currently provide a basis to define a parametric specification for organic PM_v emissions.

See Appendix A8 for additional discussion:

- Appendix A8 (Estimation of particulate matter emissions indices) describes the algorithmic development and assessment of representative aircraft type EI estimators for nonvolatile and volatile particulate matter.

3.1.9. Noise metrics

It is impractical to propose a simplified representation of noise generation and subsequent transmission through the atmosphere for MAIPA; this would require attention to flight-scale operational characteristics and a focus on individual airport circumstances, a resolution incompatible with the fidelity of the underlying temporal and geographic data. Consistency with the use of DNL in regulatory procedure and economic analyses recommends the use of A-weighted event metrics; MAIPA uses the sound exposure level (SEL) in dBA units to define representative aircraft type contributions to DNL exposure.[7]

Per-flight noise characteristics are based on INM version 6.0c noise-power-distance (NPD) curves. NPD curves present SEL for an overflight at prescribed minimum slant distances.[8] To specify a per-flight noise metric, two values for SEL are selected from the INM NPD specification, one using the power setting closest to the time-weighted thrust over the LTO departure take-off and climb segments, and the other similarly chosen for the LTO approach segment.

Distance is set to an altitude of ~315 m, for which departure and approach values are closest to take-off and approach certification data (cf. FAA Advisory Circulars AC 36-1H and AC 36-3H). These values are

[7] The FAA Integrated Noise Model (INM) is the mandated regulatory method for determining DNL levels near airports (FAA Order 1050.1D, Policies and Procedures for Considering Environmental Impacts; Order 5050.4A, Airport Environmental Handbook; and Federal Aviation Regulations (FAR) Part 150, Airport Noise Compatibility Planning.) The FAA INM estimates annual average noise levels based on an nominal day using the procedures outlined in SAE AIR 1845, consistent with the MAIPA estimation of fuel and emissions production over the LTO-cycle (cf. appendix 1). Two additional guidance documents also underly the INM, SAE AIR 1751 and SAE Aerospace Recommended Practice 866A, which present methods for calculating lateral noise attenuation and handling atmospheric absorption as a function of temperature and humidity.

[8] No lateral attenuation corrections or duration adjustments (assume overflight directly overhead is useful basis for comparison), and no speed adjustment (certification in contrast has these adjustments for the certification profile). INM NPD curves use the atmospheric absorption coefficients in SAE AIR 1845. NPD reference speed is 160 knots

logarithmically-averaged as an estimator for the mean per-flight noise level, on average 10% higher than certified take-off noise with a range [0, 23]%. Based on certification requirements, per-flight noise levels are specified with a lognormal distribution with a geometric standard deviation of 1.5 dBA around the mean per-flight noise level estimator.[9]

Noise inventories Qn are built using the logarithmic or energy summation over all representative aircraft types of per-flight noise in SEL dBA (q_n^j) weighted by the DOT-reported number of operations (n_{ops}^j) in equation 3.6.

(3.6) $$\hat{Q}_n = 10 \cdot \log_{10}\left[\sum_J n_{ops}^j \cdot 10^{\left(\hat{q}_n^j/10\right)}\right]$$

Chapter 6 develops the relationship between DNL exposure and SEL-based airport-specific noise inventories using the MAGENTA operations data described in earlier in this section; this means that we treat geographic variability as an uncertain parameter in the context of estimating noise impact. The contribution of an individual operation is estimated for each representative aircraft type relative to this baseline; at the margin, this is assumed to be proportional to a change in DNL.

No surrogate will perform this function perfectly; DNL contours are multi-dimensional and published aircraft noise data typically reference a single point (thus the single reference distance for per-flight noise), and the flight profile and atmospheric conditions strongly influence noise levels. Noise signatures LA(t) establish DNL noise contours as a summation over several flights; the airport-specific noise inventory developed in this thesis implicitly includes this summation, but not its geographic footprint. Chapter 4 discusses the joint probability functions used to correlate noise exposure areas for the 55-65 DNL and 65+ DNL noise contours to the airport noise inventories used in this thesis (as evaluated in equation 3.6).

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[9] SAE AIR 1845 indicates that if measurements are the source of SEL levels, they should capture at least LAmax - LA(t) ≤ 10 dBA, where LAmax [define L here] is the maximum sound level over the duration of the noise event LA(t), to yield exposure levels that will be biased by < 0.5 dB. However, not all NPD data is based on measurements and noise values may be derived using analytical correction procedures as specified in SAE AIR 1845. For operations in a study at Seattle-Tacoma International Airport, Flathers (1982 FAA-EE—82-19 Nov 1982) finds the overall mean difference between measured and calculated SEL is < 3 dBA when actual engine power settings are available.

See Appendix A9 for additional discussion:

- Appendix A9 (Noise characterization) contains the algorithmic development and analysis of per-operation noise estimates and inventories, preceded by a consideration of noise metric selection in the context of MAIPA objectives.

3.2. Capability evaluations

This section reports the evaluation of error, uncertainty, and inter-model consistency for fuel and emissions metrics. The evaluations are based on Monte Carlo simulations of 1000 samples obtained for each representative aircraft type for per-flight gaseous emissions $\hat{q}_i = f\left(\hat{q}_i, \widehat{\mathrm{EI}}_i\right)$. Simulation convergence is assessed using median RPK-weighted standard errors $\underline{\widetilde{SE}}\Big|^{N_{ops}}$.

$$\underline{\widetilde{SE}}\Big|^{\mathrm{RPK}}\left\{\hat{q}_{\mathrm{CO_2}} \wedge \hat{q}_{\mathrm{H_2O}}\right\} = 0.026 \text{ (95\% CI)}$$

$$\underline{\widetilde{SE}}\Big|^{\mathrm{RPK}}\left\{\hat{q}_{\mathrm{SO_x}}, \hat{q}_{\mathrm{NO_x}}, \hat{q}_{\mathrm{CO}}, \hat{q}_{\mathrm{VOC}}\right\} = [0.028, 0.031, 0.034, 0.024] \text{ (95\% CI)}$$

$$\underline{\widetilde{SE}}\Big|^{\mathrm{RPK}}[\hat{q}_{\mathrm{PM_{nv}}}, \hat{q}_{\mathrm{PM_v}}] = [0.029, 0.026]$$

3.2.1. Estimation error in per-flight fuel consumption

The accuracy and precision of fuel consumption rates and inventories were assessed against two sets of fuel consumption estimates, the first reported fuel consumption from F41T2 data, and the second from higher fidelity models. To summarize these results, the average of quarter-by-quarter errors is calculated for each representative aircraft type for the historical period 1991-2003. For the set of 19 representative aircraft types, these mean errors fall in the range $\varepsilon\left\langle\hat{q}_f\right\rangle = [-0.27 \;\; +0.19]$.

Figure 3.1 plots representative aircraft type per flight fuel consumption results using a format that emphasizes the magnitudes of uncertainty, relative to error, that result from probabilistic parameters as well as variability in the performance parameters describing representative aircraft types. The notes and legend attached to figure 3.1 describe the plot in more detail. Using this normalization, estimation errors for the 19 representative aircraft types are read directly on the x-axis as the difference compared to the per-flight fuel consumption calculated directly using F41T2 reported data; specifically, the error is equal to the x-axis value at the vertical black line. The numerical value above the mean line is the absolute value of the estimated per-flight fuel consumption in kilotons; the value to the right is the mean plus 1 standard deviation. The coefficient of variation is different for each representative aircraft type, thus the different widths for each bounding box. Uncertainties indicated by the coefficient of variation range from [0.22 0.61]. Table 3.2 provides detailed statistics.

Figure 3.1. Estimated per-flight fuel consumption for representative aircraft types with comparisons to F41T2 reported data and results from higher-fidelity models

- Per-flight fuel consumption calculated using F41T2 reported data:
  Results in figure 3.1 are plotted using a normalized x-coordinate that marks distance in fractions of the mean of all quarterly for per-flight fuel consumption calculated using F41T2 data. Thus, x=0 is mapped to this mean and is denoted by a black dot. The horizontal line spanning from black dot shows the range of the set of all F41T2 values from 1991-2003. These ranges extend asymmetrically from x=0 since they are relative to the mean, not median, value of the set.
- MAIPA estimated per-flight fuel consumption results:
  The one standard deviation range of the estimated per-flight fuel consumption for a representative aircraft type is plotted as an horizontal box with a black border; within the box, the mean per-flight fuel consumption is plotted with a vertical black line and the median qf with a vertical orange line.
- Absolute values of MAIPA per-flight fuel consumption results:
  The numerical values above the mean line are the absolute value of the per-flight fuel consumption in metric kilotons (kt). The value to the right at the end of the box is the mean plus standard deviation per-flight fuel consumption.
- Per-flight fuel consumption estimates from benchmark higher-fidelity inventory models:
  The colored marks plotted in figure 3.1 are the per-flight fuel consumption estimates calculated using the SAGE and NASA-Boeing inventory models. Section 3.5 discusses comparisons with these data in the context of overall inventory errors.

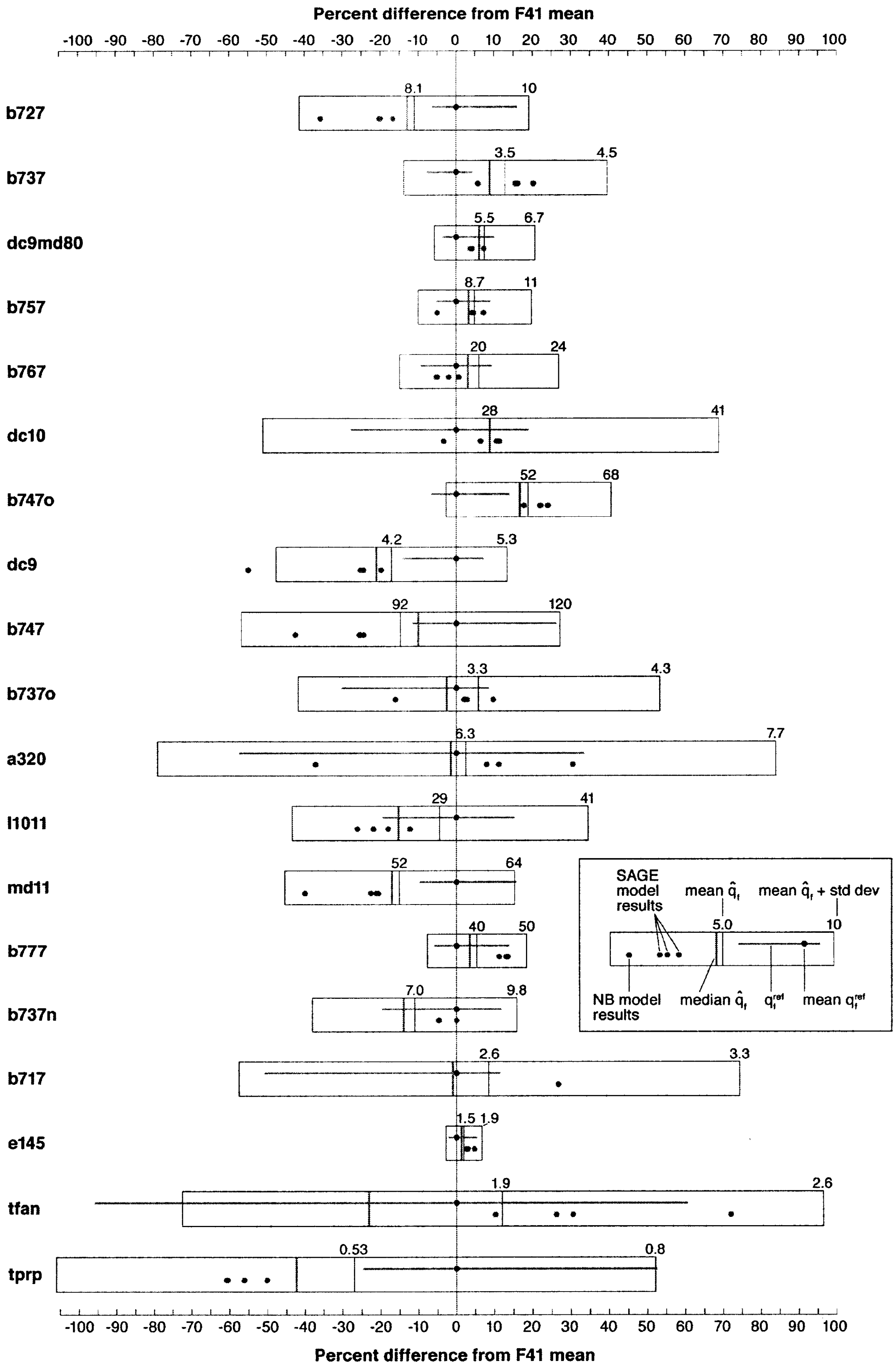

Percent difference from F41 mean
-100 -90 -80 -70 -60 -50 -40 -30 -20 -10 0 10 20 30 40 50 60 70 80 90 100
b727 8.1 10
b737 3.5 4.5
dc9md80 5.5 6.7
b757 8.7 11
b767 20 24
dc10 28 41
b747o 52 68
dc9 4.2 5.3
b747 92 120
b737o 3.3 4.3
a320 6.3 7.7
l1011 29 41
md11 52 64
b777 40 50
b737n 7.0 9.8
b717 2.6 3.3
e145 1.5 1.9
tfan 1.9 2.6
tprp 0.53 0.8
SAGE model results
mean $\hat{q}_f$
mean $\hat{q}_f$ + std dev
5.0
10
NB model results
median $\hat{q}_f$
q_f^{ref}
mean q_f^{ref}
Percent difference from F41 mean

Table 3.2. Estimated per-flight fuel consumption for representative aircraft types

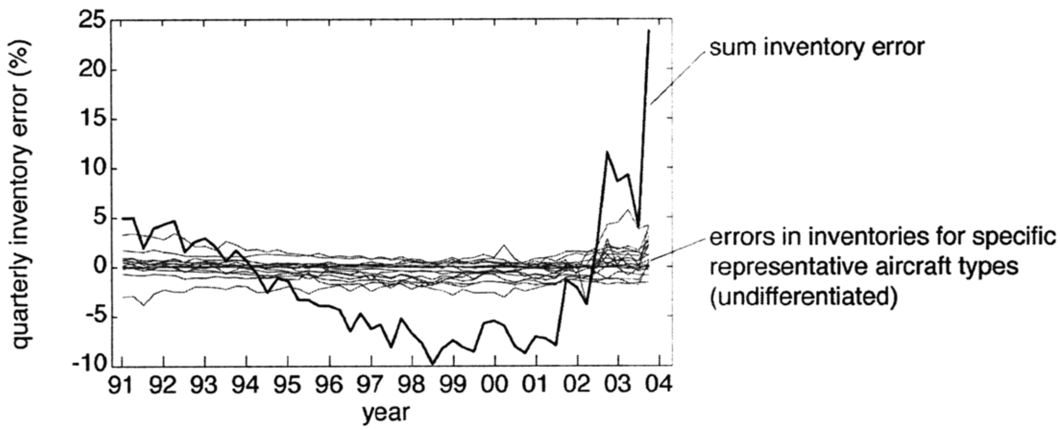
representative aircraft type	per-flight fuel use		error
	mean (t)	CV	F41T2
b727	8.1	0.23	-11
b737	3.5	0.29	13
dc9md80	5.5	0.22	8.4
b757	8.7	0.21	4.8
b767	20	0.22	6.2
dc10	28	0.46	9.1
b747o	52	0.31	19
dc9	4.2	0.26	-17
b747	92	0.33	-15
b737o	3.3	0.29	6.3
a320	6.3	0.22	2.7
l1011	29	0.43	-4.5
md11	52	0.24	-15
b777	40	0.25	5.5
b737n	7.0	0.40	-11
b717	2.6	0.26	8.4
e145	1.5	0.23	2.2
tfan	1.9	0.50	12
tprp	0.53	0.51	-27

3.2.2. Estimation error in fuel consumption inventories

The importance of errors in per-flight fuel consumption estimates is ultimately judged in the context of their propagated influence on downstream metrics. To estimate errors in the MAIPA fuel consumption inventory, an truncated inventory for the years 1991 through 2003 was constructed using only F41T2 operations and accounting for operations covered only by the 19 representative aircraft type. The operations-weighted average of representative aircraft type errors are propagated to provide an estimate of inventory error.

The unweighted average of quarterly errors over the historical period 1991-2003 summarizes the quality of the fuel consumption estimate. The 19 representative aircraft types (plus generic specification for operations not categorized) reconstruct historical fuel consumption with the mean errors $\underline{\varepsilon}\left\langle \hat{Q}_f \right\rangle$ = -1.6% ± 0.12%. Quarterly errors are plotted in figure 3.2. The interquartile range of the fuel consumption inventory error is $R - \varepsilon\left\langle \hat{Q}_f \right\rangle$ = [-9.8 +12]% from 1991-2003q3. For 2003q4, the error increases to $\varepsilon\left\langle \hat{Q}_f \right\rangle$ = +24% for 2003q4.[10]

Figure 3.2. Estimated fuel consumption inventory errors



Representative aircraft type performance models are most accurate for a limited interval of flight distance due to the normative nature of the underlying performance description. A strong correlation between error and flight distance (> 0.8) is exhibited for most representative aircraft types, suggesting that over the historical period considered, errors are associated with variation in market application of aircraft types relative to the performance specification of the MAIPA model.

3.2.3. Difficulties with interpretation of F41T2 data between 2002q1 and 2003q4

Between 2002q1 and 2003q4, the F41T2 reported fuel consumption data for several representative aircraft types are inconsistent with fuel efficiencies and load factors calculated using other F41T2 operations data. These inconsistencies affect the estimated inventory errors presented in the previous section, tending to place an upward bias on the error in estimated fuel consumption metrics.

The anomalies appear in data for representative aircraft types that exhibit lower correlation between error and flight distance—the *b727, dc10, b747o,* and regional types. For these types, data quality issues are observed in the F41T2 data; in particular per-flight fuel consumption is anomalous when utilization is low

[10] Aggregating representative aircraft types into the national inventory shows a reduction in the range of error compared with the per-flight fuel consumption errors presented previously; errors are less than 5% against the cumulative fuel consumption from 1991-2003 with an operations weighted average between [0.34, 0.46] over the 52 analysis quarters (convergence errors are O(10) smaller). The tendency towards centrality is characteristic of large, well-defined systems.

(upward bias on fuel economy) and when a representative aircraft type is used largely for freighter purposes (downward bias on fuel economy). These anomalies increase the absolute value of the estimated error in per-flight fuel consumption for the types affected. Thus, the fuel consumption inventory error also increases. Footnote 11 presents further elaboration. The cause of these data discrepancies remains unresolved. They are also a primary uncertainty in apportionments to freighter service estimated using public DOT data; it is for this reason that fuel consumption and emissions metrics for freighter service are not isolated in this thesis. Passenger and freighter services are combined to estimate fuel consumption and emission metrics for representative aircraft types.

3.2.4. Estimation of parametric sensitivities in fuel consumption metrics

Estimating per-flight fuel consumption through MAIPA introduces parameters at several points in the process of deriving fuel consumption rates and time-in-modes. Appendix A1 provides a more detailed discussion of the linear method used here to estimate variance and mean-shift contributions.

In order to better elucidate these inherited dependencies and assign contributions to uncertainties introduced at successive steps in the calculation of per-flight fuel consumption, a similarly staged approach was exercised to determine parametric sensitivities. In this application, three linear models are

[11] *Additional notes on inconsistencies in reported operations data:* Examples of problems with the reported data are shown in figure E3.1. Illustration (a) plots l1011 data showing apparent underreporting of fuels issued in 2003q4; t this type of error causes fuel efficiency (EU) to improve precipitously for the dc10, dc9, a320, and l101; summing their contributions to $\hat{\varepsilon}_{q_f}$ suggests that approximately 40% of error in the last quarter of 2003 is spurious.

A closer look at the aircraft types that do not have a strong $\hat{\varepsilon}_{q_f}$ -d_{sl} correlation revealed two additional sources of error. Illustration (b) shows a systematic deviation over time away from baseline values for the *b727*. Several other types disappearing from the fleet, the *b747o* and *dc10*, also exhibit this trend. Note the decline in the average number of seats and passengers per aircraft. This suggests an emerging predominance of freighter usage for the type. This is not a problem in itself, but the trend is inexplicably linked with a large, systematic increase in fuel efficiency. Changes in dSL do not appear to explain this trend. The result is akin to the effect of a large dSL range over the historical period, where there is a resulting large range in reported EU.

The MAIPA model, since it is limited in the extent to which it can react to changing historical usage, has problems in representing this operational space due to its normative performance description (which is oriented towards passenger service). There is currently a limited provision to simulate freight service within the confines of using F41 and F98; the division of passenger and freight utilization must be inferred from the base input data based on declines in N_{seat} and N_{psgr}. For the b727, consistency in load factor for the historical period indicates that freight utilization occurs primarily using separate flights rather than a mixed freight/passenger mode. In sum, increasing freight utilization biases $\hat{\varepsilon}_{q_f}$ increasingly upward for later quarters in the historical period where the deviation is greatest.

Erratic reporting also impacts variance. Newer type introductions also show this aspect, including the *b737n*, and *e145*. A link can be speculated to recent introductions and relatively small fleet numbers during the historical period, but more specifically, the example of the *l1011* suggests that reporting anomalies might be associated with low overall utilization. For the regional types, reporting methods suggest two additional contributions to variance. Illustration (c) shows the impact for the *tfan* type. First, air carriers operating these aircraft may or may not be required to report on F41. Second, the frequent entry and exit of regional carriers adds variance, particularly for the *tfan* and *tprp* representative aircraft types for which characteristics result from the agglomeration of many aircraft models. The e145 and *b717* fuel use estimates could suffer from any of the above sources of erratic parameter behavior.

estimated as shown in equation 3.3, each using a parameter set successively closer to the final output values (and each matched to a point of input in the estimation process), to estimate contributions to variance and mean-shift coefficients ($\delta_y = \beta_{x_i} \cdot \mu_{x_i} / \mu_y$ mean-shift coefficients are essentially linear sensitivities.).

$$
\begin{aligned}
\hat{q}_f &= \beta_{\hat{d}_{SL}} \hat{d}_{SL} + \beta_{\hat{h}_{alt}} \hat{h}_{alt} + \beta_{\hat{h}_{trht}} \hat{h}_{trht} + \beta_{\hat{h}_{grnd}} \hat{h}_{grnd} + \beta_{\hat{T}_{amb}} \hat{T}_{amb} + \beta_{\hat{P}_{amb}} \hat{P}_{amb} + q_{fo} \\
\hat{q}_f &= \beta_{W_{ap}} W_{ap} + \beta_{W_{to}} W_{to} + \beta_{F_{ap}} F_{ap} + \beta_{F_{cl}} F_{cl} + \beta_{F_{to}} F_{to} + q_{fo} \\
\hat{q}_f &= \sum_{N_{seg}=1}^{9} \beta_{\dot{m}_f^n} \dot{m}_f^n + \sum_{n=1}^{N_{seg}=9(\neq 5)} \beta_{t_n} f_n + \beta_{S_{cru}} S_{cru} + q_{fo}
\end{aligned}
\tag{3.3}
$$

The results of each regression are assumed to explain variance as measured by the coefficient of determination as it increases with uncertainties introduced at each calculation step matched by an equivalent regression model. For example, using the first model, the coefficient of determination may equal 0.21, the second may equal 0.43, and the third 0.89; the portion of the variance attributed to the parameters whose uncertainties are introduced at step 1 of the calculation would equal 0.21, at step 2 0.43-0.21=0.22, and at step 3 0.89-0.43=0.46. The remainder (0.11) is unexplained by the analysis.

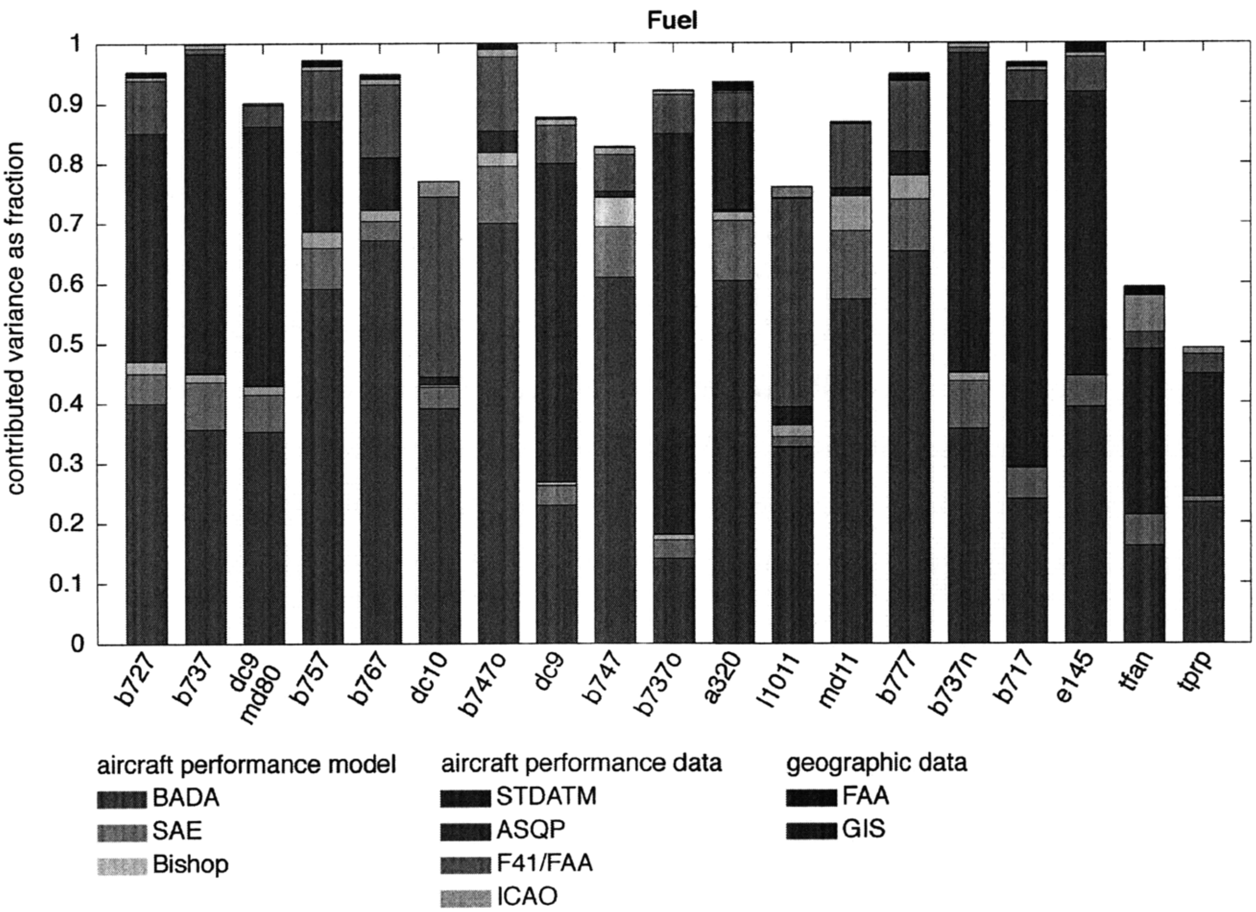
3.2.5. Contributions to per-flight fuel consumption variance for representative aircraft types

The uncertainty analysis highlights the effect of input data fidelity, specifically in restricting the achievable resolution of technological performance (e.g. number of aircraft models aggregated in a representative aircraft type) in representing the aircraft fleet. These restrictions affect both accuracy and precision in $\hat{q}_f$ and are expressed primarily through the performance models used for non-LTO segments of the nominal representative aircraft type flight.

To make clear the structural sources of variance, figure 3.3 shows the results of the linear variance analysis, plotting for each representative aircraft type the fraction of variance accounted by parameters grouped under headings that relate to particular modeling elements of the MAIPA.

Figure 3.3. Parametric contributions to variance in estimated per-flight fuel consumption

Effects broken down by aircraft performance model (BADA, SAE 1845, and Bishop weight selection), standard atmosphere parameters (STDATM), ground time data (ASQP), flight distance and weight data (F41/FAA), fuel consumption data (ICAO), and airport location data (FAA and GIS).



Effects are broken down by aircraft performance model (BADA, SAE 1845, and Bishop weight selection), standard atmosphere parameters (STDATM), ground time data (ASQP), flight distance and weight data (F41/FAA), fuel consumption data (ICAO), and airport location data (FAA and GIS). Only significant variables are included and note the linear models are not comprehensive. The coefficient of determination (i.e. height of the bars) is less than one in to the absence of higher-order terms in the estimation models. A primary component of this unaccounted variance is in apportionment of operations among component aircraft models for each representative aircraft type. This can result in a modal per-flight fuel consumption distribution when there are aircraft models with sufficiently different performance characteristics.

The *tfan* and *tprp* types manifest this most significantly and thus the low R^2; two factors account for this: (1) these representative aircraft types aggregate a large number of types that cover a wide range of operational characteristics and (2) each component model occupies a small portion of the operational service in any one year. Other representative aircraft types are also influenced by these factors, specifically the *b747, b737n, l1011, dc10* types. This is a resolvable uncertainty, but only if enough information were available to characterize the more narrowly defined types. See appendix 2 for further discussion of representative aircraft type construction methods.

Of the portion of explained variance, figure 3.3 shows that the largest structural contributor to variance in per-flight fuel consumption is the specification of non-LTO flight performance. More specifically, these uncertainties are located in a few parameters; in order of importance these are: (1) flight distance, (2) ambient temperature, and (3) in the flight performance (i.e. fuel consumption rate and time-in-mode) estimated for climb and cruise segments. For each of these parameters, variance is primarily a product of the performance diversity of component aircraft models for each representative aircraft type: (1) flight distance is related to the range of market usage among models, (2) ambient temperature is an amplification of the differences among component model service altitude specification, and (3) fuel consumption rate inherits variability in the randomized choice among engine types assigned to a given model. Low power idle and taxi segments are increasingly significant as flight distance decreases, specifically pointing to the ASQP database that specifies time-in-mode.

3.2.6. Sensitivities of mean per-flight fuel consumption to input parameters

Sensitivities indicated by mean shift parameters suggest a similar conclusion. As an example representative of all representative aircraft types, we step through the results for the *b757* type. The first model in equation 3.3—regressing 6 parameters of which only flight distance and flight altitude are significant—exhibits the positive influence of distance on fuel use and the negative influence of altitude. Mean shift coefficients are $\delta_{\hat{q}_f}^{d_{SL}} = 1.0$ and $\delta_{\hat{q}_f}^{h_{alt}} = -0.26$ respectively, each expected considering the performance equations. The intercept q_{f_0} is statistically zero, suggesting this first model is a relevant description of error propagation.

For the second and third models—regressing 5 and 18 parameters respectively— $q_{f_0} \neq 0$ statistically, a result of differences in performance across the aircraft models aggregated as representative aircraft types.

The second model indicates the importance of take-off thrust (F_{to}) and take-off weight (W_{to})—mean-shift coefficients are $\delta_{\hat{q}_f}^{F_{to}}$=0.43 and $\delta_{\hat{q}_f}^{W_{to}}$=0.62 respectively. These parameters determine the normative performance specifications above and below the reference mixing height,

In the third model, cruise speed (S_{cru}) is the most influential parameter on propagated error—mean-shift coefficient $\delta_{\hat{q}_f}^{S_{cru}}$=-0.95. While $\delta_{\hat{q}_f}^{S_{cru}}$ indicates that increasing cruise speed (S_{cru}) tends to decrease fuel flow, note the cruise speed in MAIPA is really a stand-in for time-in-mode. The remaining significant fuel consumption rate and time-in-mode parameters in the third model are positive in their effect on $\delta_{\hat{q}_f}$ as any increase in these parameters leads to an increase in fuel use.

These sensitivities are essentially structural uncertainties associated with the use of a low complexity approach for MAIPA, imposed in maintaining consistency with the resolution of input data. Representative aircraft types are defined systems and work best as models for the aircraft they represent if input data are consistent. MAIPA uses a parametric specification that works best over the range of flight distance closest to the operating condition expressed in the nominal performance specification. For performance above the mixing height, this is a static specification; thus, in cases that require estimates over a particular city-pair or similarly finer operational resolution, dynamic models would be recommended. However, as the next section discusses, there is no clear indication of improvements from application of higher-fidelity simulation at the level of national inventories.

3.2.7. Consistency of fuel consumption inventories with higher-fidelity models

This is a somewhat loose requirement. MAIPA errors are compared to those reported for inventories estimated by two structurally-different higher-fidelity models: (1) the NASA/Boeing 1991 global inventory (NB) (Baughcum et al. 1996); and (2) the FAA System for Assessing Global Emissions global 2000-2002 inventories (SAGE) (FAA 2003b; Lee 2005).

There are similarities among the MAIPA, NB, SAGE approaches to inventory estimation. But the primary differences with finer resolution of operational activity and higher-fidelity specification of aircraft performance.[12]

- SAGE employs radar data and aircraft performance correlations to estimate inventories by aggregating estimated fuel burn and emissions on a flight-by-flight basis.[13] Computed inventories are deterministic and regionally-differentiated; only the US component is used for comparison.[14]
- The NB method is also deterministic, employing proprietary performance models to estimate fuel use and emissions based upon an idealized, rule-based flight procedure applied to scheduled flights for a selection of representative aircraft types

Comparisons to available results for the NB and SAGE inventory models are shown in table 3.3. These comparisons indicate that when aggregated to the coarser geographic and temporal resolution of MAIPA, MAIPA inventory evaluations have a fidelity similar to those reported previously for higher-fidelity models.

Table 3.3. Comparisons of estimated fuel consumption inventory with results higher-fidelity inventory models

model	MAIPA 91	NASA 91	MAIPA 00q3	SAGE Oct 00
error	+4%	-18%	-1.4% +/- 0.1%	-4.4% +/- 0.9%
basis	DOT reports	DOT reports	DOT reports	airline data

---

[12] The technological resolution of the NB approach is similar to MAIPA, differing primarily in the choice of models aggregated into a representative aircraft type, source of activity data (reported versus scheduled), and performance model. Conversely, MAIPA uses the same performance characterizations to the SAGE approach, but the implementation in MAIPA is nominal, parametric, and static as opposed to the dynamic, functional, flight-by-flight implementation for SAGE. Given overlaps in data and modeling, it is important comparisons between MAIPA and these more sophisticated inventory estimates show consistency.

[13] These radar data measure the actual flight profile, typically above a nominal mixing height. The U.S. Federal Aviation Administration (FAA) Enhanced Traffic Management System (ETMS) records flight position using a flight-identifier, encoded radar position reports, and filed flight plans. Flight data can provide similar information, but is often unavailable due to its proprietary nature.

[14] In figure 3.1, the comparison is made using fuel efficiency (EU as defined in appendix A2.). This was the consistent measure obtainable from all sources plotted.

The first comparison, between NB and MAIPA, is indicated by the gray columns. Daggett et al. (1999) estimates an error of 18% in a NASA-Boeing inventory against the same DOT fuel consumption reports benchmarking MAIPA errors; for 1991, the MAIPA mean inventory error is 4.0%. There is no equivalent direct comparison with the SAGE results. Errors reported by the FAA are for specific comparisons with proprietary data. The last two columns make a comparison to SAGE inventory results with airline data on fuel consumption flights during October 2000. This comparison finds mean error of -4.4% ± 0.9%; the MAIPA inventory shows an error for the third quarter 2000 of -1.4% ± 0.1%.[15] These values are consistent, but not definitive reflections of error; the F41 error benchmark is not equivalent to the activity data underlying SAGE results.

3.2.8. Consistency of per-flight fuel consumption metrics with higher-fidelity models

There is relatively more information available through comparisons of per-flight fuel consumption for a more concrete evaluation of consistency. These comparisons are shown in figure 3.1 introduced previously; on figure 3.1, the colored marks indicate the values of comparable estimates of per-flight fuel consumption evaluated using the NB and SAGE higher fidelity performance models. As a point of reference, proprietary, detailed models of airframe-engine parameters developed by manufacturers for a particular aircraft provide estimates of fuel flow to within absolute error bounds of ±2% for any operating point over a given mission, provided that the state of the aircraft is well-known.[16]

Except for SAGE results for the *b717* and N/B results for the *dc9,* SAGE and NB results fall within the standard deviation of MAIPA distributions for per-flight fuel consumption.

- In comparison to NB, fractional differences are within the range [0.02 0.32] for representative aircraft types other than the *dc9*. MAIPA estimates are consistently higher for the aircraft where the representative aircraft type aggregations are similar (i.e. types *except for* the *tfan* and *tprp*). This is an expected result sourced to the comparison of a probabilistic approach, which accounts for extremities, and a deterministic approach which does not.
- While the per-flight fuel consumption estimated by SAGE is single-valued, its evaluation accounts for flight performance variability and the comparison is less biased. In comparison to SAGE, MAIPA

[15] The MAIPA coefficient of variation is expectedly lower than for SAGE; this stems from the finer flight-by-flight resolution of fuel consumption estimates aggregated to for the SAGE inventory.

[16] M. Schofield, Rolls-Royce, plc., personal communication.

mean fuel consumption over all representative aircraft type is within the range –0.14 to +0.21 for a three-year period results, 2000-2002, with the exception of the b717 for which the mean difference is $\mu_{q_f}^{\text{SAGE}} = 1.8\hat{\mu}_{q_f}$; a cause for this difference has not been identified.

- A more specific comparison, based on comparisons with flight data provided by NASA for a B757-200 research aircraft, the SAGE method predicts actual fuel burn over a single operation to within a mean 0.36% with a coefficient of variation of 0.29 (FAA 2003b). The MAIPA b757 estimate has a higher error of -4.8% but a lower coefficient of variation = 0.24.[17]

This is another instance highlighting the importance of structural uncertainty. This analysis suggests structural uncertainties are similar among fuel use estimation approaches; a higher-fidelity approach provides relatively low value in improving trade-off assessments. Structural uncertainties in resolution are reducible, but is refining the analysis in this way most important? While are indications that heterogeneity plays a role in determining environmental impact, it remains that much of the information offered by detailed assessments is incommunicable as decision material in the context of national policy.

3.2.9. Estimated NO_x, CO, and HC emissions metrics

Tables 3.4 and 3.5 enumerate statistics for estimated $EINO_x$, EICO, and EIHC (as VOC) in g-emissions per kg-fuel for each representative aircraft type averaged over the entire flight profile and also averaged over only the portions of the flight within the atmospheric boundary layer.

[17] Additional benchmarks are available from fuel use errors reported for the Eurocontrol Advanced Emission Model (AEM3) (Carlier and Smith 2004; Jalinek et al. 2004). AEM3 estimates fuel use based on performance inputs derived from either a flight deck recorder (FDR) or flight plan data. FDR data constitute a parametric history of a particular flight where the flight plan is an expectation of how the flight will proceed. The MAIPA fuel use estimation is akin to the latter methodology while the SAGE approach is similar to the former. Using FDR data, AEM3 errors for the b733, b737, a319, a320, and a321 fall within a range of [–2 : +9]%. Average errors calculated on the AEM3 flight plan basis for the b733, b735, b742, b744, a319, a320, and a321 were within the range [12 : 45]%, which is similar in magnitude but biased upward compared to the range of error estimated for MAIPA. Information was not available to identify the origin of this difference.

Table 3.4. Estimated NO_x, CO, and HC emissions indices for representative aircraft types

Statistics tabulated: (1) emissions index for all flight activity, (2) emissions index only for flight activity within the atmospheric boundary layer.

	NOx		CO		HC (as VOC)	
	emissions index (g-NOx/kg-fuel)		emissions index (g-CO/kg-fuel)		emissions index (g-HC/kg-fuel)	
REPACT	all flight activity	only flight activity within atmospheric boundary layer	all flight activity	only flight activity within atmospheric boundary layer	all flight activity	only flight activity within atmospheric boundary layer
b727	10	15	0.94	2.2	3.0	10
b737	10	16	0.32	1.4	4.6	21
dc9md80	10	15	0.0002	0.001	3.6	9.2
b757	13	24	0.16	0.78	2.2	11
b767	13	21	0.28	1.3	2.1	13
dc10	14	26	1.1	9.2	4.9	27
b747o	12	25	1.2	12	3.8	30
dc9md80	11	14	1.0	3.3	2.8	11
b747	14	22	0.24	1.3	1.0	11
b737o	9.7	15	1.3	3.4	3.3	11
a320	12	19	0.44	1.7	2.4	14
l1011	14	27	5.1	29	15	48
md11	13	22	0.19	1.7	1.7	15
b777	17	32	0.084	0.65	1.0	9.6
b737n	10	16	0.7	3.2	6.4	23
b717	13	18	0.028	0.083	4.0	11
e145	11	15	0.65	2.5	4.2	18
tfan	7.3	10	0.81	2.6	6.0	24
tprp	10	10	0.25	0.27	4.5	4.5

Table 3.5. Estimated per-flight NO_x, CO, and HC emissions for representative aircraft types

Statistics tabulated: (1) median per-flight emissions, (2) coefficient of variation for per-flight emissions.

	NOx		CO		HC (as VOC)	
REPACT	per-flight emissions (kg)	coefficient of variation (as fraction)	per-flight emissions (kg)	coefficient of variation (as fraction)	per-flight emissions (kg)	coefficient of variation (as fraction)
b727	83	0.47	7.6	1.4	24	1.1
b737	35	0.47	1.0	1.0	16	0.62
dc9md80	57	0.40	0.00089	12000	19	0.56
b757	110	0.46	1.3	1.9	18	0.87
b767	250	0.51	5.5	1.4	43	0.82
dc10	360	0.75	22	1.5	120	0.53
b747o	600	0.68	61	2.6	190	1.9
dc9md80	43	0.38	4.2	1.2	11	1.6
b747	1300	0.73	20	1.7	91	1.1
b737o	30	0.37	4.2	1.5	10	1.9
a320	72	0.56	2.6	1.5	14	3.8
l1011	350	0.76	140	1.4	380	1.3
md11	670	0.53	9.9	1.2	88	0.57
b777	640	0.57	3.2	2.7	40	0.95
b737n	68	0.68	4.4	3.6	38	3.9
b717	34	0.45	0.071	0.74	10	0.39
e145	16	0.42	0.91	0.78	5.8	0.80
tfan	12	0.74	1.2	1.2	9.6	0.92
tprp	4.8	0.66	0.12	0.66	2.1	0.67

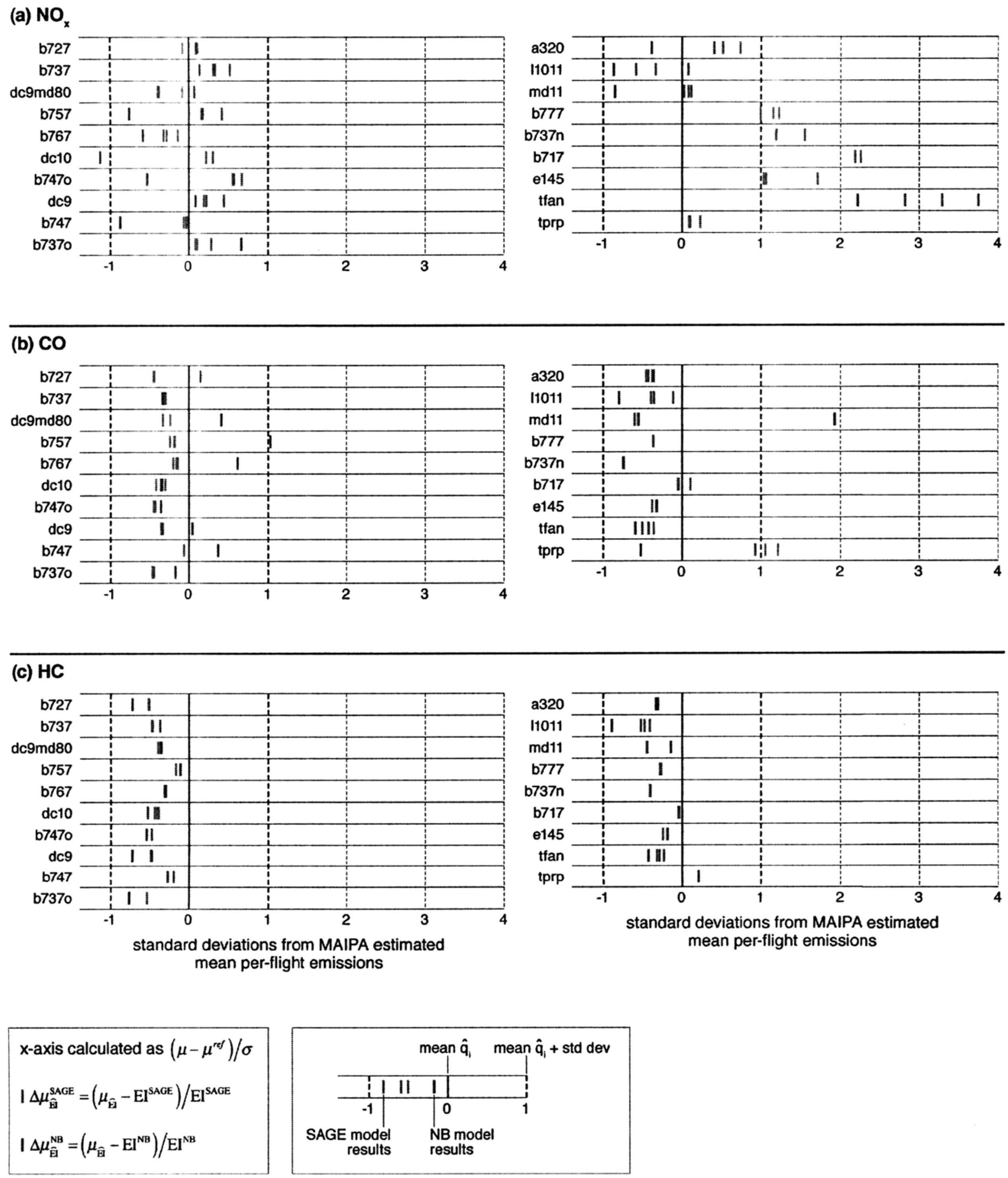
$\widetilde{CV}_{q_i}$ values range from [0.37 0.76], [0.39 3.9], and [0.66 3.6] for $EINO_x$, EICO, and EIHC, respectively. Compared to per-flight fuel consumption, these ranges reflect the additional uncertainties associated with estimating EIs.

As an additional benchmark, using standard, single-valued time-in-mode estimates defined in EPA (1992), the ICAO EEED reports emissions estimates below the reference mixing height of ~915 m for each engine with propagated measurement uncertainty. The ranges of $\mathrm{CV}\langle \mathrm{EI}\rangle = \sigma\langle \mathrm{EI}\rangle / \mu\langle \mathrm{EI}\rangle$ are [0.001 0.25], [0.005 0.38], and [0.02 0.54] for NO_x, CO, and HC, respectively. The equivalent statistics from MAIPA are [0.01 0.32], [0.12 0.82], and [0.20 1.9], which reflects the additional uncertainties in time-in-mode and throttle setting, and the variability due to the random selection of engines within a single representative aircraft type.

3.2.10. Consistency of NO_x, CO, and HC emissions indices with higher-fidelity models

Figure 3.4 plots the distributional statistics for estimated $EINO_x$, EICO, and EIHC listed in table 3.5 with comparisons to estimates calculated using the NB and SAGE inventory methods for the same quantities. To emphasize that MAIPA results are probabilistic estimators, output parameters are denoted with a caret, e.g. $\widehat{EINO}_x$. The presentation is the same as in figure 3.1 but here, normalized $\widehat{EINO}_x$ $\widehat{EICO}$, and $\widehat{EIHC}$ are shown simultaneously for each representative aircraft type as indicated in the key.

Figure 3.4. Comparisons of estimated NO_x, CO, and HC emissions indices with results from high-fidelity estimation methods



The SAGE and NB approaches to EI estimation are essentially the same for NO_x, CO, and HC as in MAIPA, using the same measurement data and method of relating EI to aircraft performance. Parametric uncertainties in EIHC are greater than EICO which is greater than $EINO_x$; conversely, distribution skew increases with higher EI, thus skew in $EINO_x$ is greater than EICO which is greater than EIHC. Corresponding to the chemical processes that lead to pollutant formation, which are mathematically multiplicative, distributions for EIs are specified using lognormal distributions. Thus, both the magnitude and form of the distributional specifications tend to increase the mean (and median) for MAIPA relative to SAGE and NB, more so for CO than NO_x, and again for VOC over CO.

Using the same measure $\left|\hat{\mu} - \mu^{ref}\right| \leq \hat{\sigma}$ to examine structural differences in emissions estimation, comparisons among $\widehat{\mathrm{EI}}_i$ and EIs estimated using SAGE and NB in figure 3.5 suggest consistency among methods for aircraft with a long history of operation. In contrast, systematic differences among MAIPA and SAGE methods are evidenced in the comparison of $EINO_x$ results in figure 3.1 for more recently introduced aircraft (between 1991 and 2003). One source of this discrepancy is that there is relatively more data applied in SAGE estimates (flight-by-flight) as opposed to MAIPA (quarterly averages). These differences also appear to be affected by bias in the estimated power setting over a flight, with MAIPA having overall lower values than SAGE or NB. Widely different estimates for the *tfan* and *tprp* types are due to the differences among their performance descriptions.

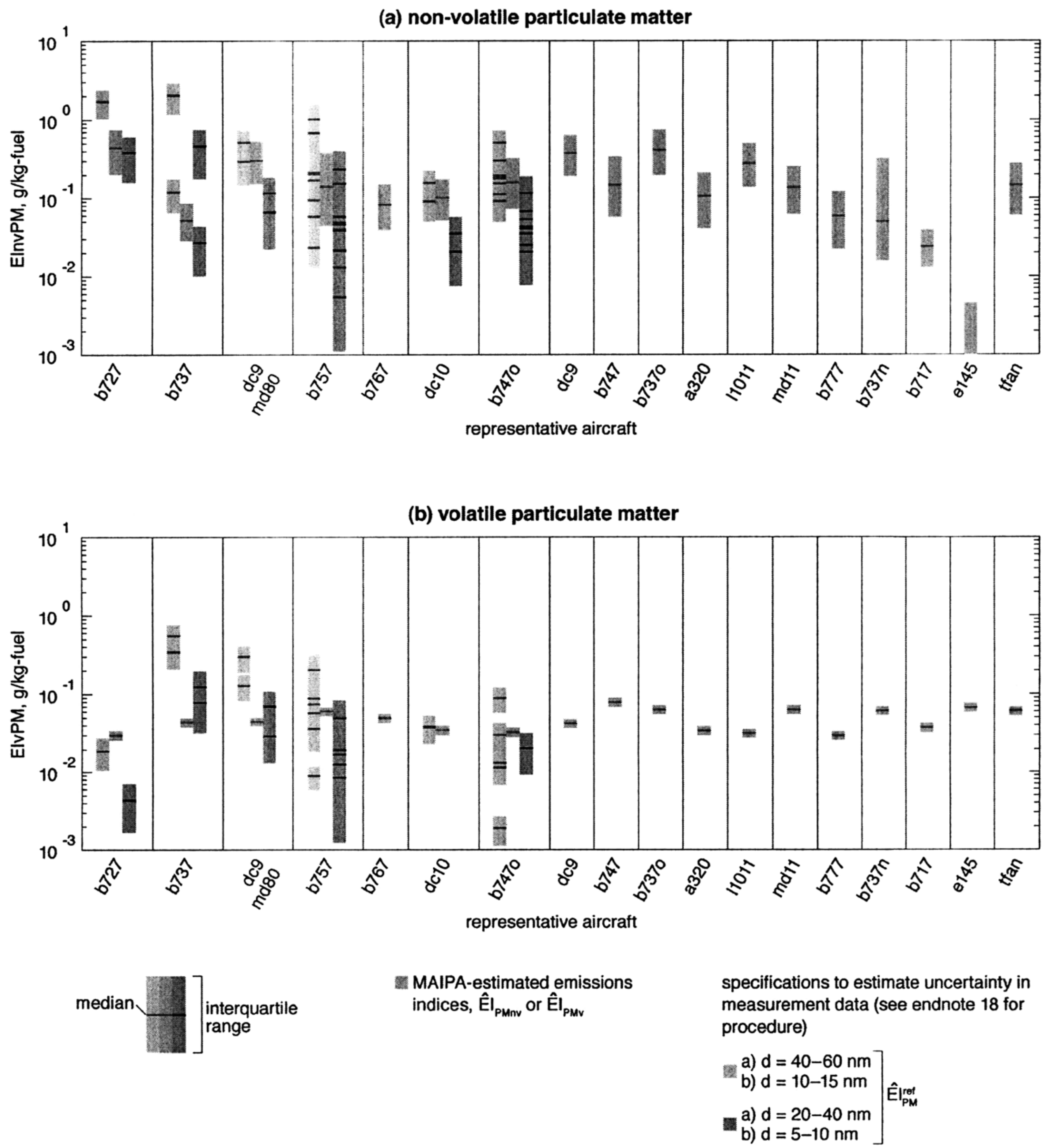
Overall, these influences result in $\widehat{\mathrm{EINO}}_{\mathrm{x}} < \left(\mathrm{EINO}_{\mathrm{x}}\langle\mathrm{SAGE}\rangle \vee \mathrm{EINO}_{\mathrm{x}}\langle\mathrm{NB}\rangle\right)$, $\widehat{\mathrm{EICO}} > \left(\mathrm{EICO}\langle\mathrm{SAGE}\rangle \vee \mathrm{EICO}\langle\mathrm{NB}\rangle\right)$, and $\widehat{\mathrm{EIHC}} > \left(\mathrm{EIHC}\langle\mathrm{SAGE}\rangle \vee \mathrm{EIHC}\langle\mathrm{NB}\rangle\right)$. The consequence is that MAIPA, for an equivalent activity basis, reports lower NO_x inventories, but higher CO and HC inventories.

3.2.11. Consistency of particulate matter emissions indices with measurement data

Figure 3.5 compares the medians and interquartile ranges of cruise $\widehat{\mathrm{EIPM}}_{\mathrm{nv}}$ and $\widehat{\mathrm{EIPM}}_{\mathrm{v}}$ for each representative aircraft type with comparisons to probability distributions for $\mathrm{EIPM}_{\mathrm{nv}}$ and $\mathrm{EIPM}_{\mathrm{v}}$ estimated using published measurements. MAIPA estimates are higher than the fleet average published in the IPCC Special Report on Aviation and the Atmosphere (1999). At the end of 2003, $\tilde{\mu}_{\mathrm{EIPM}_{\mathrm{nv}}}$ = 0.07 g/kg-fuel and $\tilde{\mu}^{bl}_{\mathrm{EIPM}_{\mathrm{nv}}}$ = 0.13 g/kg-fuel; IPCC (1999) suggests $\tilde{\mu}^{\mathrm{IPCC}}_{\mathrm{EIPM}_{\mathrm{nv}}}$ = 0.04 g/kg-fuel characterizes the global aviation fleet. This is not a comparison of equivalents however; the IPCC values were based on the few engine measurements available at the time and extrapolated to European air traffic.

Figure 3.5. Estimated nonvolatile and volatile particulate matter emissions indices

Comparison of estimated particulate matter emissions indices (green) with in situ measurement data at cruise (orange and gray).



Few measurements of emitted particulate properties exist for in-service commercial aircraft engines. This database is growing, but it remains difficult to apply as a basis for EI estimation due to relative paucity of measurements, lack of performance information attached to these measurements, and uncertainty in combining measurements taken with different techniques. The measurements plotted are from in situ sampling of primarily older in-service aircraft at cruise and are appropriate to the near-field plume (< 1s downstream of the engine exit). See footnote 18 for a summary of these data.

Measured data imprecisely specify nonvolatile particulate matter emissions indices with the range [0.002, 2] g/kg-fuel and a median value of ~0.2 g/kg-fuel. The upper and lower bounds account for the 95% CI. In comparison, at the end of 2003, MAIPA estimates a mean fleet $EIPM_{nv}$ of 0.07 g/kg-fuel in the free troposphere and 0.13 g/kg-fuel within the atmospheric boundary layer. Soot emission uncertainties are higher than for the regulated pollutants; the range of $\widetilde{CV}\left\langle \widehat{\mathrm{EIPM}}_{\mathrm{nv}} \right\rangle$=[0.70 4.2]. In comparison, the range of $\widetilde{CV}\left\langle \widehat{\mathrm{EIPM}}_{\mathrm{v}} \right\rangle$= [0.13 0.24], is relatively lower due to the deterministic treatment of representative engine cycles in the estimation of EI (see appendix 8 for further detail on EIPM estimation methods).

However, it is apparent that the spread of EIs inferred from measurements is O(10)-O(100) larger than MAIPA distributions. Thus, while these comparisons indicate consistency with measurement data, they cannot be considered definitive. The comparatively high uncertainties in measurement data indicate physical phenomena affecting PM_{nv} and PM_v in the early plume are not yet identified; this is a structural deficiency in the EIPM estimation methods.

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[18] Aircraft emit non-volatile carbonaceous particles (soot) with a size and charge distribution established over millisecond time-scales within the combustor (Dakhel et al. 2007). In situ sampling of older in-service aircraft at altitude suggests soot emissions at cruise are characterized by a lognormal distribution with a geometric mean diameter in the range of 10-60 nm and a geometric standard deviation on the order of 1.5-1.75 nm.

Probability distributions for $\widehat{\mathrm{EI}}^{ref}_{\mathrm{PM_{nv}}}$ were estimated for published number EI (nEI) measurements using Monte Carlo simulations over the above distributions for diameter and distributional parameters ($\hat{\mu}_{\mathrm{g,PM}} = \mathrm{P}_{unif}\left[10,60\right]$ and $\hat{\sigma}_{\mathrm{g,PM}} = \mathrm{P}_{unif}\left[1.5,1.75\right]$), accounting for measurement uncertainty and assuming soot density as $\hat{\mu}_{\rho_{\mathrm{PM}}} = \mathrm{P}_{unif}\left[1500,1800\right]$ in kg/m3. Altitude measurements report number EIs (nEI) in the range 0.1E15 - 6E15 particles/kg-fuel (Konopka et al. 1997; Anderson et al. 1998a; Anderson et al. 1998b; Hagen et al. 1998; Miake-Lye et al. 1998; Pueschel et al. 1998; Brock et al. 2000; Schumann et al. 2000b). Simulating over uncertainties, these data imprecisely specify $\widehat{\mathrm{EI}}^{ref}_{\mathrm{PM_{nv}}}$ with the range [0.002, 2] g/kg-fuel and a median value of ~0.2 g/kg-fuel. The upper and lower bounds account for the 95% CI in $\widehat{\mathrm{EI}}^{ref}_{\mathrm{PM_{nv}}}$.

In situ measurements at altitude indicate that volatile particles are smaller than non-volatiles, but similarly described by a lognormal distribution, with diameters in the range 1-15 nm, standard deviation on the order of 1.5, and nEIs 10-100 times greater than for soot (Konopka et al. 1997; Anderson et al. 1998a, 1998b; Hagen et al. 1998; Miake-Lye et al. 1998; Brock et al. 2000; Schumann et al. 2000). Fuel sulfur levels for these measurements are in the range 200-700 ppm, consistent with reported kerosene composition. Using a similar Monte Carlo procedure to propagate uncertainties, calculations specify $\widehat{\mathrm{EI}}^{ref}_{\mathrm{PM_{v}}}$ (as H_2SO_4) with a range [0.0007, 0.7] g/kg-fuel and median of ~0.07 g/kg-fuel. References for altitude in situ data were initially compiled in reports by Miake-Lye (2002a, 2002b, 2004), subsequently extended.

3.2.12. Resolution of EIPM performance dependencies

The analytical scaling of soot chemistry with altitude suggests ground measurements should report higher emissions indices for similar combustor inlet conditions. There are few ground measurements to reference, none of which provide direct comparisons to equivalent altitude measurements, but those that exist do indicate the trend. Recent programs report ground $EIPM_{nv}$ = 0.02-0.35 g/kg-fuel for a wide range of turbofan engines operating in the fleet, with the lower end relevant to low power conditions and vice versa. MAIPA estimated $EIPM_{nv}$ shows a similar range with medians 0.0008-0.36 g/kg-fuel among representative aircraft types.

The MAIPA does not resolve the power dependence exhibited by measurements. This is due to the constrained use of sparse SN data as described in appendix 2. Rules used to select SN from available data tend to result in EIs upward biased over lower powers since these data are typically reported as maximums rather than populating the four certification power settings (e.g. PW engines). However, note that some engines exhibit opposite trends to those observed through engine measurements (e.g. CFM56-5B series and GE-90 series engines), tempering any power trend for a representative aircraft type. Also, the MAIPA employs altitude corrections for all segments; these preferentially reduce calculated EI for higher power modes, further weakening the trend.

3.2.13. Error propagated into emissions inventory from per-flight fuel consumption estimates

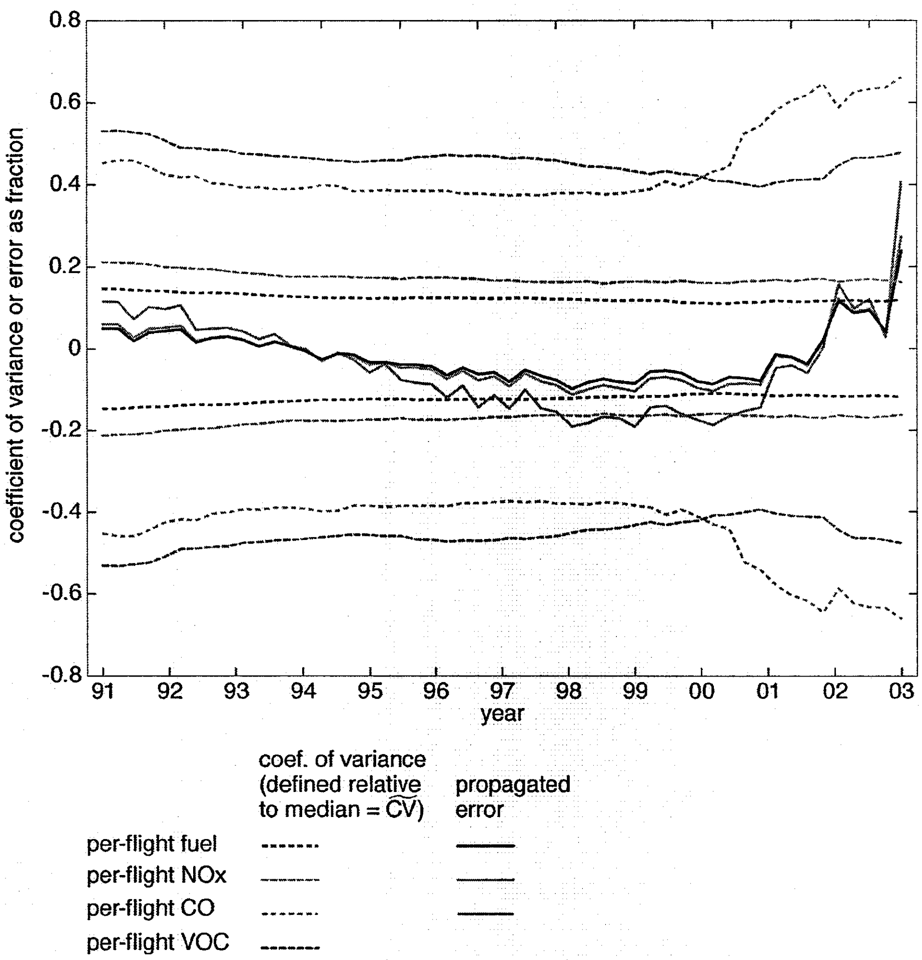
As described by equation 3.6, we can use mean-shift coefficients estimated for per-flight fuel consumption to propagate the per-flight fuel consumption error, $\hat{\varepsilon}_{q_f}$, and obtain a lower-bound estimate for the errors in per-flight emissions metrics (equation 3.6); the results are shown in figure 3.6.

(3.6)
$$\hat{\varepsilon}_{q_i} = \delta_{q_f} \cdot \hat{\varepsilon}_{q_f}$$
$$\hat{\varepsilon}_{Q_i} = \sum_{N_{type}=1}^{19} \left(\hat{\varepsilon}_{q_i} \cdot N^{i}_{ops}\right) \big/ N_{ops}$$

The salient point of this exercise is to show that per-flight emissions errors, $\hat{\varepsilon}_{q_i}$, are accentuated for all representative aircraft relative to all $\hat{\varepsilon}_{q_f}$, but that these errors fall within $\widetilde{CV_{q_i}}$. In figure 3.6, $\hat{\varepsilon}_{q_{VOC}}$ is not shown for lack of significant $\delta_{\hat{q}_f}$, but it is likely that $\hat{\varepsilon}_{q_{VOC}} < \hat{\varepsilon}_{q_{CO}}$ judging from the ratios of significant $\delta_{\hat{q}_f}$ over all representative aircraft types.

Figure 3.6. Propagation of fuel consumption errors into emissions estimates

(1) propagated errors as fraction and (2) comparison to coefficient of variation computed relative to median



3.2.14. Estimation of parametric sensitivities in emissions metrics

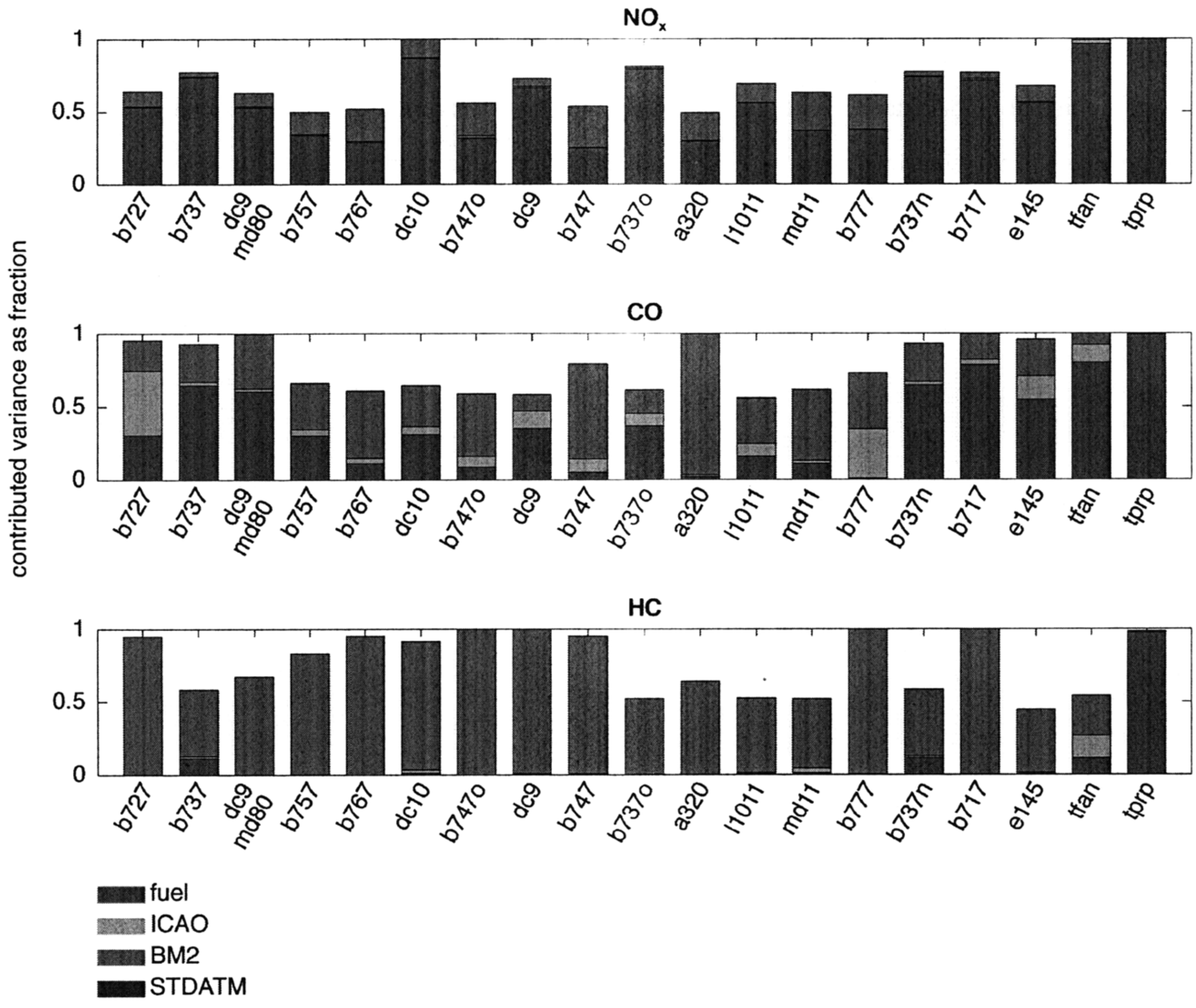
Figure 3.7 shows the results of a linear variance analysis calculated using equation 3.7 for NO_x, CO, and HC per-flight emissions.[19] Effects are broken down by estimated fuel consumption rate (fuel), standard atmosphere parameters (STDATM), ICAO emissions data (ICAO), and Boeing Method 2 corrections to altitude (BM2).

[19] The fleet median $\widehat{EI}_{SO_x}$ calculated through MAIPA is 0.95 g/kg-fuel as SO2, Since $\widehat{EI}_{CO_2}$, $\widehat{EI}_{H_2O}$, and $\widehat{EI}_{SO_x}$ multiply $\hat{q}_f$ by a factor chosen over a narrow probability function (i.e. $\widehat{EI} = P_{unif}\left[\min(EI_i),\max(EI_i)\right]$ for $i = CO_2 \vee H_2O \vee SO_x$), $\underline{\widetilde{CV}}\big|^{N_{ops}}$ for these species are $\cong \underline{\widetilde{CV}}\big|^{N_{ops}}\left(\hat{q}_f\right)$.

$$\hat{q}_i = \sum_{N_{seg}=1}^{9} \beta_{\mathrm{EI}''_f} \mathrm{EI}''_f + \beta_{\hat{q}_f} \hat{q}_f + q_{io} \tag{3.7}$$

Figure 3.7. Parametric contributions to variance in estimated NO_x, HC, and CO per-flight emissions

Effects broken down by estimated fuel consumption rate (fuel), standard atmosphere parameters (STDATM), ICAO emissions data (ICAO), and Boeing Method 2 corrections to altitude (BM2).



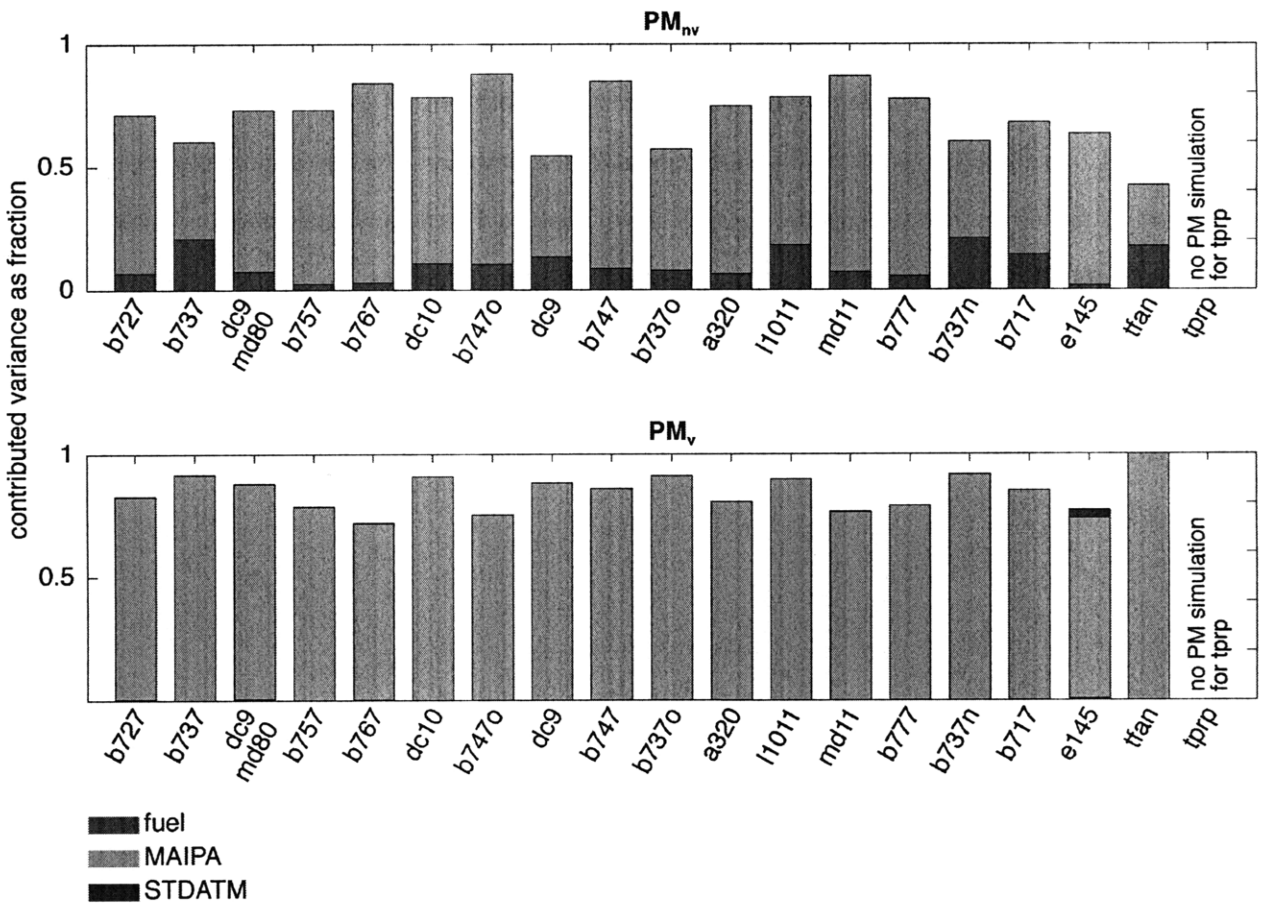
For the regulated pollutants, variance in $\widehat{\mathrm{EI}}_i$ becomes significant (i.e. parametric uncertainties in certification data and altitude corrections and variability in engine assignment). Representative aircraft type with the longest nominal flight distance show consistent influence from the cruise specification of the emissions index.[20] As flight distance decreases, trends in the estimated emissions index with engine power become statistically significant for species with the most uncertain $\widehat{\mathrm{EI}}_i$; approach segments become significant for CO and HC.

Again using equation 3.7, figure 3.8 shows variance analysis results for PM_{nv} and PM_v per-flight emissions. Effects are broken down by estimated fuel consumption rate (fuel), standard atmosphere parameters (STDATM), and estimated particulate matter indices (MAIPA). Variance in estimated $EIPM_{nv}$ is the most important contributor to variance in nonvolatile PM for representative aircraft types with the longest nominal flight distance. As flight distance decreases, uncertainties in emissions indices at cruise become significant, contributing to $\hat{\sigma}_{\mathrm{PM}_{\mathrm{nv}}}$ similarly to $\hat{\sigma}^2_{q_f}$. For the shortest range representative aircraft type, uncertainties in $\widehat{\mathrm{EI}}_i$ at idle are additionally significant. For PM_v, $\hat{\sigma}^2_{q_f}$ is the only significant contributor over all representative aircraft type.

[20] As a check on the variance analysis, for emissions that scale directly with fuel burn, results expectedly shows that the most important contributor to $\hat{\sigma}^2_{\mathrm{CO}_2}$, $\hat{\sigma}^2_{\mathrm{H_2O}}$, and $\hat{\sigma}^2_{\mathrm{SO}_2}$ is $\hat{\sigma}^2_{q_f}$.

Figure 3.8. Parametric contributions to variance in estimated nonvolatile and volatile particulate matter per-flight emissions

Effects broken down by estimated fuel consumption rate (fuel), standard atmosphere parameters (STDATM), and estimated particulate matter indices (MAIPA).



3.3. Trends in US air transport pollution 1991-2003

Trends in fuel, emissions, and noise inventories are statistically-discernible, despite uncertainties; nonparametric K-W hypothesis tests are significant (p=0.05) indicating a cross-sectional resolution that allows us determine trends over the historical period. This section presents a selective analysis of correlations among fuel, emissions, and noise metrics.

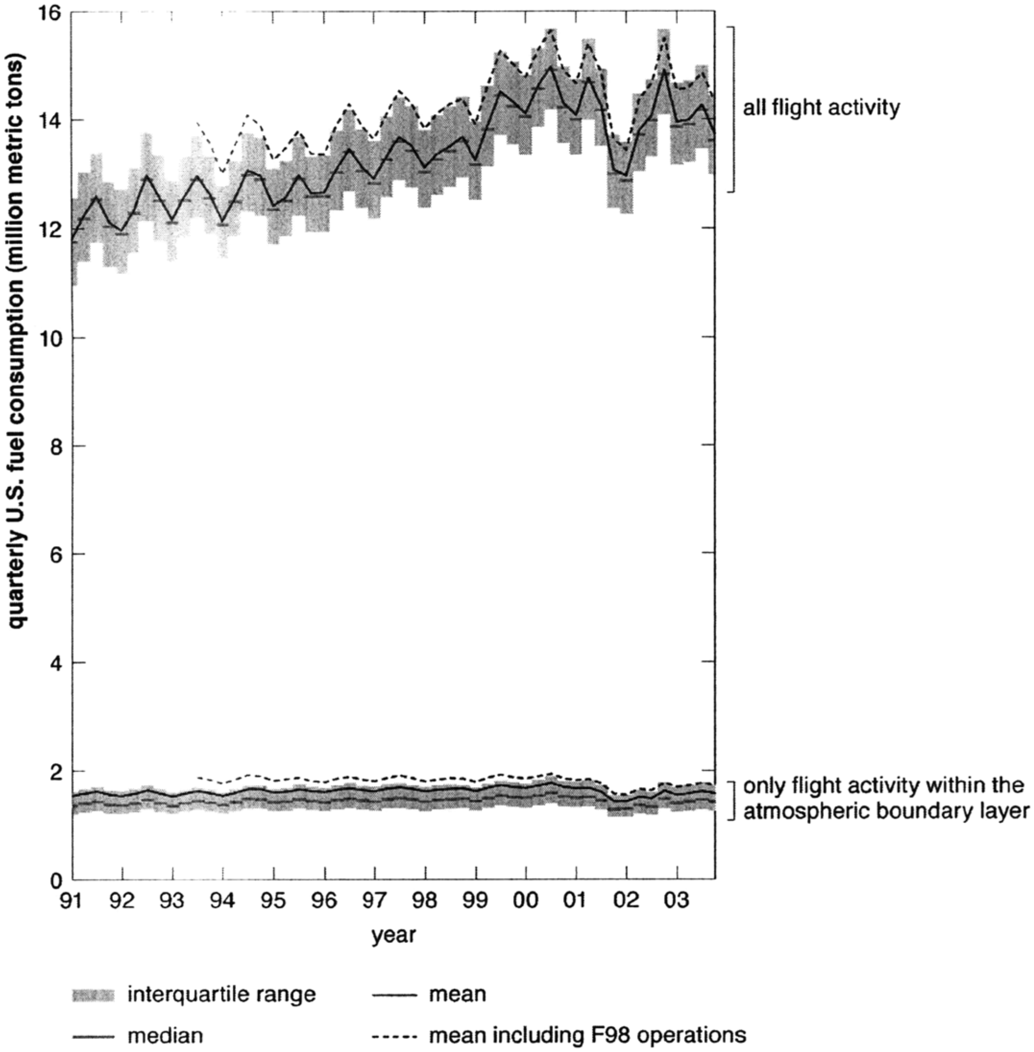
Three observations are discussed; analyses examine the impacts of shifts in service preferences, the effectiveness of emissions standards, and the relationship of noise and PM trends. Each of these examples speak to how developmental trends change the character of air transport environmental impacts. Most importantly, these results suggest shortcomings in current regulatory approaches toward source control that may be usefully addressed in future revisions of technology standards or by alternative regulatory methods.

3.3.1. Fuel consumption of US commercial aircraft 1991-2003

Figure 3.9 plots the fuel consumption by US commercial aircraft from 1991-2003. Propagated uncertainties are similar from quarter-to-quarter and the median trend moves proportionally within the uncertainty bands illustrated. From 1991 to 2003, mean fuel use increases with a compound quarterly growth rate of 0.29%; this includes the relatively sharp decline in fuel consumption from 2001-2002 that essentially negates the net growth over the previous decade.

Figure 3.9. Estimated fuel consumption by US commercial aircraft 1991-2003

Emissions for (1) all flight activity and (2) only flight activity within the atmospheric boundary layer.



3.3.2. Operational frequency and the distribution of air quality impacts

A second trend plotted in figure 3.9 runs from the third quarter of 1993 through 2003; this inventory includes the fuel consumption estimated for regional operations reported to the US DOT through Form 298A1.[21] With F98A1 operations included, the ratio of growth rate to the growth rate for F41T2

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[21] For F98A1 operations, which are not reported with aircraft type information, a split between tprp types and tfan types is assumed in order to assign fuel flow factors. For the purposes of the plot, the split is set to the value realized from F41T2 operations.

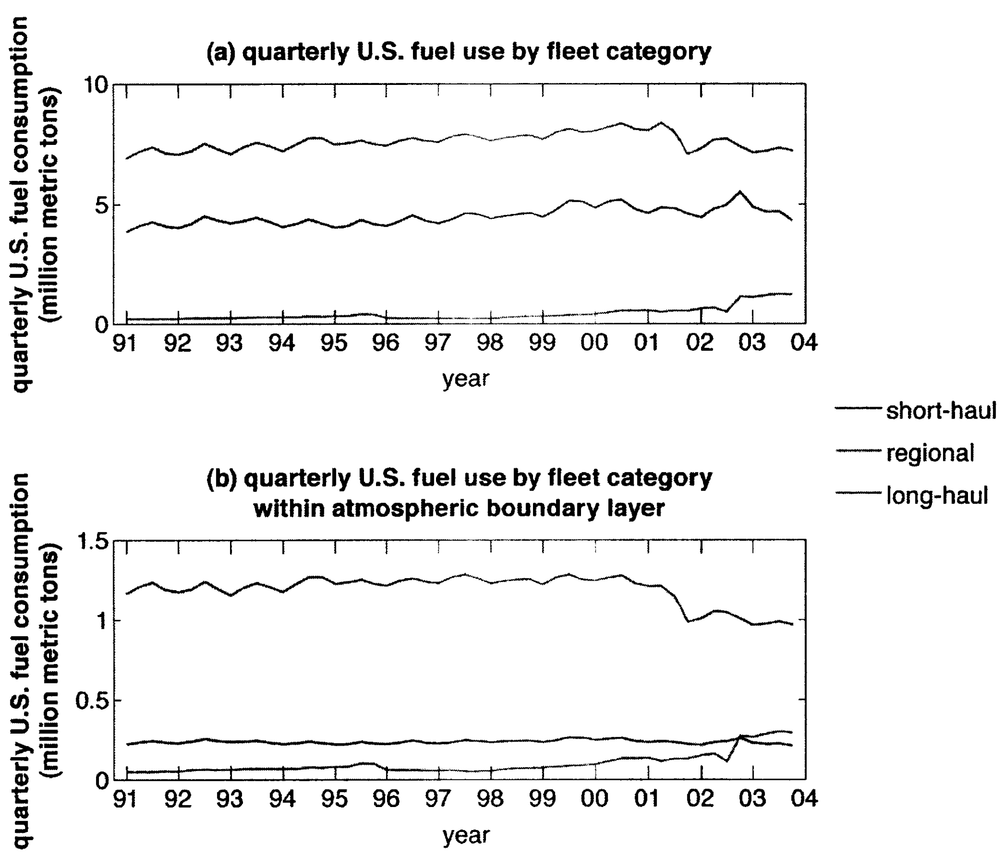
operations only is $r_g^{+F98}/r_g = 0.005$, essentially showing static fleet fuel consumption. The contribution of F98A1 operations declines with time, from 18% to 9% compared to the F41T2 inventory.

For the purpose of calculating inventories for air quality assessments, a variable mixing height based on radiosonde measurements across the United States is used to define the relevant estimators for fuel consumption and emissions inventories within the atmospheric boundary layer. Note that the mixing layer height estimated by measurements is higher than the nominal mixing height used to define the landing-takeoff cycle. The median fraction of the inventory that is expended within the atmospheric boundary layer is estimated at 10-14% for 1991-2003.

Figure 3.10 provides a further breakdown by service category (short-haul, long-haul, and regional). As shown in figure 3.10(b), fuel consumption by the regional fleet within the atmospheric boundary layer increases at $r_g^{rg} = 3.5\%$, moving from 2% to 10% of the fuel consumption inventory, largely due to shifts from the short-haul inventory; growth rates from 1991-2003 are $r_g^{sh} = 0.09\%$ and $r_g^{lh} = 0.22\%$ for the short-haul and long-haul fleets respectively. Regional operations are a relatively small portion of the fuel consumption inventory, but they account for a fraction statistically-equivalent to the long-haul fleet. Only a few of the largest regional aircraft are subject to current technology standards currently.

Figure 3.10. Estimated fuel consumption by US commercial aircraft 1991-2003 by service category

Emissions broken down by service category (short-haul, long-haul, and) regional service).



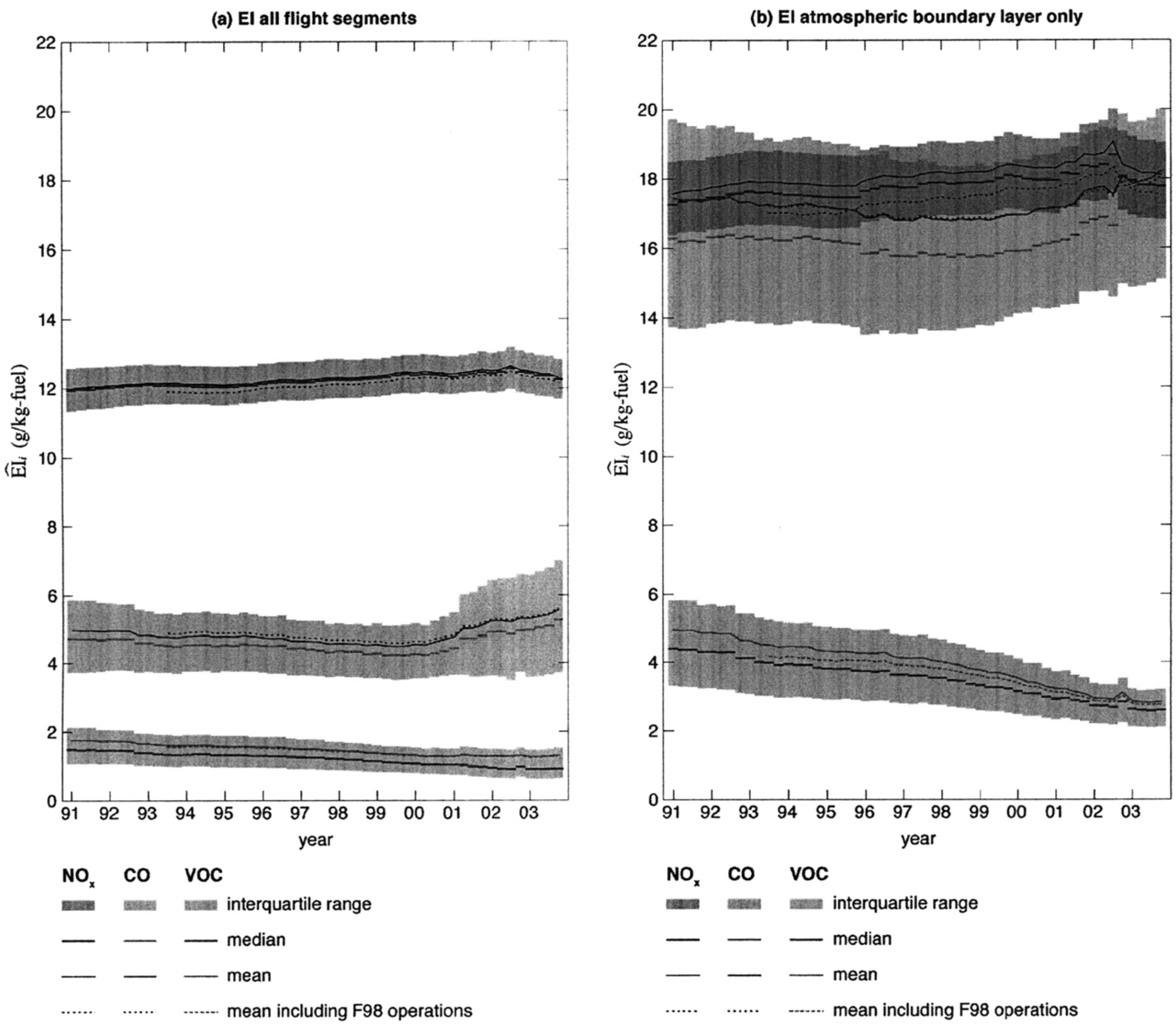
3.3.3. Influence of pollutant control on emissions above and below the mixing height

Between 1991-2003, the growth rate in the fuel consumption inventory within the atmospheric boundary layer is ~17% of the growth rate of total fuel inventory. It might be inferred that this relieves air quality burdens, but the picture cannot be fully appreciated using this statistic. Recent analyses have indicated that a portion of emissions above the mixing height are mixed down into the boundary layer, leading to a factor 2-3 increase in ground-level pollutant concentration. While emissions below the mixing height retain a regional influence on pollutant concentrations, predominantly within distances of ~100 km, those mixed from the upper troposphere influence air quality on the continental scale. Thus, the comparative trends in fuel inventories within and without the atmospheric boundary layer represent a transfer of environmental damages from airport-local populations to the general population of the United States. This also suggests a lower efficacy of landing-takeoff fuel efficiency improvements in reducing air quality impacts.

Historical trends in fuel consumption are reflected in emissions inventories, mediated by trends in emissions indices as plotted in figures 3.11-3.14 following. Trends in CO_2, H_2O, and SO_x are not shown, but they are directly proportional to fuel consumption. Consider first a comparison of trends for the regulated pollutant emissions of NO_x, HC, and CO. Figure 3.11 shows trends in $EINO_x$, EIHC, and EICO.

Figure 3.11. Estimated NO_x, HC, and CO emissions indices of US commercial aircraft 1991-3003

Emissions indices for (1) all flight activity and (2) only flight activity within the atmospheric boundary layer.



Emissions indices for NO_x, CO, and HC are influenced differently by power setting. The NO_x emissions index has an opposite trend to EICO and EIHC as a function of power setting, increasing as engine output increases rather than decreasing in a nonlinear fashion. Since proportionally more time is spent at high power below the mixing height than above, and because $EINO_x$ decreases with altitude for the same power condition, $EINO_x$ within the atmospheric boundary layer is on average higher than $EINO_x$ averaged over the entire flight profile. Over all representative aircraft type (except for *tprp* for which EIs have the same value over all flight segments by MAIPA definition), $\widehat{EI}_{NO_x}/\widehat{EI}^{bl}_{NO_x}$ = [0.49 0.74].

The same result is obtained for EICO and EIHC despite an increase in both of these parameters with altitude for a similar power condition. The comparison of EIHC and EICO trends by altitude exhibits the influence of idle and taxi at the airport. Over all representative aircraft types, the ratio of emissions indices without to within the atmospheric boundary layer are generally less than one; for CO, $\widehat{EI}_{CO}/\widehat{EI}^{bl}_{CO}$ = [0.12 0.57] and for HC, $\widehat{EI}_{VOC}/\widehat{EI}^{bl}_{VOC}$ = [0.13 1.8] with the *a320* and *dc9md80* accounting for $\widehat{EI}_{VOC}/\widehat{EI}^{bl}_{VOC} > 1$. Since the change in EICO with altitude is stronger than for EIHC— $\left(dEI_{CO}/dP_{eng}\right) > \left(dEI_{VOC}/dP_{eng}\right)$ —there is a larger relative difference between $\widehat{EI}^{bl}_{CO}$ and $\widehat{EI}_{CO}$ than between $\widehat{EI}^{bl}_{VOC}$ and $\widehat{EI}_{VOC}$.

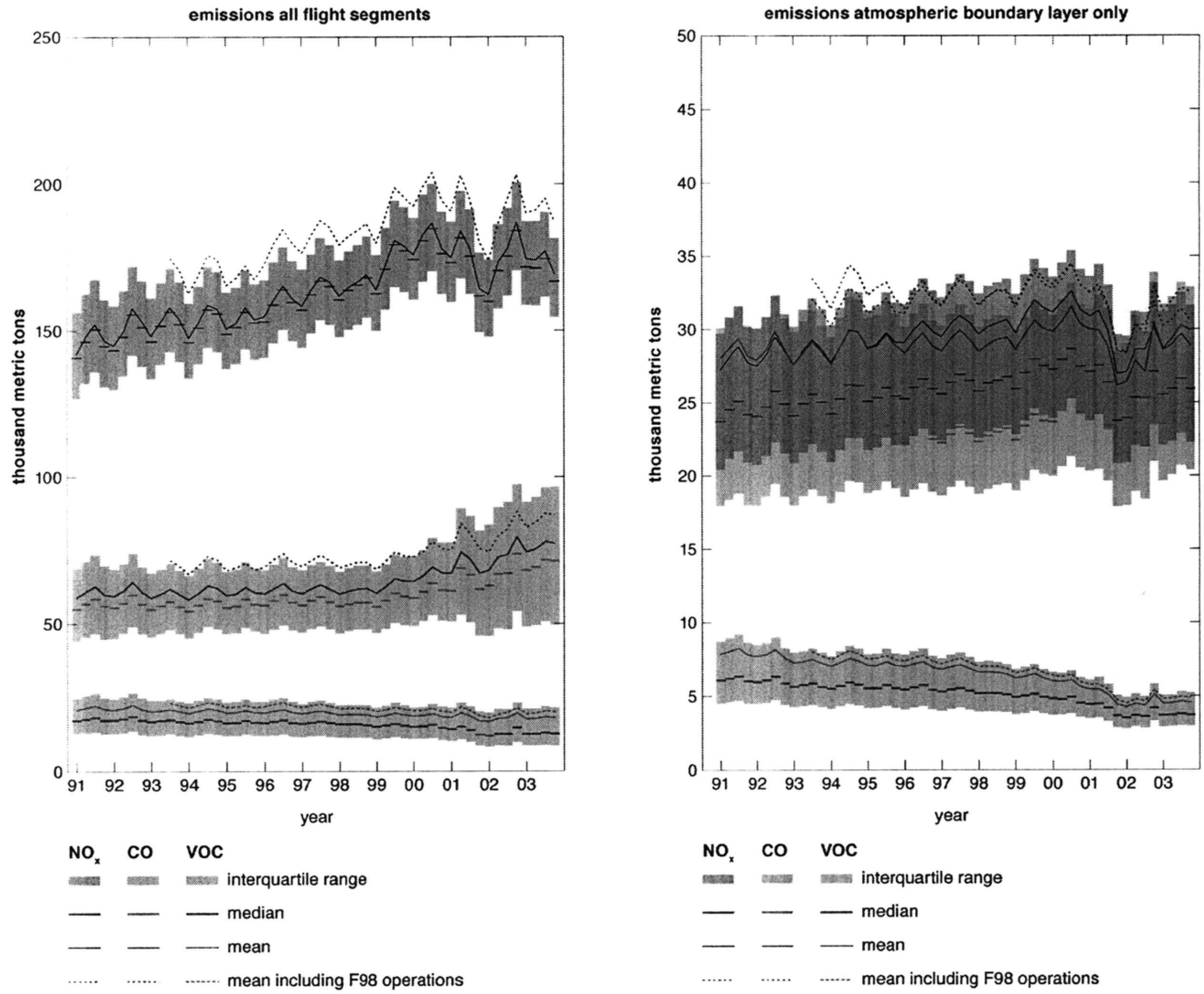
For aircraft emissions, the legal framework provides specific guidance to base regulatory action on the state of technology, both with regards to safety and developmental capability. Regulatory actions have focused on NO_x reduction through EI standards over the landing-takeoff cycle. With the influence of free tropospheric emissions on air quality noted in the previous section, the consequence is that emissions controls that attend only to the landing-takeoff cycle, as current regulations are setup to accomplish, have a lower efficacy by as much as 85% for CO and HC, and 50% for NO_x.

3.3.4. Efficacy of emissions standards in controlling total emissions loads

The inefficiency of LTO controls can be seen from a different angle looking at emissions inventory trends. Figure 3.12 plots trends in NO_x, CO, and HC emissions inventories.

Figure 3.12. Estimated NOx, HC, and CO emissions from US commercial aircraft 1991-2003

Emissions for (1) all flight activity and (2) only flight activity within the atmospheric boundary layer.



Correlation coefficients (ρ_{corr}) indicate that growth in fuel consumption within the atmospheric boundary layer is responsible for the increase in NO_x emissions between 1991-2003; correlation coefficients of the NOx boundary layer inventory with $EINO_x$-bl and Qf-bl are $\rho_{corr}\left(\hat{Q}^{bl}_{NO_x}\middle|\widehat{EI}^{bl}_{NO_x}, \hat{Q}^{bl}_{f}\right) = \{0.35, 0.98\}$ respectively. However, without the boundary layer, $EINO_x$ and Qf are similarly influential on the NO_x emissions inventory; $\rho_{corr}\left(\hat{Q}_{NO_x}\middle|\widehat{EI}_{NO_x}, \hat{Q}_{f}\right) = \{0.89, 0.99\}$. Similar results are found for CO; $\rho_{corr}\left(\hat{Q}^{bl}_{CO}\middle|\widehat{EI}^{bl}_{CO}, \hat{Q}^{bl}_{f}\right) = \{0.24, 0.78\}$ and $\rho_{corr}\left(\hat{Q}_{CO}\middle|\widehat{EI}_{CO}, \hat{Q}_{f}\right) = \{0.66, 0.70\}$.

These correlations suggest that technology standards have been more successful in limiting emissions within the atmospheric boundary layer than overall emissions, which has been their intention. They also reinforce the limited efficacy of EI controls in stemming emissions growth. The exception may be HC emissions; for HC emissions EI is the primary influence, $\rho_{corr}\left(\hat{Q}^{bl}_{\mathrm{VOC}}\middle|\widehat{\mathrm{EI}}^{\mathrm{bl}}_{\mathrm{VOC}},\hat{Q}^{bl}_{f}\right) = (0.97, 0.14)$ and $\rho_{corr}\left(\hat{Q}_{\mathrm{VOC}}\middle|\widehat{\mathrm{EI}}_{\mathrm{VOC}},\hat{Q}_{f}\right) = (0.94, -0.56)$. However the striking result is the negative correlation of fuel consumption and HC emissions. To understand this, we need to take a wider view of pollutant trends in US commercial air transport.

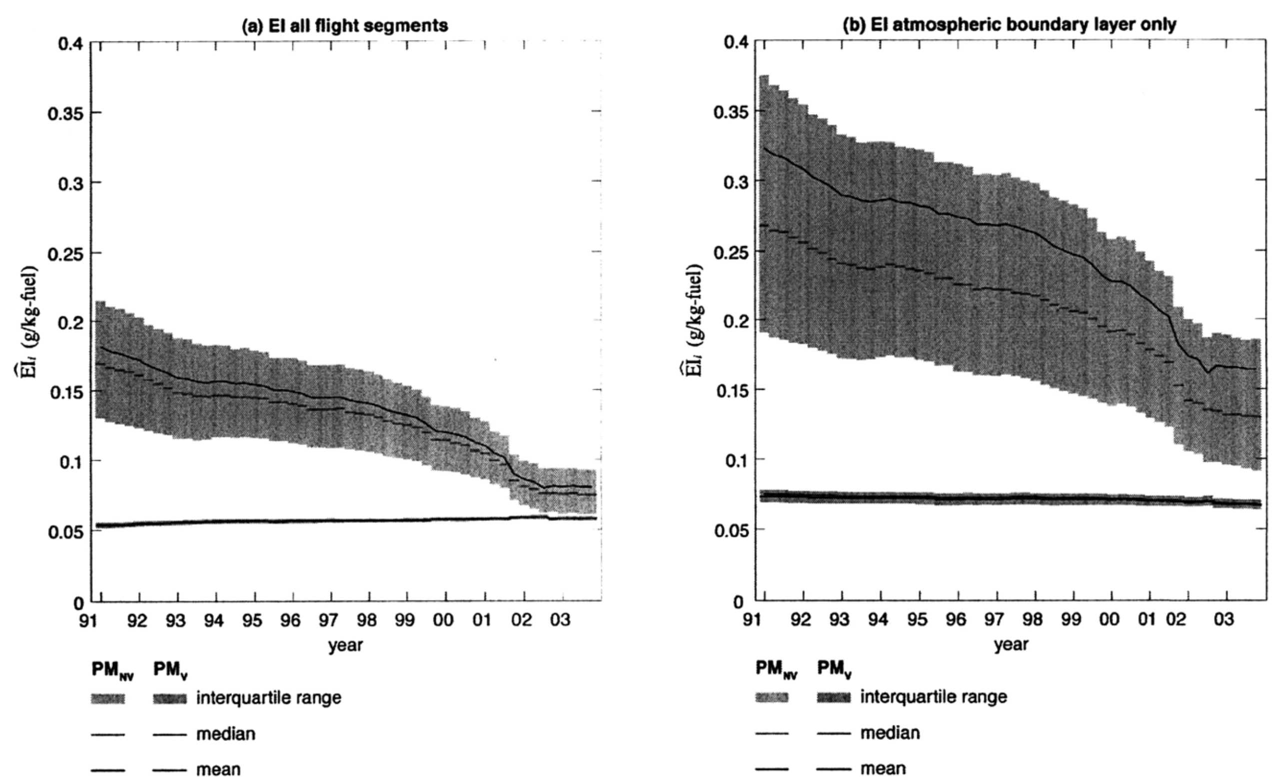
3.3.5. Crossover effect of noise regulation on emissions control

The influence of improved engine efficiency has the opposite effect on $\widehat{\mathrm{EI}}^{\mathrm{bl}}_{\mathrm{NO}_x}$ and $\widehat{\mathrm{EI}}_{\mathrm{NO}_x}$; as peak engine temperatures increase, the tendency to produce NO_x increases.

The PM inventories estimated in this thesis permit an important addition to this picture. Figure 3.13 illustrates trends in $\widehat{\mathrm{EI}}_{\mathrm{PM}_{nv}}$ and $\widehat{\mathrm{EI}}^{bl}_{\mathrm{PM}_{nv}}$, showing fleet $\widehat{\mathrm{EI}}_{\mathrm{PM}_{nv}} > \widehat{\mathrm{EI}}^{bl}_{\mathrm{PM}_{nv}}$ and declines both above and below the mixing height.

Figure 3.13. Estimated nonvolatile and volatile particulate matter emissions indices of US commercial aircraft 1991-2003

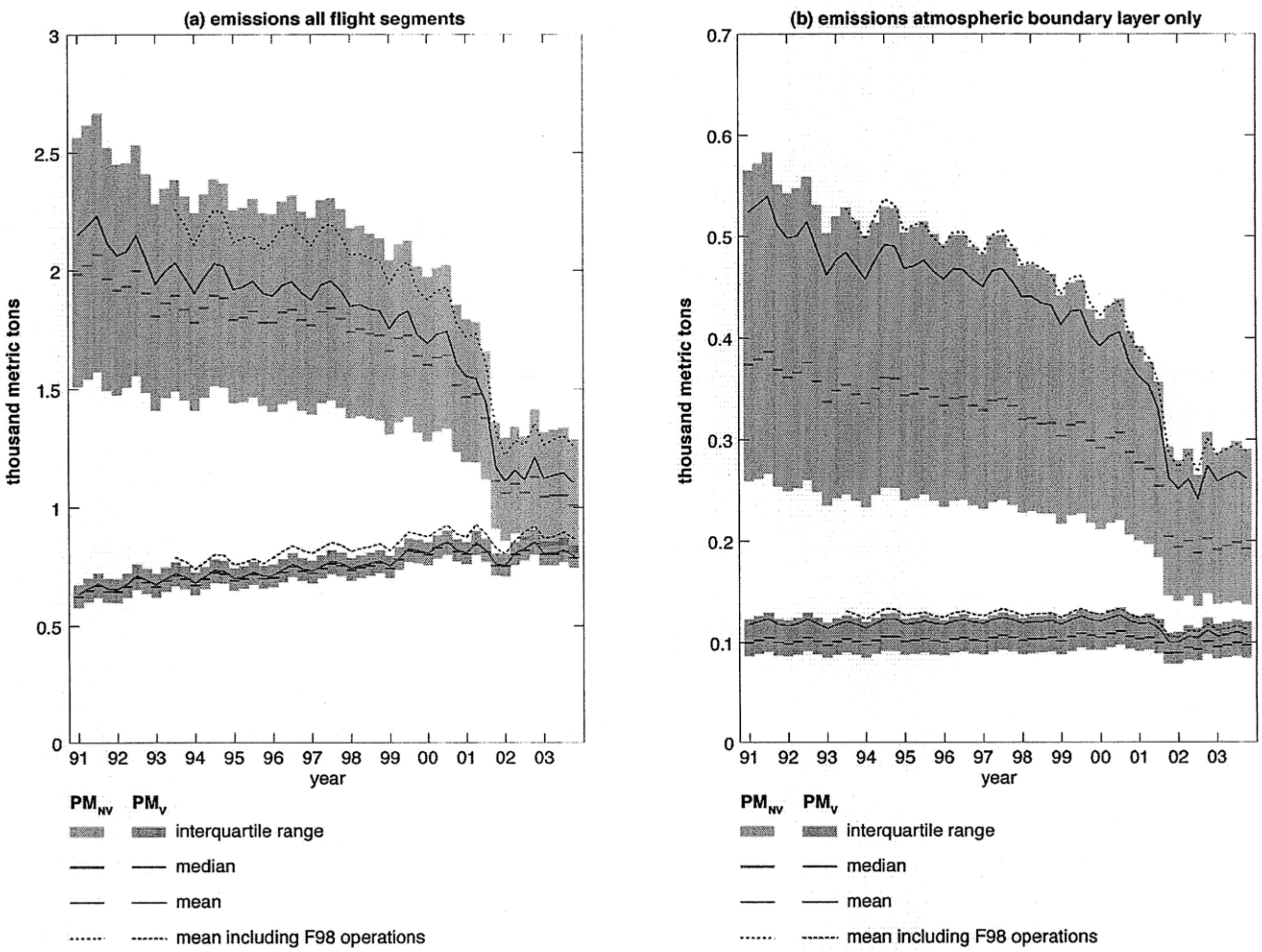
Emissions indices for (1) all flight activity and (2) only flight activity within the atmospheric boundary layer.



The rate of growth in the nonvolatile PM emissions index is negative at -1.56%. These trends are strong enough to offset increases in fuel use to produce reductions in the nonvolatile PM inventory between 1991-2003 as shown in figure 3.14.

Figure 3.14. Estimated nonvolatile and volatile particulate matter emissions from US commercial aircraft 1991-2003

Emissions for (1) all flight activity and (2) only flight activity within the atmospheric boundary layer.

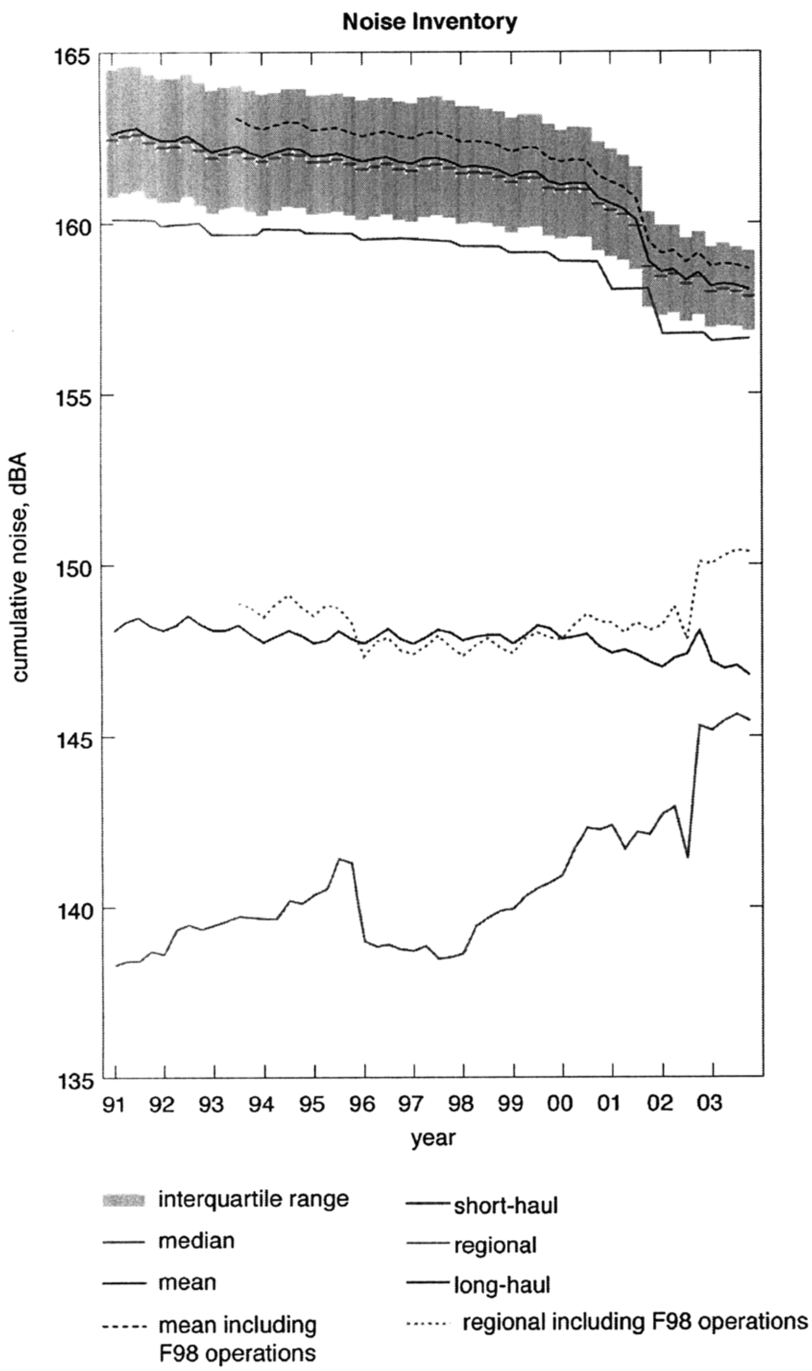


The 1991-2003 noise inventory is plotted in figure 3.15 for all fleet operations. The orange-shaded area is the interquartile range of Qn, plotted quarterly, around the median shown in the darker orange for F41T2 activity only; the orange dashed line adds F98A1 and F41T2 activity.

To understand the effect of scaling operations as described in section 3.2, compare this to the noise inventory denoted with a green line, which is constructed using only the scaled 96 airport MAGENTA operations sample ($\hat{Q}_n^{aprt}$) and plotted yearly. The important observation for the purposes of this analysis is that trends are similar over the historical period, but comparatively, $\hat{Q}_n > \hat{Q}_n^{aprt}$. This difference is due the relative division of operations among representative aircraft type in the two sets of baseline activity data (explained previously) as well as a finer attribution of per-flight noise characteristics possible for MAGENTA operations, where aircraft types are specified by individual models (e.g. by specific airframe-engine combination— q_n^{INM} —rather than the aggregated representative aircraft type q_n used to calculate of Q_n).

Figure 3.15. Noise from US commercial aircraft 1991-2003

Noise for (1) all flight activity and (2) breakout of noise contributions by service category (short-haul, long-haul, and regional service)



Over the period 1991-1999, fleet dBA SEL declined by 34% as the Stage 2 phase-out progressed.

However, this was not the primary source of noise reduction between 1991-2003. An economic shock led to a larger decline; during the period 1999-2003, a much more significant noise reduction occurred during 2001q3, resulting in a further decline of 48%. These two events dominate the overall retirement of 4.6 SEL dBA, equivalent to 65% of the noise inventory in 1991; over 90% of this decline resulted from technology turnover in the short-haul fleet.

Of interest from a policy standpoint is the strong correlation between: (1) noise and nonvolatile PM trends with correlations showing $\rho_{corr}\left(\hat{Q}_n(t):\hat{Q}_{\mathrm{PMnv}}(t)\right)$ = 0.98; and (2) noise and HC emissions where $\rho_{corr}\left(\hat{Q}_n(t):\hat{Q}_{\mathrm{VOC}}(t)\right)$ = 0.89. Reductions the $\mathrm{PM_{nv}}$ emissions inventory from 1991-2003 came almost exclusively from reductions in $\mathrm{EIPM_{nv}}$ as indicated by correlations $\rho_{corr}\left(\hat{Q}_{\mathrm{PM_{nv}}}\middle|\widehat{\mathrm{EIPM}}_{\mathrm{nv}},\hat{Q}_f\right)$ = (0.90, 0.20).[22] $\mathrm{PM_{nv}}$ emissions are nominally controlled by smoke regulations; however, these regulations were not changed between 1991-2003; the reason for this reduction was the retirement of aircraft through the Stage 2 phaseout. This also underlies the negative correlation of fuel consumption and HC emissions discussed earlier. Reductions in EIHC-bl achieved their highest reductions through the retirement of 1960s and 1970s era aircraft spurred by noise phaseout rules.

Figure 3.16 illustrates the retirement of these aircraft and suggests that as the oldest aircraft are retired, additional gains will be realized. Highlighted are those representative aircraft types that contain Stage 2 aircraft models retired from the fleet—*b727*, *b737o*, *dc9*, and *747o*—which are denoted by dashed lines. Representative aircraft types that at some point in the 1991-2003 period exceed 5% of the noise inventory are denoted by colored lines, a group accounting for 80% of the noise burden in 2003 that includes the *b727*, *b737o*, *b737*, *b737n*, *dc9*, and *dc9md80*. Reflecting the operational frequency influence on emissions mentioned earlier. the *tfan* is also in this group, the first regional type to acquire this distinction.

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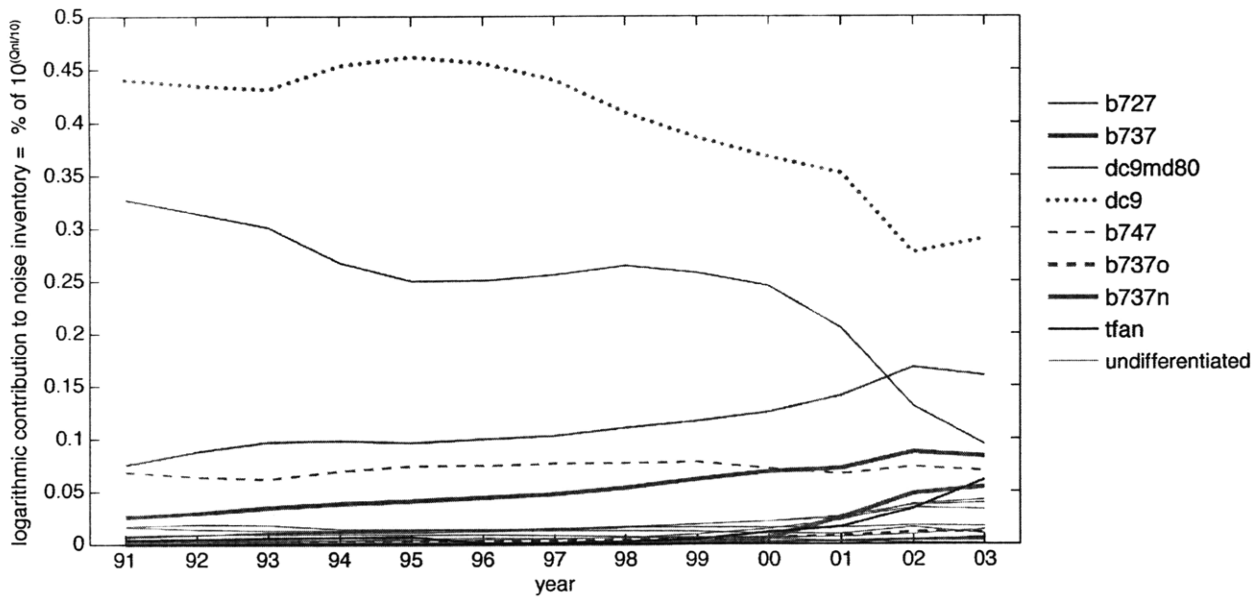
[22] Technological improvements are the root of this trend, but operational changes have an undetermined influence. MAIPA $\mathrm{EIPM_{nv}}$ values are a factor 2-3 higher than estimates made for the global commercial fleet (~0.04 g/kg-fuel). The difference in MAIPA U.S. and the global value reported in IPCC (1999) may reflect the on average shorter flight distances flown globally as compared to the United States. Similarly, the increases in service frequency and routes served since deregulation may have contributed to the trend. The time resolution of MAIPA is inadequate to quantify this operational contribution.

Figure 3.16. Contributions to noise inventory by representative aircraft type 1991-2003

Noise for all flight activity broken down by representative aircraft type highlighting only major contributions with other contributions undifferentiated in gray.



3.3.6. Inventory metrics as measures of environmental damage

The trends estimated for PM_{nv} from 1991-2003 are favorable with respect to aggregate health impacts, but the picture is complicated by the morphological characteristics of PM, for which a trend cannot yet be established. Regulation controls smoke number, a PM-related visibility metric. The historical effect of this incentive may have been to reduce the mean size of soot particulates emitted from the emissions along with total mass.

It has been suggested that this is how smoke reductions were accomplished in the development of low-smoke combustors in the 1970's. From a health perspective, migration to smaller size particulates may constitute an increase in risk, even though regulations do not resolve the gradation below the 2.5 mm size. Furthermore, because the aerodynamic diameter is reduced, these smaller particulates are more likely to persist in the atmosphere, providing more opportunities for exposure.

It is also important to recognize that any approach to reducing the effects of PM2.5 will require a joint plan addressing PM_{nv} as well as the secondary formation of nitrates, sulfates, and organics from NO_x, SO_x, and HC processing in the atmosphere. Total NO_x and SO_x inventories increased from 1991-2003 with $r_g = (0.28, \sim 0.01)$ $r_g^{bl}/r_g = (0.11, 0.17)\%$—both of which strongly influence atmospheric concentrations of PM2.5. However, inventories do not provide a complete measure of impact; air quality and climate effects are multiples of emissions so any observations here will be accentuated.[23] We will return to this point in chapter 5.

[23] A comparison of the NO_X, HC, and CO emissions inventories to reported EPA inventories for all sources (National Emissions Inventory) indicates that $\hat{Q}_i^{bl}/Q_i^{\text{EPA}}$ range from $[0.44, 0.59]\%$ for NO_X, $[0.078, 0.11]\%$ for CO, and $[0.13, 0.15]\%$ for VOC. These comparisons are for reference and are not appropriates measure of impact since the management of an externality is not a function of emissions contribution relative to other industries.

4. Damages from global climate change

Aircraft emissions have a role in several chemical and microphysical mechanisms that change the radiative properties of the atmosphere, leading to changes in temperature at the planetary surface and at altitude, and through subsequent physical mechanisms, changes in other climate variables such as sea level and precipitation. The goal in developing a climate impact model was to develop a methodology to estimate impacts on the global climate from US commercial air transport and to assess the factors that determine climate damages. Specifically, the objectives were: (1) to establish and demonstrate an approach that distinguishes the value of reducing CO_2 emissions versus non-CO_2 emissions using a metric that can be correlated with damages; and, (2) to understand the influence of different model parameters and components on uncertainty in estimated damages and relate policy implications.

The efficacy of policy options for mitigating or abating the impact of aviation emissions on environmental quality depends on the ability to compare the value of reducing CO_2 emissions versus non-CO_2 emissions in the context of climate change. To fulfill this requirement, the assessment needs to account for different timescales (and thus geographical scales) among types of perturbation to the atmosphere. This is necessary to distinguish between the longer-lived direct impacts of CO_2 emissions and the indirect impacts of other short-lived microphysical and chemical processes, such as the production of ozone or decrease in methane residence time associated with the emission of NO_X.[1] The approach implemented here provides these capabilities.

Contribution 4.1. An impulse response model of changes in surface temperature inclusive of radiatively-active species with different atmospheric lifetimes

Instead of a detailed atmosphere-ocean general circulation model, MAIPA employs an impulse-response approach to calculate probabilistic estimates of marginal, present-value climate change metrics inclusive of radiatively-active species with different atmospheric lifetimes (cf. Joos et al. 1996). The model has been evaluated and implemented in APMT.

Subsequent to the methodological development and results reviewed in this thesis, the analysis was refined and published in Marais et al. (2008) with different operational inputs that extend the analysis

[1] These differences in timescale portend differences in how effects can influence climate by virtue of the extent to which emissions are mixed in the atmosphere before their impact on radiative properties is felt.

beyond the United States and focus on aviation growth scenarios. A procedural review of the approach was prepared as a description of the APMT prototype (cf. Mahashabde et al. 2006 and FAA 2007).

The metric of climate change used to assess this effect needs to be valuable in the sense that damages can be correlated to changes in the metric. For this study, global surface temperature (T_s) is the metric of climate change. Based on analyses of damages resulting from the estimated changes in surface temperature caused by aircraft emissions, the following characterizations of air transport climate impacts emerge:

Contribution 4.2. Identified that the most important factor determining changes in damages over time is the dependence of emissions damages on the background environmental sensitivity.

Emissions impacts of US commercial aircraft are dictated by the progress in controlling emissions from other sources. The attribution of trends to parametric inputs shows that air transport emissions impacts are predominantly determined by the background environmental sensitivity, indicating that the growth of air transport emissions relative to other sources is the key factor that determines damage costs. The trend in the whole of anthropogenic carbon emissions is the primary determinant of air transport damage trends; these background trends are more influential than the course of the commercial aircraft carbon inventories cataloged previously in chapter 3.

Result 4.1. Short-lived versus long-lived climate effects

Air transport environmental decision-making is often differentiated in the context of non-CO_2 effects; however, the analysis suggests that non-CO_2 effects are a relatively small component of climate damages. From an economic perspective, only at high discount rates do cloud formation and interactions become a distinguishing focus.

Result 4.2. Influence of parametric versus scenario versus structural uncertainties

Parametric, scenario, and structural uncertainties contribute similarly to uncertainty in cost estimates. Managing the climate risks of aviation emissions is as much dependent upon (a) normative decisions underlying the specification of intergenerational wealth distribution as on (b) scientific questions of carbon-cycle and climate processes as on (c) propagated parametric uncertainties.

Result 4.3. Choice of climate impact metric

Using physical quantities as decision metrics gives significantly more weight to mitigating short-lived effects than would be recommended by an economic analysis. Measured by the ratio of non-CO_2 to CO_2 impact metrics—commonly used to describe the impact of air transport relative to other sources—is approximately 3 using instantaneous radiative forcing, while the same ratio is approximately 1.1 using a marginal cost metric. Whereas instantaneous radiative forcing is a useful measure of climate influence, marginal costs are a closer measure of risks to well-being of people exposed to climate change.

Section 4.2 discusses the models and caveats in estimating damages as a function of global surface temperature. Appendix 10 supplements the discussion with further methodological detail and background context. Section 4.3 details the relationships that compose the impulse-response approach to estimating climate impact including: the response of atmospheric CO_2 concentrations X_{CO2} to CO_2 emissions; the response of global surface temperature T_s to X_{CO2}; and the response of T_s to non-CO_2 emissions. Appendices 11-13 supplement with further methodological detail and background context. Sections 4.4-4.6 apply the model developed in Section 4.3 to assess important dynamics that characterize the impact of US commercial air transport through changes in the global climate, drawing policy implications from results.

4.1. Damages as a function of change in global surface temperature

In the case of climate, changes in environmental variables are valued directly such that estimating welfare change is not explicit; instead, damages are estimated directly as a function of a metric of environmental change (T_s) where global surface temperature change is a function of emissions. This section discusses the content and form of the damage-T_s relationship; section 4.3 then addresses the environmental modeling component.

4.1.1. Economic and equity impacts of climate change

The development of environmental damage assessments relevant to climate change lags progress in both scientific understanding as well as policy design. Climate damage estimates are highly speculative and the research literature provides no consensus on the appropriate integration of component social cost assessments. These studies integrate a number of individual reports that address different, overlapping subsets of the types of market and non-market damages that may result from climate change, primarily a combination of welfare effects resulting from changes in agricultural output, sea level, migration, water supply, and resulting shifts in the macroeconomy.

There is significant controversy over how to aggregate damages across regions. In the developed countries (e.g. the US and EU), estimated climate damages are smaller relative to the national economy than in the developing countries where estimated climate damages are a larger percentage of GNP, most of which results from the historical carbon emissions of the global regions that experienced earlier industrialization.

A second difficulty arises from the application of damage assessments conducted for the developed countries; to a developing country, where incomes are lower, the use of willingness-to-pay measures such as the value of a statistical life depress damage estimates relative to a country with a higher per capita income. Climate damages estimated in this chapter include both the portion endemic to the US as well as the global impacts associated with US national emissions, but do not further address these issues. The aggregation of regional estimates with adjustments for national income or wealth (which may be done for equity reasons) leads to an increase in the valuations relative to those presented here.

4.1.2. Climate damage as a function of global surface temperature

To relate climate damages to changes in surface temperature, Cline (1992) initially surmised (based on his own analysis) a nonlinear relationship as $C \propto \Delta\hat{T}_s^{1.3}$ relevant to long-term warming (i.e. multiple centuries). MAIPA utilizes a damage function estimated by Nordhaus and Boyer (2000), subsequently updated by Nordhaus (2008), which reflects the limited number of studies completed that assess the environmental costs of climate change. Equation 4.1 defines the climate damage function.

$\hat{C} = \beta_0 + \beta_1 \cdot \Delta\hat{T}_s + \beta_2 \cdot \Delta\hat{T}_s^{\alpha}$ = climate damage damage function

units	$\hat{C}$ as $\langle$% global GDP$\rangle$ $\quad \Delta\hat{T}_x$ in °C or K
(4.1) *define*	$\Delta\hat{T}_s$ = change in surface temperature relative to an historical reference $= \langle \hat{T}_s - \hat{T}_s^{ref} \rangle$
specify	reference year: $\hat{T}_s^{ref}$ = surface temperature in 1990 coefficients: $\beta_0 = 0 \quad \beta_1 = 0 \quad \beta_2 = P_{norm}(\mu = 0.0028, \sigma = 0.0013)$ exponent: $\alpha = 2$
source	Nordhaus and Boyer (2000) and Nordhaus (2008)

The estimated fit to these results derived in the Nordhaus studies expresses a more severe dependence on surface temperature as $C \propto \Delta\hat{T}_s^2$ and generally parallels the direction of the climate science (cf. IPCC SAR, TAR, FAR), but it is by no means definitive. Equation 4.1 fits the summation of sector-specific damages in six impact categories: agricultural patterns (cf. Darwin et al. 1995, Shimmelphennig 1996), sea level rise (cf. Yohe and Schlesinger 1998), disease incidence (cf. Murray and Lopez 1996), shifting ecosystems and human institutions (endogenous to the authors focusing on WTP to preserve associated capital), changes in markets (endogenous to the authors, primarily as related to forestry and energy production), and non-market impacts (endogenous to the authors based on time use for leisure).

Nordhaus and Boyer also attempt to account for climate dynamics that stray from a smooth transition model, such as catastrophes or surprise climate events. To add this component, they estimate a probability of climate catastrophe using a survey conducted among experts to elicit likelihood. To derive equation 4.1, a ~1% random chance of catastrophe is specified for a 2.5°C warming, and a ~7% random chance for a 6 K warming. The expected loss for the catastrophic scenario is 30% of global GDP.

4.1.3. Climate damages from US air transport emissions

To calculate climate damages, we take the difference between: (a) damages in response to the temperature change resulting from all anthropogenic emissions; and, (b) damages in response to the temperature change resulting from all anthropogenic emissions minus US aircraft fleet emissions (equation 4.2). Damage streams are then summed to net present value using the discretization in equation 4.3.

$\hat{C}_i(t) = \hat{C}^{all}(t) - \hat{C}^{all-av}(t)$ = stream of climate damages

(4.2)

definitions

$\hat{C}^{all}(t) =$ damages due to the sum forcing of global carbon emissions inventory $\langle Q_C^{all} \rangle$

$\hat{C}^{all-av}(t) =$ damages when US aircraft emissions are removed from global, $\hat{T}_s(t) = f\langle Q_C^{all} - \hat{Q}_i \rangle$

$\hat{Q}_i =$ emissions inventory of species $\langle i \rangle$ for year $\langle t_0 \rangle$

procedure

$\hat{C}(t) = f\langle \Delta\hat{T}_s(t) \rangle \equiv$ damage function in equation A.9.1

$\hat{C}_i(t)$ calculated for each year in the computational period $\langle t_0 - t_f \rangle$

$\hat{C}_i$ = present value (PV) of a stream of estimated climate damages

(4.3) *procedure*

$$\hat{C}_i \equiv \hat{C}_i(t_o) = \mathrm{PV}\left[\hat{C}_i(t)\right] = \sum_{t=t_0}^{t_f} \frac{\hat{C}_i(t - t_o)}{(1+r)^{(t-t_o)}}$$

damage stream $\langle \hat{C}_i(t) \rangle$ estimated as in equation A.9.3

specify r = discount rate $\in \{3,5,7\}\%$

4.1.4. Marginal climate damages from aircraft emissions

The marginal damage $\hat{c}$ is estimated by the derivative of equation 4.1 converted to \$/t as in equations 4.4a and 4.4b. Marginal damages for non-CO_2 emissions are pegged to the ratio of marginal to average carbon costs (equation 4.5), using the assumption that the damage function for the non-CO_2 emissions is functionally similar to equation 4.1. The marginal damage of a unit fuel consumption described by equation 4.6 is the weighted sum of the marginal emissions damages estimated by equations 4.4 and 4.5.

$\hat{c}_C^{all}$ = marginal climate damage of a unit emission of carbon

(4.4a) $$\hat{c}_C^{all} = \gamma \cdot \frac{d\hat{C}}{d\Delta\hat{T}_s} = \gamma \cdot \left(\beta_1 + \alpha \cdot \beta_2 \cdot \Delta\hat{T}_s\right) \text{ for } \hat{C} \text{ as in equation A.9.1}$$

units

$\hat{c}$ as $\langle$% global GDP$\rangle$/K

γ = converion factor to mass units $\langle \cdots \rangle/\mathrm{K} \rightarrow \langle \cdots \rangle/\mathrm{kg}$

$\gamma \succ$ backward difference approximation of the change in median values

$$\gamma = \frac{d\Delta\hat{T}_s}{d\hat{Q}_C^{all}} \cong \frac{\Delta\tilde{\mu}\langle \Delta\hat{T}_s \rangle\Big|_{t-1}^{t}}{\Delta\tilde{\mu}\langle \hat{Q}_C^{all} \rangle\Big|_{t-1}^{t}}$$

note: conversion $\langle \cdots \rangle$/kg-C $\rightarrow \langle \cdots \rangle$/EI(C) as in equation A.9.6

$\hat{c}_{\mathrm{C}}^{av}$ = marginal climate damage of unit USCAO carbon emission $\cong \hat{c}_{\mathrm{C}}^{all}$

(4.4b) *assumes* | small climate perturbation from USCAO carbon $\langle \hat{Q}_{\mathrm{C}}^{av} \rangle$ relative to global $\langle \hat{Q}_{\mathrm{C}}^{all} \rangle$
| criterion $\langle \Delta T_s^{av} \ll \Delta T_s^{all} \rangle$

$\hat{c}_i$ = marginal climate damage due to unit emissions of species $\langle i \neq C \rangle$

(4.5)

$$\hat{c}_i = \gamma \cdot \hat{c}_{\mathrm{C}} \cdot \left(\frac{\hat{c}_i^{avg}}{\hat{c}_{\mathrm{C}}^{avg}} \right)$$

define

$\hat{c}_i^{avg} = \dfrac{\hat{C}_i}{\hat{Q}_i}$ = average climate damage of species $\langle i \neq C \rangle$

$\hat{c}_{\mathrm{C}}^{avg} = \dfrac{\hat{C}_{\mathrm{C}}}{\hat{Q}_{\mathrm{C}}}$ = average climate damage of carbon

units

γ = conversion factor $\langle \cdots \rangle/\text{kg-}i \rightarrow \langle \cdots \rangle/\mathrm{EI}(i)$

$\gamma = 1000 \cdot \hat{Q}_f$ for EI(i) in $\langle \text{g-}i/\text{kg-fuel} \rangle$

$\hat{c}_f$ = marginal climate damage due to a unit mass of fuel consumption

(4.6)

$$\hat{c}_f = 0.001 \cdot \sum_i^I \hat{c}_i \cdot \widehat{\mathrm{EI}}i$$

units | EIi $\langle \text{g-}i/\text{kg-fuel} \rangle$ $\hat{c}_i$ $\langle \$/\text{kg-}i \rangle$

$\hat{c}_{\mathrm{C}}^{eqv} = \gamma \cdot \hat{c}_f =$ marginal climate damage of carbon equivalent

units

γ = conversion factor $\langle \cdots \rangle/\text{kg-fuel} \rightarrow \langle \cdots \rangle/\text{kg-C}$

$\gamma = 1000/\widehat{\mathrm{EIC}} \Rightarrow \gamma\left(\hat{\mu}_{\mathrm{EIC}}\right) \cong 1.2$

4.1.5. Estimation of change in global surface temperature

To estimate $\Delta\hat{T}_s$ we first determine how the atmospheric composition of radiatively active species responds to aircraft emissions, then estimate the change in radiative balance, and finally the impact on T_s (although the process is applicable to any output variable that measures change in the climate system). The work of (Sausen and Schumann 2000) first introduced the basic approach outlined in section 4.3 toward estimating T_s effects associated with commercial aircraft operations. Instead of a detailed atmosphere-ocean general circulation model, MAIPA employs an impulse-response approach to calculate probabilistic estimates of marginal, present-value climate change metrics inclusive of radiatively-active species with different atmospheric lifetimes (cf. Joos et al. 1996). The model has been evaluated and implemented in APMT.

4.1.6. Air transport damages in the context of other source emissions

The observed trend in aviation damages is intimately tied to activity in the rest of the economy where emissions grew exponentially from 1991-2003. With CO_2 RF a logarithmic function of its atmospheric concentration, a temperature response linear with CO_2 RF, and costs quadratic in T_s, damage costs from the whole of anthropogenic emissions rise less than the emissions trend where there is a less than an exponential exp(x) growth in emissions. Because $\mathrm{RF}_{\mathrm{CO_2}} \propto \ln\left(X_{\mathrm{CO_2}} / X_{\mathrm{CO_2}}^{ref}\right)$, $\lim_{t\to\infty}\left(\mathrm{dRF}_{\mathrm{CO_2}}/\mathrm{dt}\right) = 0$ and only for exponential growth will $\mathrm{RF}_{\mathrm{CO_2}}$ grow linearly. In contrast, for declines in carbon emissions, costs decline at a faster rate. However, this is not the primary factor determining trends in damages from air transport.

The perturbation of the aircraft emissions impulse (equal annual emissions for a given year) to background emissions resolves mathematically as climate damages proportional $C \propto \left(T_2^2 - T_1^2\right)$ where T_2 is the temperature response to all source emissions and T_1 is the response subtracting aviation. This proportionality indicates two characteristics governing how air transport damages change over time. First, climate damages respond more than proportionally to year-to-year increases in aircraft emissions with positive RF, but less so for declines (the reverse applies to cooling effects). Of the contributing species, only PMnv emissions decline over the 1991-2003 period. Second, and more importantly, because the effect of air transport is relatively small, the trend in the rest of anthropogenic carbon emissions is more influential on the air transport damage trends than the course of the commercial aircraft carbon inventories presented in chapter 3.

4.2. Response of global surface temperature to CO_2 emissions

Carbon dioxide mixes in the atmosphere on a timescale that is small compared to the duration of its radiative effect. To develop the response model, a linear, time-invariant representation of a carbon-cycle model—the impulse response or Green's function, *g(t)*—is convolved with a forcing, *f(t)*, as carbon emissions, to obtain an output, atmospheric CO_2 concentrations in this case. This is shown generally in equation 4.7. In the following discussion, we will use X_{CO2} to indicate the atmospheric CO_2 concentration.

(4.7)
$$\phi(t) = f * g = \int_{t_0}^{t} f(\tau)\, g(t-\tau)\, d\tau$$

The impulse response is determined by harmonic analysis, solving the system identification problem with a known input function and decomposing the model response with the Fourier series as given in equation 4.8.

(4.8)
$$g(t) = \sum_{j}^{J} \alpha_j e^{u_j t}$$
$$u_j = \lambda_j + i\omega_j$$

More complex transfer functions can be generated by other forcings, so while the impulse response is shown here as a single parameter model in time, which is appropriate to well-mixed CO_2, we could represent spatial dimensions with additional parameters.

4.2.1. Estimation of marginal change in atmospheric CO_2 due to aviation CO_2 emissions

Executing the convolution gives, for any particular year emissions $\hat{Q}_{CO2}$ as the forcing, a response X_{CO2}. Few complex models of the carbon-cycle have been projected in a linear form for use in response analyses. To assess variability associated with different carbon cycle model constructions, this study examines results from five linear carbon-cycle response models derived from two different base inorganic general circulation carbon-cycle models. The response models differ in the carbon-cycle pathways included in the base model for their derivation. Table 4.3 in section 4.6 summarizes the differences.

See Appendix A10 for additional discussion:

- Appendix A10 (Linear response models of the climate) describes the linear response models of the carbon cycle used in the analysis, including a brief history of their development.

The change in atmospheric CO_2 concentration ($\Delta\hat{X}_{CO2}$) due to air transport emissions is estimated by the difference between the baseline (all anthropogenic emissions) and the perturbed baseline (all anthropogenic emissions minus US aircraft fleet carbon emissions) as given by equation 4.9. Note that in contrast to equation 4.8, a constant has been added to g(t) in equation 4.9 to better represent long-term finite atmospheric CO_2 uptake.

$$\Delta\hat{X}_{\mathrm{CO2}} = \left(X_{\mathrm{CO2}} - \hat{X}'_{\mathrm{CO2}}\right) = \text{ change in atmospheric } \mathrm{CO_2} \text{ due to USCAO emissions}$$

(4.9)

$$\begin{array}{l|l} \textit{definitions} & \begin{array}{l} X_{\mathrm{CO2}} \\ \hat{X}'_{\mathrm{CO2}} \end{array} \\ \hline \textit{procedure} & \begin{array}{l} X_{\mathrm{CO2}} = g(t)\cdot Q^{ref}_{\mathrm{CO2}} \\ \hat{X}'_{\mathrm{CO2}} = g(t)\cdot\left(Q^{ref}_{\mathrm{CO2}} - \hat{Q}_{\mathrm{CO2}}\right) \end{array} \\ \hline \textit{specify} & g(t) = \alpha_1 + \sum_{j=2}^{J} \alpha_j e^{-t/\tau_j} \end{array}$$

4.2.2. Functional relationship between CO_2 concentration and radiative forcing

Observations find radiative forcing to be proportional to the logarithm of current X_{CO2} relative to an unperturbed state; the unperturbed state is defined as the pre-industrial X_{CO2} taken here to be the year 1750 (equation 4.10).[2] This relationship results from the the infrared CO_2 absorption bands being close to saturation (Myhre et al. 1998).

(4.10)
$$\mathrm{RF}_{\mathrm{CO2}} \propto \ln\left(X_{\mathrm{CO2}} / X^{ref}_{\mathrm{CO2}}\right)$$

The temperature response to CO_2 radiative forcing is estimated using a normalized index of radiative forcing (RF*) referencing the doubling of atmospheric CO_2 concentrations ($2\cdot X_{\mathrm{CO2}}$) relative to the preindustrial X_{CO2}. This is a convenience facilitating the incorporation of a common benchmark of atmospheric-ocean general circulation model (AOGCM) prediction, the radiative forcing estimated for a $2\cdot X_{\mathrm{CO2}}$ calculation. In 4.11, RF* equals 1 at the doubling level.

[2] Radiative forcing measures the thermodynamic imbalance in the system defined by the Earth's land mass, oceans, and atmosphere as the result of compositional changes in the atmosphere which alter its opacity to either incoming or outgoing solar radiation.

$$\text{(4.11)}\qquad \mathrm{RF}^*(t) = \frac{\ln(X_{\mathrm{CO_2}}(t) / X_{\mathrm{CO_2}}^{ref})}{\ln(2)}$$
$$\mathrm{RF}(t) = \mathrm{RF}_{\mathrm{ref}} \times \mathrm{RF}^*(t)$$

4.2.3. Estimation of marginal temperature change due to aviation CO_2 emissions

As with the atmospheric carbon dioxide concentration, MAIPA uses impulse response functions, derived from AOGCM simulations, to estimate changes in global surface temperature. For a given year. the radiative forcing index RF* is specified as the forcing f(t) to derive the surface temperature response using the convolution in equation 4.7. The estimated impulse response that results gives the model in equation 4.12 for change in global surface temperature.

$$\text{(4.12)}\qquad T_s^{ref}(t) = \mathrm{RF}^* \cdot \sum_{j=1}^{J} \alpha_j e^{-t/\tau_j}$$
$$\hat{T}_s^{ref'}(t) = \mathrm{RF}^* - (\mathrm{RF}^*)' \cdot \sum_{j=1}^{J} \alpha_j e^{-t/\tau_j}$$
$$\delta \hat{T}_s = T_s^{ref} - \hat{T}_s^{ref'}$$

MAIPA employs three linear surface temperature response models to construct scenarios. For the scenarios treated later in this section, each of the five linear carbon-cycle response models are paired with an era-consistent model for temperature response. Jointly, these cases are termed hereafter linear climate response models (CRM). These CRMs represent the evolution in climate model construction over a 15-year period that includes the historical MAIPA analysis for the years 1991-2003.

See Appendix A10 for further discussion:

- Appendix A10 (Linear response models of the climate) describes the linear response models for global surface temperature used in the analysis, including a brief history of their development.

4.2.4. Effect scales of non-CO_2 climate perturbations

Radiative forcing from aircraft operations results from perturbations to both well-mixed gases such as CO_2, for which the primary sinks occur through centurial processes, and perturbations to radiatively active species and particles that occur over timescales of hours to days as the result of faster chemical or microphysical processing towards their eventual sinks. This latter category of perturbations results in regional scale change that may cause locally different climate impacts.

However, the summation of globally-averaged RF from both regional and non-regional effects tracks globally-averaged surface warming to first-order (Cox et al. 1995; Ramaswamy and Chen 1997) Ramswamy 2001, cf. IPCC 2007 FAR WG1). Thus, long-lived and short-lived perturbations are typically compared using an instantaneous RF metric.[3] This would be fine if regional and global RF perturbations lead to the same climate sensitivity, but they do not. This section discusses how MAIPA accounts for these differences.

4.2.5. Estimation of short-lived climate effects

Following (Sausen and Schumann 2000) and (Lee and Sausen 2003), we represent aviation short-lived effects through a scaling of RF* for a different climate response relative to CO_2—related to spatial heterogeneity—and proportionally account for changes in emissions indices from a reference year. In this formulation, shown in equation 4.13, each subsequent unit of short-lived emissions or effect causes the same change in radiative forcing; for exponential growth, ΔRF_{short} grows exponentially, and similarly, for linear growth, ΔRF_{short} grows linearly. Thus, for longer and longer periods of time over which the emissions accumulate, the marginal radiative effects of aviation CO_2 are reduced relative to those for short-lived emissions and effects.

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[3] The present day cumulative impact of aviation emissions suggests that today, the mixture of exhaust species discharged from aircraft perturbs RF 2 to 3 times more than if the exhaust was CO_2 alone. In contrast, the overall radiative forcing from the sum of all anthropogenic activities is estimated to be a factor of 1.5 times CO_2 alone.

Enhanced forcing due to aircraft compared with ground-based sources originates in different physical (*e.g.* contrails and impacts on clouds) and chemical (*e.g.* ozone formation/destruction) effects resulting from altered concentrations of participating chemical species and different atmospheric conditions (IPCC 1999; Schumann 2003).

Total RF from aircraft for 1992 is ~4% of the direct RF from other anthropogenic sources combined. RF from additional CO_2, CH4, N2O, CFCs / methyl chloroform/carbon tetrachloride, and HCFCs / HFCs are estimated to be, respectively in W/m2, 1.46, 0.48, 0.15, 0.32, and 0.09 (IPCC 2001 TAR, cf. IPCC 2007 FAR for 2005 values).

$$\mathrm{RF}^{*}_{short}\left(t_k\right)=\lambda^{*}\cdot\alpha\cdot\frac{\mathrm{RF}^{ref}_{short}}{\mathrm{RF}_{2\cdot X_{CO_2}}}\cdot\frac{Q_i\left(t_k\right)}{Q_i^{ref}}$$

(4.13)
$$\lambda^{*}=\frac{\lambda_{short}}{\lambda_{CO_2}}$$

$$\alpha=\frac{Q_{CO_2}\left(t_k\right)}{Q_{CO_2}^{aviation}}$$

As with CO_2, short-lived effects do not grow instantaneously over time, but have a timescale of increase followed by decay (Prather 2002). Since the unit reference for an emissions impulse in MAIPA is a year, dictated by the resolution of the activity data, the rise is not simulated.

In equation 4.13, $\mathrm{RF}_{2\cdot X_{CO_2}}$ is the equilibrium radiative forcing for a doubling of atmospheric CO_2, t_k refers year k, λ_{CO_2} is the climate sensitivity to CO_2 or other well-mixed perturbations, and λ_{short} refers to the climate sensitivity for short-lived effects; λ and τ are closely related.

4.2.6. Specification of reference parameters for short-lived effects

Reference RF, EI, yearly fuel consumption, and yearly emissions (RF^{ref}, EI^{ref}, and Q^{ref}) correspond to the same reference year, here 1992. Applying the benchmark results reported in the IPCC third assessment report (IPCC), $\mathrm{RF}_{2\cdot X_{CO_2}}$ in equation 4.13 is specified using a triangular probability function with endpoints at 3.5 and 4.1 W/m^2 and likely value at W/m^2, i.e. P(*trig*; 3.5, 3.7, 4.1) W/m2; we use the TAR results since the AOGCMs behind the impulse response functions for temperature change were built with this reference. Section 3.4 of chapter 3 discusses the evolution of EI_i over the period 1991 to 2003.

For the purposes of this study, we use the estimates for $\mathrm{RF}^{ref}_{short}$ in (Schumann 2003) (cf.(Sausen et al. 2005), which update the estimates published IPCC special report on aviation and the global atmosphere (1999) based on a review of recent literature. Climate sensitivities in equation 4.13 are specified as distributions based on a literature review detailed in appendices A12 and A13.

See Appendices A12 and A13 for additional discussion:

- Appendices A12 (Ozone and effects related to nitrogen oxide emissions) and A13 (Aerosol and cloud effects related to water, soot, and sulfur emissions) discuss the specification of climate sensitivity and instantaneous radiative forcing for short-lived effects.

4.2.7. Estimation of NOx perturbation to the lifetime of atmospheric methane

The production of ozone via NOx also leads to a decrease in the lifetime of atmospheric methane, a radiatively active greenhouse gas, as a result of hydroxy radical (OH) production mediated by CO. While the proper mechanistic representation would directly account for this change in lifetime as a perturbation to the methane cycle (Fung et al. 1991; Lelieveld et al. 1998), it is not in the scope of this analysis to develop a reduced-order representation suitable for MAIPA as we have for carbon dioxide.

Instead, $RF^{ref}_{CH_4}$ is derived by scaling the value of $RF^{ref}_{O_3}$ as summarized in equation 4.14. The approximation of the integral ratio assumes a lifetime << 1 yr for O3 (few days to a few weeks) and drops the upper limit of the integral for CH4 as a negligible contribution.

$$\frac{\int_0^{100} RF_{O_3}(t)dt}{\int_0^{100} RF^{ref}_{CH_4}(t_0)e^{-1/(\tau_{CH_4})}dt} \approx \frac{RF^{ref}_{O_3}}{\tau_{CH_4} RF^{ref}_{CH_4}} = \frac{RF^{yr}_{O_3}}{RF^{yr}_{CH_4}} \qquad (4.14)$$

$$RF^{ref}_{CH_4} = \frac{RF^{ref}_{O_3}}{\tau_{CH_4}} \frac{RF^{yr}_{CH_4}}{RF^{yr}_{O_3}}$$

In MAIPA, RF effects due to methane destruction persist with an uncertain decay timescale of specified as a uniform probability function with endpoints at 10 and 14 years, i.e. P(*unif*; 10, 14) , after the initial emission (cf. Prather 1996). Since methane is a well-mixed gas like CO_2, the climate sensitivity ratio $\lambda^*_{CH_4} \sim 1$; (Hansen et al. 2005) estimates $\lambda^*_{CH_4} = 1.1 \pm 0.02$. This value is implemented in MAIPA as $\lambda^*_{CH_4} = 1.1$. Values for RF-yr are uncertain; section 4.5 examines three alternative specifications for $RF^{yr}_{O_3}/RF^{yr}_{CH_4}$ as scenarios to represent different model formulations that have been proposed to estimate this ratio (Stevenson et al. 2004, Derwent et al., 2000 Wild et al. 2002) The values estimated by Stevenson et al. (2004) are used in the baseline case, but this does not represent a preference, only chronology ($RF^{yr}_{O_3}$ = 0.0051 W/m2 and $RF^{yr}_{CH_4}$ = -0.0042 W/m2.

4.3. Economic development, carbon damages, and uncertainty

The initial published estimates for the marginal damage cost of carbon that emerged in the early 1990s reference a particular change in climate state, commonly an equilibrium $2 \cdot X_{CO_2}$ atmosphere (cf. SAR table 6.1 IPCC 1996). These state-change analyses calculate the marginal damage cost as total climate damages, discounted over a projected evolution of the climate and then divided by total carbon emissions to derive a monetary value per unit mass emitted. These estimates imply that marginal damages are equivalent to average damages, $c \cong \bar{c} = C/Q$. More recent considerations move away from state-change analyses to explicitly consider damages as a function of a climate change metric. The damage function in equation 4.1 at the beginning of this chapter is of this generation.

4.3.1. Normative context of the discount rate

To assess the role of uncertainties related to preferences for mitigating climate change and the unpredictable course of global development, MAIPA exercises two sets of exogenously specified scenarios that influence damage costs ($\hat{C}$) via economic parameters: (a) two different assumptions for economic development, the IS92a and IS92e scenarios, and (b) three different specifications of the discount rate $r = [3, 5, 7]\%$.

This thesis does not express an opinion as to the appropriate discount rate—this is pragmatically a political decision. Instead, a range of discount rates are evaluated—$r = [3, 5, 7]\%$—via the computational scenarios described in the following section. Note that US EPA guidance has changed and now recommends evaluating $r = [2, 3, 7]\%$; MAIPA damage results for a 3% discount rate are used as the baseline for the purposes of analysis comparisons.

See Footnote 4 for additional discussion:

- This note provides a brief description of the theoretical underpinnings in specifying the discount rate and relevant definitions.

There are two prototypical approaches to the (much debated) specification of the discount rate for climate economic analysis. One approach assumes that risk in climate assets is similar to existing assets. For example, Nordhaus (2008) employs observable returns on corporate capital to specify the discount rate directly (r = 6➤4.5% where the arrow indicates the decline in expected real rates of return due to uncertainty). Assuming a time preference (r_g = 1.5%) and making endogenous projections of consumption growth determines the consumption elasticity ($\theta \sim 2$); collectively, these parameters emphasize a more limited investment horizon.

A second approach places greater emphasis on distributional considerations. For example, the Stern report (2007) specifies the time discount rate (r_g = 0.01%) and a consumption elasticity ($\theta = 1$) that incline towards increased income for the future economy, arriving at a lower discount rate (r = 3.5➤1.5%) that implies mitigating climate change is a lower risk investment than traditional capital. The Stern report (2007) also presumes long-term consumption growth at ~1.5%. Thus, there is a difference in the assumed

[4] *Note on the discount rate:* With the positive assumption that policies are designed to improve the living standards of both current and future generations, the equation $r = r_t + \theta \bullet r_g$ specifies the discount rate, or the real rate of capital return, as a function of the rate of consumption growth in the context of the welfare economic theory of intergenerational discounting developed by Ramsey (1928, cf. Groom et al., 2005).

In contrast to the capital depreciation (and thus lower return on capital) of housing associated with aircraft noise effects—observable in property markets—investment return as it pertains to mitigation of climate impacts includes a fundamentally normative decision about the distribution of welfare among generations. This component, incorporated in equation 6.22 as the time discount rate (r_t), reflects our preference for welfare today versus welfare for future generations. Increasing r_t shifts welfare to the present generation.

The second normative parameter used to estimate capital return is the elasticity of the marginal utility of consumption θ—the rate of change, with respect to income, in the utility derived from a change in consumption. This parameter measures our aversion to leaving future generations poorer; as such, it measures the curvature of the utility function and specifies risk aversion (Guo et al., 2006). For $\theta = 1$, additional income for a future generation with twice the consumption will provide that generation with half the utility. Thus, increasing θ translates into less aversion. The consumption elasticity and time discount rate are distributional concepts. The rate of consumption growth (r_g) derives from assumptions about economic development and is historically positive as real incomes have increased over time.

There is a decline in expected real rates of return over time that results from uncertainty about the future evolution of the discount rate. For example, if a range of discount rates is equally plausible, the longer the time-horizon of a project, the more the expected return will deviate from the return calculated using the midpoint of the range (Newell and Pizer 2003, cf. Weitzman 2001, Ainslie 1991). In other words, because of the uncertainty of a return for long-term projects, such as climate investments, one is more likely to apply a lower discount rate, assuming risk aversion (i.e. certainty equivalents decline over time). This is sometimes used as an argument for applying a lower (constant) discount rate in benefit-cost analyses for intergenerational environmental issues as opposed to current goods rates. A declining discount rate has the effect of lengthening the tail of the value stream.

trajectory of discount rate decline with time, but the main contradiction exemplified by this comparison regards decisions about intergenerational welfare.

4.3.2. Relationship between growth and discount rate

Whereas differences in climate response affect the magnitude of damage costs, growth and the discount rate, which are related, additionally discriminate perturbations by their effect lifetimes. The use of IS92 scenarios is intended to qualitatively understand the impact of economic development uncertainty—that is, uncertainty in the baseline anthropogenic CO_2 emissions—and should not be interpreted as an investment in any particular projection of societal change. The IS92 economic assumptions reflect consumption growth at r_g = [0.032→0.019; 0.042→0.027], respectively, decaying over a 100-year period 2000 to 2100. Carbon emissions over the last decade have tracked the IS92e scenario most closely.[5] Climate analyses conducted for the IPCC TAR and FAR use a different set of scenarios to evaluate economic development uncertainties (see IPCC SRES 2000). These IPCC scenarios are used in current APMT climate assessments.

Decreasing r_g, represented by the IS92 scenarios, will conversely decrease the magnitude of the cost streams. Higher rates of consumption growth mean higher anthropogenic emission rates and thus higher temperature change. Different values for r represent a constrained set of specifications for r_t and θ, but pairs are not unique to any one specification $[r, r_g]$. The discount rate and growth rate are separated here to convey the difference in preference variability versus variability in economic projections. In application, only the IS92c scenario can be meaningfully associated with r=0.01 while the IS92a and IS92e scenarios cannot.

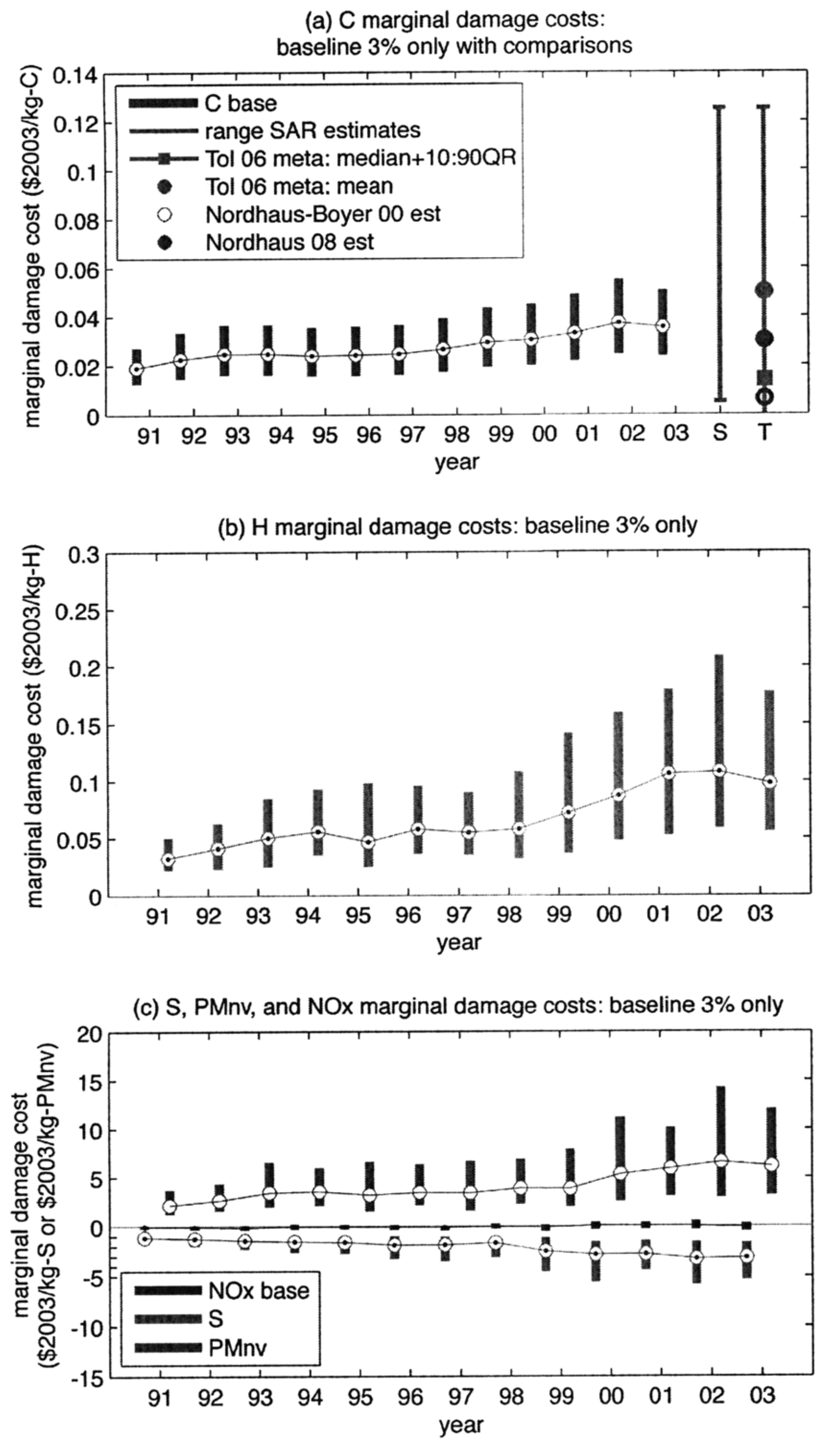
4.3.3. Estimated marginal climate damages 1991-2003

Figure 4.1 plots trends in MAIPA-estimated marginal climate damages from 1991 to 2003 for carbon, water vapor (as H), NO_x, sulfur, and PM_{nv} emissions in \$2003/kg. The baseline case (r_g = 3%) plotted in figure 4.1a is consistent with published estimates of the social cost of carbon (without equity weighting)

[5] Pepper, W.J., Xiaoshi Xing, Robert S. Chen, and Richard H. Moss (Eds.), Intergovernmental Panel on Climate Change (IPCC) Scenarios 1992 (IS92), A to F, Digital Version 1.1, 2005, Palisades, NY: CIESIN, Columbia University. Available at http://sedac.ciesin.columbia.edu/ddc/.

and shows similar statistics; from 1991-2003, the median in \$2003 increases from \$[19->30]/tC with a coefficient of variation = 0.53 (the 10:90 quartile range for 2003 = \$[5 140]/tC).

Figure 4.1. Estimated marginal climate damages of US commercial aircraft emissions 1991-2003



Nordhaus (2008) estimates marginal damage cost in the context of the Dynamic Integrated model of the Climate and the Economy (DICE). In DICE, the economy and climate are linked through emissions and carbon price feedbacks. DICE employs a simplified climate model which includes: (a) a linear three-reservoir (atmosphere, upper ocean, lower ocean) carbon-cycle model calibrated to match the Bern model; and (b) a three-reservoir heat transfer model to estimate temperature change following the construction in Schneider and Thompson (1981) with parameters calibrated to mimic results from the Model for the Assessment of Greenhouse Gas Induced Climate Change (MAGICC) as formulated for the TAR and FAR ($\lambda = 3.0$, ref MAGICC 2007).

DICE estimates the marginal benefit from abatement (shadow price of carbon) as the carbon tax necessary to keep emissions on an optimal trajectory, described by policy constraints. In the case of a Pigouvian tax, this optimization calculates a trajectory of carbon prices that efficiently reduces total social costs over a given period. In an optimization framework, the price of carbon will necessarily be less than marginal cost estimates.

Nordhaus (2008) exercises a number of policy scenarios that exhibit similar marginal damages in the initial years of the computation where the marginal damage cost of carbon— ~ $30/tC for 2005 (2003 dollars)—is near the optimal carbon price, progressively diverging thereafter and differentiating policies by their total costs (cf. Tol 2002b, 2002a for a comparative proposal for dynamic representation using dynamic cost-benefit models). As shown in figure 4.1a, the social cost of carbon estimated using the impulse-response methodology is consistent with the Nordhaus computations.

Tol (2005) reviews a number of published estimates for the social cost of carbon, summarizing the literature with a frequency distribution for the marginal damage cost of carbon. This summary is most appropriately interpreted as an expert elicitation similar to the SAR range identified previously (cf. IPCC WG2 Report, 2001, Table 19-4). Figure 4.1 plots the 10:90 percentile range = $[-2 125]/tC, mean = $50/tC, and median $14/tC of one presentation of this distribution, selecting only peer-reviewed studies that provide marginal estimates, but including a range of analyses that variously do and do not exercise equity weighting.

These studies use time discount rates ranging from ~0% to 3%. Higher marginal damage values are associated with higher discount rates and the application of equity weighting. Most of the studies reviewed in Tol are deterministic; since variability among these results due to modeling differences is to some extent endogenous to the propagated parametric uncertainties in MAIPA, this similarity indicates that the uncertainties captured through MAIPA are broadly characteristic of the parametric choices made in these studies for economic and physical specifications.

For instance, for the marginal carbon damage in this study, higher discount rates (5% and 7% for the baseline case) result in a factor of ~4 decrease in the median marginal carbon damage, similar to the variation in the median of the Tol distribution as the time discount rate is increased from 0-3% (factor 5 decrease). Note that removing equity-weighted studies from the distribution essentially culls values that form the extreme upper tail, changing the risk profile, but not the central tendency. For MAIPA, structural uncertainties related to the specification of the climate response model have a smaller effect on marginal damages (linear versus quadratic dependence on T_s).

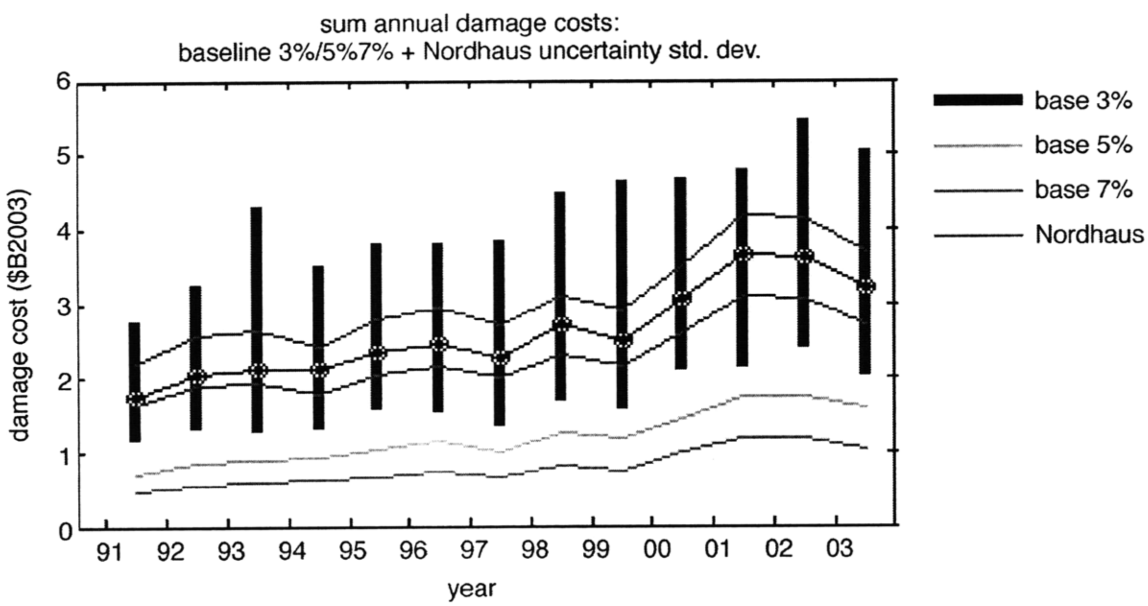
4.3.4. Uncertainty in damage function parameters

Managing climate risks requires a way to weigh the characteristics of these options to determine not just magnitude of benefit, but also how likely it is that we can attain that potential; the damage function estimated in this thesis inherits parametric uncertainty along the entire impact vector, from source through economic repercussions.

In contrast to the diminishing marginal damage functions underlying noise (cf. chapter 6) and air quality (cf. section 4.5) for commercial aviation, climate marginal damages are an increasing function of deterioration in environmental quality. However, the exacerbating tendencies in each one of these marginal damage functions corresponds to the 1991-2003 evolution of the source inventory component that dominates the annual damage estimate—i.e. dBA SEL, NO_x/SO_x, and CO_2/H_2O respectively. As a result, the marginal damages of emissions and noise also increase over this period. The marginal damages of fuel consumption quantify this trend directly; chapter 7 presents estimates of the marginal damage cost of fuel consumption and discusses implications for the realization of benefits through fuel efficiency efforts.

The coefficient error noted in equation 4.1 is not an equivalent to this chained evaluation; instead the error expresses uncertainty in the statistical fit to the sector-specific damage assessments that constitute equation 4.1. As such, the coefficient error is the final parametric uncertainty applied for MAIPA climate damage assessment. Figure 4.2 plots annual damage costs as a function of time for the baseline scenario at $r = 3\%$ for each of these emissions. Figure 4.2 compares the portion of the interquartile range due only to uncertainty in the climate damage function error against the distributions for the carbon damage cost with all parametric uncertainties propagated. Over the period 1991-2003, the coefficient error alone generates a distribution with 30-50% of the damage cost IQR. This reinforces the highly uncertain quantifications of climate damages currently available and suggests that it is important to communicate a range of results using functional forms; updates to this literature are important to consider.

Figure 4.2. Estimated climate damages of US commercial aircraft emissions 1991-2003



4.4. Comparative emissions contributions to climate damages

4.4.1. Estimated climate damages 1991-2003

Figures 4.3 and 4.4 plot the uncertain damage costs (in 2003 dollars) resulting from 1992 US commercial aircraft emissions for the nominal baseline CRM at a 3% discount rate. The three distributions plotted in figure 4.3 are the present value summations of the stream of future damages due to all emitted species and their breakdown into CO_2 effects and non-CO_2 effects for the baseline scenario.

There is a sharp distinction between CO_2 effects and non-CO_2 effects; climate perturbations with long lifetimes last O(10)-O(100) times longer than short-lived perturbations (see footnote 6 for an illustration of the damage streams). The ratio of non-CO_2 to CO_2 damages is 0.32 with an interquartile range of [0.15 0.67].

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[6] *Note on damage cost streams:* Figure F4.1a plots the uncertain stream of damage costs (in 2003 dollars) resulting from 1992 US commercial aircraft emissions for the nominal case. Although the figure illustrates just one year, the shapes of these functions are characteristic of all cases. To emphasize details, only the first 150 years of the calculation are plotted, although the calculation extends 380 years after the emissions impulse to cover the characteristic timescales of the sinks that control CO_2 atmospheric lifetime.

Figure F4.1b plots the stream of future damages due to all emitted species and their breakdown into CO_2 effects and non-CO_2 effects for the baseline scenario; the three distributions plotted in figure F4.1a in the main text are the present value summations of these damage streams respectively.

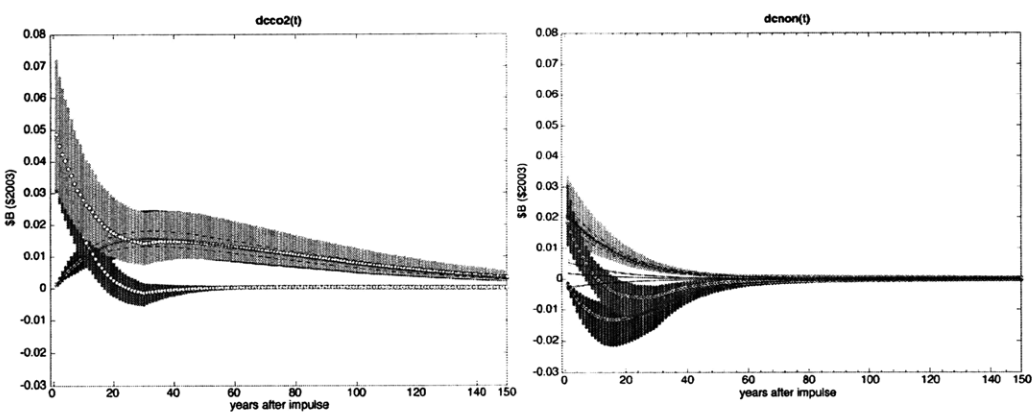
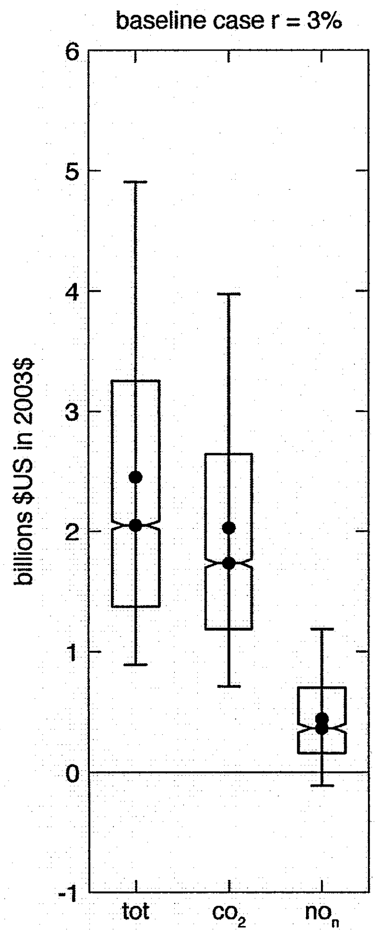


Figure F4.1

Figure F4.1b details the non-CO2 effects, showing damage streams for six perturbations: NO_x-related ozone production; NO_x-related reduction in CH4 lifetime; changes in cloud cover and properties, summing contrail and cirrus impacts; sulfate, a component of PMnv related to the release of fuel sulfur; non-volatile PM originating from incomplete combustion; and the increase in water concentrations due to H2O emissions.

Figure 4.3. Comparison of estimated 1992 CO_2 versus non-CO_2 climate damages

This figure summarizes climate damages for 1992 using the baseline scenario CRM with a 3% discount rate). The sum of all contributions to damage costs amounts to $2.0B with interquartile range $[1.3 3.2]B, CV = 0.92, and SE = 0.03.

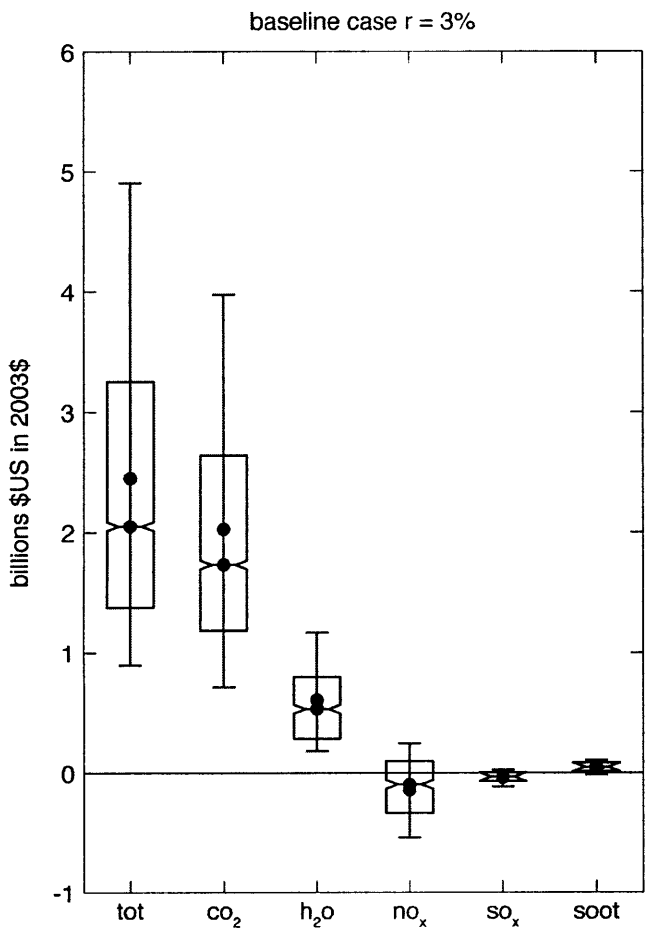


The detail breakdown of damages among emissions for 1992 is plotted in figure 4.4. Component damage costs are organized by emitted species rather than effect. We do this primarily because decomposing NO_x or cloud-contrail effects obscures the fact that they cannot be decoupled by changes to an aircraft system.

Figure 4.4. Breakdown of estimated 1992 climate damages by source emissions

This figure details climate damages for 1992 using the baseline scenario CRM with a 3% discount rate. The breakdown is as follows:

species	fractional contribution to annual sum climate damage cost	
CO_2	0.79	interquartile range (IQR) = [0.65 0.87]
H_2O	0.31	IQR = [0.22 0.39]
NO_x	-0.09	IQR = [-0.15 +0.06]
S	-0.04	IQR = [-0.04 -0.02]
PM_{nv}	0.02	IQR = [0.01 0.03]

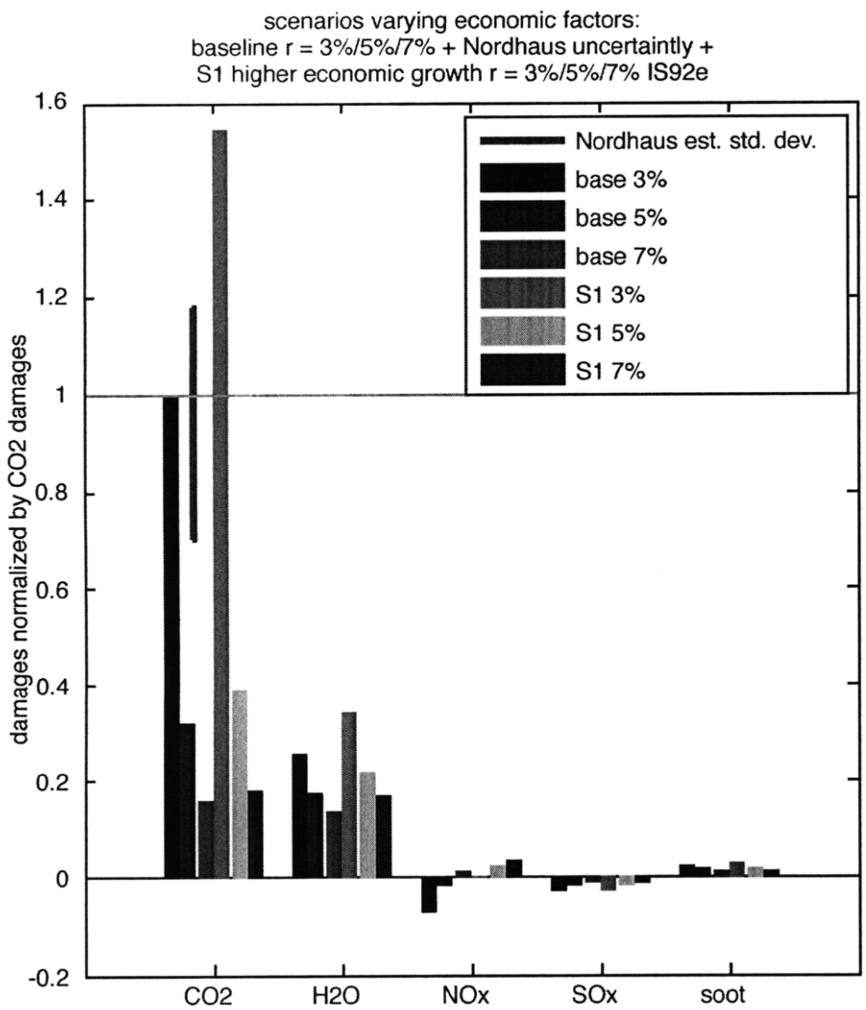


4.4.2. Effect of discount rate

A decreasing discount rate disproportionately increases magnitude of the cost streams and the extent to which future costs impact present value by changing the decay as $e^{-t/\tau}$. Short-lived perturbations are a significant factor in annual damage costs only at high discount rates where water vapor (i.e. clouds) becomes a primary effect equivalent to CO_2. Referring to figure 4.5, increasing the discount rate to r = 0.07 reduces total damages by a factor of ~6 and shifts the balance of costs in favor of non-CO_2 effects—

non-CO_2/CO_2 = 1.1, but it is also the case that the sum annual damages declines by a factor of 3. The range in total costs across discount rates is similar to the range of parametric uncertainty for the baseline estimate, the latter of which accounts for propagated uncertainties in emissions inventories, radiative forcings, and climate sensitivity. Primary combustion products are the most significant source of climate damages. Non-CO_2 emissions are a less than 10% contribution to marginal and total costs at a 3% discount rate. This increases to ~60% at a 7% discount rate, but over 90% of the non-CO_2 contribution is attributed to cloud effects. Cloud effects in MAIPA are attributed to water vapor emissions.

Figure 4.5. Impact of discount rate on CO_2 versus non-CO_2 damage contributions



Also observe that increases in economic growth are disproportionately more influential as the discount rate decreases; at $r = 3\%$, the ratio of damages for the IS92a versus IS92e scenarios is 1.6 compared to 1.1 at $r = 7\%$. In sum, estimates of annual climate damages strongly point to energy use and fuel choice as central to efforts to reduce impacts; the primary combustion products are the first-order instigators of climate change due to commercial aircraft operations.

4.4.3. Decision-making consequences of choice of climate impact metric

This is the context in which the marginal damage estimates of this study differ from existing reports intending to quantify aircraft climate damages (Pearce and Pearce 2000; Wit et al. 2003 other CE documents). These analyses apply directly the range of carbon valuations given in the IPCC SAR = $2003 [7 175]/tC (CPI inflated from $1990) to calculate total damage costs. Although the SAR range for carbon marginal damage cost was never intended to suggest any distributional form but rather a statement of the state of research. Using the average of this range results in a marginal cost ~3 times the MAIPA estimate without the context of climate physics or economic development.

These prior reports then account for short-lived effects by multiplying carbon damage costs by the ratio of the source emission instantaneous radiative forcing to that of CO_2. This is inappropriate; use of an instantaneous radiative forcing ratio as a comparative metric is analogous to basing decisions on sunk costs. Applying this technique determines that non-CO_2 impacts are overwhelmingly important. On the contrary, accounting for future effects emphasizes that CO_2 is a relatively more important influence on welfare.

Because first-order relationships have been established that connect radiative forcing (RF) with atmospheric concentrations of CO_2 and mean surface temperature change (through the climate sensitivity), RF has become a convenient comparative metric when considering historical anthropogenic influences on climate (Forster et al. 2000).[7] Ratios of instantaneous radiative forcing (RFI = RFNOx/RF CO_2), for example, provide a comparative picture of how various effects have contributed to the current

[7] In considering impacts on future welfare, changes in environmental variables at the surface are of primary importance. However, in order to assess the impact of radiative changes at the surface, large systems must be evaluated. Radiative models typically assess changes to the system consisting of the mixed layers of the ocean, land mass, and troposphere (which tends to be well-mixed) and report forcing values for the tropopause, adjusted for any changes in boundary conditions that result from alterations to stratospheric processes. This choice derives from the fact that models have determined a simple relationship between stratospherically-adjusted, tropopause RF and global mean equilibrium surface temperature

climate state. Published accounts of the estimated radiative impact associated with commercial aircraft operations, which are conventionally published as RF, communicate the cumulative role of emissions from the beginning of commercial activity about 50 years ago to the present (Brasseur et al. 1998; IPCC 1999; RCEP 2002; Schumann 2003).[8]

Integrating the marginal impact of a radiatively active gases over time, as in the global warming potential (GWP), provides a picture of comparative effects over a given time window (Lashof and Ahuja 1990). GWP is an incomplete comparative metric, especially where the effects of multiple gases are concerned (Reilly et al. 2003). One difficulty in this definition is the use of a uniform timescale of integration, typically defined for GWP at 100 years, for gases with variable lifetimes; if to address this issue the timescale were selected to be infinite, the questionable practice (and necessity) of physical discounting arises (Schmalensee 1993).

GWPs are also global averages, appropriate for well-mixed gases (i.e. long atmospheric lifetimes), but problematic in application to secondary or indirect emissions effects that may be regional in influence (i.e. short atmospheric lifetimes), delayed relative to the onset of long timescale direct effects, or generally unrelated to the radiative impact of the offending emission.

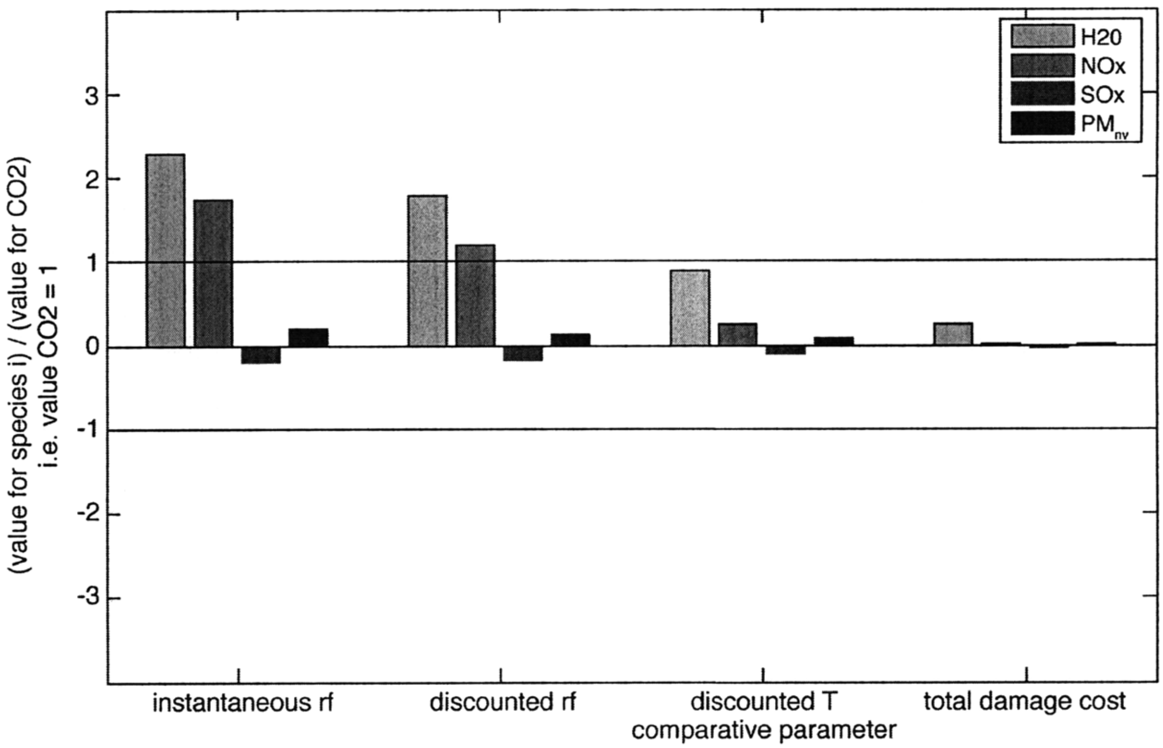
The GWP also lacks consideration of the opportunity costs of a change in RF associated with the emission or a change to atmospheric composition. In particular, such valuations will generally vary over time as economies change (Eckaus 1992). One essential fault is that RF is an indirect metric of the motivation for system change, not of the outcome of this change. This is a basic handicap in addressing questions of welfare impact (Hammitt et al. 1996).

Here, we are concerned with the impact of the next unit of emission; this is what we can influence with an increment of technological or operational change. It also requires a change in perspective from what has been cumulative, historical, and physical, to a marginal, future, and economic viewpoint. There have been several suggestions for alternative, welfare-based metrics of comparative impact (cf. Reilly and Richards 1993; Kandlikar 1996); this is also the goal of this study.

[8] These references extensively review the mechanisms of climate impact associated with commercial aircraft operations; this section assumes this background material.

Figure 4.6 makes these points graphically. On the far left of the plot is a comparison of effects using the instantaneous RF metric normalized to CO_2. The group on the right is the same comparison, but using MAIPA annual damage results as the comparative normalization.

Figure 4.6. Comparative metrics of climate impact



Whereas RF ratios suggest non-CO_2 effects are a factor of ~2.7 of that of CO_2 alone (non-CO_2/CO_2 ~ 2.7), under some assumptions total cost ratios show the opposite, with CO_2 effects a factor of 3-4 that of non-CO_2 effects (non-CO_2/CO_2 ~ 0.25-0.35).

It has been informally suggested that a discounted temperature ratio as a multiplier on equilibrium carbon costs can be employed, but there is a similar flaw in this compromise as shown in figure 4.6. The critical observation is that moving away from physical metrics towards metrics that account for risk preferences effect an important change in perspective as to choosing options that best reduce environmental risks of air transport. Taking this one step further, the ratio non-CO_2/CO_2 damages as a portion of the marginal damage cost of a unit fuel consumption is approximately 0.1 at the median. Marginal climate damages averaged over the entire analysis period 1991-2003 are summarized in Table 4.1.

Table 4.1. Estimated marginal climate damages per unit emissions and per unit fuel consumption

	climate impact vectors ($/metric ton)
species	median [interquartile range]
fuel	21 [14 32]
C	25 [16 37]
H	6.4 [3.9 11]
SO2	120,000 [60,000 200,000]
NO_x	(73) [(340 160]
PM_{nv}	3,500 [2,000 7,000]

4.5. Uncertainties in physical models

The calculations presented previously consider the relative effects of three parametric influences on the location of the baseline median value; the discount rate (baseline) and changes in r_g as represented by two alternative IS92 scenarios, lower growth (IS92a) and higher growth (IS92e). To reiterate, the discount rate and growth rate are separated here to convey the difference in preference variability versus variability in economic projections.

Differences in the construction of climate response models (CRM) and the physical processes they represent (structural uncertainties) also affect the magnitude of damage costs. With the results discussed in this section, the MAIPA analysis of climate damages finds that parametric, scenario, and structural uncertainties contribute similarly to uncertainty in cost estimates. Managing the climate risks of aviation emissions is as much dependent upon (a) normative decisions underlying the specification of intergenerational wealth distribution as on (b) scientific questions of carbon-cycle and climate processes as on (c) propagated parametric uncertainties.

4.5.1. Structural uncertainty in specification of climate models

Table 4.2 summarizes calculation specifications to assess structural uncertainties; at the close of this section, we consider the comparative roles of parametric, scenario, and structural uncertainties in the estimation of climate damages from air transport.

Table 4.2. Case specifications for evaluation of structural uncertainties

economic	climate					
	CRM1	CRM2	CRM3	CRM4	CRM5	CRM6
IS92a	S2 r=[0.03, 0.05, 0.07] 1992 only	S3 r=[0.03, 0.05, 0.07] 1992 only	S4 r=[0.03, 0.05, 0.07] 1992 only	S5 r=[0.03, 0.05, 0.07] 1992 only	baseline r=[0.03, 0.05, 0.07] 1991-2003	S6 r=[0.03, 0.05, 0.07] 1992 only

The nominal case uses the economic development assumptions of IS92a and combines the CO_2 impulse response of equation A10.4 (cf. appendix 10) and the impulse response function in equation A11.3 (cf. appendix 10) for surface temperature using a 3% discount rate. Table 4.3 provides a comparison of the CRMs identified in table 4.2.

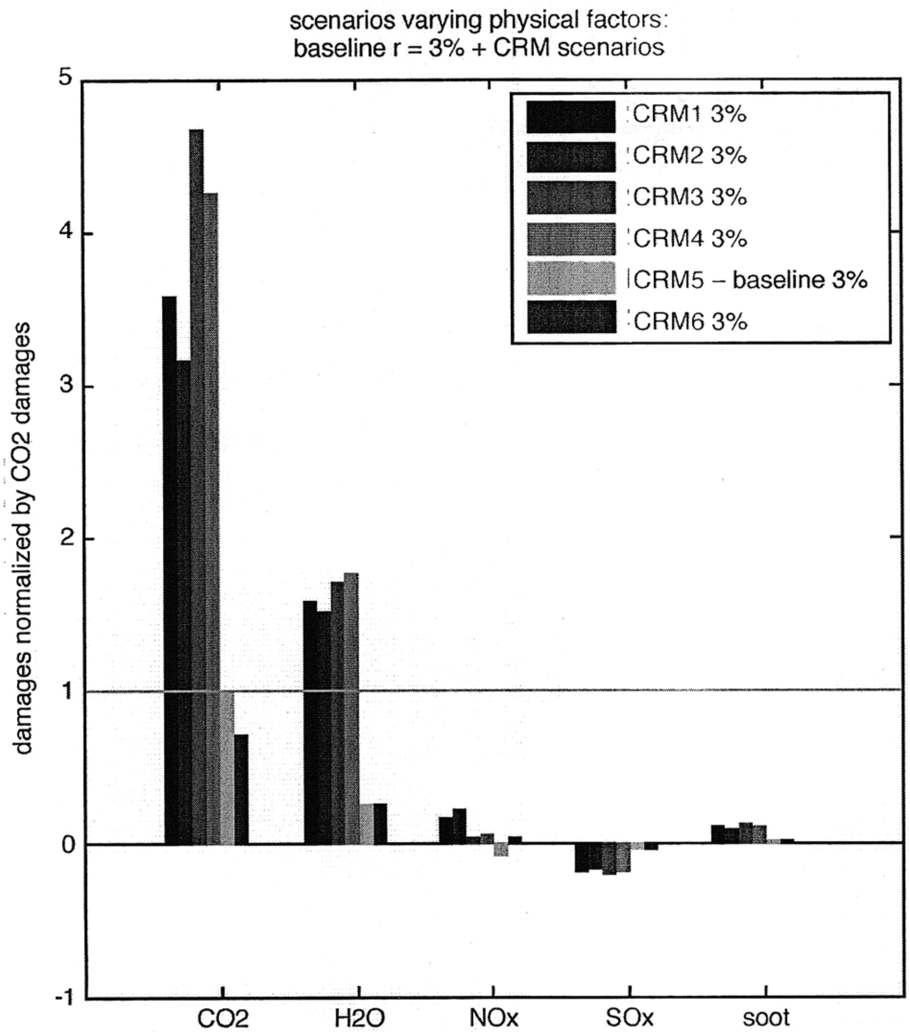
Table 4.3. Climate response models for evaluation of structural uncertainties

temperature response model	carbon cycle response model				
	C-MRH87a	C-MRH87b	C-MR93	C-Ho01	C-BernTAR
T-Ha93	CRM 1	CRM 2	CRM 3		
T-Ha97			CRM 4		
T-Ho01				CRM 5	CRM 6

designation	comments
C-MRH87a	HAMMOC inorganic ocean-circulation carbon cycle model for $1.25xCO_2$ step input
C-MRH87b	HAMMOC inorganic ocean-circulation carbon cycle model for $2xCO_2$ step input
C-MR93	HAMMOC with addition of oceanic biota and sediment sinks
C-Ho01	HAMMOC with addition of nonlinear uptake of carbon in surface waters, but no biota or sediment sinks
C-BernTAR	HILDA inorganic ocean-circulation carbon cycle model with 4-pool terrestrial biota model as specified for the IPCC TAR

The six cases outlined employ climate response models that represent two eras of development. (cf Appendices A10 and A11 for a more detailed account of these models). Figure 4.7 compares estimated annual damages for each of the cases described by tables 4.2 and 4.3. The baseline case is CRM5.

Figure 4.7. Structural uncertainties relating to climate response model formulation



In the first set are CRMs 1-4 which reflect model development leading up to the second assessment report. CRMs 1-3 are differentiated by the extent to which they capture the various processes which control the carbon cycle, represented by the amplitudes and characteristic timescales of the constituent modes. In comparison to CRM3, CRM4 reflects a decomposition of the temperature response into three different modes (as opposed to one) with the same climate sensitivity, the primary difference being in the additional identification of a relatively lower amplitude, but longer timescale perturbation.

As in the comparison of CO_2 and non-CO_2 effects previously, the relative importance of these modes is a function of the discount rate. The difference in CRM1 and CRM2 is in the input used to identify the impulse response. The key distinction is that the 2x CO_2 input of CRM2 results in a slower long-term relaxation of CO_2 but with a smaller amplitude than identified for the 1.25x CO_2 input of CRM1. Under discounting in the baseline case, these tendencies counteract, but essentially negate one another leading to approximately the same result (CRM1/CRM2 ~ 1.1).

The baseline case and scenario S6 reflect model development leading up to the IPCC Third Assessment Report. In the baseline case, the carbon-cycle representation does not account for key carbon sinks and accounts for nonlinear processes that reduce the inorganic solubility pump, resulting in a relatively higher X_{CO2} in a shorter timeframe than in CRM1-CRM4. The additional terrestrial and biospheric sinks in the Bern model counter this bias. However, the primary influence is the difference in surface temperature response which reflects a lower median climate sensitivity and while dominated by a long-term component in aggregate, is consequently a lower amplitude. Note that the Bern CCM – ECHAM case (CRM6) is the closest to the specification used by Nordhaus (2008).

Higher discount rates would further accentuate the amplitude differences; with additional carbon sinks accounted in scenario S3, the timescales of carbon uptake are shorter across all modes, but the amplitude of longest wavelength mode is higher, leading to relatively higher damages under discounting (CRM1/ CRM3 ~ 0.7). In absolute terms, the range damage costs due to these differences in carbon cycle representation $X_{CO_2}^{ref}$ are ~0.5 of the annual damage interquartile range for 3% discount rate. In comparison to scenario CRM3, CRM4 reflects a decomposition of the temperature response into three different modes (as opposed to one) with the same climate sensitivity, the primary difference being in the additional identification of a relatively lower amplitude, but longer timescale perturbation. Under discounting, the change in amplitude of the dominant mode again determines the change in response, here a decline relative to CRM1.

Discounted, the damages are weighted towards a relatively small (~4%) short-term component resulting in a factor ~4 smaller damage than in CRM1-4; in absolute, the difference between set 1 (CRM1-4) and set 2 (CRM5-baseline and CRM6) is ~2 times the baseline interquartile range at 3%. It is important to highlight that the difference between the comparisons among physical scenario and the account of propagated uncertainties is that the scenarios reflect changes in the structure of the response while the uncertainties in climate sensitivity control magnitude; the point is that with discounting, the former is at least as influential as the latter.

4.5.2. Uncertainty in specification of altitude NO_x effects

Competition among short-lived and long-lived perturbations leads to an ambiguous conclusion as to the effect of NOx emissions, with different estimates of regional versus global perturbations making it uncertain whether net damages are positive or negative. The net damages are the sum of two components —short-lived ozone production and methane destruction occurring over a longer timescale—each of which are O(10) larger than the net effect.

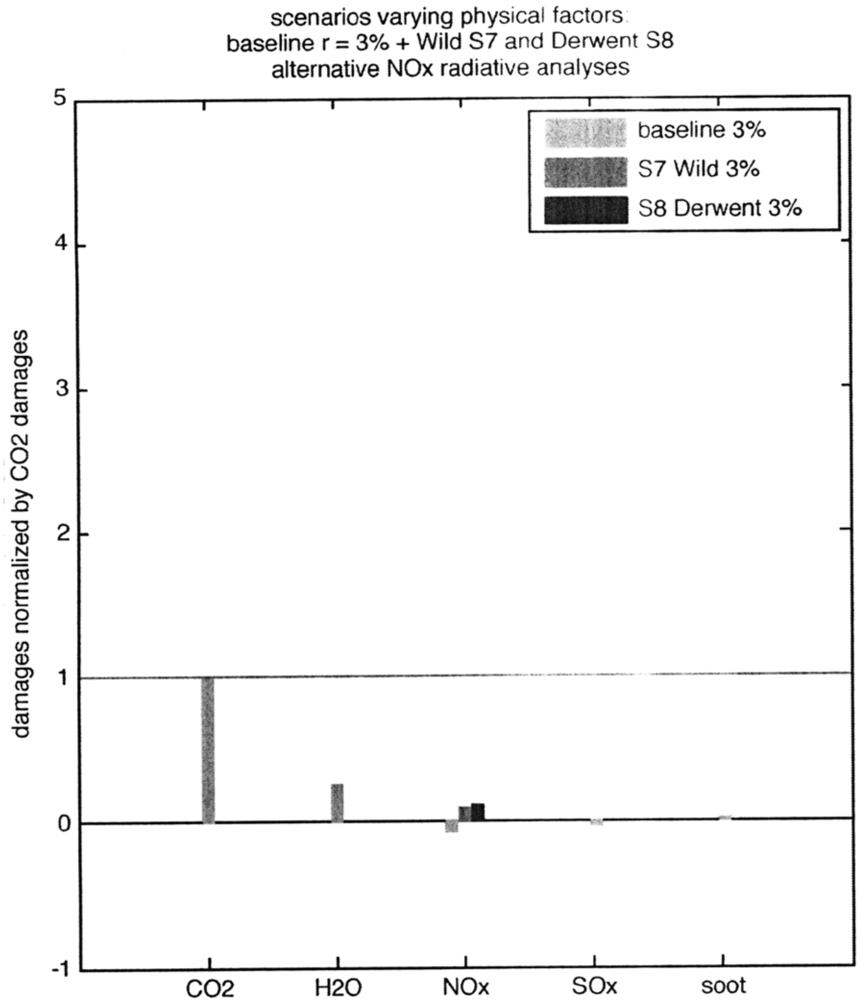
Table 4.4. Case specifications for evaluation of uncertainty in effects of air transport NOx emissions

economic	IS92a	IS92a Wild	IS92a Derw
CRM5	baseline r=[0.03, 0.05, 0.07] 1991-2003	S7 r=[0.03] 1991-2003	S8 r=[0.03] 1991-2003

For the baseline case, the analysis estimates a median net cooling effect from NOx emissions, but its magnitude is on the order of the computational resolution and thus our ability to differentiate from zero is tenuous. Relative to the median climate damage, ozone production is ~0.25•C while the methane effect is -0.34•C, leading to a net –0.09 times the median climate damage cost. Because of the difference in the perturbation lifetimes of these components, the balance of effects depends on the discount rate, the rate of economic growth, and more influentially on different specifications for the climate response. Any of these factors can determine whether NOx emissions lead to net cooling or warming.

Two alternative analyses of radiative forcing from aircraft NOx at altitude (Derwent et al. 2000, Wild et al. 2002) were also evaluated as shown in figure 4.8 compared to baseline results at $r = 3\%$. These analyses differ from Stevenson et al. (2004) primarily in the warming estimated to occur as a result of ozone production—in Wild et al., $RF_{O_3}^{yr} = 0.0079$ W/m2 and $RF_{CH_4}^{yr} = 0.0046$ W/m2 and in Derwent et al., $RF_{O_3}^{yr} = 0.0086$ W/m2 and $RF_{CH_4}^{yr} = 0.0046$ W/m2.

Figure 4.8. Alternative analyses of radiative forcing from aircraft NOx



The outcome is a net warming with a higher confidence of being distinct from zero than in the baseline case analysis (reject at p=0.1) but only for the period 2000-2003; relative to propagated uncertainties, this variability does not significantly resolve the ambiguity of effect observed for the baseline case. Uncertainties associated with economic parameters (discount rate and consumption growth) or the construction of climate models have an O(10)-O(100) larger effect on outcome in the baseline analysis. More fundamentally, NO_x damages are an absolute O(10)-O(100) less than primary pollutant (i.e. CO_2 and H_2O) damages regardless of the specifications for economic parameters. The same conclusion is evidenced for sulfur and PM_{nv} emissions.

5. Damages from reduced air quality

In the regulatory context, complex models of the chemistry and fluid dynamics of the lower troposphere are exercised to make detailed assessments of air quality to determine nonconformity and demonstrate attainment (cf. EPA 07). These models have a spatial resolution appropriate to the transport scales of urban plumes and point sources in keeping with the NAAQS. These models are designed to capture temporal scales ranging from minutes to days, focusing primarily on the analysis of episodic conditions (e.g. summertime high ozone events related to stagnated air masses, lasting on the order of a week). Regulatory assessments typically report yearly results.

Our interest, however, lies in using models to determine how efficient it is to alter the emissions characteristics of the aircraft fleet. In contrast to models of aircraft noise exposure, there is less assessment experience in direct evaluation of exposure to the major criteria pollutants affected by aircraft emissions, ozone and PM2.5. The atmospheric processing of aircraft emissions is typically not addressed in an assessment, relying instead on comparative inventory metrics.

The goal in developing an air quality impact model was to develop a methodology to estimate air quality impacts from US commercial air transport and to assess the factors that determine air quality damages. Specifically, the objectives were: (1) to develop an approach that distinguishes the value of reducing NO_x, SO_x, HC, PM_{nv}, and CO emissions that accounts for the formation of ozone and particulate matter; and, (2) to understand the influence of different model parameters and components on uncertainty in estimated damages and relate policy implications.

Contribution 5.1. A measurement-based estimation methodology to model changes in atmospheric pollutant concentrations

As opposed to a detailed chemistry and transport model, a parametric approach estimates changes in the atmospheric concentrations of the criteria pollutants (NO_2, CO, SO_2, O_3, and PM2.5). The central challenge in developing the air quality model is the representation of the nonlinear chemistries and microphysics that control ozone and secondary particulate matter production. For ozone and PM2.5, observational data are used to specify additional linear transformations salient to the chemistry and transport controlling formation processes. Changes to pollutant exposure patterns are established using scaling arguments that draw on measurement data to represent geographic patterns as locational variability.

Contribution 5.2. Identified that the major source of reducible uncertainty in emissions damages stems from the assumed extent of ozone and particulate matter production in the engine exhaust plume.

In the context of MAIPA, variance in VSL controls precision estimated damages, but choices as to how to construct the air quality analysis are equally if not more influential: specifically, homogeneous element of scale (timescale comparisons), heterogeneous element of population and meteorology, source accounting, and decisions about benefit transfers

The most significant uncertainties are tied to our understanding of how exhaust plumes evolve in the lower troposphere and interact with other emissions sources, specifically the extent to which photochemistry and particulate formation in the exhausted air mass are distinct from the physics and chemistry of the surrounding atmosphere.[1] Currently, large scale complex air quality models such as the EPA CMAQ are applied to estimate aircraft air quality impacts with the assumption that plume processes are not significant.

Comparisons of MAIPA results with recent air quality assessments, including results from an study of US commercial aircraft effects mandated by the Energy Policy Act of 2007, suggest the an O(2) impact on estimated damages from secondary particulates. The impact is more significant on ozone; the estimated impacts of NO_x emissions are divided 70:30 among ozone and PM2.5 impacts. If photochemistry is not influential as suggested by CMAQ calculations, the average median annual damage for 1991-2003 would fall from $4.5B to approximately $2B.

Result 5.1. Air quality impacts of US commercial aircraft emissions between 1991-2003

Marginal damage estimates suggest that SO_x, HC, PM_{nv} emissions play a role in air quality as important as NO_x, and instead of ozone, impacts on ambient PM2.5 emerge as the predominant air quality concern. The average median annual damage for 1991-2003 is estimated to be $4.5B (CV= 58% and rSE = 2.9%). Emissions of SO_x, NO_x, and VOC constitute 99% of the annual damage costs with the remainder due to

[1] The evolution occurs on the scale of kilometers and is uniquely distributed; a vertically-oriented, linear, buoyant air mass mixing into the atmospheric mesoscale. The smaller photochemical and microphysical time scales are relative to mixing and transport, the higher the estimated damages.

CO and PM_{nv}. Of PM2.5, 55% of annual damages is attributable to SO_x, 30% to NO_x, and 15% to VOC. These results suggest that considering PM2.5 as an equivalent air quality source control priority to ozone, specifically evaluating options for SOx, HC, and PM_{nv} emissions controls along with NO_x, would be a positive step toward improved decision-making.

Result 5.2. Comparative benefits from reductions in fuel and emissions

Emissions impacts of US commercial aircraft are dictated by the progress in controlling emissions from other sources.[2] An important distinction is that where a change in air quality can be affected immediately through a change in emissions, climate change lags emissions. The consequence is that it is almost certain that the marginal climate damages will increase year-to-year for the foreseeable future (at least over the lifetime of an aircraft generation in the commercial fleet). However, efforts to improve air quality can relatively quickly affect marginal air quality damages and change the conclusions of an assessment.

Without the ability to control the growth of marginal emissions damage costs (since they are chiefly dependent on the larger pool of sources that contribute to background emissions), it is important to make sure that the most effective mitigation approaches are taken. The following results provide some comparative guidance based on the air quality and climate analyses in this thesis.

- On average, reducing a kilogram of emitted sulfur gives approximately the same reduction in health damages as reducing ~2 kilograms of NOx or VOC, and ~5 kg of PM_{nv}.
- Reducing EI(S) provides an expected net reduction in damages with greater than 95% confidence.
- Reducing EI(NO_x) above the mixing height has an ambiguous benefit, statistically indistinguishable from zero. The magnitude of this uncertainty is O(10) smaller than marginal damages from air quality impacts.
- Marginal reductions in fuel consumption below the mixing height (e.g. the landing-takeoff cycle) have an O(10) larger benefit than in the free troposphere (e.g. cruise)—see figures 1.3(a), 1.3(b) and 1.3(c).
- Finally, reducing EI(PM_{nv}) consistently over the entire flight profile increases per-unit benefits by a factor of 2.

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[2] In contrast, noise marginal damages (discussed in the next chapter 6) decline as total noise energy increases and are determined solely by aircraft noise.

5.1. Damages as a function of disease and mortality

Health-based ambient concentration limits established by the Clean Air Act (CAA) of 1970—the National Ambient Air Quality Standards (NAAQS)—organize national air quality policies. Attaining these standards is the province of state planning, with EPA oversight and additional regulatory measures to control certain source categories, with an emphasis on ozone and particulate matter. Aircraft are one of the sources over which specific regulations have been instituted to control emissions. The CAA provides the legal framework for establishing and enforcing national emissions standards for aircraft engines (42 USC 1857), prohibiting supersession by state or local regulations (42 USC 1859), and giving specific guidance to base regulatory action on the state of technology with regards to safety and developmental capability.

5.1.1. Metrics relevant to national air quality policies

Initial regulations promulgated in 1973 (38 FR 136) to limit fuel venting, smoke, HC, CO, and NO_x emissions were founded on an EPA determination that airports would be major contributors to emissions inventories and that attaining the NAAQS (at that time addressing photochemical oxidants and smoke) would require controls on aircraft engines. Controls are based on a landing-takeoff (LTO) cycle that extends to an altitude of ~915 m (3000 ft). Emissions above 915 m, where an aircraft spends most of its time in flight, are not controlled (40 CFR 87).[3] These regulations extended to foreign civil aircraft, provided no other obligations with foreign states were affected.

Inventories are the primary assessment metric used to evaluate environmental impacts. CAA conformity requirements require federal actions, like the construction of airport infrastructure, to be consistent with state implementation plans for the control of air quality. Significant contributions to regional air quality, those that require general conformity determinations, are defined against area emissions inventories, with action required for projects adding >10% regardless of whether the sum total contribution is below *de minimus* levels (FAA and EPA 2002). Conformity determinations consider NO_2, CO, SO_2, VOC, and

[3] In compliance with the process set up by the CAA, the FAA promulgated in 1974 Special Federal Aviation Regulation (SFAR) 27 implementing certification requirements for aircraft engines (38 FR 211). Following reanalyses of local air quality problems around airports prompted by requirements in the 1977 CAA amendments, questions of technological feasibility, major new economic studies, and development delays for new combustor technology, emissions regulations saw major revisions in 1978 and several postponements of compliance dates through the late 1970s and early 1980s. In 1990, certification requirements were codified as 14 CFR 34.

primary PMnv as inventory contributions.[4] In the case of air transport operations, changes to area inventories as a result of infrastructure investments rarely result in a conformity determination.[5] The air quality analysis developed in this section shows how these metrics can be a poor indicator of the relative contributions of aircraft emissions to air quality change.

5.1.2. Estimation and valuation of changes in disease and mortality risks

In contrast to climate and noise impacts, where damages are estimated as a function of changes in environmental metrics, risks are explicitly estimated to determine damages due to emissions impacts on air quality. The approach is to estimate changes in mortality and disease incidence and then annual damages as a function of these risks. With the linear assumptions of air-quality analysis developed subsequently, statistical restrictions force us to consider the marginal air quality damages of aircraft emissions equivalent to average damage costs.

Equation 5.1 outlines the computation. In equation 5.1, n_m is the change in the incidence of a health effect m. The damage from health effect m due to emissions of species i is the product of I and the marginal damage cost c of one incident. The sum of these damages over all health effects M gives the annual air quality damage cost C. Marginal costs are then estimated by dividing annual air quality damage cost by the emissions inventory Q for species i.

$$\begin{aligned} \hat{C} &= \sum_{i=1}^{I} \hat{C}_i \\ \hat{C}_i &= \sum_{m=1}^{M} \Delta\hat{n}_m \cdot \hat{c}_m \\ \hat{c}_i &= \hat{C}_i / \hat{Q}_i \end{aligned} \qquad (5.1)$$

[4] The first formal air quality analysis of aircraft as emissions sources was conducted in 1959 by the Los Angeles County Air Pollution Control District. UBA (2004) reports the subsequent history of airport air quality model development in the United States and Europe (cf. Platt et al., 1971 , LAAPCD 1971, EPA, 1972, Rote et al, 1973 , Norco et al. (1973), Whitten and Hogo (1976) Kitagawa (1977), Duewer and Walton (1978) Yamartino et al. 1980b, Stern and Scherer (1982), and Timm and Lühring (1988). The EPA currently evaluates air quality using the Community Multiscale Air Quality model (CMAQ) (ref 40CFR pt51 appW, CMAQ v4.6 operational guidance doc).

[5] For these regulatory purposes, the FAA Emissions Dispersion Modeling System (EDMS) is the currently sanctioned method estimating airport emissions inventories (16FAR18068) (Segal and Hamilton, 1988; Segal, 1991; Moss and Segal, 1994 + add a recent reference, Hall et al., 2003).

Impacts considered in the air quality model include premature and sudden mortality, chronic respiratory illness (e.g. chronic bronchitis), hospital admissions and emergency room visits for respiratory (e.g. asthma, pneumonia) and cardiovascular diseases (e.g. chronic obstructive pulmonary disease, congestive heart failure, dysrhythmia, ischemic heart disease), and minor symptomatic illness as well as reduced activity that may be associated with illness. From a scientific standpoint, mortality studies are overwhelmingly important in the evaluation of health effects.

Estimating the occurrence of morbidity and mortality associated with changes in air quality entails the selection of epidemiological studies that statistically evaluate the relationship between pollutant exposure and health effects in human populations (controlling for the potential effects of and synergies among other pollutants, spatial and temporal exposure patterns, and confounding factors that may influence risk).[6]

Changes in health effect risks are estimated from calculated perturbations in the ambient concentrations of the criteria pollutants using one of several possible regression models, or concentration-response (C-R) functions. Equations 5.2 show the linear, log-linear, or logistic formulations typical of the regressions selected for use in the air quality analysis. In equations 5.2, the change in ambient concentration is denoted by ΔX_i; Δr_m is the change in incidence for health effect m and is a function of ΔX_i given the regression type. The number of expected cases of health effect m, n_m as in equation 5.1 above, is the product of Δr_m and the population vulnerable to the air quality change $n_{pop}{}^m$. Also in equations 5.2, beta is a risk rate, or the number of cases expected per change in ambient concentration. For the log-linear and logistic models, r_{m0} is a baseline incidence specific to health effect m.

$$\hat{n}_m = n_{pop}^m \cdot \Delta\hat{r}_m$$

(5.2)

$$\text{linear} \qquad \Delta\hat{r}_m = \beta \cdot \Delta\hat{X}_i$$

$$\text{log-linear} \qquad \Delta\hat{r}_m = r_0^m e^{-\beta \cdot \Delta\hat{X}_i} - 1$$

$$\text{logistic} \qquad \Delta\hat{r}_m = \frac{r_0^m}{\left(1 - r_0^m\right) e^{-\beta \cdot \Delta\hat{X}_i} + r_0^m} - r_0^m$$

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[6] Toxicologic assessments of disease burdens are impractical in application to large populations. A toxicological approach requires detailed information about pollutant composition, mechanism of biological effect, and specific exposure patterns. Toxicological studies do provide important information for epidemiological investigations such as which health outcomes to investigate, potential confounding factors, and populations at increased risk. factors to control.

Congruent with current regulatory practice and the recommendations the National Academy of Engineering review and critique of health effects analysis and its application in EPA decision-making (NAE 2002, cf. GAO 2006).[7] Recent EPA impact assessments are the basis for selection of the epidemiological studies used in this analysis. Concentration-response functions used in the Section 182 (§182) benefits analysis of the Clean Air Act and Amendments (CAA and CAAA, and EPA 1997, 1999) are the basis for evaluating morbidity endpoints.

The §182 study employs a clear set of selection criteria that mirror NAE recommendations, and the scope of the §182 study is consistent with the geographic and temporal boundaries of MAIPA (cf. chapter 2). Adhering to these criteria, modifications are made to update the assessment of mortality using more recent epidemiological studies of premature mortality associated with PM2.5 that are consistent with the EPA Environmental Benefits Mapping and Analysis Program (BenMAP). The air quality analysis also examines the impact of sudden mortality associated with O_3 as a source of structural uncertainty; these C-R functions are also consistent with BenMAP.

Parametric uncertainties, pooling, and data sources. Statistical errors in the C-R relationships are propagated through MAIPA to derive uncertainties in incidence of health and welfare impacts. No effect thresholds are assumed and mortality lags are set to zero.[8] A fixed effects model for weightings (1/var) pools studies with the same endpoint, thus assuming reported risk rates estimate the same effect value. Variance-weighting reduces the overall uncertainties in the incidence of health effects, by emphasizing, where averaging is required, studies where parametric uncertainties are smaller. C-R functions are pooled by individual population strata according to the categorization used in the 2000 U.S. Census, then pooled for species-specific effects, and finally aggregated across species to calculate endpoint-specific incidences. The stratification of population data from the Census is sometimes different from the subject population in the original study, particularly for those focusing on children; equivalencies are specified in tables A14.2-A14.6 (cf. appendix 14). Similarly, available air quality indicators are applied where a study measure is not available.

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[7] Other important sources that broadly treat air quality issues include the criteria documents and related staff papers, which form the technical basis for NAAQS development.

[8] This results in an overestimate of the cost since all incidences of death will occur in the year associated with the reference pollution level.

See Appendix 14 for additional discussion:

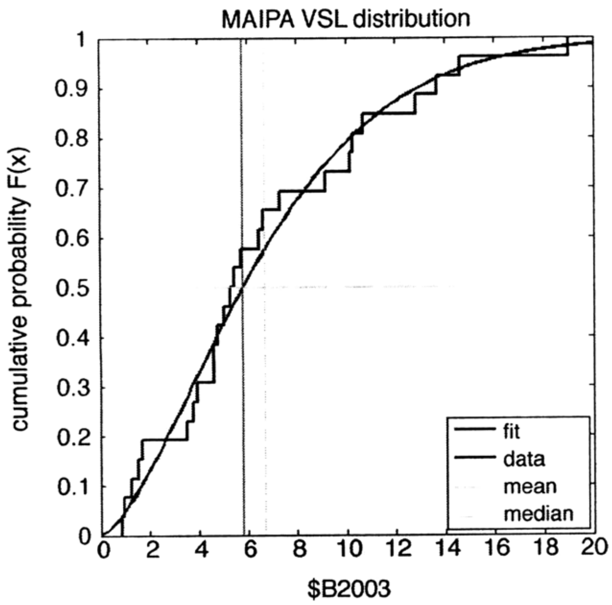
Table A14.1 in Appendix 14 (Concentration-response functions) reproduces the §182 criteria for selecting studies to specify concentration-response functions. In sum, forty-two studies published between 1980 and 2003 are employed to address the health effects of NO_2, CO, SO_2, O_3, and PM2.5. Tables A14.2-A14.6 in Appendix 14 summarize their functional forms, statistical parameters, population applicability, pooling categorization, and their sources.

5.1.3. Estimates of willingness-to-pay for reduction of health risks

The estimates of air quality damages in this thesis account for the microeconomic impacts of changes in the incidence of the mortality and morbidity endpoints described in the previous section. Marginal willingness-to-pay (MWTP)—given as the marginal damage cost c in equation 5.1—is specified differently for mortality and chronic respiratory disease as opposed to other morbidity impacts. In the case of mortality, we use estimates of the value of small changes in mortality risk, commonly known as the value of a statistical life or VSL. Such valuations are not intended to measure the worth of a life per se, but rather societal preferences for the economic compensation required to offset the increased risk; it is a marginal estimate.

A Weibull probability function fit to the mean estimates of VSL from twenty-six studies is used to characterize the variability in VSL estimates derived from two methods of estimating WTP: the hedonic price and contingent valuation methods. Figure 5.1 plots the cumulative distribution and the Weibull fit.

Figure 5.1. Cumulative distribution function for the value of a statistical life



The value of a statistical life is controversial, but it is an assessment of societal preferences that responds to the objective of this thesis to weigh options to reduce environmental impacts. It is not the intent of this study to address the more fundamental equity, and to some extent philosophical, issues that arise in the application of VSL and similar estimates of WTP for reductions in mortality risk. However, these are important questions to consider within the broader decision-making process. While it is common practice to present these risks as a number of deaths, we can as easily present these valuations on a per unit risk basis; both statistics are given in the results of this chapter.

For increased morbidity other than chronic respiratory disease, we rely on cost-of-illness (COI) estimates —a measure of out-of-pocket expenditures—instead of one of the more appropriate economic accounts derived from WTP approaches. COI aggregates observed expenditures for medical treatment and loss of wages. Since these expenditures occur after the impact of air pollution has been realized, they do not express preferences. As such, they are not welfare measures and although we can say they are underestimating surrogates for WTP—they do not account for suffering and the like—their relationship to the actual valuation is not known in the instance of their use. One estimate places medical care expenditures for ozone-related morbidity at 50% of the estimated welfare impact Gerking (1991). Despite these potentially large underestimates, they are of relatively minor consequence to damage cost estimates

—damages associated with changes in mortality risk are O(100) larger than morbidity impacts. Distributions for morbidity valuations—including chronic and symptomatic morbidity associated with respiratory and cardiac conditions, as well as the welfare effects of restricted activity that can result—are taken directly from EPA section 182 benefit analysis as summarized in table A17.8 (EPA 1999).

Benefit transfer problems exist in virtually all applications of C-R relationships. In MAIPA, as with similar studies, C-R relationships are assumed to apply for populations different from the subject population studied. Since the underlying epidemiological studies often focus on a particular region (in some cases over time, e.g. cross-sectional data), evidence for an upward or downward bias on disease incidence due to benefit transfers is not clear.

5.2. Response of ambient pollutant concentrations to aircraft emissions

Consistent with the geographic resolution of the available air quality data, MAIPA uses a box estimate (i.e. a well-mixed volume of the atmosphere bounded by the county boundaries and the mixing height) of the perturbations relative to the ambient baseline concentration measured for a county. Physically, this implies that the timescale of secondary pollutant formation (O_3 and PM2.5) is much less than the mixing and transport timescales referencing the primary dimension of the typical county.

As described in the previous section, the estimator ΔX_i determines changes in disease incidence and mortality risk. To estimate the extent to which emissions and their atmospheric derivatives affect populations, we construct a distribution for perturbations to baseline ambient concentrations that result from airport emissions inventories. Equation 5.3 shows this probability distribution for ΔX_i as the union of distributions calculated for each airport location l. Chapter 3 discussed the application of aircraft operations for 96 airports used with the MAGENTA model in MAIPA; these 96 airports are the sample L in equation 5.3, each of which is associated with a set of distributions of demographic and environmental data to estimate ΔX_i with the methodology developed in this section.[9]

(5.3) $$P(\Delta X_i) = \bigcup_{l=1}^{L=96} P(\Delta X_i^l)$$

[9] More detailed considerations appropriate to higher resolution modeling efforts can be found in Seinfeld (1986, 1989, 2004) and EPA criteria documents (ref criteria documents).

5.2.1. Change in ambient concentrations due to NO_2, SO_2, HC, and CO emissions

Changes in ambient concentrations of criteria pollutants are estimated with the assumption that the sum of chemical and microphysical processes linearly scale with small changes in initial conditions. With this assumption, changes in the ambient concentrations of NO_2, SO_2, HC, and CO are estimated proportionally to the ratio of aircraft emissions inventories (NO_x as NO_2, CO, HC as VOC, and SO_x as SO_2) to area emissions inventories, e.g. equation 5.4. This is a standard approach in EPA air quality analyses that assumes the chemical lifetimes of NO_2, SO_2, HC, and CO are large relative to the time scales of transport and diffusion over the reference geographic area.

(5.4) $$\Delta\hat{X}_i = X_i^{amb}\left(\frac{\hat{Q}_i}{Q_i^l}\right) \quad \text{for } i = \left(NO_x \wedge SO_2 \wedge VOC \wedge CO\right)$$

Monitor data and emissions inventories are resolved to county geographic areas; congruently, airports are treated as additional county sources. EPA National Emissions Inventory (NEI, ref NEI and EPA 1997, 1999b) specify county emissions Qil, accounting for all emissions sources. Baseline ambient concentrations Xiref are taken from the EPA AirData information system. These data summarize monitor measurements input to the EPA Air Quality System. The first year for which data for all criteria pollutants are available is 1997; PM2.5 was not reported prior and it is assumed that baseline ambient concentrations during 1991-1996 were the same as in 1997. Also, the EPA AirData database does not record VOC measurements as it does for the criteria pollutants. In lieu, ambient VOC concentrations are specified as P(unif: 0.32, 3.4) ppm based on the data reported in EPA (1986).

The central challenge in developing the air quality model is the representation of the nonlinear chemistries and microphysics that control ozone and secondary particulate matter production. For ozone and PM2.5, observational data are used to specify additional linear transformations salient to the chemistry and transport controlling formation processes. The next two sections describe the approach taken in MAIPA.

5.2.2. Parameterization of secondary ozone formation

Ozone kinetics depend directly on absolute and relative precursor concentrations as well as meteorology, thus varying with geography. To parameterize these dynamics, the analysis draws from the substantial literature that considers measurement indicators of ozone sensitivity. The ozone model references *in situ* measurements to specify an ozone production efficiency (OPE) as defined in equation 5.5 (Liu et al 1987, Zaveri et al. 2003). When OPE is mentioned in this section, note that it is an integrated measure encompassing the history of photochemistry in the aircraft plume through mixing at the atmospheric mesoscale.

$$\Delta X_{O3} \simeq \mathrm{OPE} \cdot \Delta X_{NOx} \text{ where } \mathrm{OPE} = \frac{dX_{O3}}{dX_{NOx}} \tag{5.5}$$

As with the underlying ozone chemistry described previously, the OPE is nonlinear, inversely proportional to NO_x and dependent on the ratio of reactive organic gases to NO_x (ROG/NO_x); in the case where ozone is limited by the availability of NO_x, the relationship is essentially linear. Regulatory analyses of national air quality programs completed over the last decade indicate values for OPE between 1-2 are estimated by air quality analyses using EPA models. Analyses using measured air quality data appear to support this range of OPE estimates. Kasibhatla et al. (1998) estimate the regional accumulation and removal of ozone using monitor data that track ozone chemistry at the resolution of EPA air quality models; they find OPE = 1-3 over the eastern US, consistent with the OPE calculated in regulatory applications of air quality models.

However, these conclusions may not be apropos of ozone formation sourced to aircraft emissions. Photochemistry may be substantial in the exhaust plume before the emissions are mixed to the atmospheric mesoscale resolution of regulatory air quality models. Observational evidence suggests that in pollutant emissions flows well-defined against the atmospheric background (such as an aircraft engine exhaust plume), ozone production may be substantially more vigorous than suggested by regional analyses using air quality models.[10] The measurement literature, accounting only studies that include evaluation of loss rates, reports OPE in the range 1-7.[11]

There is no quantitative understanding of ozone production in aircraft exhaust plumes and the scope of the present analysis cannot include the research required to understand photochemistry in this complex

[10] For plumes, initial compositional and fluid dynamical conditions affect the course of the photochemistry.

[11] This range excludes OPE estimates for the Houston area (cf. Reyerson et al., 2002; Berkowitz et al. 2004) that exceed this range; these values are due to uniquely high VOC emissions (petrochemical industries) relative to other urban locations.

reacting flow. An examination of two literatures published over the last two decades—(1) studies of ozone precursor chemistry in aircraft exhaust plumes, and (2) studies of ozone formation in plumes generated by non-aircraft emissions sources—suggests that ozone production likely occurs within the plume before precursors are dispersed to the atmospheric mesoscale.

Based on these evaluations, the existing data is applied to specify a distribution for OPE in equation 5.5. Uncertainty in ozone production as it relates to aircraft plumes—a function of both geographic variability and parametric uncertainty—is incorporated using a triangular probability function Ptrig[1:2:7] that encompasses reported summertime measurements in urban and rural areas with the likely value biased to peak at OPE = 2. The selected likely value is is intended to reflect current regulatory assessments.

As a preliminary check on the validity of this distribution, Aerodyne Research agreed to conduct computations of the passive photochemistry in an aircraft plume flow to get a sense of the possible values for OPE. The results suggest instantaneous OPE ~ 4-6. This is only suggestive of a possible downward bias in the OPE probability function as specified for MAIPA. However, it points to a significant structural uncertainty that is not currently addressed in air quality modeling of aircraft impacts on ozone air quality.

See Appendix for additional discussion:

- Appendix 13 (Tropospheric ozone production) reviews the substantial literature that considers measurement indicators of ozone sensitivity and the role of NO_x and VOC precursors in ozone formation.
- Appendix 13 also reviews the available measurement studies of ozone production in non-aircraft plumes; these studies cover a range of environmental conditions and the spectrum of ozone sensitivity to NO_x and VOC precursors.

5.2.3. Parameterization of secondary particulate matter formation

The activity of precursor chemistry in the plume also has an impact on secondary particulate matter formation. A reduced-order, bottom-up assessment of change in PM2.5 due to secondary PM is difficult. Unlike ozone chemistry, the primary factor governing PM2.5 contributions is the ratio of the chemical time scale of precursor production to the physical timescale of deposition. If this ratio is >>1, then oxidation is faster than removal and an addition close to a proportional contribution to area inventories is realized. However, this assumption is reasonable only for primary PM emissions; for SOx emissions, the ratio is O(10), for NO_x and VOC it is O(1). Interactions among precursors further complicate the evaluation of secondary PM, primary of which is the competition between NO_x and SO_x oxidants for atmospheric ammonia; a brief summary of the relevant chemistry and microphysics is given in footnote 12.

As we did for ozone, we look to measurements to estimate an parameter to represent the local atmospheric chemistry, here the conversion of emissions to secondary PM. Detailed measurements of PM2.5 concentrations at 13 sites across the United States are used to define the composition of ambient PM2.5.[13] The total change in PM2.5 is the sum of sulfate, nitrate, organic PM_v, and PM_{nv} resulting from aircraft emissions as well as any mass that may be added by reactions with ambient ammonia to form ammonium sulfate and ammonium nitrate.[14] Equation 5.6 describes the first part of a procedure to estimate the change in ambient PM2.5, estimating sulfate, nitrate, and organic PM_v; the second step is to account for the additional PM2.5 mass resulting from sulfate and nitrate reactions with ambient ammonia as given by equation 5.7.

---

[12] The ultimate fate of NO_2 in the atmosphere is irreversible conversion to nitric acid (HNO_3) via OH and, during the night, via hydrolysis of dinitrogen pentaoxide (N_2O_5). The formation of HNO_3 continues over several hours at the surface. Nitric acid has a high vapor pressure and does not readily nucleate or condense. The primary particulate sourced to NO_x emissions is ammonium nitrate (NH_4NO_3), a salt formed in the reaction of HNO_3 and ammonia (NH_3) in the atmosphere.

SO_x-related contributions to particulate matter can be in the form of acids or salts. Sulfate precursors have low saturation vapor pressures that result in near-field nucleation and condensation in the presence of water vapor and soot particulates. Sulfuric acid also competes effectively with HNO_3 for ammonia so that if sulfur emissions are present, NH_3 preferentially binds with H_2SO_4 to form ammonium sulfate (NH_4HSO_4) or ammonium bisulfate ($(NH_4)_2HSO_4$) to the point of stealing ammonium from NH_4NO_3.

The formation of organic aerosol from VOC emissions is comparatively less well understood than for inorganics. Condensation of high carbon number organic products of ozone photochemistry, uptake of polycyclic aromatic hydrocarbons (PAHs) on soot or other solid particles, and dissolution of soluble hydrocarbons are the three primary pathways.

[13] This data is specific for ~10% of the counties where airports exist.

[14] This procedure is similar to the Speciated Model Attainment Test (SMAT) used by the EPA to determine concentration changes for PM2.5 that result from area emissions. Whereas the SMAT uses air quality models to determine a change in measured concentrations, MAIPA assumes that the addition of aircraft emissions is small enough to perturb the existing atmospheric chemistry linearly.

$$
(5.6) \quad
\begin{aligned}
&\Delta\hat{X}^{\text{no-NH3}}_{\text{PM2.5}} = \Delta\hat{X}_{\text{prim-PM}} + \sum_{j=1}^{3}\Delta\hat{X}_j \\
&\textit{specify}: \left| \begin{array}{l} j = \{\text{sulfate, nitrate, organic PM}_v\} \\ i = j: \; i = \{\text{SO}_x, \text{NO}_x, \text{HC}\} \end{array} \right. \\
&\textit{define}: \left| \begin{array}{l} \Delta\hat{X}_j = \dfrac{\alpha_j \cdot \hat{X}^l_j}{Q^l_i} \cdot \hat{Q}_i \\ \textit{where}: \quad \alpha_j = \left. \dfrac{\hat{X}_j}{\hat{X}_{\text{PM2.5}}} \right|_k \\ k = \text{references apportioned PM2.5 measurements} \\ \text{from 1 of 13 US sites randomly selected} \end{array} \right.
\end{aligned}
$$

In equation 5.6, alpha is the fraction of the local PM2.5 constituted by species i, either sulfate, nitrate, organic PM_v. Similar to OPE, the ratio $\left(\alpha_j \cdot X^l_j\right)/Q^l_i$ is a linear representation of the local chemistry and microphysics of secondary PM formation, a particulate production efficiency or PPE given in units of ppm / tonne. As with the treatment of ozone production described in the previous section, in employing these data we assume that the addition of aircraft emissions would not change the manner in which area source emissions result in PM2.5 mass nor its speciation.

PM2.5 compositional data is specified with a random selection of one of the 13 sites measurement sites at each iteration of the Monte Carlo simulation. This specifies the ratio alpha in equation 5.6.[15] To estimate the increase in ambient PM2.5, the ratio of apportioned mass to source inventory is considered constant, and it is assumed that SO_x emissions affect the sulfate component of PM2.5, NO_x emissions affect nitrate, and HC emissions affect organic carbon, and PM_{nv} emissions perturb elemental carbon. Thus, for each of NO_x, SOx, VOC, and soot, the percentage increase in area inventory due to aircraft emissions is as a multiplier on the apportioned mass of sulfate, nitrate, and organic PMv.

Recent work to develop a response surface model (RSM) for the effects of aircraft operations on air quality in the US supports the linear assumptions underlying equation 5.6. Drawing from a regulatory impact assessment in support of revisions to the PM2.5 NAAQS, Masek (2008) develops an RSM using air quality computations conducted with the Community Model of Air Quality (CMAQ). CMAQ is a three-dimensional Eulerian model of atmospheric chemistry and transport and is the EPA's preferred

[15] This overestimates the variability in fractional contributions, but to an unknown extent.

regulatory analysis tool for attainment demonstrations and regulatory analyses. Masek (2008) finds that a linear surface fit correlates to PM2.5 perturbations estimated using CMAQ with a coefficient of determination equal to 0.99.

Sulfate and nitrate and compete for ambient ammonia in forming ammonium sulfate and ammonium nitrate. The formation of ammonium sulfate is thermodynamically favored over ammonium nitrate when sulfate and nitrate are both present (footnote 12 provides more detail on the relevant chemistry). The air quality model uses this characteristic of the formation chemistry to partition ammonia between sulfate and nitrate. Equation 5.7 shows the calculation.

$$\Delta\hat{X}_{PM} = \Delta\hat{X}_{\text{PM2.5}}^{\text{no-NH3}} + \sum_{j=1}^{2} \Delta\hat{X}_j$$

(5.7) *specify*:

$$j = \{\text{ammonium sulfate, ammonium nitrate}\}$$
$$i = j:\ i = \{\text{sulfate, nitrate}\}$$

define:

$$\Delta\hat{X}_j = \kappa_i \cdot \Delta\hat{X}_i$$

$$where:\quad \kappa_j = \frac{\left(1-\alpha_{\text{NH}_3}\right)}{1-\alpha_{\text{NH}_3}\cdot\left(\frac{\text{MW}_{i(\text{NH}_3)}}{\text{MW}_i}-1\right)}$$

$$\alpha_{NH3} = \left.\frac{X_{\text{NH3}}}{X_j}\right|_k$$

k references apportioned PM2.5 measurements from 1 of 13 US sites randomly selected

Ammonia is first partitioned to sulfate until it is either depleted or until the sulfate is completely converted to ammonium sulfate. If ammonia is left over, the remaining ammonia is assumed to further combine with nitrate to form ammonium nitrate until either the ammonium or nitrate are consumed. Any remaining ammonium does not contribute to the PM2.5 concentration estimate.

5.3. Characterization of air quality impacts of US commercial aircraft

Table 5.1 shows the estimated ambient concentration perturbations using the model described above in section 5.3 and the change in the county all-source emissions inventories accounted by commercial aircraft emissions. The values tabulated are straight averages for the period 1991-2003. For each of these parameters, table 5.1 enumerates three distributional statistics—median, coefficient of variation relative to the median, and the relative standard error—to summarize estimated changes in air quality due to US commercial aircraft emissions.

Table 5.1. Estimated changes in all-source emissions inventories and pollutant levels across airport-resident counties due to US commercial aircraft emissions 1991-2003

Statistical summary of changes in all-source inventories and background ambient concentrations due to commercial aircraft emissions in airport-resident counties 1991-2003

	change in all-source county inventory (average 1991–2003)			change in concentration of criteria pollutant		
ambient species	**median**	**coefficient of variation = IQR/median**	**relative standard error = SE/CV**	**median**	**coefficient of variation = IQR/median**	**relative standard error = SE/CV**
NO2	1.8%	1.5	0.043	0.28	2.0	0.048
SO2	0.24%	1.9	0.061	8.8	2.0	0.057
VOC	0.28%	1.9	0.075	4.5	2.2	0.078
CO	0.24%	1.7	0.049	0.047	2.4	0.072
O3	0.56%	2.2	0.092	0.61	2.1	0.052
primary PM2.5	0.093%	1.8	0.057	0.0020	1.9	0.056
secondary PM2.5	0.38%	1.9	0.055	0.052	2.0	0.056

Consider the change in NO_2 air quality resulting from NO_x emissions as an example interpretation of these statistics. Reading from the table, the air quality model estimates that commercial aircraft operations account for a median 1.8% increase in county inventories over the period 1991-2003, with a coefficient of variation (CV) of 1.5 or 150% (equal to the interquartile range divided by the median) and a relative standard error of 0.043 or 4.3% (equal to CV divided by the standard error). In the air quality analysis, changes in precursor concentrations are proportional to changes in emissions inventories. Thus, the estimated inventory change leads to the same fractional increase in ambient NO_2 concentrations. An increase of 0.28 ppb (CV = 2.0 and rSE = 0.048) in ambient NO_2 concentrations is estimated.

The largest sources of uncertainty in the estimated ambient concentration changes are the variability in air quality data across the 96 airport-resident counties and, in the case of PM2.5, the randomized application of apportionment data from the 13 sites across the US. Part of this uncertainty is reducible by implementing geographic specificity, i.e. removing the blind attribution of operations to airports. For the primary pollutants, the major component of variance in the ambient concentration changes of NO_2, SO_2, CO, and PM_{nv} is the variance in the ratio of Q_i/Q_i^I (see equation 5.6). In MAIPA, Q_i/Q_i^I is a function of both county and airport inventories, but its variance is almost completely dictated by variance in all-source emissions across the airport-resident counties used for the analysis.

For the change in PM2.5 concentrations, the primary source of variance is similar, related to uncertainty in apportionment among nitrates, sulfates, organics, and carbonaceous particulates. Variance in ozone concentration change is additionally influenced by OPE, but while the variance in OPE has a significant effect on uncertainty, it is an O(10) smaller influence than variance in the ratio Q_{NOx}/Q_{NOx}^I. However, as would be expected from the linear analysis, OPE, the change in NO_2 concentration, and Q_{NOx}/Q_{NOx}^I have equivalent mean-shift coefficients.

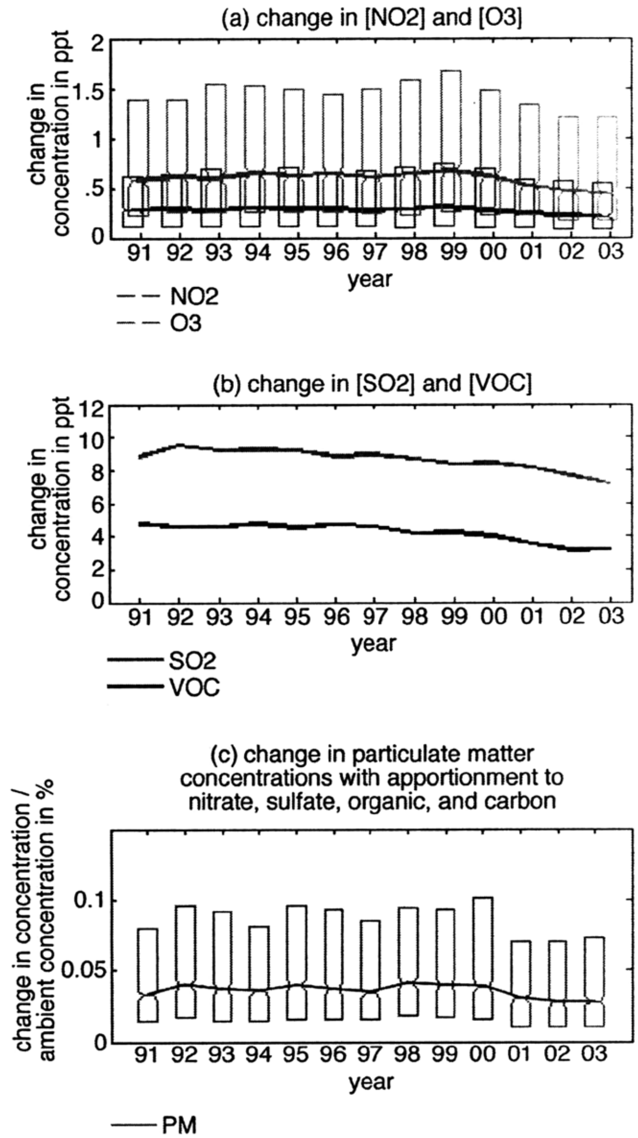
Note that the change in the county all-source inventory is the metric relevant to conformity determinations; these determinations address only the inventoried precursors NO_x, SOx, HC, PMnv, and CO. Using the inventory metric, the results in table 5.1 suggest NO_x to be relatively more consequential to air quality than other species, with SO_x, HC, and CO having a similar but secondary impact, and PMnv to be a distant tertiary contribution. This ordering is similar to the attention given by technological standards controlling aircraft engine emissions. Marginal air quality damages suggest a different comparative picture of source control priorities where SO_x, PM_{nv}, and HC emissions play a role equally important or more so compared to NO_x. Section 5.5 returns to this comparison

5.3.1. Effects of changes in background ambient air quality

Figure 5.2 plots the ambient concentration change due to US commercial aircraft emissions from 1991-2003 for the major sources of air quality damages, O_3 and PM2.5, as well as their gaseous precursors NO_2, SO_2, and VOC (summary statistics for PM_nv and CO can be found in table 5.2).

Figure 5.2. Estimated trends in the air quality impacts of US commercial aircraft operations 1991-2003

Concentrations in the units of the concentration-response functions used to estimate changes in mortality risk, the predominant source of damage costs.



The trend lines in figure 5.2 track the median perturbation to ambient concentrations from 1991-2003; the yearly evolution results from interactions among rates of change in aircraft inventories, county inventories, ambient pollutant levels, and uncertainty in these quantities.[16]

[16] Nonparametric K-W hypothesis tests indicate that only the net over an interval (different for each pollutant) between a year from the period 1991-2000 and a downturn year 2001, 2002, or 2003 is significant (p = 0.05), implying a resolution that allows us only to state that there is a decline in the air quality impact of aircraft emissions from the period before 2001 to the period after.

Despite the linear approximations of the air quality analysis, the trends plotted in figure 5.2 reveal a nonlinear change in the response of air quality to emissions inputs. Chapter 3 reported the median trends in aircraft NO_x, SO_x, and VOC emissions within the atmospheric boundary layer were constant or decreased, with inventories changing between 1991 and 2003 at compound annual rates of {~0, ~0, -0.63}%, respectively. Trends in the median fraction of county all-source inventories accounted by commercial aircraft emissions mirror these trends, with annual rates of {~0, ~0, - 5.2}% for NO_x, SO_x, and VOCs, respectively. In contrast, the annual rates of atmospheric concentration change due to aircraft emissions are {-1.1,-1.5, ~0}%, respectively.

Importantly, these trends indicate that the impact of aircraft emissions on air quality is a function of the progress in controlling emissions from other sources. A similar conclusion was reached by the climate analysis in chapter 4. This comparison indicates that the sensitivity of atmospheric concentrations to NO_x and SO_x emissions declines from 1991-2003, leading to the net declines in $\Delta\hat{X}_i$ plotted in figures 5.2(a) and 5.2(b) for NO_2 and SO2 respectively. In the model, the changes in sensitivity appears as changes in the relationship between the source inventories and the background ambient pollutant concentrations; in other words, the PPE parameter in equation 5.6 changes over time. In the air quality model, ozone and particulate matter are multiplicative of precursor trends. The annual compound rate of change in the ozone due to aircraft emissions is -1.8% from 1991-2003; for PM2.5, the rate is also negative at -1.4%.[17]

Together, the results indicate an increasing marginal damage curve for air quality, i.e. willingness-to-pay (WTP) increases as the level of pollution increases (as measured by the concentration of criteria pollutants). Thus, WTP increases with an increase in all-source emissions, most of which are from sources other than commercial aircraft as evidenced by table 5.1. Similarly, as discussed in chapter 4, climate marginal damages increase as total emissions increase, again most of which are from non-aircraft sources.

5.3.2. Influence of computational resolution on estimated air quality change

It takes on the order of a day to distribute species from the scale of the exhaust plume to the large scales of atmospheric motion. Ozone lifetimes are of the same order. In contrast, PM can survive in the atmosphere for days to weeks after formation and can travel hundreds of kilometers on prevailing winds.

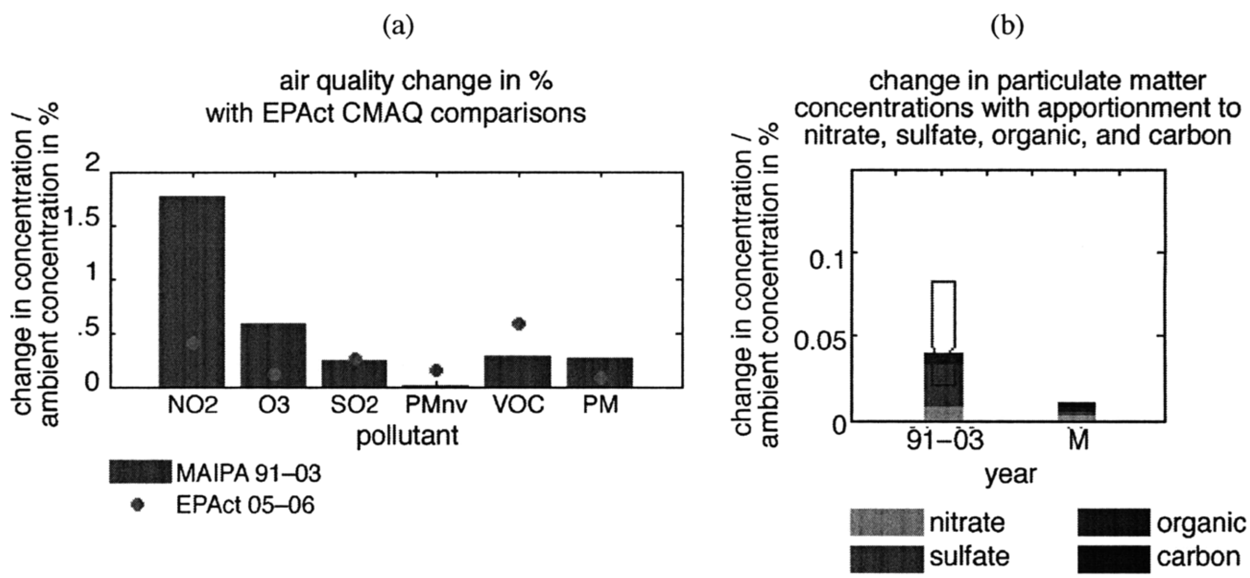
[17] Note that parametric uncertainties have an upward bias on the statistics in table 5.1, which accentuate upward trends and de-emphasize declines; however, this is an O(10) smaller effect than the median trend itself.

Wet and dry deposition processes remove PM and a portion of its precursors. MAIPA does not account for the loss of primary emissions, and assumes that secondary pollutant formation is confined to county boundaries with homogenous exposure to changes in ozone and PM2.5 concentrations.

Comparisons of MAIPA results with recent air quality assessments for the continental US suggest that differences in analysis resolution constitute a fundamental structural uncertainty in the context of modeling aircraft impacts. Figure 5.5(a) plots the estimated concentration changes normalized by the background ambient concentration listed in table 5.1 with a comparison to a recent air quality assessment based on computations performed for a report mandated by the Energy Policy Act of 2007 using the EPA CMAQ (EPACT CMAQ TSD 2007).

These comparisons are used to estimate a magnitude for the uncertainty associated with assuming O_3 and PM2.5 precursor emissions are effectively processed early in the exhaust lifetime (i.e. starting in the exhaust plume) as is assumed by MAIPA versus the assumption that precursor emissions are mixed to the atmospheric mesoscale prior to processing as in CMAQ. Based on the comparisons presented below, different physical assumptions are an O(2) impact on estimated damages from secondary particulates. For ozone, this uncertainty essentially determines whether aircraft NO_x has a role in ozone production.

Figure 5.3. Estimated concentration change normalized by the ambient concentration



The second bar in Figure 5.3a compares the change in ozone concentration estimated by the MAIPA air quality model (gray bar) and using CMAQ for the EPAct study; the MAIPA estimate (0.61 ppb or 0.56%) is 7.6 times larger than through CMAQ (0.08 ppb or 0.12%). We can explain this difference by accounting for differences in inventories, background ambient concentrations, population exposure, and treatment of chemistry.[18]

- As plotted in figure 5.3a, the MAIPA estimated change in NO_2 (0.28 ppb or 1.8%) is a factor of 4.1 larger than the EPAct analysis estimate (0.07 ppb or 0.4%).
- MAIPA uses a larger inventory (factor 1.2) and higher baseline ambient concentration (factor 1.1), but these do not fully account for this difference.
- Factored together, these differences suggest that the effective area of exposure in the EPAct analysis is approximately 2 times the sum of county areas in MAIPA.

In this respect, the CMAQ computations are more diffuse, estimating a longer NO_2 lifetime. These factors account for approximately one-half of the difference in ozone concentrations between the EPAct CMAQ and MAIPA ozone. Measurements of ozone chemistry in plumes of different sizes suggest an earlier onset of ozone production in aircraft engine plumes than for power plant or urban plumes. These studies suggest a more localized formation of O_3 than that associated with the regional scale ozone events that arise episodically downwind of large area emission sources. Footnote 19 provides a brief review of these studies. The remainder is due to the factor 1.5 times lower effective ozone production efficiency of the EPAct analysis (OPE = 1.1), at the very low end of the MAIPA OPE distribution.

MAIPA and the EPAct CMAQ also differ in their estimates of the change in ambient PM2.5 concentrations. The EPAct analysis shows an increase of 0.01 ug/m^3 (0.08%) compared to the 0.04 ug/m^3

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[18] Apart from scenario analyses, rigorous accounts of uncertainty in these computational analyses have not been reported (cf. Fine et al. 2003 for additional discussion of uncertainty assessment in air quality modeling); comparisons in figure 5.3 are made against point estimates.

[19] Nunnermaker et al. (1998) note that for the power plant plumes measured in Tennessee, photochemical lifetimes and meteorological conditions imply ozone production continues 30-100 km downstream, with longer distances related to higher emission rates (Nunnermacker et al. 2000; cf. Nunnermacker et al. 1998, Gillani et al. 1998, and St. John et al. 1998). In the urban plumes measured by Ryerson et al. (2001), maximum [O_3] was observed at larger distances between 50-170 km downstream with NO titration found early in the plume. If we consider that the aircraft source is small relative to a regional or stack plume (scale of the urban plume = 0(100) scale of urban plume and 0(10) stack plume), we may infer the same order reduction in mixing time such that aircraft plume ozone chemistry would move more rapidly towards a NO_x-limited condition. The implication is a more localized formation of O3 than that associated with the regional scale ozone events that arise episodically downwind of large area emission sources.

(0.32%) increase estimated through MAIPA, a factor 4 larger. EPAct CMAQ results did not contain information necessary to systematically isolate the sources of this difference. The primary contribution of variance in apportionment data to uncertainty in the estimated change in PM2.5 suggests a primary root of this difference lies in chemical and microphysical assumptions of the two models.

The apportionment of the MAIPA estimated change in PM2.5 among precursor contributions is indicated in Figure 5.3(b) by the stacked bar. The bar is divided to show the component contributions from NO_x (nitrate or ammonium nitrate), SO_x (as sulfate or ammonium sulfate), HC (as volatile organic PM), and PMnv emissions. Year-to-year fractional conversions are not statistically discernible; they are ratios and uncertainty is higher compared to the estimated absolute change in concentration. Thus figure 5.3(b) contains only summary figures for the entire 1991-2003. MAIPA analysis estimates median conversion percentages, averaged over the period 1991-2003, of 6%, 50%, 20%, and 4% for NO_x, SO_x, VOC, and PMnv emissions, respectively. Sulfates constitute the majority PM2.5 component, accounting for approximately 65% by mass, nitrates and organics constitute ~15-20%,[20] and carbonaceous particulate accounts for 1%.

Removing the differences in inventories and environmental data, we assume the EPAct estimates of changes in PM2.5 precursors and estimate the PM2.5 concentration change applying the MAIPA median apportionment shown in figure 5.3(b). This makes up less than 50% of the difference in estimated PM2.5 concentration change. Taking the further step of artificially altering the MAIPA apportionment to minimize the estimated change in PM2.5—essentially removing any ammonium contribution—does not make up the remainder. There are two potential explanations: first, the SMAT procedure use to determine apportionment in CMAQ may account differently for water mass; second, precursor loss mechanisms are not accounted in MAIPA. Both of these explanations again return to the question of how to treat aircraft

[20] A K-W test indicate no difference in medians at p=0.05.

plumes in the context an air quality analysis. Footnote 21 provides a brief summary of current understanding of particulate matter near airports.

5.3.3. Mortality estimates from linear versus complex air quality models

At the geographic scale over which primary emissions travel prior to the onset of photochemistry and microphysics, health effects decline at a greater rate than concentration perturbations. This suggests that damage estimates using the box approximations of MAIPA may in fact be a maximum at the limiting county resolution. In MAIPA, the population exposed equals the sum of populations over the sample counties, $N_{pop} = A^{ref} \rho_{pop}^{ref} \approx \sum_{l=1}^{L} A^{l} \rho_{pop}^{l}$.

Population data is stratified by age, sex, and race for each of the 96 counties included in the analysis. Since ambient concentration changes are for all counties together, population in equation 5.2 is the sum of these population strata over all counties. However, MAIPA does not account for the impact of population heterogeneity. Total population in this study increases from 108 to 128 million people (r = 1.3%). In comparison, there are 135 small, medium, and large hub airports in counties designated nonattainment zones by the EPA against one or more of the NAAQS. A comparison with the most recently published nonattainment status reports for O_3, PM_{10}, SO_2, CO, and NO_2, as well as proposed nonattainment status for the more recent $PM_{2.5}$ NAAQS shows that approximately 75 million people live in these counties (EPA 2004).

Since airports typically lie in high population density counties compared to those surrounding, total exposure increases in a manner at most proportional to the area of emissions influence. In the opposite sense, considering the volume bounded by a constant hmix and the county boundaries in a well-mixed limit (and that the additional area emissions from the expanded area dilute the aircraft contributions), concentration perturbations decrease in the same manner.

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[21] There is relatively more understood about particle formation in the plume than about ozone production; for example, it has been observed that fine sulfate and organic particles are in abundance in the near-field plume (cf. appendix 2 for discussion of plume particle formation). Three-dimensional air quality models indicate that there are identifiable exposures in local communities attributable to airport emissions, but do not differentiate aircraft versus other sources. Using a nested-grid computation with the tightest resolution at 4 km resolution grid, nested within a larger scale 12 km resolution grid for the Atlanta area, and further nested within a 32 km grid for the eastern US, Unal et al. 2005 estimate positive ozone and PM2.5 exposures in communities surrounding Atlanta-Hartsfield airport; similar European analyses have found comparable conclusions (cf. Moussiopoulos et al. 1997 for Athens, Pison and Menut 2004 for Paris, and Yu et al. 2004). Measurement studies of airport PM2.5 emissions are similarly suggestive of localized exposure (Barbossa 1999, Westerdahl et al. 2007, Herndon et al. papers).

Associated disease incidence diminishes at least linearly with changes in concentration but more likely at a faster rate, specifically since equation 5.6(b) is the basis of mortality incidence estimates, where $N^{m}_{pop} = \alpha^{m} \cdot N_{pop}$. By this scaling, confining ozone and PM2.5 effects to the county in which the airport resides may contribute to an upward bias in health effects per unit emissions, the quantity important to the marginal damage estimate. As comparison, estimates reported in Greco et al. suggest an increase in total health effects at a rate smaller than proportional to distance by two orders of magnitude, as ~1/x with >80% of total population exposure within 50 km.

Figure 5.4 plots the estimated increase in mortality risk as the percent increase in the national non-accidental mortality rate for 2003, associated with NO_x, SOx, VOC, and PMnv emissions. Results are compared to estimates from three recent air quality assessments for the continental US. The first assessment is the EPAct analysis introduced previously; the second uses a response surface model fit to similar computations using CMAQ but with different emissions inputs (Masek 2008); and the third applies a source-receptor methodology (Masek 2008).

Figure 5.4. Estimated mortality change with comparisons

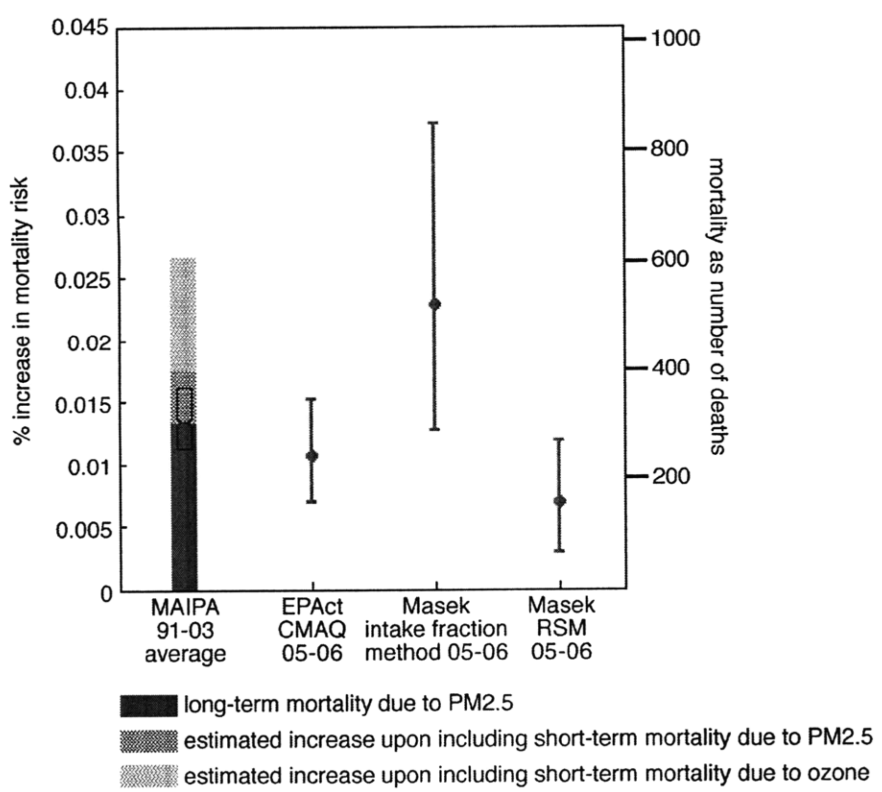


Table 5.2. Incidence of health effects and consequent air quality damages

Statistical summary of changes in the incidence of health effects and consequent air quality damages to commercial aircraft emissions in airport-resident counties 1991-2003

health endpoint	health effect incidence		
	median	coefficient of variation = IQR/median	relative standard error = SE/CV
premature mortality	400	0.22	0.0047
sudden mortality	200	0.090	0.0020
chronic respiratory disease	460	0.19	0.0045
respiratory hospitalizations	10	0.15	0.0033
cardiovascular hospitalizations	6.3	0.15	0.0032
respiratory illness	16,000	0.59	0.0073
restricted activity	11,000	0.26	0.0052

The average median change in mortality risk rate for the period 1991-2003 is 0.026% (CV = 0.38 and rSE = 0.0003). This is equivalent to an average of 400 deaths based on mortality rates for 2003.[22] This is significantly higher than the risk of death due to aircraft accidents over the same period; safety-related mortality for aircraft accidents is 0.65% of the mortality rate due to PM2.5 air quality impacts.[23] Similar findings have been calculated for road transport (cf. Kunzli et al. 2000).

Figure 5.4 also plots comparative results for PM2.5 premature mortality reported in Masek (2008) and the EPAct report to congress (2009). Two models were evaluated by Masek; an RSM developed from CMAQ computations (see previous section) and a source-receptor model (see Rojo 2007). The latter, which applies an intake fraction method to estimate exposure, reflects observations of ambient concentrations correlated to power plant plumes and, as discussed later in this section, reflects similar timescale

[22] A 0.0041% increase in the mortality rate equates to 100 deaths.

[23] National Transportation Statistics. Internet Edition. BTS, DoT. Table 2-9: U.S. Air Carrier Safety Data. http://www.bts.gov/publications/national_transportation_statistics/html/table_02_09.html; accessed 10.08.08. Sources of data: 1960: National Transportation Safety Board, Annual Review of Aircraft Accident Data: U.S. Air Carrier Operations, Calendar Year 1967 (Washington, DC: December 1968).

1965-70: Ibid., Calendar Year 1975, NTSB/ARC-77/1 (Washington, DC: January 1977).

1975 (all categories except miles): Ibid., Calendar Year 1983, NTSB/ARC-87/01 (Washington, DC: February 1987), table 18.

1975 (miles): Ibid., Calendar Year 1975, NTSB/ARC-77/1 (Washington, DC: January 1977).

1980: Ibid., Calendar Year 1981, NTSB/ARC-85/01 (Washington, DC: February 1985), tables 2 and 16.

1985-2006: Ibid., National Transportation Safety Board, Internet site www.ntsb.gov/aviation/Table5.htm as of September 2007.

assumptions to MAIPA. However, because the source receptor model is based on spatially-resolved fits to monitor measurements, it accounts for primary emissions loss. MAIPA assumes precursor chemistry initiates within the aircraft engine exhaust plume, and thus precursor loss is not accounted. In CMAQ computations, precursors are well-mixed to a 36 km^2 grid resolution prior to the onset of photochemistry or particulate formation, an assumption that no chemistry or loss occurs in diffusion to a regional scale. This results in a relatively larger exposure area compared to MAIPA.

MAIPA mortality estimates are 60-150% larger than CMAQ estimates. This is a factor of 1.6-2.5 lower than the difference in the estimated change in ambient PM2.5 concentrations reported previously. A similar result is noted in Masek (2008); mortality results from the application of the intake fraction method are 2.5 times lower than those obtained using the CMAQ RSM, similar to the comparison between MAIPA and EPAct. These differences are related to different estimates of population exposure and the use of different concentration response curves to calculate changes in mortality risk. We will consider the latter of these first.

There is essentially no difference in the mortality rate estimated in Masek and EPAct, and MAIPA estimates. Whereas the EPAct and Masek use the original Pope (2002) study to estimate mortality risk, MAIPA uses the Krewski (2000) reanalyses of the Pope studies. However, risk rates are similar among MAIPA, EPAct, and Masek.

Overlay of pollution and population distributions appear to act contrary and enhance the impact of aircraft-sourced PM2.5. From the scaling arguments discussed in the previous section, we would expect an upward bias in estimated health impacts in MAIPA. However, rather than exacerbating the differences between MAIPA and CMAQ analyses, the overlay of pollution and population distributions in MAIPA instead appears to increase the impact of aircraft-sourced PM2.5. We must conclude that the CMAQ results derive from a higher population-weighted exposure to delPM2.5.

Comparison of ozone mortality between MAIPA and EPAct leads to a contradictory conclusion. EPAct reports a small ozone mortality (~1), giving errE-mort >> errE-[o3], opposite to the comparison of PM2.5 results. There are two differences of note. First, the EPAct analysis is based on the premature mortality risk rate reported in Bell et al. whereas MAIPA considers sudden mortality with a factor 1.5 higher risk rate. Second, analysis indicates ozone destruction (e.g. NO titration near the point of emission) does not account for low EPAct mortality.

In conclusion, we are left with a lower population-weighted exposure to ozone change. We can deduce from these comparisons that geographic exposure heterogeneity is an important structural uncertainty, one that appears to act to lower exposure to delozone, but raise exposure to delPM. There are additional structural uncertainties that have a similar magnitude. Figure 5.4 divides the total mortality increase into its three components: sudden mortality due to $\Delta\hat{Y}_{O_3}$; sudden mortality due to the $\Delta\hat{Y}_{PM_{2.5}}$; and the largest component, premature mortality due to $\Delta\hat{Y}_{PM_{2.5}}$, accounting for 54% of $\underline{\tilde{\mu}}\left\{\hat{I}_m\right\}$. With the addition of PM2.5 sudden mortality, the total PM2.5 component is 72%. The remaining 18% is associated with O3 sudden mortality. A K-W hypothesis test indicates that the sum $\Delta\hat{I}_m$ trend is significant (p=0.05).

Propagated uncertainties in estimated mortality are similar to the magnitude of these structural uncertainties individually. Variances in concentration perturbation, $Var\left\{\Delta\hat{Y}_{PM_{2.5}}\right\}$ and $Var\left\{\Delta\hat{Y}_{O_3}\right\}$, are O(10) larger contributions to $\mathrm{Var}\left\{\Delta\hat{I}_m\right\}$ than uncertainty in the effect sizes β of the concentration-response functions used to calculate mortality incidence; these C-R functions are of the log-linear form as in equation 5.2(b). The pooling process favors the incidence estimates calculated using the concentration-response functions with the lowest Var(β).

Propagated uncertainties in $\underline{\tilde{\mu}}\left\{\Delta\hat{I}_m\right\}$ are attenuated due to the logarithmic transformation and are thus lower than the uncertainties in $\underline{\tilde{\mu}}\left\{\Delta\hat{Y}_{PM_{2.5}}\right\}$ and $\underline{\tilde{\mu}}\left\{\Delta\hat{Y}_{O_3}\right\}$ reported in table 5.1. However, this transformation skews $P(\Delta\hat{I}_m)$ such that the influence of propagated uncertainty on the rate of change in mortality $r_g\left\{\Delta\hat{I}_m\right\}$ is similar to the mean contribution— $r_g^{\tilde{\mu}}\left\{\Delta\hat{I}_m\right\}$ = -0.066 versus the residual $r_g^{\mathrm{Var}}\left\{\Delta\hat{I}_m\right\}$ = -0.052.

5.4. Comparative marginal benefits of emissions reductions

Comparative assessments using damage metrics suggest a different source control perspective than implied by current engine emissions regulations; the traditional regulatory approach is largely based on assessment of inventory metrics. Engine emissions standards were established primarily to address ozone with NO_X the focus.

Based on results for marginal air quality damages, MAIPA analysis presents a different perspective, suggesting that SOx, HC, PMnv emissions play a role in air quality as important as NO_x. Instead of ozone, impacts on ambient PM2.5 emerge as the predominant air quality concern. The first part of this section takes a closer look at estimated air quality damages and describes these results in further detail.

This shift in emphasis is an important result drawn from the historical air quality analysis. No specific policies are proposed in this thesis; its emphasis is on what information is needed to develop effective mitigation policies. In this context, altering air quality source control priorities from O3 to PM2.5 and evaluating options for SOx, HC, and PMnv emissions along with NO_x would be first steps toward improved decision-making.

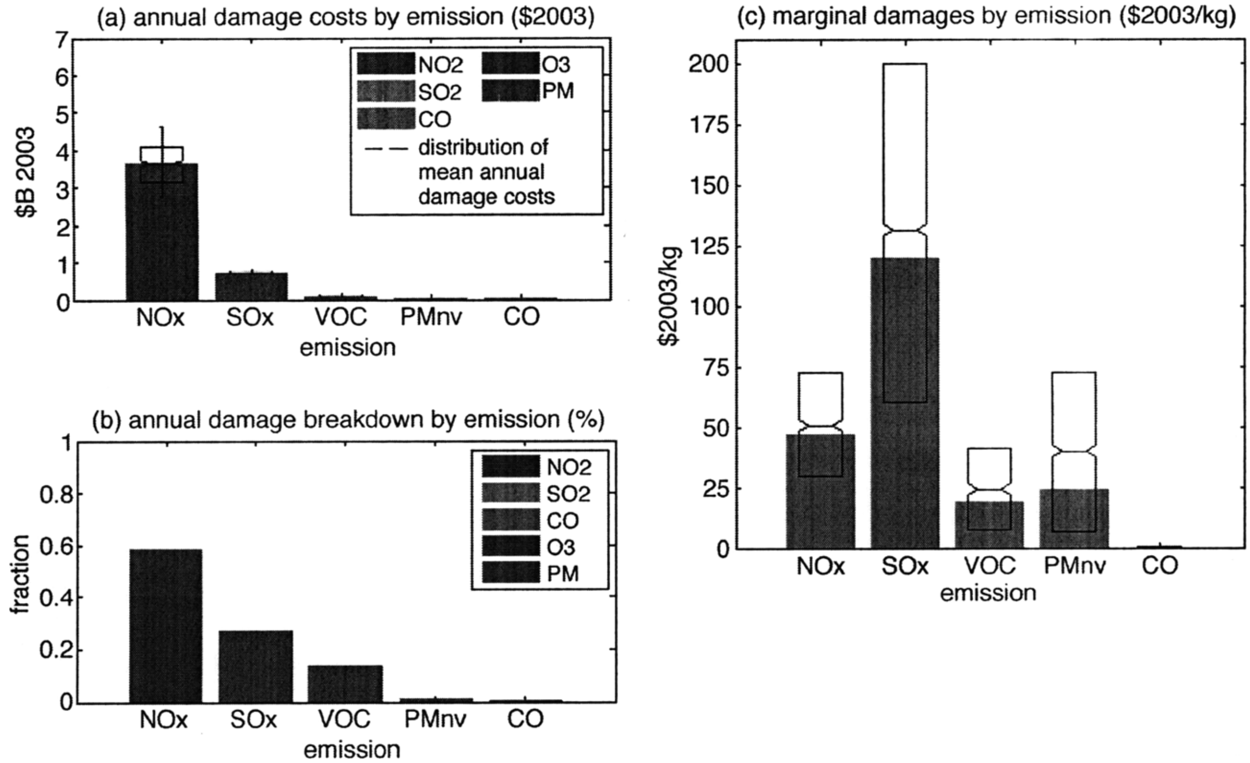
We then broaden the assessment scope to examine the extent to which decisions need to consider the multiple impact vectors initiated by several aircraft emissions (e.g. SOx and NO_x) emissions involved in multiple impacts vectors have both climate and air quality effects, such as NO_x and SOx. These comparisons are useful in understanding whether emissions controls need to extend above the atmospheric boundary layer.

The last part of this section looks at this same comparison in the context of fuel consumption; specifically, we look at the benefits of encouraging better fuel efficiency. More specifically, we are interested in how such an approach to reducing emissions impacts bundles the mitigation of climate and air quality impacts.

5.4.1. Emissions damages from US commercial aircraft

Figure 5.5 plots results summarizing the air quality damages from US commercial aircraft from 1991-2003 estimated through the MAIPA air quality model. Figures 5.5(a)-5.5(c) summarize annual air quality damages. Figure 5.5(a) plots total annual damages by criterion pollutant in billions of 2003 dollars for each impact vector evaluated in the air quality model, with the apportionment of particulate matter precursor contributions indicated for PM2.5. Figure 5.5(b) plots the same results as 5.5(a), but on a fractional scale to show relative contributions.

Figure 5.5. Air quality damages of US commercial aircraft operations



The average median annual damage for 1991-2003 is estimated to be $4.5B (CV= 58% and rSE = 2.9%). Emissions of SO_x, NO_x, and VOC constitute 99% of the annual damage costs with the remainder due to CO and PM_{nv}. For PM2.5, 55% of annual damages is attributable to SO_x, 30% to NO_x, and 15% to VOC. The impacts of NO_x emissions are divided 70:30 among ozone and $PM_{2.5}$ impacts. Figure 5.3(b) explicitly shows the importance of determining the extent of plume emissions processing. If photochemistry is not influential as suggested by CMAQ calculations, the average median annual damage for 1991-2003 would fall from $4.5B to approximately $2B.

As shown in table 5.3, virtually all annual air quality damages occur as a result of increased mortality risks, approximately 60% through premature mortality for PM2.5 and 40% through sudden mortality for ozone. For both annual and marginal air quality damages, the variance of the distribution assumed for VSL is the key primary component of variance in damages, the influence of variance in VSL is a factor of 2-10 stronger than variance in mortality incidence (wherein the range is dependent on which component of mortality risk is considered). In the context of MAIPA, our ability to reduce uncertainty associated with VSL much of this question focuses on valuation methodology.

One important difficulty in use of VSL is in the application of hedonic studies is the transfer of benefits from employees in high risk industries, who tend to be young, to the broader population affected by air quality, the most vulnerable of which are the elderly and people with predispositions due to existing health conditions. Relying solely on contingent valuation studies reduces the median VSL, and consequently damages, by a factor of 5 as compared to figure 5.5(a). This is a consideration for regulatory negotiations for air transport environmental issues; European organizations are moving away from the use of hedonic wage studies and, as a result, would tend to estimate lower air quality damages.

Table 5.3. Annual air quality damages by health endpoint

Statistical summary of changes in the incidence of health effects and consequent air quality damages to commercial aircraft emissions in airport-resident counties 1991-2003.

Morbidity costs are adjusted to \$2003 with a factor λ based on CPI considering only historical appreciation for health care costs (ref); VSL estimates are inflated using the full CPI.

	annual damage cost		
health endpoint	median	coefficient of variation = IQR/median	relative standard error = SE/CV
premature mortality	2.2	0.41	0.0076
sudden mortality	3.7	0.29	0.0062
chronic respiratory disease	0.14	0.20	0.0045
respiratory hospitalizations	0.0002	0.11	0.0025
cardiovascular hospitalizations	0.0002	0.15	0.0033
respiratory illness	0.001	0.31	0.0055
restricted activity	0.002	0.27	0.0057

As with comparative inventories, annual damages can also be a misleading indicator of the comparative worth of environmental investments. Figure 5.5(c) plots marginal air quality damages in \$2003 per kilogram. While NO_x and SOx are the major contributors to annual damages, air quality marginal damages indicate that VOC and PMnv have similar per unit impacts. The comparison in figure 5.3(c) shows SO_x marginal damages are ~3 times larger than NO_x or HC, and approximately 5 times larger than PM_{nv}. Note in particular that although PM_{nv} is a O(100) smaller component of annual damages, its marginal impact is on the same order as NO_x and HC; thus in the context of a cost-benefit analysis, there may be PM2.5 mitigation options that achieve net benefits, but the overall potential for reducing damages is much smaller than for NO_x, SO_x, or HC.

Figures 5.5(a)-(c) indicate that damages from carbon monoxide emissions are 3-4 orders of magnitude less than NO_x, SO_x, HC, or PM_{nv}. Where design approaches to reducing HC and CO emissions are similar, raising the priority of PM_{nv} management adds a different set of chemical and microphysical mechanisms that may complicate the combustor design process. Similarly, the primary challenge to removing fuel sulfur is a trade-off with fuel consumption resulting from changes in fuel composition. While the oxidation of fuel sulfur stands physically independent of NO_x, HC, and PM_{nv} formation, all of these emissions originate with combustion; for this reason, we are interested in their potential mitigation through reducing fuel consumption, a sure way to achieve reductions across NO_x, HC, and PM_{nv} emissions (with the notable exception of some NO_x-fuel efficiency engine design trade-offs). This is subject of the last part of this section.

Table 5.4 summarizes the marginal damage costs estimated in this study; the median, interquartiles, coefficients of variation, and standard errors are averaged statistics intended to summarize the entire period 1991-2003.

Table 5.4. Marginal damage costs of US commercial aircraft emissions

	air quality impact vectors ($/metric ton)	climate impact vectors ($/metric ton)
species	median [interquartile range]	median [interquartile range]
fuel	6,300 [3,800 9,400]	21 [14 32]
NO_x	47,000 [29,000 73,000]	-73 [-340 160]
PM_{nv}	24,000 [6,400 73,000]	3,500 [2,000 7,000]
VOC	19,000 [7,400 41,000]	N/A
CO	230 [140 350]	N/A
C	25 [16 37]	25 [16 37]
H	6.4 [3.9 11]	6.4 [3.9 11]
SO2	120,000 [65,000 200,000]	-930 [-1,640 -510]

As table 5.4 indicates, the conclusions drawn from the air quality analysis are robust to the inclusion of climate effects and thus can be said of emissions from US commercial air transport in general. The benefits of reducing fuel sulfur are tempered by the removal of sulfate in the upper atmosphere, but the effect is a hundredth the magnitude of the health effects associated with sulfate contributions to PM2.5. Reducing EI(S) provides an expected net reduction in damages with greater than 95% confidence.

With the current understanding of NO_x-induced ozone production and methane removal, the climate impact analysis in chapter 4 discussed the uncertainty in concluding whether NO_x emissions at altitude result in a net warming or cooling. However, the estimated benefit from reducing NO_x within the atmospheric boundary layer is O(10) stronger than the scenario or propagated uncertainties in the estimated climate impact of NO_x assessed in this study. Only in the case of soot is there a similar benefit to reductions at altitude and at ground; reducing nonvolatile particulate emissions consistently over the entire flight profile (rather than focusing solely on landing-takeoff cycles) increases the marginal benefits of PMnv mitigation by a factor of 2.

Recent analyses suggest an additional O(2) influence on air quality that originates with the intermittent entrainment of free tropospheric aircraft emissions. Aircraft fly horizontal distances of ~15 km before crossing h_{mix}. Using air quality models, unpublished results from Barrett et al. (2009 forthcoming) and Tarrason et al. (2004) estimate that aircraft emissions above h_{mix} are a factor 2-3 larger source of PMnv and NO_x than aircraft emissions within the atmospheric boundary layer.

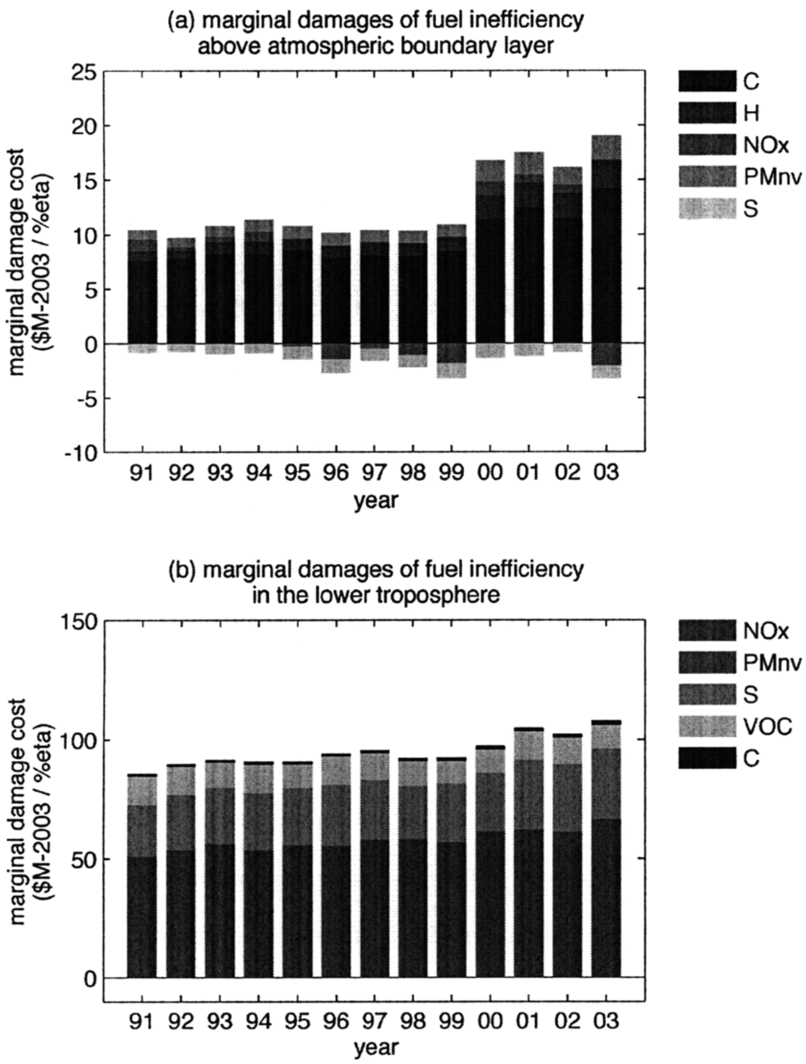
However, this perturbation is distributed continentally, over an area ~17 times the sum of airport county areas with a population ~4 times the sum of those in airport counties. With a linear scaling, transfer across h_{mix} may result in a ~12-18% increase in health effects per unit emissions in airport counties, and ~50-70% additional damages for the US as a whole. Yet there is an additional question of how this transfer changes our current estimates of the climate effects from NO_x and SO_x; whether this alters the benefit comparisons in this study remains a question.

5.4.2. Marginal damages of fuel consumption

As mentioned in the previous section, we are interested in how fuel efficiency acts as a mitigation approach. Figure 5.6 plots estimates for the marginal damage of fuel consumption in units of efficiency as

$/%eta; 5.8a considers only those portions of the flight profile within the atmospheric boundary layer and 5.8b considers only those portions of the flight profile above the mixing height.

Figure 5.6. Marginal damage of fuel inefficiency



Marginal reductions in fuel consumption below the mixing height have an O(10) larger benefit than in the free troposphere. Reducing fuel consumption is an aggregate reduction in both primary (CO_2 and H_2O) and secondary combustion products. However, the constituent damages from primary versus secondary products weight differently depending on the location of emission. As shown in table 5.4, at altitude, the marginal damage of fuel consumption is essentially equivalent to the marginal damage cost of CO_2.

In the lower troposphere, additional damages due to the impact of secondary combustion products on air quality effects result in an O(10) larger fuel marginal damage cost. Figure 5.6a shows constituent contributions, whose magnitudes are determined by EI-weighted emissions marginal damage costs. Comparatively, the benefits of fuel savings at low altitudes derive primarily from reductions in NO_x emissions whereas in the upper atmosphere, benefits derive from reductions in CO_2 emissions. Choosing a higher discount rate to estimate present value climate damages gives greater emphasis to short-lifetime perturbations, resulting in a proportionally higher contribution from cloud effects and a distinctly positive radiative influence from NO_x emissions. At the same time, marginal damages decline by a factor of 2, leading to the same comparative conclusion. Over the period 1991-2003, the median inventory-weighted marginal damage of fuel consumption is between 20-40% of the average fuel price in 2003.

Figures 5.8a and 5.8b show a jump in marginal damages during 2000; statistically, hypothesis tests indicate the most we can comment is that there is a significant difference between marginal damages before 2000 and those after. This may appear at odds with the stark reversal in fuel consumption trends in 2001; however, with the caveat that we are drawing evidence from a cross-sectional analysis, these trends are consistent with the shapes of the air quality and climate marginal damage curves.

Marginal climate damages are linear with declines in environmental quality (measured in this thesis by surface temperature), but asymptotic with emissions (due to radiative saturation in the CO_2 spectrum). However, these damages occur against a background changing under the influence of other sources. Prior to 2001, fuel consumption kept pace with or exceeded the exponential growth in CO_2 emissions in the rest of the economy; these growth rates were high enough to establish the marginal damage cost of CO_2—to which the marginal climate costs of aircraft non-CO_2 emissions are pegged—as an increasing function of anthropogenic emissions. The downturn after 2001 tempered the growth of marginal damages by changing the relative rate of fuel consumption growth against other sources.

Similarly, the relative rates of growth between air transport and other emissions sources determine temporal trends in air quality; relative to air quality impacts, emissions are an increasing fraction of declining source inventories. Regulations under the Clean Air Act and its amendments have reduced ambient concentrations of the most damaging pollutant PM2.5. The chemistry and microphysics of secondary particulate formation is asymptotic with precursor concentrations; thus, as background concentrations decline, the marginal damage of a unit precursor emission increases. A growing fractional contribution augments this tendency and the steady increase in the marginal damage of fuel consumption below the mixing height results.

6. Damages from a noisy environment

Methods for estimating noise exposure are well established. There has been less focus, however, on the explicit connection between change in these noise levels and the evolution of US fleet noise performance. More important in the context of this thesis, trends in noise impacts have not been considered in relationship to accompanying changes in emissions impacts. The first objective was to establish and demonstrate a model to estimate trends in noise exposure as a function of a cumulative noise metric. (Chapter 3 presented noise inventories in SEL dBA; the choice of this metric was not independent of the methods developed in this chapter.) The second objective was to evaluate correlations among trends in air transport noise and emissions damages; in particular, reexamine the benefits of the ANCA from the perspective of joint noise and emissions control. (Analysis of inventory trends in chapter 3 suggests opportunities exist for joint control strategies that target both emissions and noise.)

Using a probabilistic relationship between inventoried noise (cf. chapter 3) and population exposure, this chapter reports estimates of population noise exposure and consequent damage costs.

Contribution 6.1. A probabilistic model of national aircraft noise exposure

As opposed to a detailed radiative model to determine ground noise exposure for each US airport, MAIPA uses a physics-based statistical model to estimate noise-exposed populations that is built upon a coherent relationship between exposure areas (DNL contours) estimated by complex models and airport noise inventories (cf. chapter 3).

Result 6.1. Reassessed the environmental benefits of the aircraft retirements mandated by the 1990 Aircraft Noise and Capacity Act.

From December 31, 1994 through December 31, 1999, FAA mandated a scheduled phase-out of portions of the commercial fleet identified by their failure to meet a limit on noise levels (14CFR91.801-877 Subpart I: Operating noise limits). This rule, known as Stage 2 phase-out, is part of an ongoing strategy of progressive stringency to reduce noise in airport-local communities. This chapter presents results from a retrospective analysis of noise trends in the context of the Stage 2 phase-out mandated by the ANCA, the major noise regulatory action implemented within the timeframe of the MAIPA analysis 1991-2003.

Analysis results suggests that the Stage 2 phaseout was significantly less efficient in reducing noise exposure than a priori evaluations of the regulation. This reassessment finds differences are sourced to methodological issues. The resulting trends give a different picture of historical impacts than previously understood, highlighting mitigation of air quality impacts as the primary source of benefits from the ANCA. They also suggest reported reductions in noise exposure to be significantly overestimated for the period 1991-2003. Against literature estimates of the costs of the phase-out rule, this analysis suggests a positive cost-benefit ratio, i.e. a net cost. By a ratio 2:1, more of the benefit from the Stage 2 noise phase-out rule came through reductions in VOC and PMnv emissions than from noise reduction. These results point to the value of ex post assessments of aviation environmental policies toward improving mitigation efficacy, particularly in the context of providing options that obtain benefits through an appropriately bundled set of noise and emissions reductions.

Result 6.2. Noise mitigation challenges in the context of emissions impacts

Considering that: (1) an estimated $10 in damages are sourced to emissions for every $1 sourced to noise from commercial operations in the US; (2) damages from both climate and air quality impacts are driven primarily by the activities of sources outside the air transport industry; (3) further reductions in noise are expected through retirement of Stage 3 aircraft, through both economic and regulatory incentives; and (4) noise and air quality continue to be addressed through regulatory standards while an approach to controlling climate impacts has not been established, the historical analysis suggests that a realignment in resources toward emissions mitigation has merit and should be evaluated. With this in mind, the results in this chapter recommend a thoughtful reconsideration of the steps best suited to reduce noise impacts while attending to growing environmental stress from emissions.

In particular, these findings suggest the importance of a reexamination of the fundamental mechanisms of how people value reductions in environmental noise, with the goal of expanding opportunities to address aircraft noise impacts. Uncertainty analyses show the range of term/rate scenarios is greater than the parametric uncertainties propagated through the noise model, indicating that the most important factors in determining annual cost are the rate and term of the depreciation. There is, however, a fundamental question as to whether noise damages are sufficiently expressed through property values to justify the application of hedonic valuation methods.

6.1. Noise damages of US commercial aircraft operations

As in the approach to estimating climate damages, changes in noise levels are valued directly such that estimating welfare change is not explicit; instead, damages are estimated directly as a function of a metric of environmental change (DNL) where the noise exposure level is a function of fleet noise. This section discusses the content and form of the relationship used to evaluate damages as a function of DNL; section 6.3 then addresses the environmental modeling component.

6.1.1. Noise damages based on hedonic estimates of property depreciation

Noise impacts can be measured in a microeconomic sense by declines in utility, due to which people adjust their consumption patterns. The corollary to this is that there is some amount people would be willing to pay to return to the original level of utility. For this study, current progress in the development of economic methods to evaluate noise damages limits accounts of the value people place on reducing noise effects to observations of market transactions for which assumptions must be made as to how well willingness-to-pay is captured by a surrogate good.

This thesis estimates noise damages based on studies using hedonic pricing methods to estimate property depreciation via housing markets. Hedonic methods observe that certain goods and factors, such as housing, can be treated as heterogeneous, composed of at least weakly separable characteristics relating to structure and location. One of these characteristics can be environmental quality, which for noise is most often measured by DNL (or a close equivalent).

MAIPA evaluates noise damages using a meta-analysis reported by Nelson (2004), who considers explanations for such differences among 33 hedonic estimates of NDI for cities in the United States and Canada. Salient to this thesis is the regression shown in equation 6.1 where the significance of the dichotomous variables representing location and functional form—β_3 and β_4 in equation 6.1—suggests that including Canadian studies and those using linear forms lead to higher NDI estimates.

$$\text{NDI}\ (\%) = \beta_0 + \beta_1 \cdot X_1 + \beta_2 \cdot X_2 + \beta_3 \cdot X_3 + \beta_3 \cdot X_3 + \varepsilon_1 \ldots \varepsilon_4$$
$$R^2 = 0.773$$
(6.1) $$F - test : p = 0.001$$

X	$\hat{\beta}$	$SE_{\hat{\beta}}$
0: constant ≡ baseline effect size	0.5069	0.1425*
1: mean real property value (× 0.001)	−0.0001	0.0013
2: log(sample size)	−0.0140	0.0261
3: model form (linear = 1, log-linear = 0)	0.3340	0.1544*
4: country (Canada = 1, U.S. = 0)	0.3357	0.0805*

$* = (\alpha < 0.05)$

Some hedonic analyses take specific account of proximity to an airport as a benefit since it represents work and shorter commuting times.[1] There is, additionally, a difference in the time periods over which transactions are considered; some hedonic studies date to the 1960s. Nelson finds, through additional regression trials, that neither of these characteristics significantly contribute to variation in NDI. Since this thesis considers the United States and we desire NDI to reflect dependence on house price across the widest set of airport locales, both β_3 and β_4 in equation 6.1 are set to zero and we are left with β_0 as the only significant variable.

If marginal damage can be assumed constant over a sufficiently broad range of noise levels, these prices can be employed to calculate total costs relative to the noise effect threshold. The regression in equation 6.1 controls for the extent to which the hedonic estimates account for noise levels (correlated with sample size) and finds an insignificant impact, consistent with previous analyses (cf. Walters, Palmquist).

6.1.2. Application of noise depreciation index

For this study, NDI (and thus marginal damage with respect to DNL) is assumed constant over the noise levels evaluated (50-70 DNL), specified as $\widehat{\text{NDI}} = P_{norm}^{\text{NDI}}\left(\hat{\mu}_{\text{NDI}} = 0.5069 : \hat{\sigma}_{\text{NDI}} = 0.1425\right)$. Equations 6.2-6.5 summarize the MAIPA computation of annual noise damages. The total present-value noise damages ($\hat{C}_{dc}^{nois}$) (equation 6.2) are the sum of noise damages estimated for each airport locale ($\hat{C}_{dc}^{nois}\big|^{l}$) for both the 55-65 DNL and 65-70 DNL exposure areas (equation 6.3).

---

[1] The economic model of a city can be basically represented as a downtown surrounded by rings whose main characteristic is equivalent commuting time (Kolstad 2000). Land prices decrease as commuting time increases until, at some point, agricultural uses become more productive than housing. A similar idea applies to the airport. This is relevant only insofar as people in the near airport communities actually derive benefit from the proximity.

(6.2) $$\hat{C}_{dc}^{nois}\Big|_T = \hat{C}_{dc}^{nois} \cdot \frac{r}{1-(1+r)^{-\alpha}}$$

(6.3) $$\hat{C}_{dc}^{nois} = \sum_{l=1}^{L} \hat{C}_{dc}^{nois}\Big|_{55-65}^{l} + \sum_{l=1}^{L} \hat{C}_{dc}^{nois}\Big|_{65-70}^{l}$$

House price represents a present-day valuation of the expected stream of use benefits that accrue over the lifetime of investment in the house. To calculate annual damages ($\hat{C}_{dc}^{nois}\big|_T$), total property depreciation is annualized such that a change in noise levels would return an economic profit within a term α discounted at interest rate r. A range of scenarios are evaluated with α = [15:30] years, equivalent to typical mortgage terms, and rates r = [0.06:0.1], which spans the mortgage rates offered between 1991-2003 and includes the FAA recommended r = 0.07 for infrastructure project benefit-cost assessments.

Damages at a particular airport locale are the product of average noise exposure ($\bar{\theta}$), NDI, and the total affected housing capital (equation 6.4). For the calculations here, it is assumed that the average noise exposure ($\bar{\theta}$) for residents between the 55 and 65 DNL contours is the logarithmic average 62 DNL of the noise level boundaries, and for the 65-70 DNL area, the logarithmic average 68 DNL of the boundaries. The noise threshold $\hat{\theta}_x$ is specified as ggg DNL.

(6.4) $$\hat{C}_{dc}^{nois}\Big|^{l} = \left(\bar{\theta} - \hat{\theta}_x\right) \cdot \widehat{\mathrm{NDI}} \cdot \left(\lambda \cdot \hat{C}_{cap}^{house}\right)\Big|_i^{l}$$

The total housing capital affected ($\hat{C}_{cap}^{house}$) is based on the area ratio $\hat{A}_i^l / A_{county}^l$ where i = 55-65 or 65-70 DNL. Intercensal demographic data specify county average housing prices ($\bar{P}_{house}^l$) and quantity N_{house}^l (equation 6.5).

(6.5) $$\hat{C}_{cap}^{house}\Big|_i^{l} = \bar{P}_{house}^{l} \cdot N_{house}^{l} \cdot \hat{A}_i^l / A_{county}^l$$

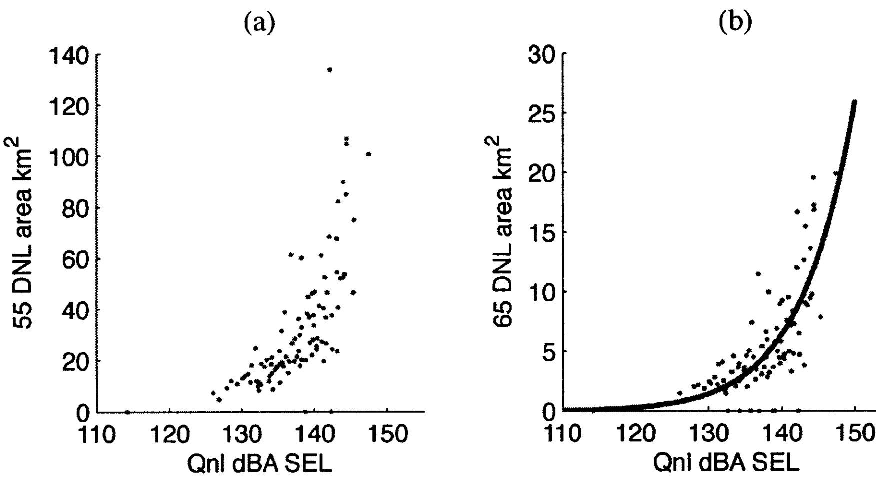
Prices are adjusted to \$2003 with a factor λ based on regionally-specific historical appreciation for census statistical areas measured by the House Price Index published by the Office of Federal Housing Enterprise Oversight, an arm of the Department of Housing and Urban Development (cf. Federal Housing Enterprises Financial Safety and Soundness Act of 1992, 12 USC. 4501 et seq.).

6.2. Probabilistic model of exposure to aircraft noise

The noise model is built on a probabilistic representation of population exposure based on a coherent relationship between exposure areas—DNL contours calculated using INM (via MAGENTA)—and airport noise inventories. Figure 6.1 shows the MAGENTA DNL contour areas A_{65} and A_{55}, respectively referring to the 65-70 DNL and 55-65 DNL contours estimated for 2002, and their functional dependence on airport inventories $\hat{Q}_n^l$ where l is the airport index ($L = 96$). Chapter 2 and appendix A3 discuss the application of the MAGENTA airport-specific operational data.

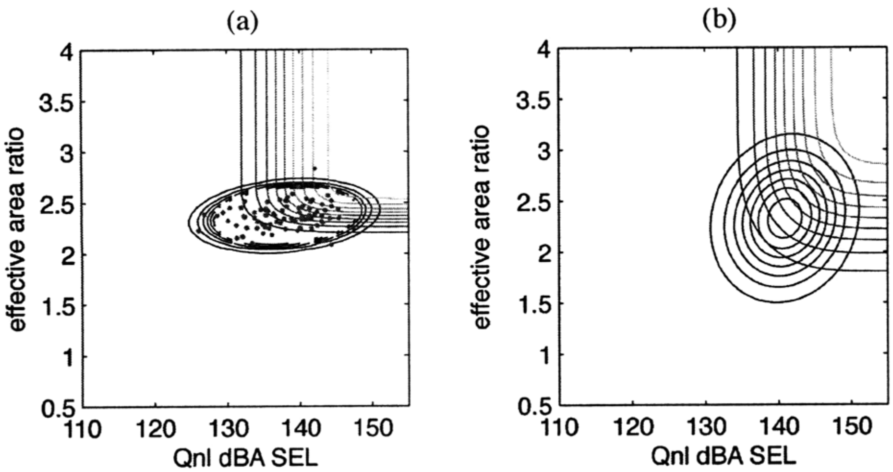
Figure 6.1. Noise exposure area as a function of airport-specific noise inventory

Relationship between MAIPA airport noise inventories and 2002 INM-MAGENTA 55-65 DNL and 65-70 DNL noise exposure areas.



A probabilistic relationship between noise exposures and airport-specific noise inventories is established using a clustering parameter. Abstracting contour areas (A) to a circle defined by an effective radius r, the clustering parameter is $\widehat{\Delta r_n} = \sqrt{\hat{A}_2} / \sqrt{\hat{A}_1}$, where A1 and A2 are contour areas and A2 > A1; figure 6.2 shows data cluster (orange dots) formed by $\widehat{\Delta r_n} = f\left(\hat{Q}_n^l\right)$.

Figure 6.2. Data clustering using ratio of noise exposure areas and Gaussian mixed model fit



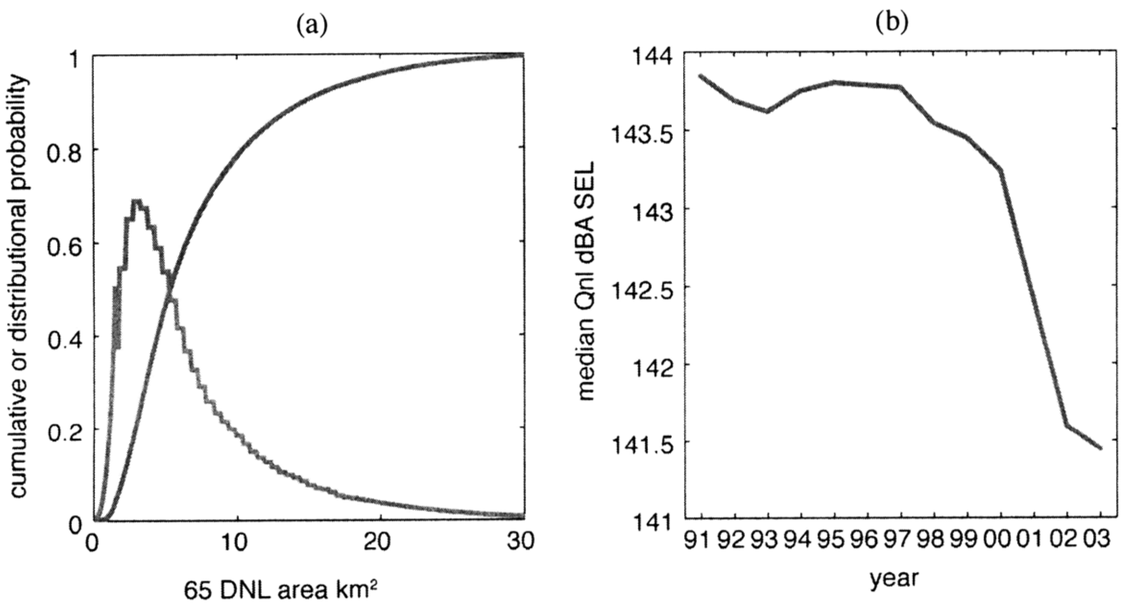
A Gaussian mixed model (GMM) is used to estimate a probabilistic function from the clustered data (with zero values censured) to define the relationship $\widehat{\Delta r_n}$ versus $\hat{Q}_n^l$: $\mathrm{P}\left(\hat{Q}_n, \widehat{\Delta r_n}\right) = \mathrm{P_{GMM}}\left[\left(\mu_{\hat{Q}_n}, \mu_{\Delta r_n}\right):\left(\sigma_{\hat{Q}_n}, \sigma_{\Delta r_n}\right)\right]$. Figure 6.2 illustrates the contours of the probability and distribution functions $\mathrm{P}_{\mathrm{GMM}}^{\widehat{\Delta r_n}}$ and $\mathrm{F}_{\mathrm{GMM}}^{\widehat{\Delta r_n}}$; of interest, over a factor of 16 in noise level, the ratio of effective radii is 1.1 with the 10-90 percentile ranges for $\hat{Q}_n^l$ and $\widehat{\Delta r_n}$ equal to [132 144] SEL dBA and = [2.2 2.5] respectively.

The GMM as applied here is a Bayesian relationship between total noise levels and noise exposure area (capturing variability among locales) that might otherwise be estimated using a regression model. Regressions might provide a clearer picture of the area-inventory relationship, however the aim here is not to uncover trends or explanations therefor, but to estimate marginal costs given a limited sample of airport activity. Also, practically, the inclusion of uncertainties in INM area estimates effectively obscures a regression. Thus, we take a different approach. The value of its application here is two-fold; the explanatory burden is low and we can improve upon the model as additional airport noise data emerges.

The FAA Integrated Noise Model (INM) resolves contour areas with a maximum accuracy of ±5 dBA for the 65+ DNL contour, dropping to ±3dB for levels 75+ DNL provided the statistical sample of days is large enough such that standard errors are relatively small (FAA 1985, and FAA 1983=AC150-5020-1). In MAIPA, this defines $\mathrm{DNL} = \mathrm{P}\left(\mathrm{norm} : \mu_A, \sigma_A = 5/3\ \mathrm{dBA}\right)$, carried through the radius ratio $\widehat{\Delta r_n}$ as uncertainties in A_{65} and A_{55}. To clarify the effect of contour and inventory uncertainties, compare the fit in figure 6.2a with figure 6.2b, the latter of which plots the GMM fit used for MAIPA based on a Monte Carlo simulation of 1000 samples for each of the 96 airport set. Here, the 10-90 percentile range for the radius ratio increases to $\widehat{\Delta r_n}$ = [1.8 2.8] with the 10-90 percentile range for the airport-local noise inventories remaining at $\hat{Q}_n^l$ = [131 147] SEL dBA. With the addition of the contour uncertainty, the effective radii increases to 1.6, while there remains a factor of 16 in noise level.

To apply the GMM to estimate noise exposure as a function of airport noise inventory, we specify an estimate for the 65-70 DNL area (A65) and then derive the 55-65 DNL area (A55).[2] In order to provide this baseline reference area (and to facilitate future improvement in precision), a nonparametric probability function is constructed for $\hat{A}_{65}$. The probability and distribution functions $P\left(\hat{A}_{65}\right)$ and $F\left(\hat{A}_{65}\right)$ are plotted in figure 6.3a, again censuring zeros; zeros are accounted separately as random switch based on the fraction of zeros in the data.

Figure 6.3. Probabilistic description of 65 DNL exposure area



Since both the reference area $\hat{A}_{65} = f\left(\hat{Q}_n^l\right)$ and the median airport noise inventory $\tilde{\mu}_{\hat{Q}_n^l}$ decline over time, as plotted in figure 6.3b, we need to adjust the probability function to account for the historical change.

---

[2] It would be preferred to map A65 against a reference area such as 75 DNL contour, which is unlikely to extend beyond the airport boundary (e.g. $\widehat{\Delta r_n} = \sqrt{\hat{A}_2}\big/\sqrt{\hat{A}_1}$ versus $\hat{Q}_n^l$ where $A_2 = \hat{A}_{65}$ and $A_1 = \hat{A}_{ref}$). This information is currently unavailable.

This is done by shifting the median of the probability distribution, approximating a yearly shift in $\tilde{\mu}_{\hat{A}_{65}}$ using the power law regression $\hat{A}_{65} = \alpha \cdot \left(\hat{Q}_n^l\right)^{\beta}$ plotted in figure 6.1b where α = 1.9e-43 and β = 20.3 ($R^2_{adj} = 0.60$).

As implemented, the noise model is exercised as follows. For each year in the study period 1991-2003, a regression is performed on the GMM to determine the conditional distribution $P\left(\widehat{\Delta r_n} \middle| \hat{Q}_n^l\right)$. Where a zero is sampled for the 65-70 DNL reference area, $\hat{A}_{65}$ is specified as $P_{unif}^{r_{65}^{ref}} = [0.05 : 0.1]$ km² to reflect the statistical uncertainty in estimating a zero exposure area.

Intercensal county statistics determine population exposure ($\hat{N}_{nois}$) based on the ratio of areas determined by $\hat{A}_{65}$ and $\hat{A}_{55}$ to county area as in equation 6.6 where l refers to the demographic data relevant to the sample airport and ρ_{pop} is population density.[3]

(6.6) $$\hat{N}_{pop}^{nois} = \left(\rho_{pop}^{ref} / \rho_{pop}^{l}\right) \cdot \sum_{l=1}^{L} \left[N_{pop}^{l} \cdot \left(\hat{A} / A^{l}\right)\right]$$

Noise exposure areas are a small fraction of total county area. Comparison of MAIPA estimated population densities to the MAGENTA 2002 exposure estimates show similar DNL 55+ population densities ($\rho^{ref} / \rho^{l} \approx 1$), but a substantially smaller population density within the 65+ DNL contour in the MAGENTA case ($\rho^{ref} / \rho^{l} \approx 0.37$). However, this bias is attenuated by a factor of ~4.5 in the total population estimate.

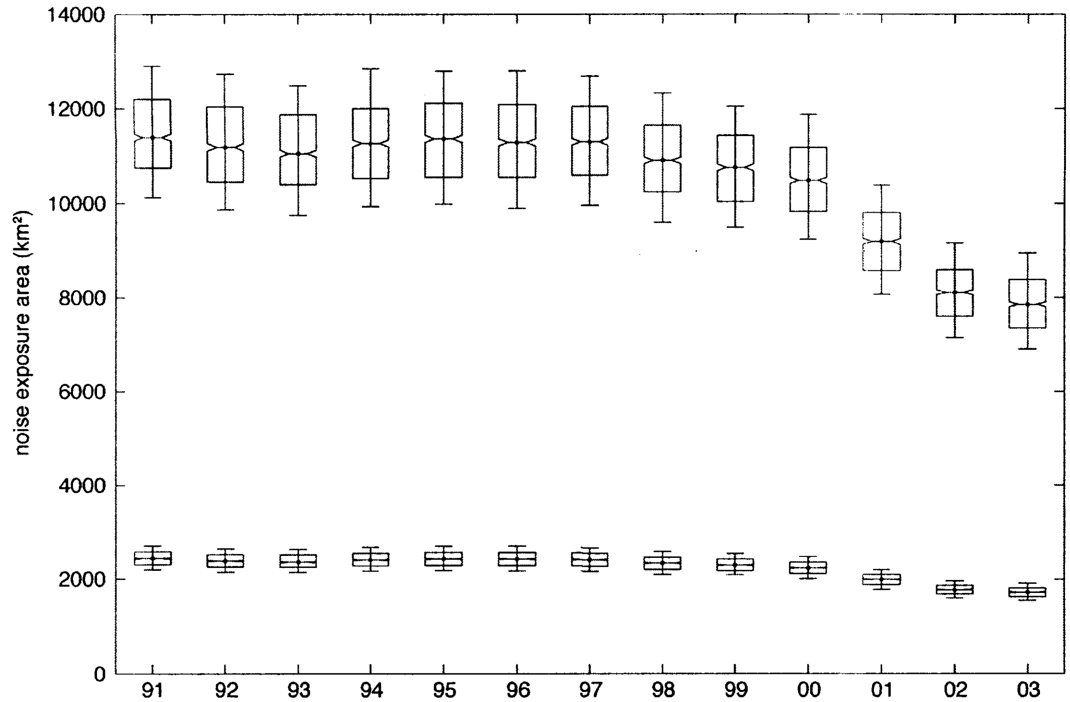
6.3. Historical trends in exposure to noise from US commercial air transport

Figure 6.4 plots distributions for the sum exposure areas $\hat{A}_{65} = \sum_{l=1}^{L} \hat{A}_{65}^{l}$ and $\hat{A}_{55} = \sum_{l=1}^{L} \hat{A}_{55}^{l}$. The coefficient of variation relative to the median and the standard error for these area estimates are $\widetilde{CV}\left\{\hat{A}_{55}\right\}$ = [0.12 0.14], $SE\left\{\hat{A}_{55}\right\} = 0.003$, $\widetilde{CV}_{\hat{A}_{65}}$ = [0.11 0.12], $SE\left\{\hat{A}_{65}\right\} = 0.0026$.

[3] References: Population Estimates Branch of the US Census Bureau intercensal state and county characteristics, population estimates with 1990-base race groups, files for internet display 6.23.03, see http:// www.census.gov/ popest/ archives/ EST90INTERCENSAL/ STCH-Intercensal.html; see also http:// www.census.gov/ popest/ archives/ methodology/ 90s-st-co-meth.txt and /90s-co-meth.txt for methodology) & Annual County Resident Population Estimates by Age, Sex, Race and Hispanic Origin: April 1, 2000 to July 1, 2006 File:7/1/2006 County Characteristics Resident Population Estimates File Source: Population Division, U. S. Census Bureau Release Date: August 9, 2007. Notes on pop data; data for Nebraska unavailable due to error on Census site (NE same as NV) thus held same at 99 levels through 00-03; 00-03 data no longer has <1 and 1-4 categories only 0-4 so infants assumed to be same proportion of 0-4 as (<1/(<1+1-4)).

Figure 6.4. Estimated noise exposure areas in the US 1991-2003

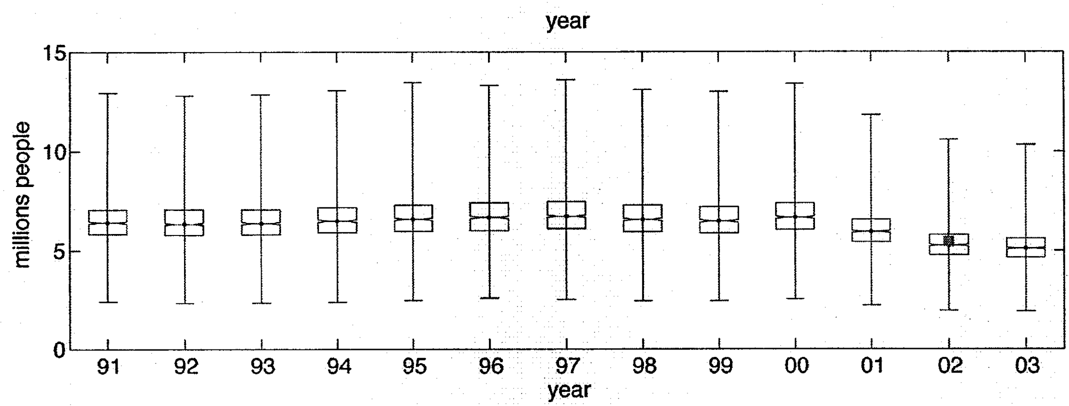
55-65 DNL and 65-70 DNL contours

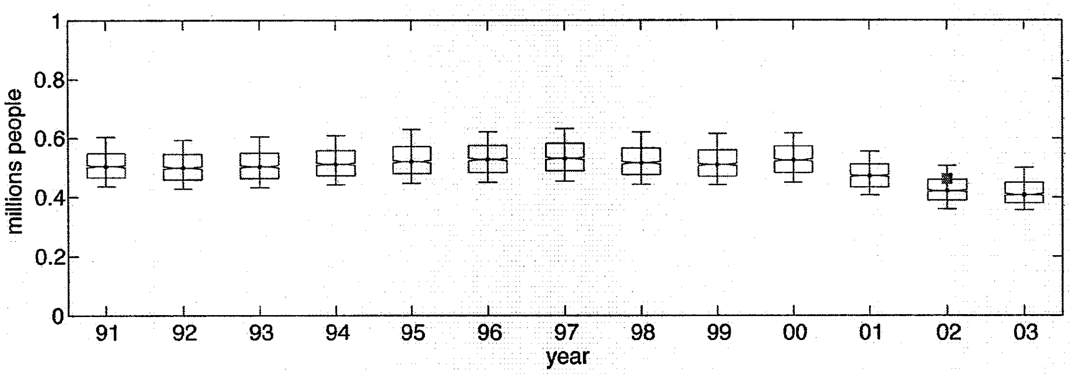


Because MAIPA propagates uncertainty through the calculation as DNL, area (and population exposure) distributions are skewed due to the squared radius dependence. Median noise exposure areas decline by $\Delta\hat{A}_{65} = \Delta\hat{A}_{55} = 0.3$ from 1991-2003, 27% of which occurred from 1991-1999 ($\Delta\hat{A}_{65}^{99+} = \Delta\hat{A}_{55}^{99+} = 0.08$). The median area values plotted in figure 6.4 are equivalent to a decline in the ratio of total exposure area to total county area $\tilde{\mu}_{\hat{A}_{55}} / A^{cnty}$ from 3.7% to 2.5% for 55-65 DNL, and from 0.8% to 0.6% for the 65-70 DNL contour.

Figure 6.5 plots population exposure, $\hat{N}_{65}(t)$, $\hat{N}_{55}(t)$, with comparison to MAGENTA INM-derived estimates for 2002 (orange dot). The coefficient of variation relative to the median and the standard error for these population estimates are $\widetilde{CV}\left\{\hat{N}_{nois}^{55}\right\}$ = [0.19 0.21], $SE\left\{\hat{N}_{nois}^{55}\right\}$ = [0.0048 0.0054], $\widetilde{CV}\left\{\hat{N}_{nois}^{65}\right\}$ = [0.16 0.17], $SE\left\{\hat{N}_{nois}^{65}\right\}$ = [0.0040 0.0046].

Figure 6.5. Exposed populations in the US 1991-2003





Trends in population exposure mimic exposure area trends, but are mitigated by population growth; over the period 1991-1999, the compound annual population growth rate is 1.6%, in contrast to the decline in exposure area at a rate of -0.6% per year; population exposure increases by $\Delta\hat{N}_{55}^{phase}$ = 3.9% and $\Delta\hat{N}_{65}^{phase}$ = 4.2% from 1991-2003. For population exposure as a whole, the contributions of mean trends in $\hat{A}_{nois}$ and $\hat{N}_{nois}$ account for $\overline{\Delta}\hat{N}_{nois}^{phase}$ = 12%, leaving a residual $\Delta'\hat{N}_{nois}^{phase}$ = -8.0% that accounts for the influence of propagated uncertainty. The sign of the residual depends on the relative directions of trends in contributing parameters and the skew in their probability functions. In contrast, from 1999-2003, a larger decline is estimated with $\Delta\hat{N}_{65} \approx \Delta\hat{N}_{55}$ = -22%; uncertainty has a smaller effect on the realized trend with $\overline{\Delta}\hat{N}_{nois}^{phase}$ = 25% and $\Delta'\hat{N}_{nois}^{phase}$ = -2.7%.

6.3.1. Comparison with historical estimates of population exposure trends

These trends are at odds with FAA estimates of exposed population change from 1991-1999; these estimates report a decline from 2.5 million to 0.4 million people (ΔN_{65}^{FAA} = -84%) (ref CAEP5 MAGENTA discussion and Connor calculations). In comparison, MAIPA estimates a slight increase from

0.50 million to 0.53 million people. To understand this difference, we parse a comparison of the estimated DNL 65-70 populations.

FAA results do not estimate population exposure directly for the DNL 55-65 exposure area, instead using a derived factor of 10 to calculate N_{55}^{FAA} based on a ratio of populations derived from the 2002 INM-based estimates $N_{55}^{MAG}/N_{65}^{MAG}$. As shown in figure 6.5, MAGENTA results for 2002 are similar to the MAIPA-estimated population exposure with $\varepsilon_{N_{55}^{MAG}}$ = -8.7%. Further, deriving a population ratio from MAIPA results gives a similar population ratio of $\hat{N}_{55}/\hat{N}_{65}$ = 13 for 2002. Referencing the regression plotted in figure 6.1b and extrapolating from a nominal median airport noise inventory of $\tilde{\mu}_{\hat{Q}_n^l}$ = 143 SEL dBA at the beginning of the historical survey period, in order to arrive at the 84% decrease suggested by FAA (which does not include change in population density) requires the median airport noise inventory to be $\tilde{\mu}_{\hat{Q}_n^l}$ = 156 dBA SEL for 1991, a difference of 13 SEL dBA or a factor 20 decrease in SEL.

In comparison, figure 6.1b shows the median MAIPA airport noise inventory declines by 1.8 dBA SEL from 1991-1999. This is consistent with (i.e. smaller than) the change in noise stringency associated with the Stage 2 phase-out rule —$\Delta Q_n < \Delta q_j = 6$ EPNdB. Note that this stringency change is the maximum possible reduction in the noise inventory; the realized noise inventory includes replacement and increases in the number of operations while the technology standard does not (~30% of aircraft were affected by phase-out). Removing the effect of population change from MAIPA results reduces the difference by 13% and does not account for the discrepancy with the FAA estimates.

The reason for the discrepancy in MAIPA trends versus FAA estimates can be traced to the modeling approach used to calculate the latter. The FAA estimates for noise contour change related to the phase-out rule, upon which population estimates are based, were calculated using the Area Equivalent Method (AEM), an FAA legacy tool used for the screening of federally-funded infrastructure improvements for the requirement of a detailed noise analysis using INM.

AEM estimates noise exposure areas using a summation of area contributions calculated for specific aircraft models. These contributions are derived from an INM calculation of the 65 DNL exposure area resulting from 100 operations at a canonical one-runway airport. Based on this estimate, a regression is obtained between operations and 65 DNL exposure area with the assumption that areas associated with other operational levels are equivalent to the logarithmic ratio of DNL levels (e.g. 106.5 /105.5 = 10, thus

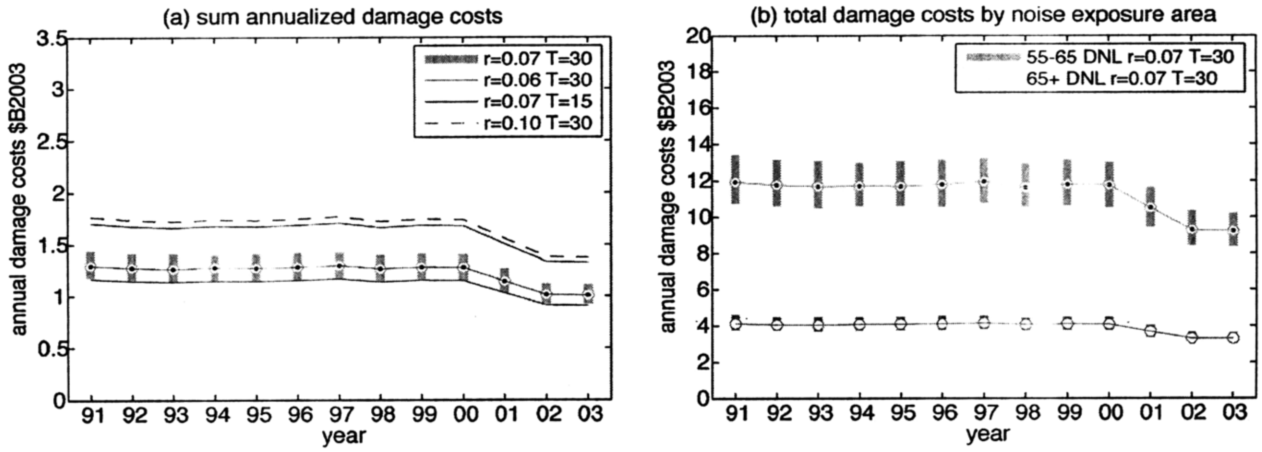
1000 flights is assumed to establish a 65 DNL exposure area equivalent to the 55 DNL exposure area calculated by the INM for the 100 flight calculation). The AEM sums these area contributions for a baseline and project-altered scenario to estimate a change in exposure area; if over 17%, FAA policy indicates that a more detailed noise analysis is required.

The FAA AEM analysis of the phase-out rule indicates a 72% decline in the 65+ DNL exposure area. We can account for the difference between AEM and MAIPA estimates of exposure area change with a comparative AEM calculation using MAIPA aircraft operations data. The result shows that, using AEM, exposure area declines by 35% while the MAIPA result reported above shows an 8% increase. This difference added to the effect of including population change accounts for the discrepancy between FAA and MAIPA population exposure estimates, a result principally due to the linear summation of exposure areas in AEM. This procedure estimates a larger change than using the MAIPA approach which is based on logarithmic summations correlated to exposure areas. Indeed, if noise exposure area was proportional to SEL rather than log(SEL), the latter referencing the regression in figure 6.1b, we would arrive at a revised change in exposure area of -34%, similar to the AEM result.

6.4. Estimated noise damages and ANCA benefits

Estimates for total and annual valuations are illustrated in figure 6.6 ($\widetilde{CV}_{\hat{C}_{dc}^{55}} \approx \widetilde{CV}_{\hat{C}_{dc}^{65}} = [0.19 : 0.21]$, $SE_{\hat{C}_{dc}^{55}} = [0.05 : 0.07]$, and $SE_{\hat{C}_{dc}^{65}} \approx 0.02$). The central estimate annuity in figure 6.6 is for a 30-year term at a 7% rate. Note that with respect to annual costs ($\tilde{\mu}_{\hat{C}_{dc}^{nois}} \approx \$1.3B$ through 2000, falling to \$1B by 2003), the range of term/rate scenarios is greater than the 10-90 percentile range of the distribution function for annual costs, indicating that the most important factor in determining annual cost is, expectedly, the rate and term of the depreciation.

Figure 6.6 Noise damage costs in the US 1991-2003



These estimates refer only to owner-occupied residential houses; direct impacts on renters or commercial property are not accounted. This suggests a downward bias in the damage estimates. In general, since renters have a shorter time horizon, valuation of noise changes are likely less than for owners of residential property. For renters under the case of no movement costs, the impact of noise change is zero, but if there are relocation costs, tenants are damaged. Similarly, quality of life issues may not matter so much for a commercial property and, thus, depreciation would be expected to be less than for owner-occupied residential housing. In a study of global aviation noise damages, Kish (2008) estimates that if rent depreciation is similar to the NDIs estimated for owner-occupied housing, and this depreciation is equivalent to capital loss for landlords, total noise damages would be a factor of ~1.4 higher than for owner-occupied housing alone.

Trends in capital loss (similar for total and annual valuations) mimic exposure area trends, but are mitigated by growth in housing units and changing house prices, more so over the period 1999-2003 than 1991-1999. This is evidenced by the comparison of correlations of capital loss and housing unit trends for these two periods, i.e. $\rho_{corr}\left(\hat{C}_{dc}^{nois}(t) : N_{house}^{l}(t)\right) = 0.97$ versus –0.003 and $\rho_{corr}\left(\hat{C}_{dc}^{nois}(t) : \bar{P}_{house}^{l}(t)\right) = 0.95$ versus –0.03. There is an estimated decrease in capital loss of \$0.23B from 1991-1999, all exposure areas summed, a change of -1.3%. The contributions of mean trends in $\hat{A}_{nois}$, $\bar{P}_{house}^{l}$, N_{house}^{l} account for $\bar{\delta}_{\hat{C}_{dc}^{nois}}$ = -6.6%, leaving a residual $\delta'_{\hat{C}_{dc}^{nois}}$ = -5.2% that accounts for the influence of propagated uncertainty. For the subsequent period 2000-2004, capital loss decreases by \$3.3B ($\delta_{\hat{C}_{dc}^{nois}}$ = -21%, $\bar{\delta}_{\hat{C}_{dc}^{nois}}$ = -18%, $\delta'_{\hat{C}_{dc}^{nois}}$ = -3.4%).

The US General Administration Office (GAO 2001) estimates abatement costs associated with the ANCA in the range \$4.0-5.2B, outweighing apparent benefits estimated in this study by a ratio of 17-23. With the generous assumption that capital gains over the period 2000-2004 can also be attributed to the phase-out of Stage 2 aircraft, the cost-benefit ratio remains positive at 1.2-1.6 (cf. Morrison et al. 1999).

In light of these results, further consideration of the ex post assessment of the Stage 2 noise phase-out rule provides an example of how a unified treatment of environmental impacts, as in MAIPA, can recommend a different set of regulatory goals. The noise analysis suggests that a \$0.23B benefit was derived from the retirement of older aircraft. As mentioned in chapter 3, correlations among trends in noise, VOC, and PMnv are high, suggesting that the phase-out reduced aircraft with lower combustion efficiency as well as noisy aircraft. If we include the air quality benefits of reducing VOC and PMnv, the phase-out benefit almost triples to \$0.64B, indicating that the majority of environmental quality improvement was from reductions in air pollution by a ratio of 2:1.

Distributions of marginal damage costs ($\mathrm{F}\left(\hat{c}_{mdc}^{nois}\Big|_T^j\right)$) and average damage costs ($\mathrm{F}\left(\hat{c}_{adc}^{nois}\Big|_T^j\right)$) are estimated using equation 6.7 for each representative aircraft type j in each year T on a per-operation basis as the set of evaluations for airports L for a change φ_j in the airport noise inventory. For $\hat{c}_{mdc}^{nois}$, this change is calculated as the fraction $\left(\hat{q}_n^j/\hat{Q}_n\right)\Big|_T^l$ and for $\hat{c}_{adc}^{nois}$, $\varphi_j=\left(\hat{Q}_n^j/\hat{Q}_n\right)\Big|_T^l$, equivalent to the logarithmic representative aircraft type contribution to the airport noise inventory.

$$
\begin{aligned}
&\mathrm{F}\left(\hat{c}_{adc}^{nois}\Big|_T^j\right)=\bigcup_{l=1}^{L}\left(\varphi_j\cdot\hat{C}_{dc}^{nois}\Big|_T^l\right) \text{ where: } \varphi_j=\left(\hat{q}_n^j/\hat{Q}_n\right)\Big|_T^l \\
&\mathrm{F}\left(\hat{c}_{adc}^{nois}\Big|_T^j\right)=\bigcup_{l=1}^{L}\left(\left(\varphi_j\cdot\hat{C}_{dc}^{nois}/N_{ops}^j\right)\Big|_T^l\right) \text{ where: } \varphi_j=\left(\hat{Q}_n^j/\hat{Q}_n\right)\Big|_T^l
\end{aligned}
\tag{6.7}
$$

Table 6.1 provides tabulated statistics ($SE_{\hat{c}_{mdc}^{nois}}\approx SE_{\hat{c}_{adc}^{nois}}<0.001$). Marginal costs are O(100) smaller than average costs.

Table 6.1. Marginal noise damage costs in the US 1991-2003

US commercial aircraft during the period 1991-2003 in 2003 $/operation. Values are averaged statistics summarizing an historical analysis of US commercial air transport activity during the period 1991 to 2003

REPACT	noise marginal damages $2003 / operation (55+)	noise marginal damages $2003 / operation (65+)
	median [interquartile range]	median [interquartile range]
b727	97 [56 190]	39 [23 77]
b737	4.5 [3.0 6.5]	1.8 [1.2 2.6]
dc9md80	12 [7.4 23]	4.7 [3.0 9.2]
b757	3.7 [2.3 5.4]	1.5 [0.9 2.2]
b767	11 [6.8 16]	4.3 [2.7 6.4]
dc10	18 [12 24]	7.0 [4.8 9.6]
b747o	35 [19 74]	14 [7.5 29]
dc9	28 [14 67]	11 [5.5 28]
b747	21 [14 35]	8.4 [5.7 14]
b737o	47 [25 99]	19 [10 40]
a320	4.3 [3.0 6.0]	1.7 [1.2 2.4]
l1011	17 [12 25]	6.9 [4.9 9.9]
md11	8.5 [6.2 12]	3.4 [2.4 4.7]
b777	6.7 [4.7 10]	2.7 [1.9 4.1]
b737n	5.4 [3.9 7.9]	2.2 [1.6 3.2]
b717	2.6 [1.8 3.7]	1.0 [0.71 1.5]
e145	0.78 [0.56 1.1]	0.31 [0.23 0.42]
tfan	2.0 [1.0 3.3]	0.80 [0.42 1.3]
tprp	0.56 [0.30 1.4]	0.22 [0.12 0.56]

Because of the shape of the damage curve, concave towards the x-axis as inferred from the logarithmic dependence, average costs overstate the value of a reduction in operations. Note that while marginal damage costs and housing capital are strongly correlated among airports ($\rho_{corr} \approx 1$), there is low correlation between total operations and housing capital ($\tilde{\rho}_{corr} \approx 0.15$); marginal costs are dependent on where the aircraft is flown and have little to do with the size of the origin or destination airports. Marginal costs rise from 1991-2003. This is because the marginal addition of a single flight to the base noise level is a function of the magnitude of the noise inventory, so changes have more impact as damages decline. This has to do with the shape of the damage curve where there is a declining marginal impact as noise level increases.

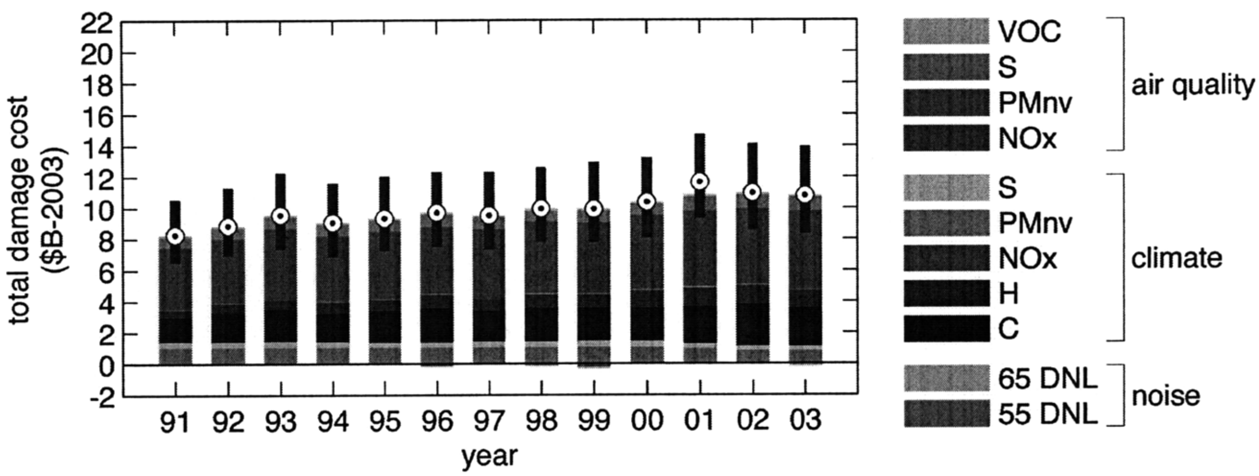
Uncertainty in marginal and average damage costs is two-dimensional. The variance sourced to the distribution across airports around the median shows coefficients $\widetilde{CV}_{\hat{c}^{nois}_{mdc}} \approx \widetilde{CV}_{\hat{c}^{nois}_{adc}} = [1.7 \;\; 1.9]$ over all representative aircraft type. Variance due to propagated uncertainties show coefficients $\widetilde{CV}_{\hat{c}^{nois}_{mulc}} \approx \widetilde{CV}_{\hat{c}^{nois}_{adc}} =$ [0.69 2.2] dominated by uncertainties in noise exposure area with an insignificant contribution from the estimated airport noise inventory (see comparison in figure 6.2). Thus, it is apparent from the analysis that uncertainty associated with noise contours controls the value of improved precision in marginal noise damages. Variance in marginal noise damages is additionally affected by uncertainty in the fraction of an airport's noise inventory attributable to the representative aircraft type; its magnitude relative to the contribution of variance in noise exposure area is thus a function of location, generally accounting for about half of the coefficient of variation.

6.5. Comparison of noise and emissions damage estimates

To close this chapter, we consider the place of noise mitigation efforts in the context of reducing air transport environmental impacts. Figure 6.7 plots historical trends in annual damage costs, decomposed by impact agent (e.g. NO_x versus SOx) and grouped by impact vector (air quality, climate, and noise). In figure 6.7, climate damages reference a discount rate of 3%, and for noise a discount rate of 7% annualizes capital losses over a 30-year term. Figure 6.7 shows estimated annual damages of ~$12-15B per year, increasing from 1991-2003; in perspective, this social cost equates to ~0.025% of US GDP, or ~13% of mean annual operating revenues, and exceeds the sum industry profit over the period 1991-2003.

Figure 6.7. Damages from U.S. commercial aircraft operations 1991-2003

All monetary values are given in 2003 dollars. The nominal scenario reports: (1) climate damages assuming a 3% discount rate and IS92 growth; and (2) annualized noise damages at a 7% rate over 30 years, 1991-2003.



The historical analysis outlines a challenge in obtaining future benefits from noise reduction:

- The largest portion of welfare losses result from air quality impacts. In any given year, NO_x, SOx, and CO2 emissions compose ~85% of total damages. Noise is a relatively small and declining portion of total estimated damages, accounting for less than 10% in 1991, and a declining contribution through 2003.

- The trends suggested by the historical analysis show emissions impacts are growing while those due to noise are declining. Yearly trends indicate that climate impacts are the fastest growing vector and, thus, CO2 is the fastest growing impact agent. Estimated median annual climate damages increase at a compounded annual growth rate of 4.1%, 4 times the estimated growth in fuel consumption (0.95%). Comparatively, air quality damages increased at 1.8%, and noise damages declined at -1.3%.

- In the context of cost-benefit analysis, comparative marginal damage results suggest a much stricter efficacy constraint on the cost of noise abatement options. In comparison to the marginal damages of NO_x, SOx, HC, PMnv, and CO2 calculated on a per-flight basis, noise marginal damages are O(1e3)-O(1e6) smaller.

- Though noise, air quality, and climate impact vectors trend in directions of increasing marginal damage costs, the rate of increase for emissions vectors is faster than for noise over the historical period, indicating that the efficacy constraints imposed by marginal damages will not alleviate in the near-term.

- As the ANCA reassessment presented earlier in this chapter indicates, technological approaches to noise mitigation may be much less efficient than conventional wisdom suggests.

Considering that: (1) an estimated $10 in damages are sourced to emissions for every $1 sourced to noise from commercial operations in the US; (2) damages from both climate and air quality impacts are driven primarily by the activities of sources outside the air transport industry; (3) further reductions in noise are expected through retirement of Stage 3 aircraft, both through economic and regulatory incentives; and (4) noise and air quality continue to be addressed through regulatory standards while an approach to controlling climate impacts has not been established, we can speculate that a significant realignment in resources away from noise reduction and toward emissions mitigation is on the horizon. Decisions like the halting of Qantas flight procedure tests discussed in chapter 1 may favor the fuel efficiency objective, and if tradeoffs are made, ground may be lost in reducing noise impacts.

With this in mind, the results in this chapter recommend a thoughtful reconsideration of the steps best suited to reduce noise impacts while attending to a growing environmental stress from emissions. In particular, these findings suggest the importance of a reexamination of the fundamental mechanisms of how people value reductions in environmental noise, with the goal of expanding opportunities to address aircraft noise impacts. The next section discusses this approach from the perspective of structural uncertainty in the noise model presented previously.

6.6. Structural uncertainties in noise damage estimates

We turn to the question of accuracy in the noise damage estimates by examining the results in the context of conclusions using other valuation techniques; in particular, we ask whether damages are fully captured by differences among house prices. The emerging literature on this subject suggests that hedonic estimates are a lower bound. The full welfare impact of noise (or emissions) changes consists of: (1) out-of-pocket costs and opportunity costs (e.g. depreciation in property values and relocation costs for households that move), which value use attributes; and (2) welfare changes due to utility impacts (including the loss of place-specific surplus for households that move, cf. Walters (1975)). We would expect annoyance to impact the last of this list, which is where revealed preference approaches to noise valuation may fall short.

6.6.1. Caution on use of surrogates

It is useful to compare these annual values to commonly cited noise costs, which are sometimes proposed as surrogates for the estimation of damages but often confuse external and internal costs. In particular, while costs to market participants should be accounted in a benefit-cost analysis, they are not external to current markets. For example, the costs of capacity limitations are sometimes used as surrogates for noise damage costs. Congestion costs estimated by DRI-WEFA (2002), which include induced (macroeconomic) costs related to the propagated economic effects of airline growth constraints, total \$9.9B (17% induced); however, these congestion costs are reflected in ticket prices insofar as they limit supply.

Additional surrogates can be proposed that are similarly accounted for within the market, such as payment of noise landing fees or payment of taxes that are then redistributed for noise mitigation purposes, but none of these are necessarily measures of welfare. A tally of government outlays for noise mitigation is essentially a non-preference method of valuing noise in that it uses actual expenditures as a substitute for determining, via observation or elicitation, resident preferences for quiet. Based on annual compensations to near-airport communities in the United States (ref AIP and PFC funding streams), this method of valuation determines a \$0.27-0.53B annual social cost. However, government outlays are tied to noise only in the manner in which they are distributed (e.g. for households in the DNL 65 contour), not in the way they are collected. Since they are politically determined, they could be either higher or lower than actual damages; these compensations amount to about half of the property capital loss estimated via MAIPA.[4]

6.6.2. Capturing noise damages through property values

Transaction costs are high in the housing market. Bartik (1988) argues that the marginal damage curve derived from second-stage analyses ignores important adjustment costs and thus underestimates benefits. Evaluation of residual costs examines the effects of relaxing assumptions in the hedonic price method, such as non-equilibrium effects, which can increase noise damages. Estimating residual noise shadow costs using happiness measures for communities around Amsterdam Schipol, Van Praag and Baarsma (2001) finds that house prices are not dependent on noise, such that the residual cost accounts for the welfare loss, a result attributed to disequilibrium in the housing market. The residual shadow cost is a declining portion of income as noise level rises (cf. Van Praag and Baarsma 2004). This indication of decreasing marginal costs with level of noise disturbance coincides with a similar conclusion in IWW/ Infras (2000) who, in reviewing the broader noise valuation literature, survey marginal costs lower than average costs (30-60% of average costs).

6.6.3. Noise damages estimated using contingent valuation techniques

Stated preference valuation techniques, such as contingent valuation, recognize the importance of quality of life issues in determining willingness-to-pay for environmental amenities (e.g. non-use attributes) and hold promise for capturing the total value of a change in environmental quality. However, these techniques are underdeveloped in application to noise and the distinction between the physical and

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[4] This is perhaps most correctly compared to an averting behavior approach to welfare evaluation, where a good that is complementary to a particular impact (such as soundproofing to noise) is identified and demand for the environmental quality of interest inferred from its consumption. Averting behavior expenditures are underestimates of willingness to pay for pollution reduction because they do not account for changes in personal utility as a result of the expenditure (i.e. there is some extra monetary compensation that would be required to restore them to their original point before the expenditure was necessary, a supplement to income). Kolstad (2000) gives a useful example related to noise that differentiates between an expenditure as an averting behavior and one as a resultant cost. As Kolstad (2000) states, "by observing expenditures on the complementary private good, we obtain a lower bound on the value of the environmental good or bad. The outside noise must cause at least as much damage as expenditures on soundproofing." There is a difference between the types of expenditures that fall into the category of travel cost or averting behavior and the category of non-preference valuations that result from a political process. For example, expenditures for soundproofing of homes is an averting cost behavior when a homeowner pays out of his own pocket. However, when the government allocates a certain amount for soundproofing of homes there is often little evidence to support connection to a welfare measure. If soundproofing or other such expenditures were categorized as an averting cost behavior, then the amounts will typically understate the true WTP for noise reduction (Kolstad 2000). Averting behavior methods have been applied to the valuation of noise reduction, but underestimate the true costs since the expenditure surrogates they employ do not represent a removal of the noise. The origins of such applications can be traced to Pearce (1991), who used owner expenditures on soundproofing to determine a cost estimate for neighborhoods around Heathrow Airport.

psychological effects of environmental change remains vague. Whereas for an increase in noise the compensating variation would be higher than the rent loss (in the economic sense), the opposite is true of a decrease in noise (Brookshire et al. 1982)(Feitelson et al. 1996). This is a subtle observation; stated preference methods estimate use and non-use values for improvements in environmental quality, but comparison with actual damages depends on the direction of quality change.

Another perspective considers that prices may be increasing or decreasing over time; in the former case, the hedonic estimate of the amenity value is an underestimate when a deterioration in environmental quality is contemplated since valuation will tend to include some time delay in adjustment. A contingent valuation methodology conducted in Tel Aviv (Feitelson et al. 1996) estimated that damages calculated using the hedonic price method may underestimate WTP to maintain quiet by a factor ~5.[5] This issue is unsettled, as broader analyses have found mixed results for transportation noise (cf. Nelson 2007 see Kish) and lower values from stated preference studies for broader applications among environmental goods (Carson et al. 1996, see Kish). (cf. chapter 5 for further discussion of contingent valuation approach in the context of air quality.)

6.6.4. Presence of wage-rent trades

The use of housing prices as a basis for valuation relies on the assumption that wage-rent trades, such that increases in pollution are reflected in land prices as well as wages, do not exist locally. Recall that the use of an hedonic price estimate of marginal damage cost assumes that disutility arising from extra noise (or lowered air quality) is reflected in property valuations as opposed to wages, implying that pollution is not a productive quantity for the airport. A group of studies applying alternative methods indicate that this may be a real difficulty in measurement. Feitelson et al. (1996) finds that WTP declines with noise level, suggesting this is because the number of people willing to buy a home decreases with noise level, thus

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[5] The accuracy of an HP result as an estimate of welfare gain requires that no compensation has been provided to resident (e.g. soundproofing). If compensation has been provided, method would tend to overestimate welfare loss due to property value depreciation to the homeowner, but still relates to a potential economic gain resulting from a reduction in noise for society as a whole. As a matter of determining social cost, the concern that the valuation estimate does not properly account for compensation provided to residents is related to equity issues, not efficiency. A program instituted to reduce noise will still result in appreciation of housing values and additional utility to residents, regardless of whether they have been compensated or not. There is net economic gain to be had, provided a sufficiently low-cost program can be designed. Thus, the question of compensation should not come into play when evaluating a decision to implement a noise reduction program, but when considering questions of equity. Compensation is a transfer of economic flows.

selecting for more noise tolerant residents rather than an explicit income effect. However, these trends are connected. For example, Palmquist finds income elasticities of willingness to pay for quiet of between 1.5-1.6, which indicate a luxury good such that higher incomes will pay disproportionately more for quiet.

Three studies have examined whether hedonic estimates are statistically similar, given differences in locational characteristics, functional model form, and time of estimation (Nelson 2003, Shipper et al. 1998, Johnson and Button 1997). Shipper et al. (1998) reports a meta-analysis for hedonic studies for airports in the US and Canada that finds an important explanatory variable is the relative mean sample house price, equal to the reported mean property value divided by the per capita income; this is a measure of real wages adjusted for housing costs (Nordhaus 1996, Nelson 2003). The coefficient on this variable is positive, suggesting that as wages increase, people will tend to increase expenditures on noise. Further, with a negative constant in the regression, it indicates that at some low income level, willingness to pay would drop to zero. Thus, there appears to be an equity issue with the selection of low income residents for high noise areas and subsequent depression of WTP.

6.6.5. Developing a more accurate picture of endpoints of noise impact

In sum, alternative approaches to estimating noise damages suggest that with the influence of annoyance trends as included in the hedonic price function (marginal annoyance and other noise measures, such as awakenings, increase with DNL exponentially), hedonic price studies may not capture the full welfare loss (disutility not captured). And in the absence of noise, the marginal damage curve may shift as the market for homes near airports expands (higher income residents enter), leading to changes on a macroeconomic level. Consider also that preferences cannot be expressed if people do not have the relevant information; Pope (2007) finds that providing information about noise to potential buyers reduced prices by an additional ~35% compared to a hedonic estimate without disclosure for Ldn of 60-70 dB areas around Raleigh-Durham. At the moment, HP studies are the available economic benchmark for this study, but their assumptions as they apply to locations in the United States are yet to be thoroughly tested. Asking the question of whether we are fully addressing the sources of noise damages may usefully expand our options for reducing these impacts.

7. Summary and conclusions

Planning technology investments to reduce the environmental impacts of aircraft operations has become increasingly complex, particularly due to increased emphasis on the multi-criteria objectives of an integrated environmental policy. The objectives of this thesis were to evaluate metrics to compare the influence of aircraft performance characteristics on environmental change and methods to incorporate integrated analysis in assessment tools. This work employs estimates of the environmental damages of emissions and noise to compare impacts of changes in climate, air quality, and community noise.

This chapter first summarizes the five primary contributions discussed in chapters 2–6. It then presents estimates for environmental trade-offs implicit in policy or design decisions for the US commercial fleet from 1991–2003. The results presented in this chapter reference a discount rate of 3% for climate damage estimates, and a discount rate of 7% to annualize noise-related capital losses over a 30-year term, with damages given in 2003-$. Given the retrospective nature of the analysis and the changing environmental costs, the estimates are not applicable to future scenarios, but they are illustrative of the kinds of analyses that should be considered for future decision-making.

7.1. Integrated approach to air transport environmental impact assessment

Developed an integrated assessment approach to prototype the FAA Aviation-environmental Portfolio Management Tool (APMT).

A probabilistic multi-attribute impact pathway analysis (MAIPA) was developed to model climate, air quality, and community noise damages. MAIPA was developed as a prototype assessment capability in a pathfinding effort that identified requirements for and contributed feasibility demonstrations to APMT. APMT is currently under development for application to regulatory decision-making in the United States.

MAIPA was used to make an historical assessment of the environmental impacts of commercial aircraft operations in the United States between 1991–2003. These impacts are characterized by inventory, environmental, risk, and economic metrics, and, ultimately, estimates of environmental damages in terms of aircraft performance parameters. The resolution of results is national and yearly, with the exception of inventories, where results are quarterly.

7.1.1. Fidelity

MAIPA models operational data, technological parameterizations, and environmental and socioeconomic conditions. In this thesis, historical performance and operational data limit scope to US commercial aircraft operations. All inputs are specified probabilistically and uncertainties are propagated using Monte Carlo methods. The analysis adheres to the observational and theoretical fidelity of the data and knowledge underlying the estimation of damages.

7.1.2. Inventories

Emissions, noise, and fuel consumption inventories were estimated using a representation of the technological and operational features of US commercial aircraft in-service between 1991–2003, with parametric inputs defined by open-source data for 19 representative aircraft types. Quarterly estimates are provided for 10 inventories: fuel consumption, gaseous emissions CO_2, H_2O, SO_x, NO_x, HC (as VOC), CO, noise (as SEL dBA), and PM (nonvolatile and volatile). This set of inventories is the first detailed characterization of pollution trends for US commercial aircraft operations for the period studied.

7.1.3. Environmental models

The scope and resolution of parametric inputs, and the goal of providing timely decision analyses, encouraged the development of reduced-order inventory and environmental models for MAIPA. Consistent with the analysis scope and resolution, this study employs models of environmental change that are reduced-order evaluations of the physical processes contained in the higher-fidelity models typically used in the regulatory context. Order reductions are possible because air transport impact agents are either, (1) a small contribution compared to other sources (emissions), or (2) the dominant environmental perturbation of its type (noise).

Models address change in climate, air quality, and noise. For climate, we use an impulse-response approach to calculate probabilistic estimates of marginal, present-value climate change metrics inclusive of radiatively-active species with different atmospheric lifetimes (cf. Joos et al. 1996). The model has been implemented in APMT. For air quality, we use a parametric approach to estimate changes in the atmospheric concentrations of the criteria pollutants (NO_2, CO, SO_2, O_3, and PM2.5). Changes in pollutant exposure patterns are established using scaling arguments that draw on measurement data to represent geographic patterns as locational variability. For noise, MAIPA uses a statistical model to estimate noise-exposed populations, based on a relationship between exposure areas (DNL contours) and airport noise inventories. MAIPA environmental models capture large-scale uncertainties, interdependencies, and dynamics that collectively provide an integrated characterization of impacts mechanisms and a description of limitations in application to policy development.

7.2. Particulate matter impacts of aircraft emissions

Introduced a treatment of air transport particulate matter emissions, environmental fate, and health impacts of particulate matter.

MAIPA shifts focus from pollution generation to the consequent risks, as contrasted with current assessment mechanisms. One important addition was to extend the scope of impact assessment with a comprehensive treatment of particulate matter emissions. To estimate particulate matter impacts, it was necessary address the absence of data and methods to construct PM inventories in mass units, the basis for epidemiological correlations with disease and mortality incidence as well as climate model estimates of radiative forcing. Estimated particulate emissions indices were applied to evaluate the first mass-based PM inventories specific to the operational performance of US commercial aircraft. This work identifies precursor emissions (NO_x, SOx, and HC) as primary sources of environmental damages through their impact on ambient concentrations of PM2.5 and resultant mortality risks.

7.3. Uncertainties in impact assessments

Identified that the major source of reducible uncertainty in emissions damages stems from the assumed extent of ozone and particulate matter production in the engine exhaust plume.

The role of the engine exhaust plume chemistry and microphysics was found to be a primary uncertainty in estimating the change in ambient ozone and particulate matter concentrations due to aircraft operations. This uncertainty derives from our limited understanding of how engine exhaust plumes evolve in the lower troposphere and interact with other emissions sources.

Currently, large scale complex air quality models such as the EPA CMAQ do not include plume production of ozone and volatile PM. Recent analyses using CMAQ, including an air transport study mandated by the 2007 Energy Policy Act, report <1% of estimated mortalities are from changes in ambient ozone concentrations. Changes in ambient PM2.5 account for all other mortalities.

The air quality model developed for MAIPA specifies an uncertain ozone productivity that is a factor of 0.7 to 5 times the ozone productivity of the CMAQ analyses. Assuming a similar difference in PM precursor production, MAIPA-estimated damages using CMAQ assumptions are reduced by approximately 60%, equivalent to one-third of the sum air quality, climate, and noise damages. This is the largest reducible uncertainty in estimated damages.

7.3.1. Reducing uncertainties

Analyses conducted for each of climate, air quality, and noise impacts uniformly highlight that parametric uncertainty in physical inputs and uncertainties in societal preferences for environmental quality lead to uncertainties in damage estimates that are similar in magnitude. For example, the analysis suggests that managing the climate impacts of aircraft emissions is as much dependent on normative decisions underlying the specification of intergenerational wealth distribution (e.g. discount rate) as on the (a) scientific understanding of carbon-cycle and climate processes, or (b) propagated parametric uncertainties. For air quality, the value of a statistical life has the largest estimated influence on mean-shift. For noise, the uncertainty in specifying the rate and term assumed to annualize damages is of the same scale as propagated uncertainty in the specification of exposure area, the dominant component of parametric uncertainty in noise damages. In this context, improved technical capabilities have a limited role in reducing uncertainties. Potential biases due to uncertainties in environmental physics, the physiologic responses and socioeconomic conditions that regulate health and livelihood risks, and the preferences of societies are the largest determinants of the magnitude and change in air transport environmental impacts.

7.4. Damages due to US aircraft operations from 1991–2003

The historical analysis suggests that, together, increases in ambient PM2.5 concentrations and surface temperature resulting from aircraft emissions were the primary components of environmental damages during 1991–2003. In any given year from 1991–2003, emissions of NO_x, SO_x, CO_2, and HC together account for ~90% of median annual damages. Of these, only hydrocarbon inventories showed a decline. Noise accounts for the remaining 10% with a declining share over the historical period.

The efficacy of climate mitigation options depends on knowing the value of reducing CO_2 emissions versus non-CO_2 emissions in the context of climate change. To do this, the assessment must account for different timescales among types of perturbation to the atmosphere to distinguish between the longer-lived direct impacts of CO_2 emissions and the indirect impacts of other short-lived microphysical and chemical processes, such as the production of ozone or the decrease in methane residence time associated with the emission of NO_x.

The analysis indicates CO_2 and H_2O emissions are the primary source factors controlling climate damages. Results show CO_2 accounts for ~90% of annual climate costs at a 3% discount rate. At a 7% rate, non-CO_2 contributions increase to ~60% of damages with an increased emphasis on short-lived atmospheric effects; over 95% is accounted by increased cloudiness, attributed to H_2O in this study.

Where results indicate that primary combustion products (CO_2 and H_2O) are the predominant agents of climate damages, the estimated damages due to secondary combustion products (SO_x, NO_x, VOC, and PM) are primarily through changes in air quality. While secondary products are also involved in climate effects, their air quality impacts are estimated to be 2–3 orders of magnitude larger than their contributions through climate vectors.

7.4.1. Factors affecting changes in damages with time

Identified that the most important factor determining changes in damages over time is the dependence of impacts on the background environmental sensitivity.

The results of this study show that emissions reductions influence neither the magnitudes nor trends in marginal damages because emissions impacts of US commercial aircraft are dictated by the progress in controlling emissions from other sources. The attribution of trends to parametric inputs shows that air transport emissions impacts are predominantly determined by the background environmental sensitivity. Results show estimated changes in environmental metrics of ~1% due to commercial aircraft emissions. Environmental sensitivities to emissions inputs in any one year are altered by an amount insignificant compared to the change in responses over time. As a result, the economic and regulatory factors controlling non-aircraft sources dictate marginal costs.

The explanation of trends in emissions damages provides another important finding. From 1991–2003, estimated annual climate damages grew at a median rate of 4% per year, twice the rate of annual air quality damages and four times that of fuel consumption. These observations are explained by the dependence of sum damages on the relative growth rates of air transport versus other source emissions. Air transport emissions affecting both climate and air quality grow faster than non-aircraft sources, resulting in growth in damages disproportionate to fuel consumption. Higher functional sensitivity on the difference in growth rates accounts for the faster growth in climate as compared to air quality damages.

7.4.2. Policy choices

Because environmental interactions among CO_2/H_2O, the group of secondary products, and noise are at most second-order to their respective damages, trade-offs are a direct function of interdependencies in the aircraft system design. As a result, policy choices take on a broader context, involving trade-offs between the large-scale performance of the air transport system toward mitigating global, long-term impacts (e.g. climate), versus the smaller-scale, component-level metrics traditionally used to address short-lived (i.e. air quality and noise) environmental impacts.

As CO_2 and H_2O emissions are proxies for fuel consumption, these results are strongly suggestive of the long-term value of improvements in fuel efficiency and reductions in carbon intensity (relative to per-unit energy). This does not imply that there is no value in mitigating short-lived effects, only that such action exerts limited leverage against long-term climate change. Unlike noise impacts, the marginal costs of air transport emissions impacts cannot be completely controlled through endogenous technological or operational change. Without the ability to control the growth of marginal emissions damage costs, decisions may be best served by identifying the most efficient combination of mitigation options. For example, some control over cloud effects may be had in the near-term with the present fleet by altering flight paths to avoid adverse weather conditions. The immediate benefits so obtained might justify the costs.

7.5. Integrated approach to impact mitigation

Reassessed the environmental benefits of the aircraft retirements mandated by the 1990 Aircraft Noise and Capacity Act.

From December 31, 1994 through December 31, 1999, FAA mandated a scheduled phase-out of portions of the commercial fleet identified by their failure to meet a limit on noise levels (14CFR91.801–877 Subpart I: Operating noise limits). A reassessment of the environmental benefits derived from that mandated phase-out of noisy aircraft during the 1990's has been conducted, showing a different result. Previous studies estimated a ~80% reduction in population exposure. In contrast, the reassessment estimates a ~2% reduction, providing benefits 17–20 times lower than published estimates of abatement costs. The difference is due to accounting for trends in both air quality and noise damages from 1991–2000 indicates emissions mitigation benefits of Stage 2 phase-outs exceeded benefits from noise reduction by a ratio of 2:1.

7.6. Trade-offs

Quantified the environmental trade-offs in decisions specifying aircraft performance for the technology in the US commercial fleet from 1991–2003.

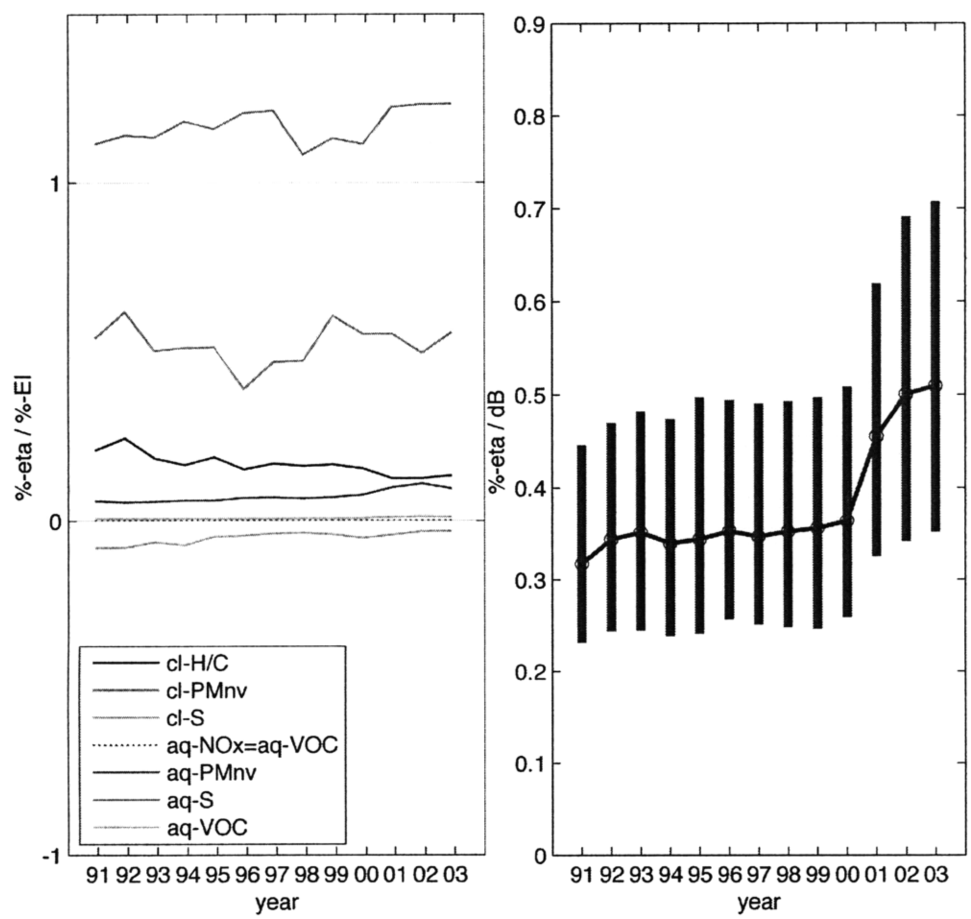
There are many trade-offs weighed during design of a new aircraft. Looking forward to the next generation of aircraft designs, the central environmental dilemma is in determining an effective balance in technology development that achieves benefits in environmental quality comprehensive of all impact vectors. In short, we desire a clear design objective.

The traditional objectives of design toward regulatory standards are marked minimum NO_x or minimum noise. A more comprehensive perspective recognizes that different sets of environmental performance characteristics can provide equivalent levels of welfare. This thesis estimates a damage function to describe sensitivities to performance changes in the US commercial fleet from 1991–2003. The damage function suggests a set of equivalencies that define a damage-neutral change to the air transport system affecting any of the fuel consumption, fuel composition, emissions, and noise characteristics of the commercial fleet. Tables 4.1, 5.5, and 6.1 show medians and interquartiles for the 28 marginal damage cost estimators as evaluated through MAIPA; emissions marginal damage costs are provided in the denominations used for the damage equation $/EI.

7.6.1. Elasticities of performance

Trade-offs are specified with comparisons among estimated marginal damage costs of emissions indices, per-flight noise level, and fuel consumption. Figure 7.1(a) shows the trade-offs among fuel consumption and emissions as elasticities—the percent change in fuel consumption that is equivalent to a percent change in the emissions index. Figure 1.1(b) similarly plots the change in efficiency equivalent to a 1 dBA SEL change noise. Damage-neutral trade-offs between noise and fuel consumption vary by a factor of 10 dependent on the policy definition of quiet—0.35 dBA SEL per 1%-η referencing the 55+ DNL contour exposure area versus 0.038 dBA SEL per 1%-η referencing 65+ DNL exposure.

Figure 7.1. Fuel-emissions and fuel-noise damage elasticities



Trade-offs quantified in this thesis are a set of probabilistic constraints on the direction of technological change intended to increase expected benefits. Trade-offs help focus opportunities for research, design, or policy to reduce these constraints. One way to avoid trade-off complexities is to identify options that bundle benefits. As example, nonvolatile PM inventories show the sharpest decline among all emissions inventories, the result of a -1.6% annual rate of change in the estimated nonvolatile PM emissions index from 1991–2003. The cause of this decline was found to be the retirements of aircraft through the phase-out mandates of the Airport Noise and Capacity Act of 1990. The technological correlation between noise, and nonvolatile PM and VOC emissions that underlies this finding still exists in the fleet today; bundling mitigation options into a portfolio-based policy can take advantage of these characteristics.

7.7. Next steps

Environmental law in the United States provides for a regulatory process designed to minimize uncertainty in the objectives of these investments. To accomplish this, rule-making is coordinated with developmental milestones to establish expectations for achievable emissions and noise mitigations. Standards are negotiated periodically in reference to these plans, favoring stringency options that provide the largest reduction in fleet emissions or noise per dollar technology cost. These procedures reduce the risk that investments will fail to achieve compliance with standards. They are successful in this respect because the requirements are controlled. Trade-offs challenge the certainty provided by this process.

Balanced technology goals will be realized by negotiating among mitigation objectives that are established from a range of perspectives. Environmental, as well as infrastructure, safety, security, and global technology leadership objectives place demands on available resources for air transport development. The traditional regulatory process is not equipped to decide the proper balance for planning. New technology development has found ways to make progress, but breaking these trade-offs remains a stubborn impediment. There has been little attempt to evaluate environmental performance characteristics specified by trade-offs. Working from an understanding of trade-offs, one branch of future work is to characterize policy designs that best encourage improvements in overall environmental quality. Findings in this thesis also emphasize the importance of understanding how such policies can be made most robust to uncertainties that are only partially reducible, and what such methods recommend for technical investigations.

In this context, it is important to continue to ask the questions: (1) with what fidelity can we provide assessment conclusions based on existing scientific understanding?; and (2) what capabilities are important to expanding the contributions of assessments to decision-making? The highest priority recommendations for future work are: (1) improving the characterization of air quality impacts, particularly to clarify the role of exhaust plume production of precursors to PM2.5 and ozone, and the apportionment of PM2.5 production among precursors; (2) evaluating the importance of extending assessment capabilities to capture spatially-variable impacts, such as heterogeneity in the distribution of radiative forcings, e.g. extend climate impulse response $g(t)$ to $g(x,t)$; and (3) determining the scale and locations of macroeconomic shifts that may result in the presence of air transport environmental damages.

There are many possible approaches to mitigation. For example: (1) attending to the ozone versus PM2.5 impacts of NO_x; versus (2) decisions to control NO_x, SO_x, HC, or PM_{nv} emissions to reduce air quality impacts; versus (3) the broader objectives for reductions in climate and noise as well as air quality; versus (4) decisions that allocate resources towards emissions versus noise issues. National policy acts at the broader end of this spectrum, while technology investment decisions are served by a higher resolution specification of design requirements. This thesis demonstrates a means to achieve consistency across these different levels of decision-making. There remains, however, the need evaluate constraints imposed by policy-making, institutional or otherwise, in order to determine where increased resolution of impacts would be most useful for decision-making.

Appendix

1. Computation and analysis conventions

This appendix details the approaches to the specification of parametric, scenario, and structural uncertainties, the computational implementation of uncertainty propagation, and the validation of results where measurements may or may not be available.

All quantitative results are presented with two (2) significant digits
cf. Notation (at beginning of the document)

The Notation section at the beginning of this document defines symbolisms for ranges, intervals, and sets, as well as all statistical parameters used in the thesis.

1.1. Specification of probability and distribution functions

To address parametric uncertainty and variability, each input variable is represented by a probability or distribution function to describe the likely assignment of values. The important qualification for these specifications is representativeness, meaning that we want to specify input distributions faithful to the manner in which uncertainty (i.e. incomplete understanding, reducible) and variability (i.e. true heterogeneity, not reducible) arise, with consideration of physical limits on parameter values (EPA 1997, 1999b).

The functional forms specified for input parameters are described throughout the text as they arise in the discussion. All of these specifications following a common set of guidelines:

Little or no information about originating processes

- Where no phenomenological information is available and where data are few, a uniform or triangular probability function suffices, the latter in cases where a likely central value can be identified.

Parametric functions

- Parametric probability functions apply where variance results from physical phenomena (e.g. normal probability functions for additive process, lognormal functions for multiplicative process, etc.).
- Physical limits can restrict variable specification to a known range of values, for which alternative

parametric functions can apply. However, such applications are limited; often we understand a process, such as random noise, but application of an unconstrained distribution in the context of a simulation will result in non-physical values.

- A decision has to be made whether to truncate or to apply a flexible parametric function that can approximate with limits on tail extremes. This is a case-specific decision and is noted as applicable.

Nonparametric functions

- Nonparametric functions are employed where data are available to characterize variability or where uncertainty results from extensive convolution of underlying physical processes.
- Statistical models (e.g. regression) are in some places employed as an alternative approach to representing trend and variability.
- Complex social processes are the primary area where goodness-of-fit parameter estimates provide a model for describing the data.

1.2. Simulation technique

With simulation, it is possible to address uncertainties through both linear and nonlinear models in a nonparametric fashion, if required. To determine the relationships between inputs and output, fewer assumptions are made about the nature of the random variables than with analytical methods. Input distributions can be of mixed character, which is a requirement for the impact pathway analysis.

Parametric uncertainties and variability propagate through the impact pathway via Monte Carlo simulation, a technique using statistical sampling to determine the distribution of model realizations resulting from random inputs.[1] A sampling procedure is chosen and exercised to select, for successive iterations, a set of inputs to run through the model. In the context of simulation, pseudo-random number generators set an input matrix drawn from object probability or distribution functions. Several iterations are accomplished using the pseudo-random draw, generating a probability distribution for the model output.

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[1] Relative to analytical methods, Monte Carlo simulation is advantageous in evaluating complex systems such as the impact pathway. Analytic methods typically require linearity assumptions to handle the propagation of uncertainties. It may be not be possible to validate such an approximation beyond a small range around a particular result. Analytical methods also require that the underlying model be differentiable near the point of interest.

For this analysis, the extent of convergence is characterized by the standard error of the mean (SE) of the output. The rate of approach is proportional to $1/\sqrt{n}$.[2]

1.3. Assumptions of Monte Carlo simulation

Assumptions underlie the Monte Carlo technique that are similar to those of linear regression. First, probability and distribution functions specified for input variables are assumed to faithfully represent the generating process. Second, inputs are assumed independent.

Strict enforcement of the independence assumption is limited by the quality of data used to specify the inputs. For example, in specifying aircraft performance, thrust and speed settings are governed and connected by a set of control rules. However, without full knowledge of those control rules, the manner of dependence cannot be specified.

In their absence, the assumption of independence is made, which introduces additional variability due to the presence of unlikely or impossible combinations of inputs. The presence of such multicollinearity is the most consequential in assessing the relationship between independent and dependent variables using either a conditional or linear technique for inferring mean-shift and variance contributions (cf. section A1.5 et seq.).

1.4. Choice of statistical descriptions of results

With a mixture of functional forms used to specify input parameters and other characterizations of variability, the sample mean and standard deviation are poor characterizations of central tendency and spread, primarily because all output parameter distributions are skewed and nonparametric. In this thesis, the median and interquartile range are the preferred statistical characterizations. The exception is in comparisons to data reported without statistical characterization or comparisons to deterministic model results. In such cases, there is an underlying arithmetic average inherent or assumed; the mean and standard deviation are more appropriate bases for comparison.

[2] More efficient sampling procedures can be implemented to quicken the rate of convergence; for example, Latin hypercube sampling.

1.5. Estimation of variance and mean-shift contributions

An important benefit of the distributional output is that we can find where reducing parametric uncertainties provides the most benefit towards reducing decision risk. Uncertainty in some inputs may be more costly to minimize than in others. Simulation allows the relative efficiencies of input variance reduction to be ranked to a degree limited by convergence error.

1.5.1. Difficulties with conditional expectation approach

Contributions to variance and mean-shift in output variables are most thoroughly explored using a conditional expectation approach. In a simulation framework, this involves observing the change in an output probability function resulting from restrictions on the uncertainty in particular inputs in random sampling. With this information, the influence of an input can be mapped over a range of model computations to determine impact on the output.

The conditional expectation approach is a large computational burden; to be complete, all conditional variations need to be evaluated, but only a finite number of simulations for a selected group of variables is practicable. However, this is largely unnecessary in MAIPA, particularly with respect to damage estimates. There is often one primary influence and in those cases where there are several, their effect on marginal damage estimates is washed out by the introduction of downstream uncertainty and variability.

1.5.2. Application of linear regression to identify important sources

Given the number of independent variables in the analysis and attendant computational limitations, MAIPA uses linear regression to assess sources of variance $\mathrm{var}\langle y \mid x \rangle$ and mean-shift $\Delta\mu\langle y \rangle$ in key output metrics. The mean-shift due to a change in xi is $\Delta\mu\langle y|x_i\rangle = \beta_x \cdot \mu_x / \mu_y$ where the beta parameters are estimated using the regression coefficients (see footnote 3 for derivation). The ability of the linearized model to capture these relationships is assessed via goodness-of-fit statistics. Confidence intervals are specified for contributions to variance and mean-shift to show uncertainties in rank.

1.6. Conventions for estimating errors and differences

Quantification of the distance between compared results (ε) are calculated relative to the reference datum or model result with the general definition as given in equation A1.1.

$$\text{(A1.1)} \qquad \text{`} \ldots \varepsilon_{q_f} = \left(\hat{\mu}_{q_f} - \mu_{q_f}^{\mathrm{F41}}\right) \Big/ \mu_{q_f}^{\mathrm{F41}} \ldots$$

Equation A1.1 translates as, '... the mean estimated per-flight fuel consumption differs from the per-flight fuel consumption calculated using U.S. DOT Form 41 reported fuel use data by $(\varepsilon \cdot 100)\%$ ....'

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[3] Referring to equations A1.3, assuming random inputs xi are independent and the output y is well behaved in the area of $y = f(x_i)$, we take a Taylor series approximation for the function $y = f(x_i)$ around means μi of the random variables xi. Keeping only the first-order terms, with the residual error less than or equal to the sum of the next higher-order terms, and using the definition of variance, a linearized model results.

$$\text{(A1.3)} \qquad \begin{aligned} y - \mu_y &= \sum_i \frac{\partial y}{\partial x_i}(x_i - \mu_i) \\ \sigma_y^2 &= \int (y - \mu_y)^2 \, p(y) \, dy \\ \sigma_y^2 &= \int \sum_i \left(\frac{\partial y}{\partial x_i}\right)^2 (x_i - \mu_i)^2 \, p(y) \, dy + \int \sum_i p(y) \, dy + \left[cross\text{-}terms\right] \\ \sigma_y^2 &= \int \sum_i \left(\frac{\partial y}{\partial x_i}\right)^2 \sigma_x^2 p(y) \, dy + \left[cross\text{-}terms\right] \\ \sigma_y^2 &= \left(\sum_i \left(\frac{\partial y}{\partial x_i}\right)^2 \sigma_x^2\right) + \left[cross\text{-}terms\right] \end{aligned}$$

The cross-term for the case of two independent variables x1 and x2 is $2 \cdot \rho(x_1, x_2) \cdot \sigma_{x1} \cdot \sigma_{x2}$ where ϱ is the coefficient of correlation. For the general case, the cross terms are $2 \cdot corr(x_i, x_j) \cdot \sigma_i \cdot \sigma_j$ for i not equal to j. If the correlations among variables is weak (i.e., the coefficients of correlation are small or identically zero on the correlation matrix off diagonals), then the cross-terms are zero and we can approximate the variance by the first term in equations A1.3.

The derivative $\partial y / \partial x_i$ is estimated by a linear multivariate regression model as $y = f(x_i)$ cognizant of its attendant assumptions (i.e., representativeness, linear independence—no multicollinearity, homoscedasticity, uncorrelated normal errors, no systematic error). Ignoring higher-order terms means that the linearized technique $\mathrm{var}\langle y = f(x_i)\rangle$ does not maintain all relationships among input variables as in the conditional expectation approach, which looks to probe effects based on $\mathrm{var}\langle y | x_i \rangle$.

The error in the variance model is the same as the error in $\partial y / \partial x_i$, the error in the regression. coupling through the impact pathway model, which, even if all variables are independent, can lead to nonlinearity. The linearized method reflects this in the cross-terms via covariance, and this could be either positive or negative.

Comparisons between results are typically presented as normalized variables to emphasize where the comparative datum or model result falls within an estimated distribution. The normalized variable most often used is written in equation A1.2 as a comparative datum or model result and the superscript *ref* indicates the reference source (e.g. $ref = \mathrm{F41}$).

(A1.2) $$\left(\hat{\mu} - \mu^{ref}\right)/\hat{\sigma}\text{, where } \mu^{ref}\text{ i}$$

Using this variable, a value of zero means the reference datum or model result equals the calculated mean estimate $\hat{\mu}$ while values of 1 and –1 demarcate the single standard deviations above and below the mean, $\left|\hat{\mu} - \mu^{ref}\right| = \hat{\sigma}$.

1.7. Descriptions and analysis of trends

- compound annual rate of growth (CAGR) for a specified period
- relative contributions of mean trends $\bar{x}$ versus residual x' accounts for the influence of propagated uncertainty
- nonparametric K-W hypothesis tests that are significant (p=0.05) indicate a cross-sectional resolution that allows us determine trends

Appendix

2. Benefit assessment in cost-benefit analysis

While the analysis and findings elaborated by this thesis derive from a formal evaluation of the welfare consequences of air travel, this study is not a cost-benefit analysis. To reiterate, this thesis employs welfare metrics to make primarily comparative conclusions to elucidate trade-offs and identify improvements to current assessment practice, including where further scientific understanding is most needed. This information is beneficial to decision makers independent of the framework chosen to conduct policy discussions. However, a comprehensive cost-benefit analysis (CBA) capability is envisioned as the next step in the expansion of AT-EIA capabilities and it is important to understand how benefit assessments are applied in the context of CBA that derive from the considerations presented in this chapter.

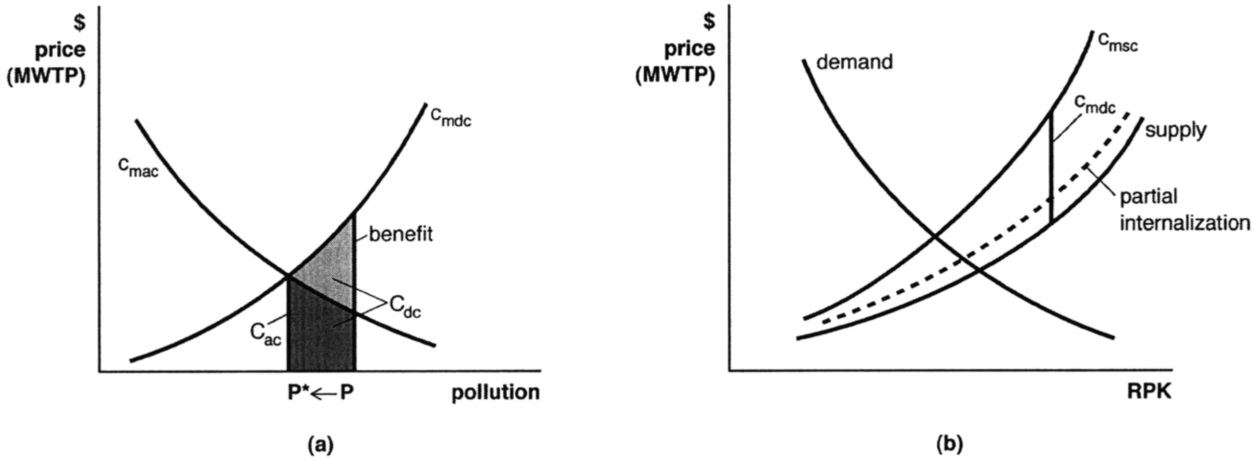
The damage function described in the previous section represents the opportunity costs resulting from aviation emissions and noise impacts on people and their resource systems. Cost-benefit analysis entails weighing environmental quality improvements, as expressed through the damage function, against the costs of achieving changes. An account of environmental costs, such as provided in this thesis, is thus a central element in any CBA conducted to assess research needs, design options, or to evaluate regulatory approaches to internalizing aviation environmental costs. Traditional CBA implies a certain conception of how we value decisions in the social aggregate. Social welfare is measured here by the value of consumption, and to the extent that well-being can be represented in terms of utility, this metric helps define the comparative desirability of decision options. CBA further implies that if total benefits exceed costs, we should pursue a given option regardless of whether some individuals are worse off than before (the Kaldor-Hicks compensation principle). This is Pareto-optimal in the sense that transfers could be made such that everyone would be better off after the option is exercised (Kolstad 2000), but practically such transfers do not necessarily occur. Nonetheless, CBA does identify distributional issues that have significant implications for policy design.

To see the damage function and its relationship to CBA, refer to figure A2.1. There are many producers (i.e. airlines) and many consumers (e.g. airport-local communities) of aviation noise and emissions.

Figure A2.1(a) shows a notional marginal damage cost curve, relating a metric of environmental pollution p and the marginal cost of an incremental change in environmental quality c.

Figure A2.1. Supply and demand for environmental resources

(a) an environmental good (e.g. quiet, clean air), and (b) a polluting good (e.g. air transportation services)



The marginal damage curve generally increases with pollution, as shown here, but other shapes are possible. Also shown is a marginal abatement cost curve, relating p and the cost of reducing emissions or noise by an incremental amount (ca). The marginal abatement curve declines with increasing p since it represents the savings to the firm by being able to pollute. For an airline, these savings may be in the form of reduced capital (Ccap) and operating expenditures (Cop).[1]

A fundamental difference between c and ca is that the marginal abatement cost addresses a marketed good for which prices are available. For example, there is a market for pollution abatement services that is enabled by the ability to divide (excludable goods) and restrict access (rival goods) to such services. Thus, the cost to reduce noise and emissions is associated with prices in the market. Comparatively, c represents non-excludable and non-rival environmental goods for which there is no market because the medium of

[1] Uncertainties, existing economic distortions, and the non-competitive market structures evident in the air transport industry can undermine an incentive policy. The availability of abatement options, technological, operational, or otherwise determines the intersection between cmdc and cmac. The impact of an emissions restriction that pushes beyond the availability of abatement options could be inefficient in the short-term. Depending on the nature of the restriction, a revenue stream could arise, for which a use would need to be determined (e.g supporting innovation), or there could be a straight economic loss. It is uncertain whether long-term options will become available (although there is an incentive to find options to be able to increase supply). These market shifts need to be accounted if abatement costs are increased and environmental benefits realized. Thus, the analysis in this thesis is necessary for policy design, but not sufficient.

impact, air for example, cannot be divided or restricted in the same manner. However, we can derive value for environmental goods by evaluating the trade-offs a consumer necessarily makes between these goods and other ordinary goods in the presence of limited resources; this is the focus of consumer demand theory. In the absence of a market, c(p) is constructed by asking consumers what they would be willing to pay for environmental quality, or by statistically inferring that amount via their actions in other markets. Thus, the marginal demand curve plots quality against marginal willingness-to-pay (MWTP), congruent with price for market goods. In contrast to ca(p), we are interested in the aggregate MWTP for a given quantity of the non-rival environmental good. The integral under the aggregate c curve represents WTP.[2]

In the air transport context, marginal damages represent externalities to the extent that they are not considered by producers (e.g. airlines) in making their supply decisions. The resulting market clearance is thus not Pareto optimal and an economic inefficiency arises. This market failure generally occurs because rights to quiet, clean air, or other environmental goods are defined poorly or not at all. This is often the impetus for government intervention. Figure A2.1(b) shows an idealized picture of this situation (cf. Schipper 2001). In figure A2.1(b), the supply curve represents the private cost of providing air transportation services at various levels (measured by revenue passenger-km, RPK, in the figure). The marginal damage cost in figure A2.1(a) is the external cost of transport services which when summed with the private cost produces the marginal social cost curve $c_s(p)$. This is the total social cost of providing services. In a perfect market, reflecting the marginal social cost in actual transactions, say via a tax or charge, would change supply and price. However, c is uncertain, which leads to impreciseness or inaccuracy in our determination of the efficient tax or charge to set. As suggested by figure A2.1, we are concerned here with the costs related directly to production in the industry, not from manufacture or disposal, and not from the presence of transport infrastructure.

In the cost-benefit framework, a public policy decision made to obtain a given change in emissions or noise would be evaluated against the minimum sum of the change in environmental costs, C—the sum of the shaded areas in figure A2.1(a)—and any costs incurred in achieving that change C_a—the dark shading

[2] This is akin to consumer surplus if the demand curve estimated is ordinary. This is a measure of welfare, although not necessarily the preferred measure. Generally, however, the valuation methods for non-market goods employed in this thesis use observed data that would result in ordinary demand and do not take steps to extend this to compensated demand where utility is constant. For a more detailed theoretical development (cf. Braden and Kolstad 1991; Kolstad 2000; Haab and McConnell 2002; Freeman 2003). To arrive at an aggregate ca(p) for the industry as a whole, we sum the abatement each producer is willing to provide at a given price. The aggregate c(p) is the sum of individual marginal damage curves at a given quantity.

in figure A2.1(a). This summation is minimized where the marginal damage cost is equal to the marginal abatement cost, which, to ensure efficiency, should be equated across producers as described by equation A2.1. The minimum is shown in figure A2.1(a) by the change p to p^*.

$$\min\{C + C_a\} \quad \text{or} \quad c = c_a \tag{A2.1}$$

This decision criteria, as well as the process developed in this thesis to estimate environmental costs, are compatible with policy guidelines in the United States, Canada, and Europe (TC 1994; OMB 1996; EAtHLG 1999; EPA 1999a, 2000; OMB 2003; FAA 2004). Note, however, that this formulation, maximizing net present value of surplus, assumes an underlying measure of welfare that is utilitarian and not necessarily in congruence with notions of sustainability (Chichilnisky 1996; Fankhauser *et al.* 1997; Pezzy and Toman 2002). In particular, the criterion in equation A2.1 does not say how costs will be distributed through society. In separating economic efficiency (or Pareto-optimality) from other desirable social objectives, such as equity, we are restricting ourselves to one framework for social choice. Other useful decision frameworks exist, but none can be perfect (Arrow 1951; Arrow 1977).[3]

For a given policy option k, the value of c_a for a firm depends intimately on its technological and operational position. Under the neoclassical assumptions, an airline will ordinarily favor proposals that maximize profit, which for pollution abatement expenditures would likely be at the lowest cost possible. On the economic balance sheet, c_a can be approximated to first order as $\bar{c}_a$, as in equation A2.1, for a change in environmental quality or impact, measured by p, but only for the case where C_a is linear. Where C_a is nonlinear, $c_a \neq \bar{c}_a$ and we must instead use the definition of marginal cost as in equation A2.3. These costs affect the supply of air transportation services and may have multiplicative effects through the broader economy, resulting from changes in resources committed to providing such mobility.

[3] We are also making the assumption that costs and benefits are finite in the formulation (i.e. not divergent with time; that is, something can be done to stem impacts (cf. Tol 2003)).

Figure A2.2. Roles of environmental decision-making and economic feedback

▩ MAIPA

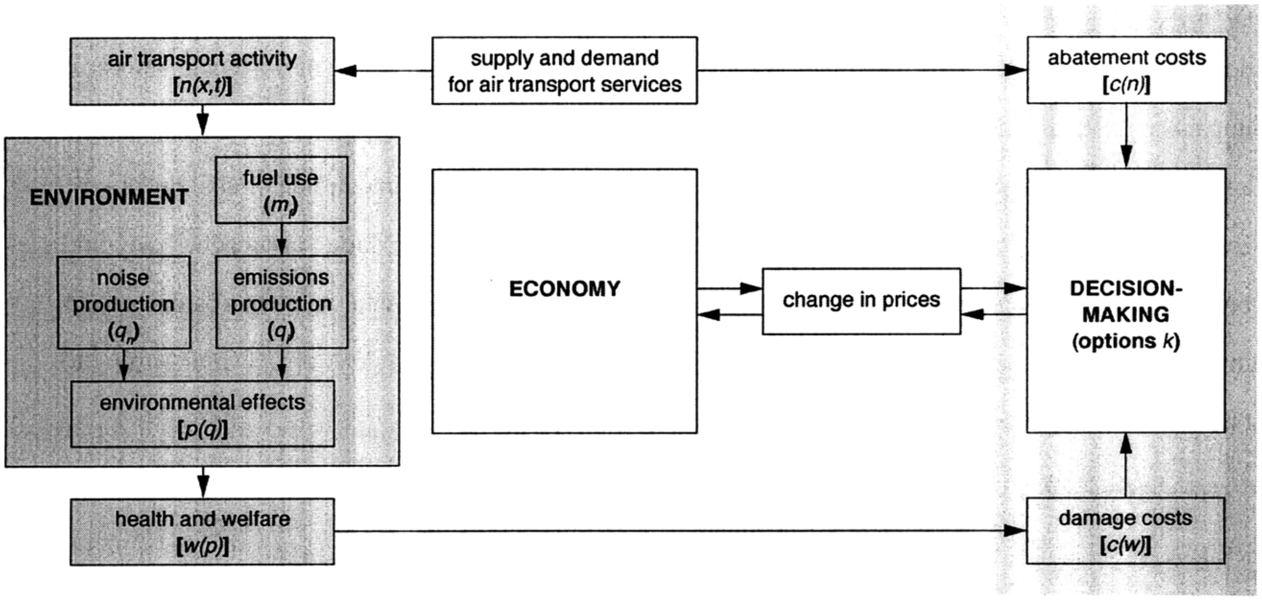


Figure A2.2 depicts MAIPA within the broad context of a comprehensive cost-benefit analysis (CBA) that considers the consequences of environmental policy decisions across a wide economic scope, showing the physical, economic, and social flows of information that constitute the interactions we seek to influence. Following the outlined paths, air transport activity (n), represented as operations or revenue passenger-km (RPK), produces emissions and noise (q), changing environmental quality (e), affecting the economic well-being of people through health and other impacts (w), such as those on ecosystems, which can be expressed economically as damage costs c(w). Policy-makers, manufacturers, airlines, and citizens may take various actions in response to these costs, considering the costs of pollution reduction, c(n). These decisions affect the economy either directly through the primary markets associated with air transportation (e.g. changing ticket prices), or indirectly through other avenues in the economy to impact supply and demand for mobility by air (e.g. changing household production so that less income is available for leisure).

These valuations must be distinguished from abatement and mitigation expenditures. Foremost, abatement costs remain to be understood; technology options remain expensive, and the low marginal costs attributed to noise and most emissions suggest a long wait between technology introductions under a traditional, cost-benefit analysis focused on Pareto optimality. Examining the comparative uncertainty in c_i, c_n, and c_a is a useful perspective from which to examine decision risks entailed in comparing policy or design options; the expected value of reducing uncertainty in the welfare value of noise and emissions impacts can be directly referenced to improved policy efficiency, enabling prioritization of research agendas. Depending on the extent of risk aversion, likely high given the magnitude of abatement costs, certainty equivalents are potentially much higher than expected benefits. Market approaches may find better use, but work in this area is limited. In the US, estimates place the total value of abatement and mitigation expenditures (for all environmental impacts) at approximately 1-2% of US GDP, similar to other industrialized countries.

Appendix

3. Low complexity models of environmental performance

This appendix looks at the application of low complexity parametric models of flight performance in the context of assessment practice. A performance model based on the Breguet equation highlights the importance of flight distance specification to model error.

3.1. Aircraft performance and the Breguet equation

Aircraft performance can be generally described with the power balance shown in equation A3.1, which relates the rate of work done by thrust (F) and drag (D) on the aircraft with the rate of change in potential energy (i.e. due to altitude change) and kinetic energy (i.e. due to speed change). This is a constructive form in the MAIPA context since it relates directly to energy change along the flight path, and thus qf. In equation A3.1, W is weight, equal to m·g (mass times the gravitational constant), h is altitude, and uo is the flight speed.

(A3.1)
$$Fu_o = W\frac{dh}{dt} + \frac{W}{g}\frac{d}{dx}\left(\frac{u_o^2}{2}\right) + Du_o$$

To first order, required F is set by aircraft size. The efficiency (η) with which F is achieved is dependent on engine thermodynamic cycle and component design. The basic relationships among fuel use, F, and η can be seen in equation A3.2, where F is specified for an engine with a single, core flow stream (e.g. a turbojet engine). The approximate equivalence is achieved for conditions where $\dot{m}_f$ is a small part of total mass flow through the engine, and where exit pressure, pe is equal to ambient pressure, po.[1]

(A3.2)
$$F = m_e u_e - m_o u_o + A_e\left(p_e - p_o\right) \cong m_o\left(u_e - u_o\right)$$

Propulsive performance is defined by specific fuel consumption (SFC, e.g. mg/N-s). SFC can be related to engine efficiency (ηo) as in equation A3.3, where LHV is the lower heating value of the fuel (e.g. MJ/kg).

[1] Thrust is produced when air moving through the engine is pushed out the exhaust at a higher speed than it entered (i.e. at the flight speed); Newton's second law, F=ma.

(A3.3) $$\mathrm{SFC}=\frac{m_f}{F}=\frac{u_o}{\mathrm{LHV}\cdot\eta_o}$$

Engine efficiency is the product of propulsive (ηp) and thermal (ηt) efficiencies. Propulsive efficiency measures conversion of mechanical power (e.g. change in kinetic energy) to propulsive power (F·uo) and increases as a smaller impulse is provided to a larger mass flow. Thus, for a given F, increasing engine mass flow relative to intra-engine velocity change will increase ηp and decrease SFC. Where equal F can be produced through a large impulse to a small mass flow or a small impulse to a large mass flow, the latter option is the choice of civil applications for reasons of efficiency and typically involves the use of a second, bypass flow stream. This can be represented in equation A3.2 by replacing ue with an exit velocity equal to the appropriate average of the core and bypass flow streams. Thermal efficiency measures conversion of fuel chemical energy to mechanical energy. Higher engine overall pressure ratio and, for imperfect (i.e. non-isentropic) components, higher turbine entry temperature increase ηt. Ideally, for a given F required, u0, h, and limitation on peak temperature of the engine (e.g. by material constraints), SFC is a function of the compressor and fan pressure ratios, and the bypass ratio of the engine design.

The Breguet range equation shown in equation A3.4 is an important simplification of the aircraft performance equations, characterizing the role of technology in fulfilling flight goals.

(A3.4) $$R=\frac{u_o(L/D)}{g\cdot\mathrm{SFC}}\ln\left(1+\frac{W_{fuel}}{W_{payload}+W_{structure}+W_{reserve}}\right)$$

The Breguet equation describes an aircraft in steady, level flight where lift (L) just equals weight (W), and thrust (F) just equals drag (D). Its typical use is to determine the maximum range (R) achievable for an aircraft design represented by parameters describing propulsive, aerodynamic, and structural performance. The specific fuel consumption (SFC) characterizes propulsive performance, u0 is the flight velocity (here true airspeed), aerodynamic performance is described by the lift-to-drag ratio (L/D), a non-dimensional parameter, and structural efficiency is described by the weight of the structure (Wstructure) relative to the weight it can carry (i.e. payload, Wpayload; fuel used during operation, Wfuel; fuel reserve, Wreserve; and the structure itself, Wstructure).[2] The resulting estimate for range using equation A3.4 is a maximum because several flight segments are not accounted (i.e. idle, taxi, take-off and climb, and descent and landing).

3.2. Breguet-based low complexity estimation of aircraft fuel consumption

A low complexity approach to estimating aircraft fuel consumption can be drawn directly from the basic Breguet formulation of the aircraft performance equations. With a slight reorganization of equation A3.4, a model equation for fuel efficiency can be derived from the Breguet equation; from fuel efficiency, an estimate for per-flight fuel consumption can be obtained. The Breguet-based specification is the least complex performance formulation that can capture the influence of both technology and service use. Considering the MAIPA approach in the context of the Breguet formulation highlights the minimum performance representation sought in its development for this thesis. Equation A3.5 shows per-flight fuel consumption as the product of fuel efficiency (ηf) and a nominal flight distance (dsl) where fuel efficiency is defined in units of fuel mass per unit distance. In the equation, nseats is the number of seats, approximated as Wpayload / Wpsgr, ηft is the flight time efficiency, an empirical operational parameter that is the ratio of minimum flight hours (related to R) to total operational hours.[3] The flight time efficiency corrects for the influence of inaccurate utilization specification, due to ground and flight delays, and non-cruise operation, but does not account for inaccuracies in fuel reserve amounts, non-reported weight elements, and variability in performance parameters during the flight. Fuel consumption inventories are then calculated with the summation of per-flight fuel consumption over all operations.

(A3.5)
$$\begin{aligned} q_f &= d_{sl} \cdot \eta_f \\ q_f &= d_{sl} \cdot W_{fuel} \cdot \frac{g \cdot \mathrm{SFC}}{u_o (L/D)} \cdot \frac{1}{\ln\left[1 + \left(W_{fuel}/W_{payload} + W_{structure} + W_{reserve}\right)\right] \cdot \eta_{ft}} \\ Q_f &= n_{ops} \cdot q_f \end{aligned}$$

[2] The flight velocity can be alternatively represented by the Mach number (M), which is a more useful parameter since cruise operation is typically specified with constant M.

[3] Fuel efficiency defined in units of energy per unit available seat-distance are typically 1-2 MJ/ASK. For regional aircraft, the range is 1.5-3 MJ/ASK. Historical trends are given in Lee et al. (2001) and Babikian et al. (2002) for large commercial and regional aircraft.

Using a database of technology parameters for 23 large aircraft types flown by U.S. airlines during the period 1991-1998, Lee et al. (2001) estimate a fleet fuel consumption accuracy of +/- 20% against the the same parameter calculated from F41 data. Babikian et al. (2002) find similar results for regional aircraft. Breguet approximations are valuable as technology trending and forecasting metrics, and can be applied to a large portion of the historical record to assess the technological and operational changes in the air transport system.[4] However, the Breguet approach lacks a flight profile definition and is thus too low resolution for ATEA practice; a distinction must be made among near-airport operations (terminal area and the landing-takeoff cycle, LTO) in order to correctly address emissions and noise impacting local communities via changes to air quality and those resulting in climate perturbations.

3.3. Importance of flight distance specifications in low complexity models

The important lesson from experience with the Breguet approach is that the specification of flight distance is crucially important to accuracy; without the flight time efficiency correction applied, equation A3.5 is highly inaccurate for short-haul flights. The flight rules selected for design evaluation using a proprietary model are often dependent on the launch customer for a new product and thus specific to a particular market application. Once introduced by an airline, an aircraft may be frequently operated off the design evaluation optimum, both with respect to stage length (or market application) as well as speed and trajectory selection. Thus, flight distance needs to be relevant to market application and the performance specification must be flexible enough to account for the changes in performance resulting in off-specification use. For MAIPA, these findings are applied in the choice of aircraft model aggregations into representative aircraft types, first specified by matching flight distance among types, then further divided within these groupings so that the spread in technology performance is minimized, constrained by the performance model fidelity (cf. next section). This second step reduces the elasticity required in the aircraft performance model and thus improves chances for reducing complexity toward the objectives discussed in chapter 2.

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[4] Lee et al. (2001) specifically assess equation A3.5 as a predictor of reported flight distance through a comparison with F41 data. Using technology parameter values gleaned from public sources (and verified by manufacturers), dSL estimates without the flight time efficiency correction were greater than reported by 10-30% for long-haul types and as much as 120% for short-haul types. Correcting for flight time efficiency, remaining deviations were reduced to ~10% over all aircraft. Babikian et al. (2002) find the uncorrected fuel efficiency results, on average, in 1.6 times higher error for regional jets compared to large types; however, applying the flight time efficiency reduces errors to similar levels.

Appendix

4. Historical operations data and representative aircraft types

This appendix describes the activity data available through DOT Form 41 and Form 298C and its use as a source characterization input to MAIPA. It also details the approach to aggregating certificated aircraft types into representative aircraft type groupings to characterize the technology operating in the historical aircraft fleet.

4.1. Operational statistics for US commercial air transportation

This study uses historical operations data collected by the U.S. Department of Transportation (DOT) Bureau of Transportation Statistics (BTS) under Parts 241 and 298 of Title 14 (Aeronautics and Space) of the U.S. Code of Federal Regulations (14 CFR 241 and 14 CFR 298), otherwise known as Form 41 and Form 298C data, respectively. Large certificated air carriers report traffic statistics via Form 41. Certificated carriers that do not meet this classification (small certificated air carriers) and passenger air taxi operators (commuter air carriers) are required to report via Form 298C; these carriers operate regionally.

Each form is divided into schedules. Of relevance to this thesis are Form 41 Schedule T-2 (F41T2), which summarizes data reported in Schedule T-100 by aircraft type, and Form 298C Schedule A-1 (F98A1). The analysis accounts for scheduled and nonscheduled domestic and international operations with at least one point of service in the United States or its territories for revenue passenger and revenue cargo service. Both F41T2 and F98A1 are reported quarterly. Military operations, as well as general aviation activity for business, recreational, or personal use, are not included. The revenue focus additionally excludes piston-powered aircraft.

Although traffic data for large certificated air carriers are available monthly via Form 41 Schedule T-100, quarterly data is used to maintain internal consistency of resolution with other data sources. Data components used in the analysis are listed in table A4.1. Only revenue aircraft-miles flown and revenue aircraft departures performed contribute to estimated costs. Fuel use and the additional data listed are employed for validation purposes only.

Table A4.1. Operational statistics provided by DOT Forms 41-T2 and 298C-A1

F41T2 U.S. Air Carrier Traffic and Capacity by Aircraft Type
921: AIRCRAFT_FUELS — Aircraft Fuels (gallons) The amount of aircraft fuels issued, in U.S. gallons, during the reporting period for both revenue and nonrevenue flights.
410: REV_AIR_MILES — All Services, Revenue Aircraft Miles Flown Revenue aircraft miles flown are computed in accordance with the airport pairs between which service is actually performed; miles are generated from the data for scheduled aircraft departures (Code 520) times the interairport distances (Code 501)
510: REV_DEP_PERFORMED — All Services, Revenue Aircraft Departures Performed The number of revenue aircraft departures performed.
650: AIR_HOURS — Total Aircraft Hours Flown (Airborne) The elapsed time, computed from the moment the aircraft leaves the ground until it touches down at the next landing. This includes flight training, testing, and ferry flights.
630 :AIR_HOURS_RAMP — Aircraft Hours, Ramp-to-Ramp The elapsed time, computed from the moment the aircraft first moves under its own power from the boarding ramp at one airport to the time it comes to rest at the ramp for the next point of landing. This data element is also referred to as 'block' and block-to-block aircraft hours.
140: REV_PAX_MILES — All Services, Revenue Passenger Miles (000), Total Computed by multiplying the interairport distance of each flight stage by the number of passengers transported on that flight stage.
240: REV_TON_MILES — All Services, Revenue Ton Miles, Total Ton miles are computed by multiplying the revenue aircraft miles flown (410) on each flight stage by the number of tons transported on that stage. (Note: sums RTM_PAX,, RTM_FREIGHT, and RTM_MAIL)
247: RTM_FREIGHT— All Services, Freight Revenue Ton Miles Equals the volume of freight in whole tons times the interaiport distance
249: RTM_MAIL — All Services, Mail Revenue Ton Miles Equals the volume of mail in whole tons times the interairport distance
320: AVAIL_SEAT_MILES — All Services, Available Seat Miles (000), Total
The aircraft miles flown on each flight stage multiplied by the seat capacity available for sale
Definition of seats available: installed seats in an aircraft (including seats in lounges) exclusive of any seats not offered for sale to the public by the carrier; provided that in no instance shall any seat sold be excluded for the count of available seats
Definition of seat-miles available: revenue: the aircraft miles flown on each flight stage multiplied by the number of seats available for revenue use on that stage
410: AVAIL_TON_MILES — All Services, Available Ton Miles, Total
The aircraft miles flown on each flight stage multiplied by the available capacity on the aircraft in tons

F98A1 Report of Flight and Traffic Statistics in Scheduled Passenger Operations
AIRCRAFT_HOURS — Aircraft Hours
AIRCRAFT_MILES — Aircraft Miles
AVAIL_SEAT_MILES — Available Seat Miles
AVAIL_TON_MILES — Available Ton Miles
DEP_PERFORMED — Aircraft Departures
REV_PAX_MILES — Revenue Passenger Miles
REV_TON_MILES — Revenue Ton Miles

There are important limitations to highlight. Complete fleet coverage, defined by airline data reporting of better than 99.995%, is available only for 1991 to present. Prior to 1991, activity and fuels-issued data are incomplete or missing for some air carriers. This, and BTS restrictions on data release, limit the analysis to the 13-year period 1991 through 2003. Additionally, only scheduled service is reported on F98A1, with no categorization by aircraft type and no record of fuel use. Since most of this activity is conducted using regional aircraft, and given the coarse representation of such technology in this study, assumptions are required to attribute emissions and noise performance parameters to these data; later sections provide details.

4.2. Representative aircraft types

This thesis constructs aggregations of certificated aircraft types to account for fleet technology based on fidelity considerations. A certificated aircraft type is a logical fundamental unit of technology with respect to environmental impact. For a commercial aircraft, the primary mission is driven by the business of efficiently moving people and goods between locations in a safe manner. This determines the design characteristics of an airframe-engine combination.

Major product features critical to this objective are fuel economy, reliability, maintainability, and environmental performance. These are offered at a cost, are to some degree correlated, and are variously traded to determine the final form of an airframe-engine combination that will satisfy demand in a particular market segment. The A380 case discussed in chapter 1 is an example of such trades as they pertain to environmental performance. Over the five decades of modern air transport, product requirements have led to a measure of design and operational standardization exemplified by the limited variability in aircraft planform and assignment of mission rules.

4.3. Construction of representative aircraft types

MAIPA represents technological performance in the fleet at a resolution suitable for comparing the magnitude of environmental costs across fleet segments. We wish to differentiate aircraft by market application—as defined by mobility, not equipment configuration (e.g. number of seats, aisles, engines)—with as complete coverage of fleet activity practicable, here measured by number of departures and kilometers-flown for the relevance of these metrics as environmental drivers. Flight distance relates directly to divisions among mobility markets; equipment decisions correlate longer flights with larger aircraft, which tend to have more engines and aisles. Larger differences among aircraft with regards to energy use are attributable to operational rather than technological differences; further discussion of this point is found in appendix A5.

Through average stage length, each representative aircraft type describes a particular market use, which may correspond to many different city-pairs. Policy and design choices are made using distinctions among technologies and categories are limited by the fidelity of input data, including activity and performance. This also makes it possible to selectively substitute for individual aircraft types in prospective analyses of changes in damage costs resulting from technology introduction.

Each individual airframe-engine combination cannot be specified uniquely. To arrive at distinct categorizations, identified by non-overlapping activity, fuel use, emissions, and noise specifications, aircraft models are aggregated. Generally, this means that the differences between representative types are characterized by significant technology advancement relevant to environmental performance in comparison to market predecessors. Specification of types at the resolution of city-pair markets is not consistent with the resolution of the input data.

To represent technology in the historical fleet, representative aircraft types were constructed from in-service technology as summarized for large types in table A4.2 and regionals in table A4.3. Other categorizations have been used for environmental analyses or other analyses pertaining to the air transport markets. Equivalencies are shown in tables A4.2 and A4.3 for commonly cited sources, including the type certification sheet designations and categorizations used by the Forecasting and Economic Sub-Group (FESG) of the ICAO Committee on Aviation Environmental Protection (CAEP).

Table A4.2. Representative aircraft type assignments for large DOT aircraft model identifiers with type certification and FESG equivalents

gray-shaded entries indicate out-of-production models

Name	F41 code / name	Certification Name	Certification Date	Equivalencies
b727	715 / Boeing 727-200/231a	B727-200	1967	FESG: 100-150
(1)	none	B727-200F	1983	
	710 / Boeing 727-100	B727/-100	1963	
	711 / Boeing 727-100c/Qc	B727C/-100C	1966	
b737	619 / Boeing 737-300	B737-300	1984	FESG: 100-150
(2)	616 / Boeing 737-500	B737-500	1990	
	617 / Boeing 737-400	B737-400	1988	
	618 / Boeing 737-300lr	n/a	n/a	
md80	655 / McDonnell-Douglas DC9 Super 80 /MD81/2/3/8	DC9-80	-81 1980; -82 1981 -83 1985; -88 1987	FESG: 100-150
	645 / McDonnell-Douglas DC9 Super 87		-87 1987	
b747o	816 / Boeing 747-100	B747-100 types	-100 1969; -100B 1979 -100B SUD 1986; SR 1973	FESG: 301-400
(3)	817 / Boeing 747-200/300	B747-200 types	-200B 1970; -200C 1973	
		B747-300	1983	
	822 / Boeing 747sp	B747-SP	1976	
b747	819 / Boeing 747-400	B747-400_types	-400 1989; -400D 1991	FESG: 301-400
(3)	820 / Boeing 747f	B747-200F	-200F 1972	
		B747-400F	-400F 1993	
b757	622 / Boeing 757-200	B757-200	1982	FESG: 151-210
	623 / Boeing 757-300	B757-300	1999	
b767	626 / Boeing 767-300/300er	B767-300	1986	FESG: 151-210
	625 / Boeing 767-200/200er	B767-200	1982	
	624 / Boeing 767-400	B767-400ER	2000	
dc10	730 / McDonnell Douglas Dc-10-10	DC-10-10	1971	FESG: 211-300
(8,9)	732 / McDonnell Douglas Dc-10-30	DC-10-30	1972	
	733 / McDonnell Douglas Dc-10-40	DC-10-40	1972	
	735 / McDonnell Douglas Dc-10-30cf	DC-10-30F	1973	
	731 / McDonnell Douglas Dc-10-20	n/a	n/a	
	none	DC-10-10F	1974	
	none	DC-10-15	1981	
	none	DC-10-40F	1976	
dc9	640 / McDonnell Douglas Dc-9-30	DC9-30 types	-31 1966; -32 1967, -34 1976	FESG: 100-150
(4)	650 / McDonnell Douglas Dc-9-50	DC9-50	-51 1975	
	630 / McDonnell Douglas Dc-9-10	DC9-10 types	-11/2/3/4 1965; -15 1966;	
	645 / McDonnell Douglas Dc-9-40	DC9-40	-41 1968	
	635 / McDonnell Douglas Dc-9-15f	DC9-15f	1967	
	none	DC9-20	-21 1968	

Name	F41 code / name	Certification Name	Certification Date	Equivalencies
a320	694 / Airbus Industrie A320-100/200	A320-100/200 series	-111/211 1988; -231 1989	FESG: 100-150
			-212 1990; -232 1993	
			-233 1995; -214 1996	
	698 / Airbus Industrie A319	A319-100 series	-112 1996	
			-111/113/114/131/132 1997	
	699 / Airbus Industrie A321	A321-100/200	-111/112/131 1995	
			-211/231 1997	
l1011	760 / Lockheed L-1011-1/100/200	L-1011-385-1 types	-1 1972	FESG: 211-300
			-1-14 (-100) 1975	
			-1-15 (-200) 1975	
	765 / Lockheed L-1011-500 Tristar	L-1011-385-3	1979	
md11	740 / McDonnell Douglas Md-11	MD-11	1990	FESG: 151-210
(10)	none	MD-11F	1990	
b777	627 / Boeing 777	B777-200	-200 1995; -200ER 1997	FESG: 211-300
		B777-300	1998	
		B777-300ER	2004	
b737n	614 / Boeing 737-800	B737-800	1998	FESG: 100-150
(5,6,7)	612 / Boeing 737-700/700lr	B737-700	1998	
	none	B737-700C	2000	FESG: 151-210
	634 / Boeing 737-900	B737-900	2001	
	615 / Boeing 737-5/600lr	n/a	n/a	FESG: 151-210
	633 / Boeing 737-600	B737-600	1998	
b717	608 / Boeing 717	B717-200	1999	FESG: 100-150

Table notes:

(1) no –200F F41 designation

(2) not a separate Boeing or type cert designation for B737-300lr

(3) –400 is similar to –300; -300 is similar to –200 with a stretched upper deck; -sp is essentially a long-range version of –100

(4) no –20 F41 designation, No activity reported for DC-9-15f over period considered.

(5) –600/700/800 all under same type cert

(6) no separate certification or Boeing designation, assumed same as –600

(7) no –700C F41 designation

(8) no –20 Boeing or type cert designation

(9) no –10F, -40F, or –15 F41 designation

(10) no –11F F41 designation

Table A4.3. Representative aircraft type assignments for regional DOT aircraft model identifiers with type certification and FESG equivalents

gray-shaded entries indicate out-of-production models

Regional types specified as groups	
turbofans (tfan): F41 code / name	**turboprops (tprp): F41 code / name**
629 / Canadair Rj-200er	456 / Saab-Fairchild 340/B
603 / Fokker 100	461 / Embraer Emb-120 Brasilia
674 / Embraer-135	441 / Aerospatiale/Aeritalia Atr-42
868 / British Aerospace Bae-146-100	416 / Cessna 208 Caravan
628 / Canadair Rj-100/Rj-100er	483 / Dehavilland Dhc8-100 Dash-8
835 / Avroliner Rj85	442 / Aerospatiale/Aeritalia Atr-72
602 / Fokker F28-4000/6000 Fellowship	469 / British Aerospace Jetstream 31
867 / British Aerospace Bae-146-200	445 / Convair CV-660
601 / Fokker F28-1000 Fellowship	405 / Beech 1900 A/B/C
866 / British Aerospace Bae-146-300	467 / Swearingen Metro Iii
	471 / British Aerospace Jetstream 41
	449 / Dornier 328
	489 / Shorts 360
	484 / Dehavilland Dhc8-300 Dash-8
	408 / British Aerospace Bae-Atp
	450 / Fokker Friendship F-27/Fairchild F-27/A/B/F/J
	550 / Lockheed L-188a/C Electra
	430 / Convair Cv-580
	435 / Convair Cv-600
turbofans (tfan): Equivalencies	**turboprops (tprp): Equivalencies**
FESG: 50-99	FESG:20-49

Regional types specified individually				
Name	F41 code / name	Certification Name	Certification Date	Equivalencies
e145	675 / Embraer 145	EMB-145 series	-145 1996; -145ER 1996 -145MR 1998; -145LR 1998 -145XR 2002; -145MP 2003 -145EP 2003	FESG: 50-99

4.4. Coverage of US commercial aircraft operations 1991-2003

There are 19 designations, 16 for large aircraft operations, 3 for regionals, with one additional generic specification. Only designations with adequate data to describe performance were considered. In addition, based on experience with computational overhead, only F41T2 designations for which revenue aircraft departures performed or revenue aircraft kilometers (RAK) flown represented >0.3% of fleet totals individually, or >0.3% as a group with other F41T2 designations, were used to construct representative aircraft types. For each representative aircraft type, there is typically a dominant aircraft model (i.e. the model that accounts for most departures or aircraft kilometers), denoted by underscore.

Collectively, representative aircraft types account for ~95% of departures performed, RAK flown, and aircraft fuels issued as reported on F41T2 for the period 1991 through 2003. The remaining 5% is specified as a fleet average, generic representative aircraft type. Data reported on F98A1 increase total departures by 25% and RAK by 7%. A summary of cumulative activity coverage is given in table A4.4; an extended tabulation of data concerning activity coverage is contained in table A4.5.

Table A4.4. Representative aircraft type coverage of US commercial aircraft activity 1991-2003

Accounted activity	F41 dep	F41 RAK	F41 fuel
large aircraft types	72%	91%	86%
regional aircraft types	22%	3%	8%
tprp portion of regionals	15%	1%	4%
tfan portion of regionals	7%	2%	4%
Total coverage	94%	94%	94%

Additional regional data added via F98A1 as % of F41T2 data	25%	not available	7%

Unaccounted activity	F41 dep	F41 RAK	F41 fuel
piston / rotor aircraft types	1%	0%	0%
other tprp types	1%	0%	0%
other tfan types	2%	1%	1%
other large types	2%	5%	4%

Table A4.5. Detailed statistics for representative aircraft type coverage of US commercial aircraft activity 1991-2003

Name	F41 code / name	F41 DEP (10^6) and % contribution		F41 RAK (10^6) and % contribution		F41 fuel (10^3 metric tons) and % contribution	
b727	Total contribution to cumulative	11156	10%	12334	10%	83575	12%
	715 / Boeing 727-200/231a	9825	88%	11120	90%	76181	91%
	710 / Boeing 727-100	1124	10%	1045	8%	6287	8%
	711 / Boeing 727-100c/Qc	206	2%	169	1%	1107	1%
b737	Total	19012	17%	18323	15%	77597	11%
	619 / Boeing 737-300	13835	73%	13251	72%	56093	72%
	616 / Boeing 737-500	3064	16%	2877	16%	11891	5%
	617 / Boeing 737-400	2082	11%	2188	12%	9566	12%
	618 / Boeing 737-300lr	30	0.16%	7	0.04%	48	0.06%
md80	Total	12818	11%	15649	12%	77910	11%
	655 / McDonnell-Douglas DC9 Super 80 /MD81/2/3/8	12759	99.5%	15599	99.7%	77659	99.7%
	645 / McDonnell-Douglas DC9 Super 87	59	0.5%	50	0.3%	251	0.3%
b757	Total	7471	6.7%	14091	11%	70334	9.8%
	622 / Boeing 757-200	7427	99.4%	13994	99.3%	69862	99.3%
	623 / Boeing 757-300	44	0.6%	97	0.7%	471	0.7%
b767	Total	2715	2.4%	9365	7.4%	59461	8.3%
	626 / Boeing 767-300/300er	1672	62%	5884	63%	37983	64%
	625 / Boeing 767-200/200er	949	35%	3223	34%	19637	33%
	624 / Boeing 767-400	95	3%	258	3%	1840	3%
dc10	Total	1694	1.5%	5143	4.1%	52926	7.4%
(1)	730 / McDonnell Douglas Dc-10-10	887	52%	2096	41%	20059	38%
	732 / McDonnell Douglas Dc-10-30	592	35%	2422	47%	25883	49%
	733 / McDonnell Douglas Dc-10-40	211	12%	615	12%	6896	13%
	735 / McDonnell Douglas Dc-10-30cf	— —		— —		— —	
	731 / McDonnell Douglas Dc-10-20	— —		— —		— —	
b747o	Total	703	0.6%	3372	2.7%	46158	6.4%
	816 / Boeing 747-100	455	65%	2123	63%	28501	62%
	817 / Boeing 747-200/300	230	33%	1132	34%	16274	35%
	822 / Boeing 747sp	19	3%	118	3%	1383	3%
dc9	Total	8593	7.7%	6157	4.9%	31485	4.4%
(2)	640 / McDonnell Douglas Dc-9-30	5976	70%	4540	74%	22550	72%
	650 / McDonnell Douglas Dc-9-50	1498	17%	804	13%	4947	16%
	630 / McDonnell Douglas Dc-9-10	771	9%	535	9%	2652	8%
	645 / McDonnell Douglas Dc-9-40	349	4%	278	5%	1336	4%
	635 / McDonnell Douglas Dc-9-15f	— —		— —		— —	
b747	Total	397	0.4%	5816	1.9%	30996	4.3%
	819 / Boeing 747-400	283	71%	2002	82%	24864	80%
	820 / Boeing 747f	115	29%	442	18%	6132	20%

Name	F41 code / name	F41 DEP (10^6) and % contribution		F41 RAK (10^6) and % contribution		F41 fuel (10^3 metric tons) and % contribution	
b737o	Total	8298	7.4%	5938	4.7%	29470	4.1%
	620 / Boeing 737-100/200	7447	90%	5413	91%	26738	91%
	621 / Boeing 737-200c	852	10%	526	9%	2732	9%
a320	Total	3806	3.4%	6523	5.2%	24674	3.4%
	694 / Airbus Industrie A320-100/200	2604	68%	4613	71%	17672	72%
	698 / Airbus Industrie A319	1105	29%	1719	26%	6171	25%
	699 / Airbus Industrie A321	97	3%	190	3%	831	3%
l1011	Total	758	0.7%	2013	1.6%	21022	2.9%
	760 / Lockheed L-1011-1/100/200	578	76%	1350	67%	14369	68%
	765 / Lockheed L-1011-500 Tristar	180	24%	663	33%	6653	32%
md11	740 / Mcdonnell Douglas Md-11	406	0.4%	1927	1.5%	18147	2.5%
b777	627 / Boeing 777	372	0.3%	2029	1.6%	16874	2.3%
b737n	Total	2002	1.8%	3190	2.5%	11913	1.7%
(3)	614 / Boeing 737-800	1072	54%	1844	58%	7140	60%
	612 / Boeing 737-700/700lr	905	45%	1301	41%	4620	39%
	634 / Boeing 737-900	25	1%	46	1%	153	1%
	615 / Boeing 737-5/600lr	— —		— —		— —	
	633 / Boeing 737-600	— —		— —		— —	
b717	608 / Boeing 717	1817	1.6%	1285	1.0%	2486	0.3%

Regional types specified individually							
Name	F41 code / name	F41 DEP (10^6) and % contribution		F41 RAK (10^6) and % contribution		F41 fuel (10^3 metric tons) and % contribution	
e145	675 / Embraer 145	311	0.3%	237	0.2%	921	0.1%

Regional types specified as groups							
Name	F41 code / name	F41 DEP (10^6) and % contribution		F41 RAK (10^6) and % contribution		F41 fuel (10^3 metric tons) and % contribution	
tfan	Total	5338	4.8%	3501	2.8%	10071	1.4%
	629 / Canadair Rj-200er	1968	37%	1402	40%	2750	27%
	603 / Fokker 100	1047	20%	804	23%	3627	36%
	674 / Embraer-135	498	9%	305	9%	671	7%
	868 / British Aerospace Bae-146-100	497	9%	213	6%	1008	10%
	628 / Canadair Rj-100/Rj-100er	441	8%	325	9%	200	2%
	835 / Avroliner Rj85	340	6%	185	5%	672	7%
	602 / Fokker F28-4000/6000 Fellowship	340	6%	167	5%	724	7%
	867 / British Aerospace Bae-146-200	123	2%	59	2%	250	2%
	601 / Fokker F28-1000 Fellowship	74	1%	36	1%	158	2%
	866 / British Aerospace Bae-146-300	11	0.2%	6	0.2%	12	0.1%

Name	F41 code / name	F41 DEP (10^6) and % contribution		F41 RAK (10^6) and % contribution		F41 fuel (10^3 metric tons) and % contribution	
tprp	Total	17139	15%	5318	4.2%	7244	1%
	456 / Saab-Fairchild 340/B	3205	19%	1048	20%	1515	21%
	461 / Embraer Emb-120 Brasilia	2285	13%	870	16%	1149	16%
	441 / Aerospatiale/Aeritalia Atr-42	1706	10%	523	10%	983	14%
	416 / Cessna 208 Caravan	1705	10%	394	7%	28	0.4%
	483 / Dehavilland Dhc8-100 Dash-8	1641	10%	449	8%	594	8%
	442 / Aerospatiale/Aeritalia Atr-72	1284	7%	466	9%	948	13%
	469 / British Aerospace Jetstream 31	1141	7%	277	5%	281	4%
	445 / Convair CV-660	890	5%	243	5%	312	4%
	405 / Beech 1900 A/B/C	848	5%	236	4%	130	2%
	467 / Swearingen Metro Iii	764	4%	224	4%	185	3%
	471 / British Aerospace Jetstream 41	506	3%	179	3%	249	3%
	449 / Dornier 328	462	3%	198	4%	281	4%
	489 / Shorts 360	260	2%	43	1%	87	1%
	484 / Dehavilland Dhc8-300 Dash-8	118	1%	37	1%	56	1%
	408 / British Aerospace Bae-Atp	116	1%	27	1%	65	1%
	450 / Fokker Friendship F-27	80	0.5%	21	0%	41	1%
	550 / Lockheed L-188a/C Electra	64	0.4%	55	1%	263	3.6%
	430 / Convair Cv-580	27	0.2%	14	0.3%	36	0.5%
	435 / Convair Cv-600	24	0.1%	10	0.2%	32	0.4%

Notes:

(1) <<1% of DC-10 activity reported for DC-10-30cf and DC-10-20 over period considered.

(2) <<1% of DC-9 activity reported for DC-9-15f over period considered.

(3) no activity reported for Boeing 737-5/600lr, 737-600 over period considered.

Absent from these lists are DC-8, A300/A310, and MD-90 models, each of which are older and/or discontinued models fading from the U.S. fleet. In addition, newer large aircraft models entering the U.S. fleet, for which limited or no activity data exists, include new A380, A330/A340, B737, and B777 models, several of which have completed certification within the last few years. There is a bias error introduced due to the lack of full fleet coverage, downward in terms of inventories and environmental costs aggregated at the national or airport level, but the sign of bias is ambiguous for fleet emission indices or marginal cost estimates.

As shown in table A4.4, most missing contributions fall into the large transport category. To account for this error, an estimated correction to inventories is introduced before deriving average or marginal quantities based on averaged emissions and noise characteristics for the portion of the fleet specified by representative aircraft types. This process provides a generic large transport representative aircraft type in a manner similar to regional turbofans and turboprops.

4.5. Treatment of regional aircraft

Data are not available to fully characterize the regional fleet, particularly where noise and emissions performance for many smaller engines have not been publicly reported. Even where such data are found, measurement techniques have been inconsistently applied across engine models, and comparisons are suspect. Also, a large portion of regional activity is reported without attribution to an aircraft model via F98A1. Some of these data issues have been addressed by Babikian et al. (2002), but only for the purpose of examining fuel use.

As identified in table A4.3, only the Embraer 145, a turbofan-powered aircraft, is treated similarly to its larger counterparts. The Embraer 145 is well-characterized by existing data using emissions and noise measurement techniques similar to those standardized for large commercial aircraft. Other regional types are represented by summary categories, one for turbofans and another for turboprops, but no piston-powered models.

MAIPA treats all regional activity reported via F41T2 as distinct from the E145, and all activity reported on F98A1 with weighted average specifications for emissions and noise performance based on these categories. Noise and emissions inventories calculated in this manner are for comparison only to determine the relative magnitude of contribution to overall damage costs. No specific policy or design decisions should be made on the basis of results for the generic category.

4.6. Aircraft categorizations by service application

4.6.1. Basic types by flight distance

For inventory calculation, types were assigned to one of three categories organized by reported mean flight distance. The categorization derives directly from the population of F41T2 quarterly reports. Table 3.1 in chapter 3 summarizes fleet categorization and assignment; the shading again designates types no longer in production. The number of departures or kilometers-traveled was used as an activity weighting to develop parameter distributions in the probabilistic analysis. When distributions for emissions are generated for a representative aircraft type, the contributions of sub-types are determined by cumulative kilometers-traveled over the period considered. This is a relevant weighting since inventory emissions depend on total fuel burn, which is a strong function of kilometers-traveled. Similarly, a departures weighting is used for determining aggregated noise characteristics. These distributions are also shown in table A4.5; the listing order is by fuels-issued.

4.6.2. Apportionment of operational activity to freight service

Freight is separated in this analysis from passenger service. F41T2 and F98A1 specify freight activity by revenue ton-miles; departures are not called out specifically. Table A4.6 estimates the portion of activity accounted by freight relative to passenger service. In calculating the ratio of freight to total mass flown, the mass of a passenger (including baggage) is assumed to be 90.7 kg (200 lbs). The portion of fuel expended in freighter operations trends downward at r_g^{lh} = -0.27% and a declining fraction of large aircraft fuel use, from 21% to 16% .

Table A4.6. Estimated apportionment of US commercial operations to freight service 1991-2003

Name	F41 code/name	Estimated fraction of cumulative activity in freight
b727	715 / Boeing 727-200/231a	18%
	710 / Boeing 727-100	78%
	711 / Boeing 727-100c/Qc	98%
b737	619 / Boeing 737-300	4%
	616 / Boeing 737-500	4%
	617 / Boeing 737-400	4%
	618 / Boeing 737-300lr	33%
md80	655 / McDonnell-Douglas DC9 Super 80 /MD81/2/3/8	4%
	645 / McDonnell-Douglas DC9 Super 87	4%
b757	622 / Boeing 757-200	14%
	623 / Boeing 757-300	2%
b767	626 / Boeing 767-300/300er	36%
	625 / Boeing 767-200/200er	26%
	624 / Boeing 767-400	22%
dc10	730 / McDonnell Douglas Dc-10-10	42%
	732 / McDonnell Douglas Dc-10-30	57%
	733 / McDonnell Douglas Dc-10-40	23%
b747o	816 / Boeing 747-100	57%
	817 / Boeing 747-200/300	51%
	822 / Boeing 747sp	34%
dc9	640 / McDonnell Douglas Dc-9-30	6%
	650 / McDonnell Douglas Dc-9-50	5%
	630 / McDonnell Douglas Dc-9-10	7%
	645 / McDonnell Douglas Dc-9-40	12%
b747	819 / Boeing 747-400	38%
	820 / Boeing 747f	100%
b737o	620 / Boeing 737-100/200	4%
	621 / Boeing 737-200c	38%
a320	694 / Airbus Industrie A320-100/200	5%
	698 / Airbus Industrie A319	4%
	699 / Airbus Industrie A321	4%
l1011	760 / Lockheed L-1011-1/100/200	18%
	765 / Lockheed L-1011-500 Tristar	17%
md11	740 / Mcdonnell Douglas Md-11	76%
b777	627 / Boeing 777	32%
b737n	614 / Boeing 737-800	3%
	612 / Boeing 737-700/700lr	4%
	634 / Boeing 737-900	3%
b717	608 / Boeing 717	1%

Name	F41 code/name	Estimated fraction of cumulative activity in freight
e145	675 / Embraer 145	0%
tfan	629 / Canadair Rj-200er	2%
	603 / Fokker 100	3%
	674 / Embraer-135	0%
	868 / British Aerospace Bae-146-100	1%
	628 / Canadair Rj-100/Rj-100er	8%
	835 / Avroliner Rj85	0%
	602 / Fokker F28-4000/6000 Fellowship	3%
	867 / British Aerospace Bae-146-200	8%
	601 / Fokker F28-1000 Fellowship	3%
	866 / British Aerospace Bae-146-300	0%
tprp	456 / Saab-Fairchild 340/B	4%
	461 / Embraer Emb-120 Brasilia	1%
	441 / Aerospatiale/Aeritalia Atr-42	0%
	416 / Cessna 208 Caravan	98%
	483 / Dehavilland Dhc8-100 Dash-8	2%
	442 / Aerospatiale/Aeritalia Atr-72	0%
	469 / British Aerospace Jetstream 31	0%
	445 / Convair CV-660	5%
	405 / Beech 1900 A/B/C	7%
	467 / Swearingen Metro Iii	35%
	471 / British Aerospace Jetstream 41	0%
	449 / Dornier 328	1%
	489 / Shorts 360	4%
	484 / Dehavilland Dhc8-300 Dash-8	0%
	408 / British Aerospace Bae-Atp	0%
	450 / Fokker Friendship F-27	32%
	550 / Lockheed L-188a/C Electra	89%
	430 / Convair Cv-580	86%
	435 / Convair Cv-600	100%

4.7. Use of airport-specific operational data

In the absence of geographic specificity for local and regional effects, airport-specific operational and demographic data are aggregated probabilistically using a sample of airports. The resulting probability function conveys geographic variability to the damage function. A unified source of airport-specific operations data does not exist for the US that differentiates by aircraft. These data are necessary for the consistent evaluation of local effects, both air quality and noise-related. Total historical airport operations since 1976 are available from issues of the FAA Terminal Area Forecast (TAF), but not at the resolution of representative aircraft types.[1] As a compromise, operations were disaggregated among major airports, and thus counties, using a distribution of operations developed for use in the FAA Model for Assessing Global Exposure to the Noise of Transport Aircraft or MAGENTA (CAEP 2001b, 2001a).

In application to MAIPA, this database is applied to partition operations reported via F41T2 and F98A1 among major airports in the US and makes it possible to localize the evaluation of changes in environmental variables. Only the 96 airports in the US (out of 1724 civil airports worldwide) for which detailed operational and route data are available are used in this analysis. Operations at these airports are categorized by aircraft type using the descriptive conventions of INM version 5.2a and derive from the scheduled passenger, cargo, and charter operations reported in the Official Airline Guide (OAG ref). Using the OAG in combination with tower operations records obtained directly from each airport, the MAGENTA database specifies the average daily number of operations for each INM-type aircraft.

One difficulty in applying the MAGENTA database is that its total operations counts are inconsistent with DOT reported data. The OAG is a near-term forecast based on expected schedules reported by airlines. This is qualitatively different from the F41T2 and F98A1 data, which are historical. At the time this analysis was completed, an aircraft-type differentiated activity database was available for 2002 only. The 96 MAGENTA airports do not cover all operations reported on F41T2 and F98A; there are 22% fewer operations reported for 2002 than for MAIPA representative aircraft types.[2] By government agreement, airports in the MAGENTA database must remain anonymous, designated only by an index, so it is not possible to fully explain these differences. This is an impetus for a generalized model of noise exposure, treating this 96 airport set as a sample for the purposes of determining marginal damages.

To associate demographic data with each of these airports, it is necessary to designate a specific location for each of the 96 indices. It is likely that the 96 airports correspond to one of the 134 large, medium, and small hubs reported in the TAF for 2002. To attribute demographic and air quality data, airports were matched to counties based on a likelihood indicator composed of the root mean square of population

[1] The TAF designates airports in the US as either large, medium, small, or non-hubs, based on the number of enplanements handled. Large hub airports each process >1% of total enplanmenents; medium hubs, small hubs, and non-hubs process 0.25-0.99%, 0.05-0.24% and <0.05% respectively. There were 32 large hubs, 35 medium hubs, 67 small hubs, and 340 non-hubs in 2002.

[2] Comparing total MAGENTA operations (including military and general aviation) with operations at the 134 TAF airports suggests that the MAGENTA airports account for approximately 62% of reported movements.

density and number of operations. Table A4.7 lists representative aircraft type assignments for MAGENTA aircraft model identifiers using this procedure. To represent the uncertainty in using this method to identify airports, data for the three airports most closely matching the MAGENTA indicator value were carried through the analysis, combined through Monte Carlo sampling to derive demographic, air quality, and operations parameters for each airport location. To estimate costs associated with changes in noise (and air quality) in a manner consistent with the DOT activity data, total MAGENTA operations were summed over all airports and scaled by a factor equal to the ratio of these operations to total F41T2/F98A1 operations for the associated representative aircraft type. This reconciliation preserves the ratio of movements among all airports for a specific representative aircraft type, and changes the ratio of movements among representative aircraft types at a specific airport by < 5%.

Table A4.7. Representative aircraft type assignments for MAGENTA aircraft model identifiers

- aircraft models are listed by their DOT Form 41 identifiers
- gray-shaded entries indicate out-of-production models

Name	MAGENTA equivalents	MAGENTA substitutions
b727	727D17 727EM1 727EM2 727QF 707QN 720 720B	707C56
b737	7373B2 737300 737400 737500	none
md80	MD81 MD83 MD83	none
b747o	747100 747200 74710Q 74720A 74720B 747SP	7473G2
b747	747400	none
b757	757PW 757RR	757300
b767	767300 767400 767CF6 767JT9	none
dc10	DC1010 DC1030 DC1040	none
dc9	DC93LW DC950 DC95HW DC9Q9 DC3 DC6 DC850 DC870 DC8QN	DC9Q7 DC86BT
b737o	737N17 737N9 737	73717A
a320	A319 A320 A32023	A321 A32123
l1011	L1011 L10115	L188
md11	MD11GE MD11PW MD9025	none
b777	777200 777300	none
b737n	737700	737800
b717	717200	none
e145	EMB145 EMB14L	none
tfan	BAE146 F10062 F10065 F28MK2 F28MK4	none
tprp	CVR580 DHC830 EMB120 SF340 DHC8 BAC111 BEC300 DHC6 DHC7 HS748A SD330	ATR42 BEC200 DO328 FK70 SAMER2 SAMER4

Appendix

5. Flight performance model

This section describes the definition of the nominal representative aircraft types flight profile in reference to flight procedures, and the parametric model of aircraft performance used to estimate flight operation and time-in-mode.

5.1. Flight distance specification

Distributions for flight distance are determined using quarterly F41T2 data from 1991-2003. Flight distance is calculated as total RAK divided by total revenue departures performed (RAD) for the F41 identifying codes corresponding to the aircraft models specified in tables A4.2 and A4.3.[1] Given the limited number of values (n = 52), it is not possible to specify a form for the long-term distribution that characterizes distance data. Thus, the probability distribution is specified based on this data as:

(A5.1)
$$P\left(d_{fl}\right) = P\left\langle \text{unif}, \min(d_{fl}), \max(d_{fl})\right\rangle$$

This restricts MAIPA to estimation of fleet-averaged performance for any representative aircraft type. This distribution applies to the average flight over time, not the flight-to-flight variability within a particular time period.

5.1.1. Corrections for deviations from great circle flight distance

Reported distance data are based on city-pair great circle stage lengths. In service, dsl will be longer as a result of air traffic controller or pilot decisions to deviate. A distribution for deviation in latitude and

[1] Scheduled rather than actual departures should be used because RAK in F41T2 is determined based on scheduled departures as detailed in table A4.1 (cf. appendix 4). However, scheduled departures are not available from the U.S. Bureau of Transportation Statistics, thus the use of actual departures. This introduces a upward distance bias into the analysis of an unknown amount.

longitude ($\Delta\hat{d}_{sl}$) is derived from an analysis of actual trajectory displacements calculated as the difference between FAA ETMS radar-tracked flight positions and great circle routes (FAA 2003b).[2]

Deviations $\hat{\delta}\langle d_{fl}\rangle$ are input as perpendicular great circle distances from the halfway point of the flight using the experimental distribution $\hat{\delta}\langle d_{fl}\rangle$= Pex[midpoint deviation]; the deviation increases with flight distance. The revised distance is calculated as the sum of two great circle routes to and from the newly located halfway point. Because the Pex[midpoint deviation] is derived from individual flight statistics, it overestimates the deviation contribution to variance by an unknown amount.

5.2. Flight profile segmentation for performance schedules

There are several options for setting flight rules given the description in equation A3.1 (appendix A3). Using aircraft control inputs, any pair of flight speed, throttle (F), and rate of climb/descent (dh/dt) can be specified to set performance, given a description of aircraft drag and weight. Flight rules are also significantly influenced by safety regulations, air traffic control, and navigational requirements.[3] These restrict possible sets of performance characteristics, particularly at altitudes below ~3 km where airspace hazards increase. From the perspective of MAIPA performance modeling, this is advantageous since it reduces variation among model types.

5.2.1. Flight profile segmentation by performance mode

For MAIPA, vertical flight profiles, h(t), are described in nine segments (nseg = 9). Four operational modes below the nominal mixing height (h<bl) are set for the landing-takeoff (LTO) cycle: idle/taxi, take-off, climb, and approach. For the LTO cycle, trajectories are specified through standardized methods reported in the Society of Automotive Engineers (SAE) Aerospace Information Report 1845 (AIR, see SAE 1986) for the FAA Integrated Noise Model (INM, see Bishop and Mills 1992; FAA 1999).

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[2] A simplified implementation of the FAA dispersion correction, which is specified as a function of distance, is used. For MAIPA, a distribution of lateral deviations is specified at only the midpoint of the flight, from which a randomized deviation is selected. The factor (λsl) is then determined using a geometric argument in equation E5.1.

(E5.1) $$\lambda_{SL} = \sqrt{\left(\hat{d}_{SL}/4 + \Delta\hat{d}_{SL}\right)/\hat{d}_{SL}}$$

[3] Within the boundaries of flight rules and physics, there is variability in flight profiles among aircraft types dependent upon market application, which correlates with aircraft size. For design and market evaluation, manufacturers conventionally use speed and throttle schedules to specify standardized flight profiles over a great circle route. Performance objectives center around achieving best fuel economy.

An additional five operational modes above the nominal mixing height (h<bl) approximate the en route portion of the flight (h>bl); two additional climb segments, cruise, and two additional approach segments. En route profile and performance follow Eurocontrol Base of Aircraft Data (BADA) schedules (cf. Eurocontrol 2003).[4]

5.2.2. Application of FAA INM and Eurocontrol BADA flight procedures

Trajectories below and above the nominal mixing height are equated, respectively, to the FAA INM and the BADA standard procedures closest to the constituent F41 models specified for each representative aircraft type. Where there are several designations that fit a constituent aircraft model, only designations referencing models flying in the analysis period are selected, using data provided by Wyle Laboratories that specific airframe-engine combinations to flight frequencies (see table A4.7 in appendix A4). If more than one model exists, a particular profile is chosen at random among the options for each simulation iteration.

Table A5.1 lists the INM and BADA identifier equivalents to F41-coded models for representative aircraft types. Figure A5.1 illustrates MAIPA flight profile definitions referenced in the following discussions of performance specification.

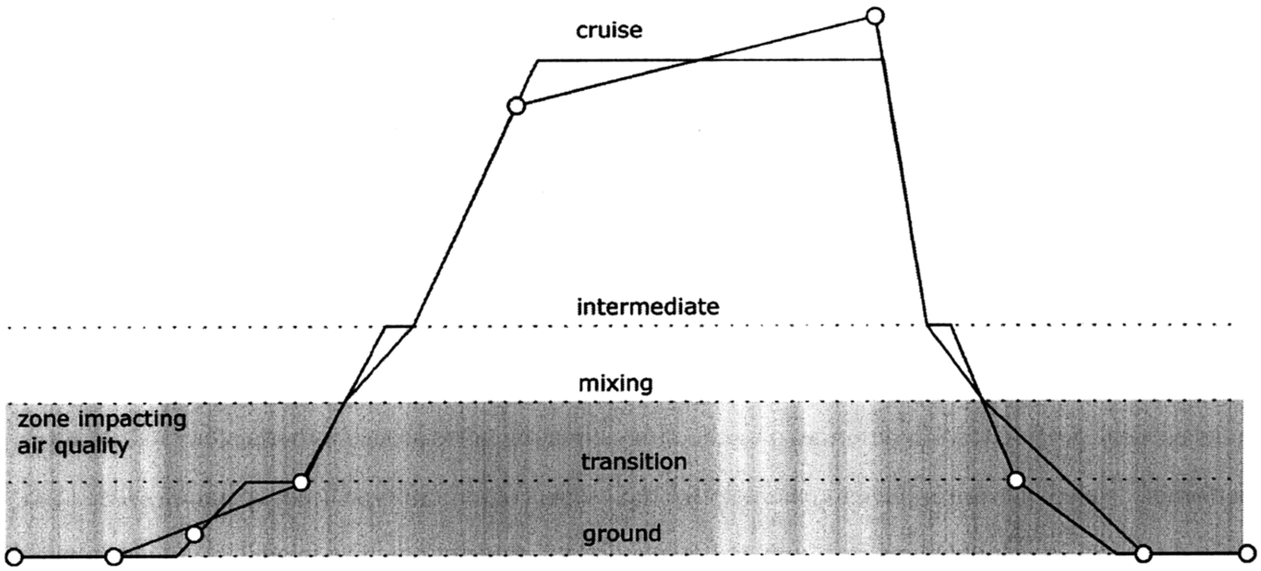
[4] INM version 6.0c and BADA version 3.5 are used for this analysis.

Table A5.1. Representative aircraft type assignments for INM and BADA aircraft model identifiers

Name	F41 code/name	BADA equivalent	INM equivalent(s)
tprp	456 / Saab-Fairchild 340/B	SF34	SF340
	461 / Embraer Emb-120 Brasilia	E120	EMB120
	441 / Aerospatiale/Aeritalia Atr-42	AT43	SF340, EMB120, DHC8, CVR580, DHC830, L188
	416 / Cessna 208 Caravan	JS31	SF340, EMB120, DHC8, CVR580, DHC830, L188
	483 / Dehavilland Dhc8-100 Dash-8	DH8C	DHC8
	442 / Aerospatiale/Aeritalia Atr-72	AT72	SF340, EMB120, DHC8, CVR580, DHC830, L188
	469 / British Aerospace Jetstream 31	JS31	SF340, EMB120, DHC8, CVR580, DHC830, L188
	445 / Convair CV-660	AT43	CVR580
	405 / Beech 1900 A/B/C	JS31	SF340, EMB120, DHC8, CVR580, DHC830, L188
	467 / Swearingen Metro Iii	JS31	SF340, EMB120, DHC8, CVR580, DHC830, L188
	471 / British Aerospace Jetstream 41	JS41	SF340, EMB120, DHC8, CVR580, DHC830, L188
	449 / Dornier 328	D328	SF340, EMB120, DHC8, CVR580, DHC830, L188
	489 / Shorts 360	SH36	SF340, EMB120, DHC8, CVR580, DHC830, L188
	484 / Dehavilland Dhc8-300 Dash-8	DH8C	DHC830
	408 / British Aerospace Bae-Atp	ATP	SF340, EMB120, DHC8, CVR580, DHC830, L188
	450 / Fokker Friendship F-27	F27	SF340, EMB120, DHC8, CVR580, DHC830, L188
	550 / Lockheed L-188a/C Electra	ATP	L188
	430 / Convair Cv-580	AT43	CVR580
	435 / Convair Cv-600	AT43	CVR580

Name	F41 code/name	BADA equivalent	INM equivalent(s)
b727	715 / Boeing 727-200/231a	B722	727200, 727Q15, 727Q9, 727D15, 727D17
	710 / Boeing 727-100	B722	727100, 727Q7
	711 / Boeing 727-100c/Qc	B722	727EM1, 727EM2, 727QF
b737	619 / Boeing 737-300	B733	737300, 7373B2
	616 / Boeing 737-500	B735	737400
	617 / Boeing 737-400	B734	737500
	618 / Boeing 737-300lr	B733	737300, 7373B2
md80	655 / McDonnell-Douglas DC9 Super 80 /MD81/2/3/8	MD80	MD81, MD82, MD83
	645 / McDonnell-Douglas DC9 Super 87	MD80	MD83
b747o	816 / Boeing 747-100	B742	747100, 74710Q
	817 / Boeing 747-200/300	B743	747200, 74720A, 74720B
	822 / Boeing 747sp	B742	747SP
b747	819 / Boeing 747-400	B744	747400
	820 / Boeing 747f	B744	747100, 74710Q, 747400
b757	622 / Boeing 757-200	B752	757PW, 757RR
	623 / Boeing 757-300	B753	757PW, 757RR
b767	626 / Boeing 767-300/300er	B763	767300
	625 / Boeing 767-200/200er	B762	767CF6, 767JT9
	624 / Boeing 767-400	B763	767400
dc10	730 / McDonnell Douglas Dc-10-10	DC10	DC1010
	732 / McDonnell Douglas Dc-10-30	DC10	DC1030
	733 / McDonnell Douglas Dc-10-40	DC10	DC1040
dc9	640 / McDonnell Douglas Dc-9-30	DC9	DC910, DC93LW
	650 / McDonnell Douglas Dc-9-50	DC9	DC950, DC95HW
	630 / McDonnell Douglas Dc-9-10	DC9	DC910, DC9Q9
	645 / McDonnell Douglas Dc-9-40	DC9	DC930, DC93LW
b737o	620 / Boeing 737-100/200	B732	737, 737N9, 737QN, 737D17, 737N17
	621 / Boeing 737-200c	B732	737D17, 737N17
a320	694 / Airbus Industrie A320-100/200	A320	A320, A32023
	698 / Airbus Industrie A319	A319	A319
	699 / Airbus Industrie A321	A321	A320, A32023
l1011	760 / Lockheed L-1011-1/100/200	L101	L1011
	765 / Lockheed L-1011-500 Tristar	L101	L10115
md11	740 / McDonnell Douglas Md-11	MD11	MD11GE, MD11PW
b777	627 / Boeing 777	B772	777200, 777300
b737n	614 / Boeing 737-800	B738	737700
	612 / Boeing 737-700/700lr	B737	737700
	634 / Boeing 737-900	B738	737700
b717	608 / Boeing 717	B712	717200
e145	675 / Embraer 145	E145	EMB145, EMB14L

Figure A5.1. MAIPA nominal flight profile definition



5.2.3. Specification of mixing height

Although ISA conditions are assumed for aircraft performance purposes, MAIPA accounts for weather variability and geography (airport altitudes) in the estimation of fuel burn and emissions. The extent to which aircraft emissions may contribute to changes in air quality is dependent on the mixing height.

The mixing height, or inversion height, is the altitude at which the near-surface temperature slope switches sign and temperature begins to increase with altitude. This inversion defines the extent to which pollutants can rise; atmospheric turbulence can transport pollutants through this layer quickly. It is important to distinguish this physical parameter from the nominal mixing height h_{bl}^{ref}; the latter is a convenience for aircraft performance specification (cf. next section).

MAIPA uses twice daily (morning and afternoon) mixing height estimates calculated using the methods in Holzworth (1972) based on radiosonde measurements taken across the United States (ref NCDC). These data are weighted in each year 1991-2003 by relative airport operations to derive equation A5.2.

(A5.2) $$\hat{h}_{mix} = \mathrm{P}_{\exp r}\left[\text{mixing height}\right]$$

5.3. Flight performance schedule within the atmospheric boundary layer

MAIPA defines performance within the atmospheric boundary layer with four segments. The four-mode LTO-cycle is often employed in regulatory emissions analyses. EPA (1985) initially suggested generic time-in-mode specifications for commercial aircraft operating out of large, congested metropolitan airports based on aircraft data from February 1980. However, these specifications are out of date and do not address expected variability among aircraft types. Times-in-mode derived using SAE AIR 1845 supersede this guidance.

In its entirety, SAE AIR 1845 describes a method for estimating noise around airports. Only its aircraft performance elements are applied to parameterize representative aircraft type flight operation. SAE 1845 procedures reflect manufacturer, operator, and regulatory standards of practice, in addition to the aerodynamic and engine performance of the aircraft for a specified weight. These are contained in unique thrust schedules from ground to heights up to ~3 km for each INM aircraft type.

5.3.1. Departure performance

Departure performance depends on the take-off gross weight (WTO) which is determined in MAIPA as a function of $\hat{d}_{sl}$ using the equivalency in Bishop and Mills (1992). Only one approach specification is available. For all representative aircraft types, takeoff weight is based on the flight-by-flight comparison in Lee (2005) against proprietary data from a major carrier, accounting for potential bias due to fuel tankering.

(A5.3)
$$\hat{W}_{TO} = P_{norm}\left[\mu_{W_{TO}}\left(\hat{d}_{SL}\right), \sigma_{W_{TO}} = 10\%\right]$$

This is a coarse description; variability in the average across flights is likely less and 10% should be considered an upper bound. For comparison, $\mu_{W_{TO}}\left(\hat{d}_{sl}\right)$ can be estimated to within $\sigma = 1\%$ with a complete flight description, an accuracy available to operators in estimating fuel required for a particular flight.[5]

[5] M. Schofield, Rolls-Royce, plc., personal communication.

5.3.2. Specification of altitude triggers for changes in operational mode

In emissions estimation over the LTO-cycle, the FAA Emissions and Dispersion Modeling System (EDMS) designates the portion of the profile between start of take-off roll and the attainment of ~300 m altitude (nominal transition height, h_{trht}^{ref}) as in the take-off mode. Operation between the nominal mixing and transition heights is attributed to climb. Nominal transition height is chosen as a conservative estimate of the lowest height above ground at which a pilot might throttle back on a typical takeoff (EPA 1992).

5.3.3. Approach performance

For landing in EDMS, the approach mode is the full segment between the h_{bl}^{ref} and landing stop. A different approach is used for MAIPA to better represent procedural variability. A more liberal estimate might place the h_{bl}^{ref} at a lower altitude, perhaps ~150 m. Given this uncertainty, the transition height is specified with equation A5.4 for all representative aircraft types.

(A5.4) $$\hat{h}_{trht} = \mathrm{P}_{beta}\left[h_{low}, h_{high}^{EDMS}, \alpha = 6, \beta = 6\right]$$

The shape parameters in equation A5.4 are set to approximate a truncated normal distribution and the reference ground altitude is an operations-weighted, annually-specified distribution of airport altitudes as in equation A5.5.

(A5.5) $$\hat{h}_{grnd} = \mathrm{P}_{\exp r}\left[\text{airport altitude}\right]$$

5.3.4. Reconciliation of flight profile and SAE AIR 1845 performance schedule resolutions

INM schedules are provided at finer resolution than suggested by MAIPA profile segment divisions. This detail is maintained only insofar as to calculate time-in-mode, after which the resolution is reduced to correspond to the fidelity of certification fuel use and emissions data. These data, reported to the International Civil Aviation Organization (ICAO), are given for institutionally determined thrust settings, including an idle point. Assuming these settings apply to the LTO-cycle, uncertainty in the fuel consumption rate are quantified directly for each of the four LTO segments. Subsequent sections return to this calculation upon introducing the underlying fuel and emissions data.

5.3.5. Application of DOT ASQP ground time data

The SAE AIR 1845 standard does not address idle/taxi; DOT Airline Service Quality Performance (ASQP) reported taxi time data specifies the distribution in equation A5.6.

(A5.6) $$\hat{t}_{id/tx} = \mathrm{P}_{\mathrm{expr}}\left[\text{ASQP taxi time}\right]$$

The value of $\hat{t}_{id/tx}$ is the sum total of all ground movement prior to take-off roll and after landing stop. An empirical distribution of delay times is used based on algorithms described in Carr et al. (2002) for 1995 operations. Only domestic operations by the US majors are covered by ASQP. It is implicitly assumed that these data characterize regional and international operations as well. There are no alternative sources of publicly-available information reporting position for ground movements.

5.4. Flight performance schedule above the atmospheric boundary layer

MAIPA defines performance above the atmospheric boundary layer with five en route segments bounded by the mixing height, an intermediate height triggering a change in climb and descent rules, and the flight altitude. Figure 5.1 illustrates these segments: climb-to-intermediate, CTI; climb-to-altitude, CTA; cruise, CRU, decent-from-altitude, DFA, descent-from-intermediate, DFI.

5.4.1. Climb and descent performance

Commercial airlines usually employ procedures specifying climbs and descents at a constant calibrated air speed (CAS) and thrust setting. Below a height of ~3000 m (10000 ft), CAS is limited to ~460 km/hr (250 knots) as a safety precaution against increased airspace hazards. This height is referred to here as the intermediate height (hinter) and marks the MAIPA transition between the en route pairs of climb (CTI-CTA) and approach segments (DFI-DFA).

Below hinter, INM performance schedules are extended from the climb and approach segments below the mixing height. CTI is conducted at the time-weighted average dh/dt and pitch of the INM climb segment. Similarly, DFI is specified using the time-weighted average dh/dt and pitch of the INM approach segment. Based on comparisons with typical flight profiles supplied by a manufacturer, the accuracy of this extrapolation over the CTI and DFI segments is better than the uncertainty in the time-weighted speeds and pitches derived from the INM profile.

Above hinter, a constant climb or descent is continued at an increased CAS and thus higher thrust. This CAS, typically ~550 km/hr (300 knots), is set to correspond to the desired performance objective, usually to maximize time at cruise altitude in order to increase overall fuel economy.[6] Nominal rates of climb and descent, as well as speeds, are specified for representative aircraft types using BADA-estimated performance. Flight rules underlying BADA profile and performance specifications are derived from inspection of airline flight manuals and represent the nominal settings for a particular aircraft type. Speeds are tabulated as performance tables, at altitude intervals of ~2000 m; climb speeds are a function of takeoff weight.

For CTA and DFA, speed tables are integrated between hinter and halt to determine time-in-mode. For CTA, climb rates are a function of weight. Where TAS needs to be derived from CAS, equation A6.6 (see following appendix 6) provides atmospheric conditions.[7] As with the performance specification below h_{bl}^{ref}, BADA profiles are reconciled to match the lower performance resolution of the MAIPA five mode en route schedule.

5.4.2. Cruise performance

For all representative aircraft type at cruise, a constant M rule is set at a constant altitude. Cruise tn is defined for constant M at a prescribed altitude over the net of dSL and all climb or descent segments. Mach number and service ceiling (hceil) are set in accordance with the BADA characterization. The probability function for M is specified with equation A5.7; this is an engineering guess and assumed to account for different performance optimization objectives as well as weather conditions.

(A5.7)
$$P_{\mathrm{expr}} = \left[M_{\mathrm{BADA}}^{ref}, 2.5\%\right]$$

5.4.3. Specification of flight altitude

Flight altitude is specified as a deviation below hceil ($\Delta\varepsilon_{h_{alt}}$) derived from the FAA analysis of ETMS-recorded flights (FAA 2003b) (equation A5.8).

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[6] At higher altitudes, past the tropopause, the governing climb and descent rules may be replaced by a constant M such that the greatest speed per unit of fuel use is obtained (thus minimizing total expenditure).

[7] Descent is practically restricted by passenger comfort and limits on the cabin depressurization rate.

(A5.8) $$\Delta\varepsilon_{h_{alt}}\,(P_{\exp r} = \left[f\left(\hat{d}_{sl}\right) < \mathrm{v} \geq 925\mathrm{m}\right])$$

In this thesis, which uses F41 operations data, the latter applies only to calculations involving regional flights.[8] In cases where the randomly chosen flight altitude is not achievable because the flight distance is too short (i.e. no cruise, truncated climb and descent profiles), $\hat{h}_{alt}$ is adjusted incrementally until climb is just completed and descent just starts.

5.5. Performance model uncertainties

5.5.1. Parametric uncertainties in flight performance schedule

Performance model uncertainties are implemented through: (1) the choice of the appropriate performance schedule based on uncertainties $\hat{d}_{sl}$; (2) uncertainty in the location of the reference altitudes dividing operational segments; and (3) uncertainties in the parameters of the particular schedule selected for the Monte Carlo iteration.

Equation A5.8 does not specify the uncertainty in the ICAO and BADA characterization of typical flight operations. Based on comparisons with published SFC data, Lee (2005) estimates a σ=10% uncertainty in the BADA specification applied to a particular flight. Similarly, measurements recently taken for a large commercial engine suggest flight-to-flight variability would be within an absolute range of 15-20% over all power settings. While these characterizations over-specify the uncertainty in the average quarterly operation and are not comprehensive for all aircraft types, they are currently the only public sources available.

The parametric uncertainties of the third element listed are introduced in the computation of times-in-mode with normal distribution. Its standard deviation is specified from comparisons of computed flight profiles with proprietary carrier data for departure and approach segments respectively (Lee 2005). As indicated in equations A5.9 and A5.10, the value of the standard deviation below the mixing height is different from its value above the mixing height.

(A5.9) $$\left[\hat{t}_{\mathrm{t/o}} \,\|\, \hat{t}_{\mathrm{c/o}} \,\|\, \hat{t}_{\mathrm{cti}} \,\|\, \hat{t}_{\mathrm{cta}}\right] = \mathrm{P}_{norm}\left[\hat{\mu} = t^{ref}_{\mathrm{SAE\|BADA}}, \hat{\sigma} = 10\%\right]$$

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[8] In their original form, the distribution functions are used to specify cruise altitude directly for SAGE OAG flights.

(A5.10) $$\left[\hat{t}_{\text{ap}} \,||\, \hat{t}_{\text{dfi}} \,||\, \hat{t}_{\text{dfa}}\right] = \mathrm{P}_{norm}\left[\hat{\mu} = t^{ref}_{\text{SAE||BADA}}, \hat{\sigma} = 25\%\right]$$

For LTO, times-in-mode $\hat{t}_{t/o}$, $\hat{t}_{c/o}$, and $\hat{t}_{\text{ap}}$ are determined by integrating the position (h and d) and speed schedule of the relevant INM profile at its original resolution. As calculated, $\hat{t}_n$ during take-off roll, which occurs at ground, is included in the summation for the take-off segment. Similarly, landing stop is included in the approach segment.

5.5.2. Structural uncertainty in SAE 1845 performance specifications

A limited validation of SAE AIR 1845 procedures was undertaken in a study of LTO profiles at Seattle-Tacoma airport (Flathers 1982), which formed the basis of profile updates included in the INM database used here (Bishop and Mills 1992). Using terminal area radar data, it is possible to make an assessment of the extent to which these flight profiles characterize the actual two-dimensional profile. There is also an issue with route dispersion as indicated previously. However, there has been no published assessment of these two sources of uncertainty.

SAE AIR 1845 references sea-level (ISA) conditions for a standard-day at 298 K (59°F) with WTO at 85% of maximum and landing gross weight at 90% of maximum. Other influential assumptions in SAE AIR 1845 include specification of 100% rated thrust and an 8 knot headwind on take-off, resulting in a potential overestimate of fuel use (and emissions) over LTO segments, particularly if the pilot elects not to depart at full throttle (or at least an unknown variability if weather conditions differ).

For example, the FAA and CSSI Inc. have estimated that cutback would typically result in a ~5-15% reduction in total NO_x emissions over the LTO-cycle for a range of aircraft types. Procedures have been developed for the FAA where noise minimization is a key driver and thrust is cutback during take-off (Bishop and Mills 1992). Cutback is often used in actual procedures, requiring additional take-off roll but lower fuel use. However, apart from representative aircraft types associated with hush-kitted INM equivalents, operational noise techniques such as cutback are not currently accommodated by MAIPA. In addition, no thrust reverse is modeled on landing.

5.5.3. Structural uncertainty in Eurocontrol BADA performance specifications

Lee (2005) determined a less than 5% absolute difference between fuel use derived using BADA performance descriptions over a profile designated using SAE 1845, versus actual fuel use as recorded by a flight computer. This is provided that weight is specified correctly and that the profile corresponds to the actual procedure. These are structural uncertainties and relate to the assumptions underlying the performance specifications.

The definition of the five en route segments is based on the availability of a reasonable mechanism through which to estimate uncertainty in performance specifications along these trajectories; roughly, en route segments are characterized by similar flight rules. MAIPA ignores step transitions, where, for example, the aircraft may change speed at a constant altitude to move to another climb point. Profile simplification in MAIPA leads to a negative bias error in the rate of fuel consumption.

Based on comparisons with data and higher fidelity models using radar position information, presented in chapter 3, these inaccuracies have a significantly lower effect on per-flight fuel consumption and emissions than their overall uncertainties. A simplified flight performance model has the advantage of reducing the complexity of performance specification where additional fidelity would represent only a marginal improvement in the estimation of costs. However, further refinement represents a benefit where the objective is an accurate and precise flight-by-flight fuel consumption estimate.

Appendix

6. Estimation of fuel consumption rate

This appendix details the application of engine measurement data to estimate fuel consumption rates based on representative aircraft type performance parameters.

6.1. Representative aircraft type assignments for engine types

Engines are matched to each aircraft model of a representative aircraft type based on the relevant FAA type certification sheet using the closest equivalents from the ICAO Engine Exhaust Emissions Data Bank (ICAO 1995), the same data source used to specify emission indices for NO_x, HC, and CO, as well as the smoke number. At every simulation iteration, one of these engines is selected randomly with equal probability. Table A6.1 shows these equivalencies.

Table A6.1. Representative aircraft type assignments for ICAO engine model identifiers

Gray-shaded entries indicate out-of-production models

Name	F41 code/name	ICAO equivalent(s)
b727	715 / Boeing 727-200/231a	1PW004=JT8D-7series(smoke fix); 1PW005-JT8D-7series(reduced emissions)
		1PW006=JT8D-9series(smoke fix); 1PW007=JT8D-9series(reduced emissions)
		1PW008=JT8D-11; 1PW009=JT8D-15(smoke fix)
		1PW010=JT8D-15(reduced emissions); 1PW011=JT8D-15A
		1PW012=JT8D-17(smoke fix); 1PW013=JT8D-17(reduced emissions)
		1PW014=JT8D-17A; 1PW015=JT8D-17AR; 1PW016=JT8D-17R
		1PW018=JT8D-217series; 4PW070=JT8D-217C
		1PW019=JT8D-219; 4PW071=JT8D-219(enviro.kit)
	710 / Boeing 727-100, and 711 / Boeing 727-100c/Qc	1PW004=JT8D-7series(smoke fix); 1PW005-JT8D-7series(reduced emissions)
		1PW006=JT8D-9series(smoke fix); 1PW007=JT8D-9series(reduced emissions)
		1PW008=JT8D-11; 1PW009=JT8D-15(smoke fix)
		1PW010=JT8D-15(reduced emissions); 1PW011=JT8D-15A
		1PW012=JT8D-17(smoke fix); 1PW013=JT8D-17(reduced emissions)
		1PW018=JT8D-217series; 4PW070 JT8D-217C
		1PW019=JT8D-219; 4PW071 JT8D-219(enviro.kit)
		3RR033=TAY651(Transply); 3RR032= TAY651(Pedhead)
b737	619 / Boeing 737-300, and	1CM004=CFM56-3-B1; 1CM005=CFM56-3B-2
	616 / Boeing 737-500, and	1CM006=CFM56-3C-1(rerated); 1CM007=CFM56-3C-1
	617 / Boeing 737-400, and	
	618 / Boeing 737-300lr	
md80	655 / McDonnell-Douglas DC9 Super 80 /MD81/2/3/8	1PW017=JT8D-209; 1PW018=JT8D-217series; 4PW068=JT8D-217(enviro.kit)
		4PW069=JT8D-217A; 4PW070=JT8D-217C
		1PW019=JT8D-219; 4PW071=JT8D-219(enviro.kit)
	645 / McDonnell-Douglas DC9 Super 87	1PW018=JT8D-217series; 4PW069=JT8D-217A; 4PW070=JT8D-217C 1PW019=JT8D-219; 4PW071=JT8D-219(enviro.kit)

Name	F41 code/name	ICAO equivalent(s)
b747o	816 / Boeing 747-100	1PW020=JT9D-7; 1PW021=JT9D-7A; 1PW022=JT9D-7F(ModV)
		1PW023=JT9D-7F(ModVI); 1PW024=JT9D-7J; 1PW029=JT9D-7R4G2
		1RR004=RB211-524Bseries(Package1)
		1RR005=RB211-524Bseries(Phase2); 1RR006=RB211-524C2
		1RR007=RB211-524D4series(Package1)
		1RR008=RB211-524D4series(Phase2)
		1GE005=CF6-45A2; 3GE068=CF6-45A2(low emis fuel noz)
	817 / Boeing 747-200/300	1PW020=JT9D-7; 1PW021=JT9D-7A; 1PW022=JT9D-7F(ModV)
		1PW023=JT9D-7F(ModVI); 1PW024=JT9D-7J; 1PW035=JT9D-70A
		1PW025=JT9D-7Q; 1PW029JT9D-7R4G2
		3GE075=CF6-50E; 3GE076=CF6-50E1; 1GE009=CF6-50E2
		3GE077=CF6-50E2(low emis fuel noz)
		1GE022=CF6-80C2B1(#1); 2GE041=CF6-80C2B1(#2)
		1RR004=RB211-524Bseries(Package1)
		1RR005=RB211-524Bseries(Phase2)
		1RR006=RB211-524C2; 1RR007=RB211-524D4series(Package1)
		1RR008=RB211-524D4series(Phase2)
	822 / Boeing 747sp	1PW020=JT9D-7; 1PW021=JT9D-7A; 1PW022=JT9D-7F(ModV)
		1PW023=JT9D-7F(ModVI); 1PW024=JT9D-7J; 1PW029= JT9D-7R4G2
		1RR004=RB211-524Bseries(Package1); 1RR005RB211-524Bseries(Phase2) 1RR006=RB211-524C2; 1RR007=RB211-524D4series(Package1)
		1RR008=RB211-524D4series(Phase2)
		1GE005=CF6-45A2; 3GE068CF6-45A2(low emis fuel noz)
b747	819 / Boeing 747-400	1PW041=PW4056; 1PW042=PW4056(reduced smoke)
		1GE023=CF6-80C2B1F(#1); 1GE024=CF6-80C2B1F(#2)
		2GE045=CF6-80C2B1F(#3); 3GE057=CF6-80C2B5F
		1RR009=RB211-524G(#1); 1RR010=RB211-524G(#2),
		1RR011=RB211-524H; 4RR037=RB211-524H-T
	820 / Boeing 747f	1PW041=PW4056; 1PW042=PW4056(reduced smoke)
		1GE023=CF6-80C2B1F(#1); 1GE024=CF6-80C2B1F(#2) 2GE045=CF6-80C2B1F(#3); 3GE057=CF6-80C2B5F
		1RR009=RB211-524G(#1); 1RR010=RB211-524G(#2) 1RR011=RB211-524H; 4RR037=RB211-524H-T
		1PW020=JT9D-7; 1PW021=JT9D-7A; 1PW022=JT9D-7F(ModV); 1PW023=JT9D-7F(ModVI); 1PW024=JT9D-7J; 1PW035=JT9D-70A 1PW025=JT9D-7Q; 1PW029=JT9D-7R4G2; 3GE075=CF6-50E
		3GE076=CF6-50E1; 1GE009=CF6-50E2; 3GE077=CF6-50E2(low emis fuel noz),
		1GE022=CF6-80C2B1(#1); 2GE041=CF6-80C2B1(#2)
		1RR004=RB211-524Bseries(Package1); 1RR005=RB211-524Bseries(Phase2)
		1RR006= RB211-524C2; 1RR007=RB211-524D4series(Package1) 1RR008=RB211-524D4series(Phase2)

Name	F41 code/name	ICAO equivalent(s)
b757	622 / Boeing 757-200	757PW, 757RR
	623 / Boeing 757-300	757PW, 757RR
b767	626 / Boeing 767-300/300er	767300
	625 / Boeing 767-200/200er	767CF6, 767JT9
	624 / Boeing 767-400	767400
dc10	730 / McDonnell Douglas Dc-10-10	DC1010
	732 / McDonnell Douglas Dc-10-30	DC1030
	733 / McDonnell Douglas Dc-10-40	DC1040
dc9	640 / McDonnell Douglas Dc-9-30	DC910, DC93LW
	650 / McDonnell Douglas Dc-9-50	DC950, DC95HW
	630 / McDonnell Douglas Dc-9-10	DC910, DC9Q9
	645 / McDonnell Douglas Dc-9-40	DC930, DC93LW
b737o	620 / Boeing 737-100/200	737, 737N9, 737QN, 737D17, 737N17
	621 / Boeing 737-200c	737D17, 737N17
a320	694 / Airbus Industrie A320-100/200	A320, A32023
	698 / Airbus Industrie A319	A319
	699 / Airbus Industrie A321	A320, A32023
l1011	760 / Lockheed L-1011-1/100/200	L1011
	765 / Lockheed L-1011-500 Tristar	L10115
md11	740 / McDonnell Douglas Md-11	MD11GE, MD11PW
b777	627 / Boeing 777	777200, 777300
b737n	614 / Boeing 737-800	737700
	612 / Boeing 737-700/700lr	737700
	634 / Boeing 737-900	737700
b717	608 / Boeing 717	717200
e145	675 / Embraer 145	EMB145, EMB14L
tfan	629 / Canadair Rj-200er	CL600, CL601
	603 / Fokker 100	F10062, F10065
	674 / Embraer-135	EMB145
	868 / British Aerospace Bae-146-100	BAE146
	628 / Canadair Rj-100/Rj-100er	CL600, CL601
	835 / Avroliner Rj85	BAE146
	602 / Fokker F28-4000/6000 Fellowship	F28MK4
	867 / British Aerospace Bae-146-200	BAE146
	601 / Fokker F28-1000 Fellowship	F28MK2
	866 / British Aerospace Bae-146-300	BAE300

Name	F41 code/name	ICAO equivalent(s)
tprp	456 / Saab-Fairchild 340/B	SF340
	461 / Embraer Emb-120 Brasilia	EMB120
	441 / Aerospatiale/Aeritalia Atr-42	SF340, EMB120, DHC8, CVR580, DHC830, L188
	416 / Cessna 208 Caravan	SF340, EMB120, DHC8, CVR580, DHC830, L188
	483 / Dehavilland Dhc8-100 Dash-8	DHC8
	442 / Aerospatiale/Aeritalia Atr-72	SF340, EMB120, DHC8, CVR580, DHC830, L188
	469 / British Aerospace Jetstream 31	SF340, EMB120, DHC8, CVR580, DHC830, L188
	445 / Convair CV-660	CVR580
	405 / Beech 1900 A/B/C	SF340, EMB120, DHC8, CVR580, DHC830, L188
	467 / Swearingen Metro Iii	SF340, EMB120, DHC8, CVR580, DHC830, L188
	471 / British Aerospace Jetstream 41	SF340, EMB120, DHC8, CVR580, DHC830, L188
	449 / Dornier 328	SF340, EMB120, DHC8, CVR580, DHC830, L188
	489 / Shorts 360	SF340, EMB120, DHC8, CVR580, DHC830, L188
	484 / Dehavilland Dhc8-300 Dash-8	DHC830
	408 / British Aerospace Bae-Atp	SF340, EMB120, DHC8, CVR580, DHC830, L188
	450 / Fokker Friendship F-27	SF340, EMB120, DHC8, CVR580, DHC830, L188
	550 / Lockheed L-188a/C Electra	L188
	430 / Convair Cv-580	CVR580
	435 / Convair Cv-600	CVR580

6.2. Specification of fuel consumption rate through performance schedules

ICAO fuel flow data are reported for sea-level static (SLS) conditions at power settings of 100%, 85%, 30%, and 7% rated output (ICAO 1995).[1] Measurement uncertainty in the reported ICAO $\dot{m}_f$ value is characterized as equation A6.1.

(A6.1) $$P_{norm} = \left[\hat{\dot{m}}_f^{\mathrm{ICAO}}, 1\%\right]$$

6.2.1. Fuel consumption rates within the atmospheric boundary layer

Over the LTO procedure, $\dot{m}_f$ for each representative aircraft type is determined based on comparison of the time-weighted average F over each segment to the certification setting using a piecewise linear interpolation to the F versus $\dot{m}_f$ relationship from the ICAO database. For MAIPA, F determined via the SAE AIR 1845 procedure is used directly rather than assuming the certification F represents actual use. This approximation is of a finer resolution than current regulatory practice for inventory development,

[1] In-service idle F and $\dot{m}_f$ may be up to 50% less in actual operation as compared to the certification setting. The impact of this variation was not examined for this thesis.

which applies the certification $\dot{m}_f$ without correction for thrust level (i.e. SAE AIR 1845 procedures are used only to calculate tn, after which certification $\dot{m}_f$ are applied). Uncertainty in F is specified by equation A6.2 based on comparisons with air carrier data (Lee 2005).

(A6.2) $$P_{norm} = \left[F^{ref}_{\mathrm{SAEvBADA}}, 10\%\right]$$

Comparison to a detailed proprietary specification of B777 flight performance indicates a range of $\dot{m}_f$ within 3% for en route segments, within 6% for take-off and climb, and within 20% for approach and idle/taxi. Based on these three analyses, $\dot{m}_f$ is specified as in equation A6.3 to account for interpolation error in the F versus $\dot{m}_f$ relationship.

(A6.3) $$\mathrm{P}_{norm}\left[\dot{m}^{ref}_f, 15\%\right]$$

6.2.2. Fuel consumption rates above the atmospheric boundary layer

Fuel flow above h^{ref}_{mix} is derived from the relevant BADA performance tabulation in a manner corresponding to the specification of speeds discussed in the previous section. BADA $\dot{m}_f$ references one engine type. For other engine types, an equivalent rated output is determined using the ICAO F versus $\dot{m}_f$ specification for the reference engine; $\dot{m}_f$ was determined using this rated output for the alternative engine. Fuel flow was integrated at the resolution of the BADA tabulation for the defined en route segments.

6.2.3. Propagated variability from performance schedule selection

Propagated variability in take-off F related to aircraft representative type and weight (WTO = [f(dSL)] is included through the choice of $\dot{m}_f$ for the particular LTO profile segment. Cruise $\dot{m}_f$ was also determined as a function of WTO. To the extent that aircraft take-off and climb at reduced power, resulting $\dot{m}_f$ will overestimate the actual. Without access to flight data records, it was not possible to assess the extent of this bias.

6.2.4. Correction to altitude conditions

With no change in flight speed, change in the required $\dot{m}_f$ for a given F is strongly correlated with ambient conditions. Sea level reference conditions can be corrected to equivalent altitude conditions at a given F. This is done by mapping changes in combustor inlet temperature (T3) as a function of h. T3 is a primary correlation parameter for emissions variability with engine conditions, but is typically proprietary. Thus, correlations are provided using $\dot{m}_f$ as the primary parameter, since data is available from the manufacturer that relates T3 and $\dot{m}_f$ from an emissions test.

This is the conceptual basis for Boeing Method 1, or BM1 (Baughcum et al. 1996), applied to correct $\dot{m}_f$ for h, M, and engine installation effects. Equation A6.4 shows the correlation and equation A6.5, the installation corrections.

(A6.4) $$\dot{m}_f^{ref} = \frac{\dot{m}_f^{alt}}{\delta}\theta^{3.8}e^{0.2M^2} \text{ where: } \delta = \frac{P_{amb}}{101325},\ \theta = \frac{T_{amb}}{288.15}$$

(A6.5) $$\dot{m}_f^{n'} = c_n\dot{m}_f^n \text{ where: } c_{t/o} = 1.010, c_{cl} = 1.013,\ c_{ap} = 1.020,\ c_{id/tx} = 1.100$$

The Boeing method, in its entirety, addresses the prediction of altitude emissions rates. Boeing Method 2, or BM2, is the more commonly employed version; the only difference with BM1 is that BM2 accounts impact of changes in M on T3. The error in using this method is unclear. The appropriate F for each engine application cannot be verified without further data. Error in using BM2 has been estimated for its this purpose and is discussed in the next appendix.

6.2.5. Atmospheric temperature and pressure

The 1976 U.S. standard atmosphere outlined in equation A6.6 specifies the variation of temperature and pressure (and humidity, see appendix A5.2) with altitude. In equation A6.6, R is the gas constant and γ is the specific heat ratio.

(A6.6) $$\left.\begin{aligned} T_{amb}^{ref} &= 288.15 - 0.0065h \\ P_{amb}^{ref} &= 101325\cdot\left(T_{amb}^{ref}/288.15\right)^{5.2579} \end{aligned}\right\}\text{below tropopause } (\leq 11000 \text{ m})$$

$$\left.\begin{aligned} T_{amb}^{ref} &= 216.65 \\ P_{amb}^{ref} &= 22619\cdot e^{-\left(\gamma/R\cdot T_{amb}^{ref}\right)\left(h-22619\right)} \end{aligned}\right\}\text{above tropopause } (>11000 \text{ m})$$

The tropopause marks the boundary between the troposphere and stratosphere; atmospheric conditions, such as the ambient temperature lapse rate, and chemical constituency change across this boundary. Like the boundary layer height, the tropopause altitude is not constant geographically or temporally; in this study, the tropopause is set at a constant height of 11000 m only to mark the change in atmospheric conditions. The ambient temperature profile is an uncertain parameter, defined by a systematic variability around the nominal profile as in equation A6.7. Thus, the profile shifts uniformly with the random variability.

(A6.7) $$\hat{T}_{amb} = T_{amb}^{ref} \pm \mathrm{P}_{unif}\left[\Delta T_{amb}\Big|_{1\%}^{99\%}\right].$$

6.2.6. Corrections to account for engine deterioration

The estimation methods described in this section for $\dot{m}_f$ are based on new production performance. To account for in-service deterioration effects, a bias error correction is applied, increasing $\dot{m}_f$. An increase in installed SFC of between 2-4% due to deterioration may be tolerated by an airline before an overhaul is contemplated. Part of this deterioration is unrecoverable. Based on the review in Lukachko and Waitz (1997), SFC is permanently degraded (i.e. unrecoverable via maintenance) by 1% upon engine use. Propagating this error through the equation for EU would lead to a 2-4% bias. It is not known to what extent the actual, in-service fleet is operating away from new aircraft performance. Lukachko and Waitz (1997) propagate this correction to an estimated change in emissions performance. Thus, this correction is applied at a later point, in the emissions inventory estimate (cf. appendix A7).

Appendix

7. Estimation of gaseous emissions indices

The following sections review MAIPA specification of emission indices (EI) as a function of flight performance. Section A7.1 addresses the estimation of EIs for species emitted in simple ratio with fuel use, CO2, H2O, and SO2. Section A7.2 presents the same for the regulated gaseous pollutants, NO_x, CO, and HC. Section A7.3 considers particulate emissions.

7.1. Emissions indices for species emitted in simple ratio to fuel flow

For species emitted in a simple ratio to fuel flow—carbon dioxide (CO2), water vapor (H2O), and sulfur oxides (SOx)—EIs were specified using fuel composition standards with corrections for engine performance, accounting for uncertainties in fuel specifications. EICO2 and EIH2O derive from typical fuel hydrogen/carbon (H/C) ratios with adjustments for combustion inefficiencies. Estimates of total sulfur emissions, EISOx as SO2, reference typical fuel sulfur levels.

7.1.1. Carbon dioxide and water vapor

For CO2 and H2O, calculation of per-flight emissions requires specification of the combustion efficiency (ηc) and the fuel hydrogen to carbon (H/C) ratio. The H/C ratio is a property of aviation fuel, standardized for certification but not in general fuel specifications.[1] ICAO specifies the H/C ratio (as % hydrogen by mass) for the purposes of aircraft emissions certification to be between 13.4-14.1% (ICAO 1993). This range is characteristic of actual aviation fuels in the U.S. and the U.K. (IPCC 1999). Since the distribution of the H/C ratio in fuel used in the fleet is unknown, calculations assume that any H/C ratio in the fuel specification range is equally possible, giving the probability functions in equations A7.1 and A7.2.

(A7.1) $$\widehat{EI}_{CO_2} = P_{unif}\left[3150, 3168\right] \text{ g-CO2/kg-fuel}$$

[1] Two specifications govern civil aviation fuels in the market: the American Society of Testing and Materials (ASTM) D1655 standard (DSTAN 2004); and the U.K. Defence Standardization Organization (DSTAN) 91-91 standard. The ASTM specification contains two relevant fuel designations (Jet A and Jet A-1), which differ only in their freezing points. The DSTAN specification addresses only Jet A-1.

(A7.2) $$\widehat{EI}_{H_2O} = P_{unif}\left[1206,1269\right] \text{ g-H2O/kg-fuel}$$

The error term represents the bias of the probability functions which assume a combustion efficiency (ηc) of 1. Adjusting for actual combustion efficiencies corrects these ranges for upward bias. Combustion efficiencies are estimated by the heating value of CO, HC, and PMnv emissions, footnote 2 steps through the computation of combustion efficiencies. For most operational conditions, the correction is << 0.1% for CO2 and H2O, small relative to the spread of their functional specifications. Over all operational conditions, the correction is less than 1% except for idle where the bias error can be up to 2% for recent designs and up to 7% for the oldest engines in the fleet.

To calculate EICO2 and EIH2O, a random EI is sampled from equations A7.1 and A7.2, and then decreased by a multiplicative factor (1- $\hat{\eta}_c$). Where data was not available, such as for en route segments, $\hat{\eta}_c$ is assumed at 99.9%, constant over all engine types. Section A7.2, following, describes estimated uncertainties in EICO and EIHC. Appendix 8 reviews the method for estimating EIPMnv. Uncertainties in EICO and EIHC are accounted prior to the calculation of the combustion efficiency. Thus, EICO2 and EIH2O are functions of operational condition only insofar as combustion efficiency changes.

7.1.2. Sulfur oxides

Total sulfur emissions are also dependent on fuel composition, but standards are less specific, limiting sulfur content to values below 0.3% by mass [EI(S) = %mass / 1000]. Fuel composition analyses suggest sulfur content is fairly constant worldwide from year-to-year at about 0.05% for commercial jet fuel, with a range of 0.035-0.07% since 1986 (EPA 1992; IPCC 1999). MAIPA implements this as in equation A7.3. Appendix 8 describes the method for estimating conversion of SO2 to volatile particulate matter (PMv).

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[2] **Note on estimation of combustion efficiencies**: CO, HC, and PMnv emissions result from combustion inefficiency, collectively representing the incomplete conversion of fuel hydrocarbons to their primary products of CO2 and H2O. Data for EICO and EIHC are available as a function of fuel consumption rate for the four, sea-level static (SLS) certification operating points (7%, 30%, 85%, and 100% rated thrust output) as specified in ICAO (1995). These data enable calculation of an approximate ηc at the four points using ratios of lower heating values (LHV) as shown in equation E7.1.

(E7.1) $$\hat{\eta}_c = \left[\widehat{EI}_{CO}\cdot LHV_{CO} + \widehat{EI}_{HC}\cdot \widehat{LHV}_{HC} + \widehat{EI}_{PM_{nv}}\cdot LHV_{PM_{nv}}\right] / \left(1000\cdot \widehat{LHV}_{fuel}\right)$$

Values for LHV are set at 10.1 MJ/kg for CO, 33 MJ/kg for soot assuming pure carbon, $\widehat{LHV}_{HC} = P_{unif}\left[48,50\right]$ based on current understanding regarding the constituency of HC emissions (Spicer et al. 1994; Slemr et al. 1998; KurteNBach et al. 2003), and $\widehat{LHV}_{fuel} = P_{unif}\left[42.9,48.2\right]$ MJ/kg for kerosene fuel, dependent on the H/C ratio, based on the ICAO, ASTM, and DSTAN fuel specifications (ICAO 1993; ASTM 2004; DSTAN 2004). In practice, it is likely that the LHV for kerosene is on the low end of the range specified.

(A7.3) $$EI_{SO_x} = P_{unif}\left[0.7,1.4\right] \text{ g/kg-fuel}$$

7.2. Regulated gaseous emissions

For the regulated gaseous emissions, nitrogen oxides (NO_x), unburned hydrocarbons (HC), and carbon monoxide (CO), EIs source to public certification data reported as a function of engine ground power setting. Boeing Method 2 (BM2), introduced in appendix A7, corrects these data to altitude conditions. Emissions data are specific to each engine model randomly selected for each simulation iteration. The estimation of EIs accounts for certification measurement uncertainty, and uncertainties in interpolating or extrapolating data to flight conditions different from the original tests. The following sections detail modeling choices and provide relevant background. The first considers the quality and breadth of available EI data. These data are often summarized in the form of statistical models to estimate values over the aircraft operating envelope. The second describes how these data and related statistical models were used to specify $\widehat{EI}_{NO_x}$, $\widehat{EI}_{CO}$, $\widehat{EI}_{HC}$.

7.2.1. Data and modeling considerations

As a result of regulatory attention, there is a broad basis of theoretical and empirical knowledge regarding the origins and technological factors controlling gaseous emissions of nitrogen oxides, carbon monoxide, and hydrocarbons. Nitrogen oxide emissions are produced primarily during the high-temperature oxidation of air nitrogen.[3] These processes are influenced in bulk by engine cycle temperatures and pressures, and depend to some extent on the details of combustor fluid mechanics, which determine variability in composition and the time history of temperatures through the combustor. Generally, EI_{NO_x} increases with overall pressure ratio and fuel-air ratio. NO_x emission indices are greatest at high power settings. In contrast, for current era engines, CO and HC emission indices are greatest at low engine power conditions where mixing of fuel and air in the combustor is less complete.[4] Further details on the chemistry of NO_x, CO, and HC, technological influences, and methods of control are available from numerous sources (e.g. Mellor 1990; Lefebvre 1999; Kuo 2005).

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[3] To a much lesser extent for aviation kerosene combustion, NO is also produced through the oxidation of fuel-bound nitrogen.

[4] Equilibrium levels of CO and HC increase with higher engine temperatures such that improving engine efficiency through higher TET and OPR may ultimately lead to increases in CO and HC. However, engine temperatures in commercial applications available currently and envisioned for the next 20 years are far below the levels at which thermodynamic limitations would influence emissions of CO and HC (Lukachko et al. 2003).

Detailed NO_x, CO, and HC emissions data for individual engines are generally proprietary, but a few open sources exist. Stationary test data at ground and altitude are available for specific engines (Frings 1908; Becker et al. 1980; Wulf 1980; Lyon and Bahr 1981; Schumann 1995; Howard et al. 1996). Comparability among these sources prohibits their integration into a unified data set due to differences in measurement technique and engine operational conditions. A consistent characterization of emissions for engines flying with the current fleet is found in the ICAO Engine Exhaust Emissions Data Bank, which records measurements required by certification (ICAO 1995). Data for EI_{NO_x} , EI_{CO}, EI_{HC}, and smoke number (SN) are available at four SLS operational points for new, uninstalled engines. In situ data is also available for a broader range of species, including PMnv and PMv, and their precursors. A more complete discussion of these data and of the use of smoke number in MAIPA analysis is given in appendix A8. The basis and quality of SN certification data are discussed in this section.

Full-scale engine tests at a range of altitude conditions are the most relevant emissions measures because of the closer simulation of flight conditions. However, measurements, including those reported in the ICAO certification database, are typically conducted at ground level. As a result, scaling laws must be employed to derive altitude emissions performance. These are usually presented as statistical correlations between emissions data and engine operating conditions. Combustor designers have historically used model parameters derived from theoretical considerations of the governing physical processes. Empirical correlations between emissions rate and engine operation reference uninstalled combustor-only and full-scale engine tests. Detailed chemical and computational flow models of the combustion process can provide more insightful descriptions of pollutant formation than empirical approaches alone, but they generally have limited predictive application due to the flow complexities of modern gas turbines. Semi-analytical statistical models represent a compromise between the full predictive capability of measurements over all temperature, pressure, and atmospheric conditions, and the still immature capability represented by full numerical reacting flow simulations.

Model equations and parameters are often related to a combustor Damköhler number (Da), the non-dimensional ratio of an appropriately defined flow residence time to a characteristic time that describes the rate of pollutant formation, $Da = \tau_{flow} / \tau_{chemistry}$. For NO_x, a useful ratio of this type is between residence time in the combustor primary zone as τflow, where temperatures are highest in the cycle, to a time scale for NO_x formation as τchemistry, which, from the law of mass action, increases exponentially with temperature and to a power of pressure. Power or exponential dependencies on compressor exit temperature (T3) and combustor exit temperature (T4), and a power dependence on the combustor pressure (P3 = P4) variously appear in the definition of these characteristic times. To construct the correlations, emissions data is obtained at sea level conditions along the engine operating line (specified by thrust, for example) to determine model constants. Additional data obtained at altitude conditions along the operating line defines performance over the aircraft operational envelope. Data needs (and expense) can be reduced if the model variables are incorporated as ratios to reference conditions, for example as the ratio of altitude to SLS conditions. Correlations can also be presented as ratios to standard atmospheric conditions for dimensional and informational convenience.[5]

Correlations have been published for specific aircraft engines and variants (e.g. Platt and Norster 1979; Prather et al. 1992; Schumann 1995). Equation A7.4 shows an example of a direct correlation for NO_x, where model parameters are specified as ratios referenced to test conditions, for the General Electric CF6-50C. In equation A7.4, Ho is the ambient specific humidity, calculated as shown, based on the saturation vapor pressure Pv, which is a function of temperature. A correction for humidity is often applied directly to the data used for correlation equations, although some, as in equation A7.4, include this explicitly in the correlation. The humidity correction is important; high humidity at the same altitude and temperature conditions suppresses NO_x formation relative to a low humidity condition. There is a significant change in specific humidity between ground and cruise altitude, leading to an increase of ~12.5% in EI_{NO_x} at 10 km for the same combustor operating conditions as sea level. Thus, it is important to specify relative humidity accurately at ground for a NO_x correlation (Madden and Park 2003).

[5] A reference formulation also removes the volumetric term, which is combustor specific and appears in the specification of residence time, from the regression analysis.

$$1.35 \cdot 0.986 \cdot \left(\frac{P_3}{1\text{ atm}}\right)^{0.4} \exp\left(\frac{T_3}{194.4\text{ K}} - \frac{H_0}{53.2\text{ g-H}_2\text{O / kg-dry-air}}\right)$$

$$\hat{H}_0 = -19\left(\frac{0.37318\hat{P}_v}{\hat{P}_{amb} - 0.6\hat{P}_v} - 0.00634\right)$$

(A7.4)
$$\hat{P}_v = 6895 \cdot 0.014504 \cdot 10^{\hat{\beta}}$$

$$\beta = 7.90298\left(1 - \frac{373.16}{\hat{T}_{amb} - 0.01}\right) + 3.00571 + 5.02808\log\left(\frac{373.16}{\hat{T}_{amb} - 0.01}\right) +$$

$$1.3816\text{E-07}\left[1\text{-}10^{11.344\left(1-\frac{\hat{T}_{amb}-0.01}{373.16}\right)}\right] + 8.1328\text{E-03}\left[10^{3.49149\left(1-\frac{373.16}{\hat{T}_{amb}-0.01}\right)} - 1\right]$$

As part of the New Emissions Parameter Covering the Whole Aircraft Operation (NEPAIR) program work effort, Madden and Park (2003) suggest the models in equation A7.5 for NO_x emissions performance at altitude, depending on the type of combustor. For some concepts, such as lean burn configurations, a FAR correction may be required as in equation A7.5(b). In these cases, the correlation is combustor specific, and exponents m and n are determined uniquely.[6]

(A7.5)
$$\text{EI}^{alt}_{\text{NO}_x} = \text{EI}^{ref}_{\text{NO}_x} \cdot \left(\frac{P^{alt}_{31}}{P^{ref}_{31}}\right)^{0.4} e^{19(0.00634 - H_0)} \qquad (a)$$

$$\text{EI}^{alt}_{\text{NO}_x} = \text{EI}^{ref}_{\text{NO}_x} \cdot \left(\frac{P^{alt}_{31}}{P^{ref}_{31}}\right)^{m} \left(\frac{\text{FAR}^{alt}}{\text{FAR}^{ref}}\right)^{n} e^{19(0.00634 - H_0)} \qquad (b)$$

Similar to NO_x, EI correlations for CO and HC can be related to Da. The preferred form based on theoretical considerations is a ratio of the combustor bulk residence time to the time-scale of kerosene consumption in the combustor (hydrocarbon combustion). The inverse of this ratio is commonly referred to as the combustor loading parameter (LP), presented in terms of power per unit volume. The LP has been used primarily as a correlating parameter for combustion efficiency; by definition, LP is related to

[6] For equation A7.4, thermal NO_x formation kinetics suggest a pressure ratio exponent of 0.5, but the presence of uncorrelated prompt NO_x formation (which has no pressure dependence), mixing, and primary zone temperature changes (related to combustor FAR distribution) will generally lower this value. In the NEPAIR study, 0.4 was found to minimize error over a range of rig and engine test results.

EI_{CO} and EI_{HC}.[7] The characteristic time for hydrocarbon combustion is the sum time for droplets of liquid fuel sprayed into the combustor to evaporate, mix with oxidizer (air), and combust. Several model formulations can be posited, including plug flow based comparison of flame velocity and a combustor reference velocity (related to mass flow), stirred reactor assumptions (well-mixed assumption), and various perturbations based on the presence of evaporation and/or mixing limitations (Lefebvre 1983). Concentrations of CO and HC correlate with ($1-\eta c$), and thus (1-LP), or more generally (1/LP), can usefully serve as the basis of EI_{CO} and EI_{HC} regressions with engine parameters. In this light, the model equations and parameters suggested for EI_{NO_x} are only minimally different. To first order, differences are constituted primarily of changes in the assumed exponential dependence on temperature, which is expected due to the different reaction mechanisms considered. Developing correlations with LP based on gaseous kerosene consumption implies no important dependence on droplet evaporation (e.g. evaporation time << combustion time). Dopelheuer and Lecht (1998) suggest that at altitude, the evaporation time becomes important, and propose correlating EI data with a quadratic form equation based on (1/LP) corrected by a parameter that accounts for the change in droplet evaporation time with altitude. This is shown in equation A7.6.

(A7.6) $$EI_{CO} \vee EI_{HC} = f\left(\frac{\dot{m}}{P_3^{1.8} \cdot e^{(T_3/300)}}\right) \cdot \left[\frac{T_3}{T_{3_{ref}}} \frac{P_{3_{ref}}}{P_3}\right]^{0.4}$$

Problematically, equation A7.6 requires knowledge of T3 and P3. For constant component efficiency, these can be estimated by changes in ambient conditions, but estimation of combustor inlet conditions generally requires an often-proprietary engine deck. As a result, correlations of this type are only available publicly for either older technology or for experimental combustion systems.

To reduce reliance on proprietary information, methods which correlate fuel flow with the emissions index have been proposed that are built to use certification data, for which fuel flow and EI is available at four operating points. Boeing Method 2 (BM2) is the most widely implemented formulation of this

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[7] CO, HC, and particulate (soot) emissions are related to the combustion efficiency. At the most basic, EI estimation for these emissions can proceed from an assumption for the combustion efficiency at a particular operating condition and an assumption as to how the inefficiency is split among these species. This approach requires, at a minimum, information on burner efficiency (ηc), both at ground and altitude. Only the former can be estimated with certification data.

approach. For NO_x estimation. BM2 uses certification data to establish EI_{NO_x}, EI_{CO}, and EI_{HC} as a function of fuel flow, corrected for installation effects. Fuel flow at altitude is corrected to an SLS reference function relating EI to fuel flow based on changes in ambient pressure and temperature, and Mach number. Changes in P3 and T3 relate to these parameters and thus to the kinetic influences on NO_x, CO, and HC formation. In this vein, the corresponding reference is obtained, which is then corrected back to altitude. Equation A7.7 summarizes altitude corrections for EI_{NO_x}, EI_{CO}, and EI_{HC} using BM2. BM2 does not completely get away from the use of proprietary information; correction exponents were derived from undisclosed Boeing data.

(A7.7)

$$EI_{NO_x} = EI_{NO_x}^{ref} \cdot e^{19(0.00634-H_0)} \left(\frac{\delta^{1.02}}{\theta^{3.3}} \right)^{0.5}$$

$$EI_{CO} = EI_{CO}^{ref} \left(\frac{\theta^{3.3}}{\delta^{1.02}} \right)$$

$$EI_{HC} = EI_{HC}^{ref} \left(\frac{\theta^{3.3}}{\delta^{1.02}} \right)$$

7.2.2. Estimating NO_x, HC, and CO emissions indices

MAIPA estimators $\widehat{EI}_{NO_x}$, $\widehat{EI}_{CO}$, $\widehat{EI}_{HC}$ are based on engine emissions certification data[8]; probabilistic simulation for EI distributions depends on characterizing uncertainty in this data. There are several uncertainties accounted for EIs along the LTO portion of the flight profile: measurement error, uncertainty due to performance variability among new engines of a specific type, aging bias accounting for in-service operation, and error sourced to fits of the LTO EI data to obtain EIs at thrust settings different from the certification values. The effects of uncertainty in LTO flight profile are carried through the EI calculation as variance in aircraft performance parameters. Each representative aircraft type consists of several airframe-engine combinations; EIs are calculated for N randomly selected combinations, where N is the number of simulation iterations.

Most engines are reported using measurements from 1 to 3 engines (ICAO 1995); the minimum requirement is a single engine measured three times. Based on historical experience with engine

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[8] Emissions for turboprop engines are unavailable; in MAIPA, turboprop EIs uniformly specified using the values provided in Boeing (1992) in the construction of inventories for the IPCC (1999) report.

variability, averaged certification measurements are divided by a compliance factor, β, to ensure a 90% confidence in meeting regulatory limits for untested engines placed into service. Values for β are listed in table A7.1 (FAA 2003a). These factors quantify engine-to-engine variability. For example, based on a single engine sampled three times, 90% confidence intervals for EI_{NO_x}, EI_{CO}, EI_{HC}, and SN for any engine picked out of a fleet are ±16%, ±23%, ±54%, and ±29%, respectively (e.g. $(1/\beta)-1$, where the value for β is taken from the first line of table A7.1.

Table A7.1. Emissions certification compliance factors

(FAA 2003a)

Number of engines tested (i)	β CO	β HC	β NOx	β SN
1	0.8147	0.6493	0.8627	0.7769
2	0.8777	0.7685	0.9094	0.8527
3	0.9246	0.8572	0.9441	0.9091
4	0.9347	0.8764	0.9516	0.9213
5	0.9416	0.8894	0.9567	0.9296
6	0.9467	0.899	0.9605	0.9358
7	0.9506	0.9065	0.9634	0.9405
8	0.9538	0.9126	0.9658	0.9444
9	0.9565	0.9176	0.9677	0.9476
10	0.9587	0.9218	0.9694	0.9502
More than 10	$1-(0.13059/i)$	$1-(0.24724/i)$	$1-(0.09678/i)$	$1-(0.15736/i)$

These intervals define an upper bound standard deviation characterizing variability. To specify the engine-to-engine variability, MAIPA assumes the minimum testing requirement to calculate a coefficient of variation, $\hat{\sigma}/\hat{\mu}$ using the definition in equation A7.8. In equation A7.8, $\hat{\sigma}$ is the sample standard deviation, n is the number of samples, t is the t-distribution value with n-1 degrees of freedom, and $\hat{\mu}$ is the sample mean.

(A7.8)
$$\frac{\hat{\sigma}}{\hat{\mu}}=\frac{(1/\beta)-1}{t_{\alpha/2}^{n-1}/\sqrt{n}}$$

Using β, and recognizing that pollutant formation is at least a multiplicative process, $\widehat{\mathrm{EI}}_{\mathrm{NO_x}}$, $\widehat{\mathrm{EI}}_{\mathrm{CO}}$, $\widehat{\mathrm{EI}}_{\mathrm{HC}}$, and SN are specified as $\mathrm{P}_{\log n}\left[\mu_g = f(\hat{\mu}), \sigma_g = f(\hat{\sigma})\right]$ where $\hat{\sigma}$ is 9%, 14%, 32%, and 17% respectively. As a check on the interpretation of equation A7.8, data on engine-to-engine $\mathrm{EI}_{\mathrm{NO_x}}$ variability compiled in Lukachko and Waitz (1997) show reported standard deviations for a range of engine types and operating points that are within the standard deviations derived from certification data. Table A7.2 reproduces a portion of the data presented.

Table A7.2. Variability in the NO_x emission index of new, post-checkout engines

(Lukachko and Waitz 1997)

engine type	tests *	% rated output	σ
CF6-50C2 [1]	6	30%	6.5-6.9%
CF6-50C2 [2]	12	30%	4.8-4.9%
CF6-6 [1]	7	30%	4.0-4.2%
CF6-50C2 [1]	6	85%	1.4-2.6%
CF6-50C2 [2]	12	85%	5.7-6.2%
CF6-6 [1]	7	85%	2.5-3.3%

* All measurements corrected for compressor exit pressure and temperature, humidity, and fuel mass flow

References: [1] (Lyon et al. 1980) using Jet A fuel; [2] (Lyon et al. 1980) using JP-4 fuel.

These results employ a larger number of tests than typically conducted for certification and thus the smaller estimate for standard deviation. These standard deviations are not indicative of the precision error in the measurement of a particular engine, which is typically less than 1% for all certification points.[9] For MAIPA, measurement error in the reported EI is characterized by $\mathrm{P}_{norm}\left[\widehat{\mathrm{EI}}, 1\%\right]$.

Because engine aging alters performance, emissions can also change. (Lukachko and Waitz 1997) found a maximum, but partially recoverable change in average $\mathrm{EI}_{\mathrm{NO_x}}$ with aging of between -1% and +4% for typical utilization scenarios. For MAIPA, this bias is represented as $\mathrm{P}_{beta}\left[6,6,-1\%,4\%\right]$. For NO_x emissions, studies indicate that variability in $\mathrm{EI}_{\mathrm{NO_x}}$ among new or old engines does not appear to change (AEMTG 2004). Sensitivities of SFC and combustor flow parameters to component aging are enhanced by increases in cycle temperatures and pressures. This ultimately results in a higher sensitivity of NO_x emissions to engine degradation for cycles representative of more advanced technology. Similar analyses

[9] P. Madden, personal communication.

for other species such as CO and HC are not available and no correction is applied. Transient operation also has a similar biasing impact on emissions performance. It has been observed qualitatively that transient engine operation (e.g. engine start-up, power change, etc.) can temporarily increase CO, HC, and particulate emissions far beyond levels suggested by steady-state measurements. These biases have not yet been fully characterized and are not included in the $\widehat{\mathrm{EI}}$ evaluation.

$\widehat{\mathrm{EI}}$ is adjusted for the time-weighted average thrust setting calculated from the SAE AIR 1845 performance specification. This adjustment uses the same factor specified for fuel flow to account for departures from certification settings. For BM2, there is the question of how to interpolate between points for the ground level reference. To fit empirical data taken over several more points than reported in certification, the accepted practice is to use a bilinear fit on a log-log scale, connecting the lowest two certification power settings with one line, and then a second line using a best-fit curve for the two highest settings (AEMTG 2004). The intersection of the first and second lines creates a kink in the case of CO and HC and could cause some difficulties in the region of the kink and perhaps an overestimate in EI for low power. This is not the case for NO_x where a single best fit line is used on the log-log scale. (AEMTG 2004) has estimated that these curve fit methods predict actual values to within 10% for EI_{NO_x}. Most results for EI_{CO} and EI_{HC} also fall within this error bound. MAIPA assumes the AEMTG value to be $2\,\hat{\sigma}$, and applies $\varepsilon_{fit}^{\mathrm{EI}(\dot{m}_f)} = \mathrm{P}_{norm}\left[0{,}5\%\right]$ to define fit uncertainty. Larger deviations are reported for dual-annular combustor (DAC) designs. In such staged combustion systems, different regimes of operation (pilot-only, pilot + main burner, etc.) result in data discontinuities that are difficult to capture using curve fits as described. Published estimates of correlation uncertainties do not consider the effect of changing technology. These systems are, however, relatively less frequent in the historical fleet.

All $\widehat{\mathrm{EI}}$ are corrected for altitude effects using BM2. Altitude fuel flows estimated using the performance model are corrected to SLS conditions; these corrected fuel flows are equated to an EI using linear fit models, and then corrected back to altitude using the factors summarized in equation A7.7. Relative humidity is a function of $\hat{T}_{amb}$. Uncertainties in correlations used for altitude corrections are not well defined; more often, precision rather than accuracy is addressed, and reports address NO_x almost to the exclusion of CO and HC. Lister et al. (1995) considered the applicability of nine possible $\widehat{\mathrm{EI}}_{NO_x}$ correlation equations of the types described in the previous section, derived for four different turbofan engine types. Using a straight average, they find variability across correlations referencing ground level certification data is 18%. In MAIPA, the median variability of 15% defines accuracy in $\widehat{\mathrm{EI}}_{\mathrm{NO_x}}$ correlation model $\varepsilon^{\mathrm{EI}}_{alt.corr}=\mathrm{P}_{norm}\left[\widehat{\mathrm{EI}},15\%\right]$ (another model form chosen for the BM2 could result in a different correction). Less information is available to gauge the equivalent variability in $\widehat{\mathrm{EI}}_{\mathrm{CO}}$ and $\widehat{\mathrm{EI}}_{\mathrm{HC}}$. Based on CO altitude chamber measurement data from a Pratt & Whitney PW305 engine, Dopelheuer and Lecht (2001) suggest a larger variability sourced to model choice, finding that without a correction factor for evaporation, prediction errors could be as much as ~50%. Dopelheuer (2001), comparing the correlation against altitude chamber measurement data from a military engine, suggests that such a correction can lower prediction error to below ~25%. Given the relatively greater variability in HC measurements, correlations of this type for $\widehat{\mathrm{EI}}_{\mathrm{HC}}$ would be at best equal to these comparisons. Based on these reports, uncertainty in the altitude EICO or EIHC is $\varepsilon^{\mathrm{EICO \vee EIHC}}_{alt.corr}=\mathrm{P}_{norm}\left[\widehat{\mathrm{EI}}_{\mathrm{CO}} \vee \widehat{\mathrm{EI}}_{\mathrm{HC}},25\%\right]$.

The precision of fuel-flow based correlations (e.g. BM2) has not been directly assessed. Activities undertaken by ICAO (see IPCC 1999) found agreement between $\mathrm{EI}_{\mathrm{NO_x}}$ correlations referencing engine parameters and fuel flow correlations within $\sigma = 6\%$ and a maximum error of 13%. Correlations referencing engine parameters directly as in equation A7.4, are typically the most precise model for estimating emissions performance. The $\mathrm{EI}_{\mathrm{NO_x}}$ test data in table A7.2, which is based on performance variability among new engines, provides an indirect measure of such correlations as a representation of any particular engine in the fleet; a limiting precision of $\sigma < 5\text{-}7\%$ is inferred, if models are based on adequate operating data. Madden and Park (2003) address the accuracy of ground referenced correlations that employ engine parameters finding a similar predictive uncertainty (to within 11%) via the model described in equation 7.5. For MAIPA, these uncertainties are stacked to define BM2 $\widehat{\mathrm{EI}}_{\mathrm{NO_x}}$ precision as $\varepsilon^{\mathrm{EI}}_{\mathrm{BM2}}=\mathrm{P}_{norm}\left[\widehat{\mathrm{EI}},6\%\right]\cdot\mathrm{P}_{norm}\left[\widehat{\mathrm{EI}},6\%\right]$, first applying an error to account for the difference in BM2 and engine parameter based correlations, and then applying a second error to account for the uncertainty in applying direct correlations to a fleet of engines (the resolution of MAIPA). It is important to recognize that prediction errors are rooted in model formulation and thus structural uncertainty; the correlations models described capture a single dominant spatial or time scale. Thus, despite success in deriving statistical models for $\mathrm{NO_x}$, CO, and HC, test parameters do not accurately account for changes in combustor flow mixedness and correlation attempts suffer losses of accuracy since global pollutant kinetics are employed. This is an important question in determining whether precision errors described for

$\widehat{\mathrm{EI}}_{NO_x}$ extend to $\widehat{\mathrm{EI}}_{CO}$ and $\widehat{\mathrm{EI}}_{HC}$. However, lacking information to make such an assessment, MAIPA assumes the same uncertainty for $\widehat{\mathrm{EI}}_{CO}$ and $\widehat{\mathrm{EI}}_{HC}$ model precision as for $\widehat{\mathrm{EI}}_{NO_x}$.

Appendix

8. Estimation of particulate matter emissions indices

A comprehensive particulate matter emissions inventory requires an account of emitted non-volatile particulate matter (PMnv, primarily soot), as well as volatile particulate matter (PMv) sourced to chemical conversion processes originating with gaseous precursor emissions formed within and immediately downstream of the engine (i.e. sulfate, nitrate, and organics). This section describes procedures for estimating near-field PMnv (section A8.1) and PMv (section A8.2) emissions indices. Characterizing particulate matter with a mass-based metric is a limited account of the properties consequential to environmental assessments. A more detailed account of PM in the near-field includes descriptions of particle phase, multi-species particle composition (e.g. volatile coatings on soot, sulfate-nitrate-organic-water constituency), electronic state, and morphology. Specification of these attributes is more appropriate for a physical model of plume dynamics, microphysics, and chemistry.

8.1. Non-volatile particulate matter emissions indices

This section is specific to the MAIPA approach towards characterizing PM emissions using the mass-based EI metric. It does not present an exhaustive survey of previous estimation methodologies. See (Wayson et al. 2003) for additional review of historical developments in data acquisition and their application to PMnv specification for commercial aircraft.

Soot PMnv emissions are generally a function of fuel properties (types of hydrocarbons present), equivalence ratio, liner pressure drop (mixing), and combustor inlet pressure and temperature (Lefebvre 1999). While empirical correlations have been successfully developed for NO_x, CO, and HC to predict emissions as a function of engine performance, their plausibility is uncertain in application to aerosol precursors and soot, particularly because easily measured cycle parameters are less influential in comparison to the phenomenological details of combustor design. Thus, historically, PM emissions from commercial aircraft have been crudely estimated using, at various times, diesel emission factors, data from a small sample of older commercial engines, or limited tests undertaken for older military aircraft. Results discussed in chapter 3 demonstrate that these estimation methods can lead to inaccurate PMnv characterization for commercial aircraft operating during the last decade.

Note that condensed matter can further increase through heterogeneous nucleation of sulfuric acid on soot and metal surfaces, activated by adsorption of oxidized sulfur, in the presence of water vapor (Zhao and Turco 1995; Karcher et al. 1998). This analysis does not address such interactions between non-volatile PM and aerosol precursors.

To reiterate, particulate measurements are not a viable basis for EI estimation. For PMnv, smoke number (SN) measurements mandated for certification comprise the only comprehensive and methodologically consistent characterization of carbonaceous particulate emissions from aircraft (DuBois 2001). These data are the basis for the estimation method introduced in this section. The MAIPA estimation procedure estimates ground-level concentration (Cnv) over the aircraft operating envelope through correlations with SN, corrects for altitude, and then converts to an emission index. Smoke number is a visibility metric, and as such, has no direct physical relationship to a measurement of non-volatile particulate mass.[1] Uncertainty in correlating SN to Cnv is not trivial; the fundamental difficulty is that they express the tendency of sampling technique to capture particles as a function of particle size and measurement test time. The smoke number is a complex function of the pattern of collection on the paper and soot morphology. The particles obtained in a smoke measurement tend to be on the high end of the emitted size distribution. As a result, soot concentration characteristics can be expected to change from engine to engine.

Thus, the form of a statistical relationship between SN and Cnv cannot be easily posited a priori based on physical understanding. Attempts to date have exclusively used simple polynomial or power fits derived from simultaneous measurements of SN and Cnv. To relate engine performance to SN, DeChamplain et al. (1995; 1997) developed a physics-based correlation between Cnv and engine parameters, and then employed a polynomial model proposed by Odgers and Magnan (1988) to correlate Cnv and SN.[2] DeChamplain et al. estimate a 40% standard deviation in prediction of SN from engine performance. Correlating parameters relating engine performance to Cnv derive from a soot production model using

[1] The smoke number is derived from an optical measurement initiated by passing exhaust gas through a white paper filter. Soot is collected on the paper, changing its optical characteristics. Light is then passed through the filter paper and the intensity of reflection is determined. This intensity is compared to a pristine filter and the resulting ratio recorded. SN is presented on a scale of 0 to 100, with the lowest value equivalent to an unsoiled filter. The accuracy of this measurement is +/-3 on the SN scale as specified by the SAE Aerospace Recommended Practice (ARP) 1179 (SAE 1997).

[2] The correlation reported by Odgers and Magnan was derived from a different source of emissions than that analyzed by DeChamplain et al.

fundamental time scale ratios of the type discussed previously for CO, HC, and NO_x. The formulation is complex and dependent on specific knowledge of the studied engine and combustor, information which is partially lacking for MAIPA representative aircraft types. DeChamplain et al. conclude that the sensitivity of PMnv emissions to small changes in combustor design or operation make it unlikely that a universally applicable correlation not referenced to ambient or test conditions can be developed, at least for the rich primary zone combustion systems that dominate the current fleet.

Citing the same concerns, Dopelheuer and Lecht (1999) and Dopelheuer (2001) suggest a similar method to relate engine parameters to Cnv employing the less information-intensive, reference-type correlation in equation A8.1 to estimate PMnv emissions over the entire operating envelope from cycle-derived engine parameters at sea-level static. In equation A8.1, model coefficients were estimated using PMnv measurements from laboratory flames and combustor tests. Reference values refer to the SLS condition at the same T3 and P3 of the aircraft operating condition in flight (e.g. at altitude). Tfl is the combustor flame temperature. Cnv is in units of mg/m3.

(A8.1) $$C_{nv} = C_{nv}^{ref}\left(\frac{\phi}{\phi_{ref}}\right)^{2.5}\left(\frac{p_3}{p_{ref}}\right)^{1.35}\frac{e^{\left(-20000/T_{fl}\right)}}{e^{\left(-20000/T_{fl,ref}\right)}}$$

To relate SN and Cnv, Dopelheuer et al. combine three statistical correlations—equation A8.2 parts (a), (c), and (f) in units of mg/m3—to derive sea-level static concentrations from SN certification data. Comparisons to four in situ PMnv measurements—one helicopter, two commercial aircraft, and one supersonic engine (Pueschel et al. 1998; Petzold et al. 1999)—were in agreement to predictions within absolute bounds of +/- 30% where an engine model was available to estimate parameters. This estimate does not account for the variability among SN-Cnv correlations; a single relationship was derived from averaging the three aforementioned equations.

$$C_{nv} = 3.25\text{E-}06 \bullet SN^4 - 1.27\text{E-}04 \bullet SN^3 + 3.22\text{E-}03 \bullet SN^2 + 8.76\text{E-}02 \bullet SN + 0.14 \quad (a)$$

$$C_{nv} = 4.05\text{E-}06 \bullet SN^4 - 3.01\text{E-}04 \bullet SN^3 + 8.17\text{E-}03 \bullet SN^2 + 6.32\text{E-}02 \bullet SN + 0.0944 \quad (b)$$

$$C_{nv} = 0.001 \bullet \left[\frac{0.177 \bullet SN}{(100\text{-}SN)^{0.472}} \right] \quad (c)$$

(A8.2)

$$C_{nv} = 1.63\text{E-}04 \bullet SN^3 - 5.03\text{E-}03 \bullet SN^2 + 0.134 \bullet SN - 0.055 \quad (d)$$

$$C_{nv} = \begin{cases} 3.232 \bullet \left[1 - \sqrt{\dfrac{SN}{19.58}} \right] & \text{for SN} \le 18.7 \\ 0.0027511 \bullet SN^{2.319} & \text{for SN} > 18.7 \end{cases} \quad (e)$$

$$C_{nv} = \begin{cases} 0.0694 \bullet SN^{1.23357} & \text{for SN} \le 30 \\ 0.0297 \bullet SN^2 - 1.802 \bullet SN + 31.94 & \text{for SN} > 30 \end{cases} \quad (f)$$

The MAIPA estimation method extends the works of DeChamplain et al. (1995; 1997), Dopelheuer and Lecht (1999), and Dopelheuer (2001). Importantly, where previous work has dealt with deterministic results, MAIPA outputs probabilistic estimates that account for technological variability and uncertainties in both data and models. MAIPA applies the reference correlation in equation A8.1 to correct for altitude, with uncertainty as estimated by DeChamplain et al. (here interpreted as $\varepsilon^{EI}_{alt.corr} = P_{norm}\left[\widehat{EI}, 30\%\right]$), and employing engine parameters derived from a new set of cycle analyses chosen to match a wide range of existing models in the commercial aircraft fleet Han (2003). Table A8.1 categorizes the cycles developed and summarizes parameters important for MAIPA. Table A8.1 also summarizes MAIPA matches of representative aircraft types with engine cycles. Engine parameters are deterministic.

Table A8.1 Cycle parameter specifications

Class (kN)	<100		100-200		200-400		>400	
Thrust (kN)	60	65	100	170	205	230	250	420
Case #	1	2	3	4	5	6	7	8
Take-off								
P3 (atm)	16	15	23	25	20	27	27	37
T4 (K)	1350	1200	1520	1510	1400	1560	1540	1940
EINOx (g/kg-fuel)	35	19	55	17	36	30	27	20
Climb								
P3 (atm)	14	12	19	22	17	24	23	31
T4 (K)	1300	1140	1440	1450	1340	1510	1480	1840
EINOx (g/kg-fuel)	29	16	42	14	27	24	22	16
Cruise								
P3 (atm)	6	5.7	8	9	7.1	7.3	9.7	13
T4 (K)	1170	1030	1340	1340	1270	1210	1370	1710
EINOx (g/kg-fuel)	5.1	7.1	7.1	6.3	8.2	7.6	9.4	5.9
Approach								
P3 (atm)	6	5.7	7.1	9.2	7.6	10	9.8	13
T4 (K)	960	870	1010	1080	1010	1140	1110	1440
EINOx (g/kg-fuel)	9.3	8.4	10	6	7.3	10	11	5.5
Idle								
P3 (atm)	2.6	2.6	3.3	3.7	3.3	4.4	4	5.3
T4 (K)	750	710	840	870	830	940	900	1220
EINOx (g/kg)-fuel	3.7	8.4	6	4	2.9	4.2	4.7	2.4

Notes: EINOx as NO2 and EIS = 0.5 g/kg-fuel. EICO and OFAR different for each cycle and condition. Common initial species conditions are: (a) NO/NOx = 1; (b) SO/SOx = 0; (c) no HC or H2. Note combustion efficiency consistent with EICO.

The MAIPA approach employs statistical correlations to relate Cnv and SN, but with the intention of capturing the variability in estimates of SLS Cnv from SN certification data by randomly selecting, with equal probability, from the six published correlations relating SN and Cnv shown in equation A8.2 [(a) Champagne 1971; (b) AGARD 1982; (c) Odgers and Magnan 1988; (d) Hurley 1993; (e) Colket et al. 2004; (f) Hurley 2005]. Random selection among the published correlations in equation A8.2 is done in order to represent the space over which SN and C might be related in the fleet, understanding that each correlation is specific to the soot generating apparatus on which it is based. Care is taken to limit the application of a correlation to the range of SN used in its derivation. Dopelheuer and Lecht (1999) and Dopelheuer (2001) estimate typical uncertainty in these correlations of +/- 7% individually, which is applied in MAIPA specifying $\varepsilon_{alt.corr}^{C(\mathrm{SN})} = \mathrm{P}_{norm}\left[\widehat{C}, 7\%\right]$.

MAIPA implements a hierarchical approach to the selection of SN data from the certification database. Certification in reported data for SN is sparse for many engines. MAIPA applies the maximum available data available from using rule protocol, applying SN data reported for individual certification points when available, averaging when the SN data is incomplete, and employing reported bounds when individual specifications or averages are not possible. Error in SN measurement is specified using $\varepsilon_{SN} = P_{unif}[-3,3]$. SN uncertainties attributed to typical variability in new engine performance in the fleet around the certification value were described previously in this appendix. No aircraft degradation due to use is included in the estimation of PM emissions.

Soot concentration can be equated to an EI using equation A2.9 where R is the gas constant. Ambient conditions (subscript amb) are specified relative to the flight profile and the fuel-air ratio (FAR) is derived from the aforementioned engine cycles. Corrections to Cnv for SN measurements made on mixed flow turbofan engines apply as needed.

(A8.3) $\widehat{\mathrm{EI}} = \left(\hat{C}_{nv}/1000\right) \cdot R \cdot \left(\hat{T}_{amb}/\hat{P}_{amb}\right) \cdot \left(1/\mathrm{FAR} + 1\right)$

MAIPA parameters are specified for steady-state conditions; the analysis does not account for transients. Engine start-up and changes in operational condition may change the combustion efficiency and emissions performance temporarily during transients. Typically, an increase in both fuel flow and emissions index results.

8.2. Near-field volatile particulate matter emissions indices

Aircraft emissions of trace sulfur and nitrogen oxides contribute to the generation of fine volatile particulate matter. Resultant changes to ambient PM concentrations and radiative properties of the atmosphere may be important sources of aviation-related environmental impacts. Fine particles are emitted and generated in aircraft engine exhaust in size ranges constituting the nucleation, Aitken, and accumulation modes of a typical PM size distribution. Direct and indirect atmospheric effects from aviation-sourced PM constitute environmental risks of an uncertain magnitude (EPA 1996; Brasseur et al. 1998; IPCC 1999).

Sulfate and nitrate production is initiated by gas phase oxidation of SO2 and NO that begins in the post-combustion intra-engine flow. This gives a unique role to trace species chemical processing through the combustor dilution zone, turbine, and exhaust nozzle that is as important to the formation of volatile PM emissions as is the influence of combustor fluid mechanics on total NO_x ($NO + NO_2$), CO, and HC emissions. Precursor emissions of trace nitrogen and sulfur oxides are formed within the engine over time scales on the order of 10 ms. These emissions contribute to the generation of volatile fine PM formed in the engine plume at longer time scales (10 ms to < 1 s) compared to soot formation in the combustor.

Various mechanisms lead to the production of particulate precursors within the engine. Oxidized nitrogen species originate primarily from the high temperature oxidation of atmospheric nitrogen in the combustor. Total sulfur emissions are predictable functions of fuel composition and emerge from the primary zone as SO2 in lean conditions. Sulfur emissions are thus controlled by fuel consumption to a greater extent than NO_x. Formation of precursors to volatile PM, including SO3 and HONO, initiates within the combustor and continues downstream through the turbine and exhaust nozzle (Brown et al. 1996a; Lukachko et al. 1998; Tremmel and Schumann 1999; Starik et al. 2002). The response of trace chemistry to the temporal and spatial evolution of temperature and pressure through the turbine is complex and presents both computational and experimental challenges. Total emissions are related to the technological characteristics of the aircraft (weight, aerodynamic efficiency, and engine overall efficiency), its operational use, and details of the combustor, turbine, and nozzle design.

Experimental and modeling studies have highlighted the role of trace emissions of SO3 in the formation of high number densities of fine aerosol particles observed in the exhaust streams of several aircraft (Hofmann and Rosen 1978; Hofmann 1991; Reiner and Arnold 1993; Miake-Lye et al. 1994; Fahey et al. 1995a; Zhao and Turco 1995; Brown et al. 1996b; Karcher 1996; Schumann et al. 1996; Brown et al. 1997; Karcher and Fahey 1997; Petzold et al. 1997; Anderson et al. 1998a; Anderson et al. 1998b; Karcher et al. 1998; Miake-Lye et al. 1998; Schröder et al. 1998; Yu and Turco 1998; Schröder et al. 2000a; Schumann et al. 2002). Particle concentrations are correlated with the level of oxidized fuel sulfur in the exhaust (Busen and Schumann 1995; Schumann et al. 1996; Miake-Lye et al. 1998). At exhaust temperatures and lower temperatures, SO3 converts to H2SO4 in the presence of exhaust water vapor (Kolb et al. 1994; Reiner and Arnold 1994). In the plume, new volatile sulfate particles can be formed by binary homogeneous nucleation of H2SO4 with emitted water vapor (Karcher et al. 1995), accentuated by concomitant chemiion emissions (Freznel and Arnold 1994; Yu and Turco 1997; Arnold et al. 1998;

Sorokin and Mirabel 2001). Nucleated sulfate particles grow via coagulation and uptake of water vapor (Brown et al. 1996c; Karcher 1998).

Because nitrous and nitric acids have high saturation vapor pressures relative to sulfuric acid, they have a lesser tendency to nucleate new particles and thus contribute to PM primarily via uptake on existing particles. Hydrocarbons can also contribute via uptake, and may additionally nucleate as an independent PM source (Karcher 1997, 1999), but thermodynamic conditions are not favorable for this process at the HC levels typically emitted (Dakhel et al. 2005). Both HNO3 and H2SO4 are emitted in concentrations orders of magnitude smaller than SO3 and HONO, thus evaluation of volatile aviation PM should focus on the latter species.

The results of our analysis show that production of H2SO4 from emitted SO3 is greater in magnitude than new sulfuric acid production in the plume.

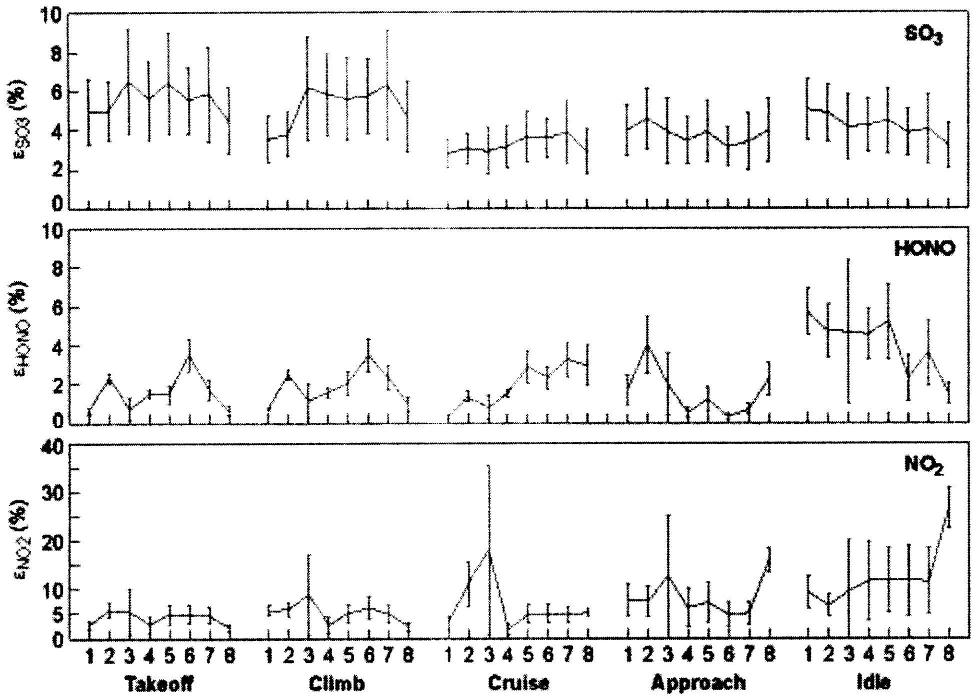
Because nucleation rates are high for H2SO4 given typical fuel sulfur levels, modeling investigations of the microphysical processes that lead to the formation of volatile aerosols emphasize the development of oxidized sulfur through the aircraft plume and wake. These investigations find that known gaseous pathways yield only 1-2% oxidation within the near-field plume (< 1 s after emission) for a range of aircraft engine configurations, physical approximations, and chemical assumptions (Miake-Lye et al. 1993; Quackenbush et al. 1993; Miake-Lye et al. 1994; Anderson et al. 1996; Brown et al. 1996c; Karcher et al. 1996; Hanisco et al. 1997). Modeling investigations have also shown that compared to the plume, sulfur oxidation can be more vigorous within the aircraft engine as a result of gaseous chemical processes through the combustor dilution zone, turbine, and exhaust nozzle. Upper bound chemical kinetic analyses indicate that SO3 formation via atomic oxygen is less than 6% of SOx within the combustors used in aircraft (Brown et al. 1996a) and industrial applications (Hunter 1982). Previous studies have also suggested SO3 formation via OH and O may result in an upper limit 10% oxidation through the turbine and nozzle (Brown et al. 1996a; Lukachko et al. 1998; Tremmel and Schumann 1999; Starik et al. 2002; Wilson et al. 2004).

Lukachko et al. (2005) is the technical basis for estimating PMv emission indices for representative aircraft types. This paper addresses aircraft engine design and operational impacts on aerosol precursor emissions of sulfur (SOx) and nitrogen (NOy) species; estimates for EI_{PM_v} in the near-field exhaust of current, in-service commercial aircraft are presented as a function of operational mode. Volatile particulate mass is specified using the conversion efficiencies determined by Lukachko et al. (2005). For the LTO cycle, conversions are assigned as identified in Lukachko et al. (2005) for the certification power settings. En route climb and approach settings are set at the corresponding value determined for the relevant portion of the LTO cycle. Efficiencies were determined independently for the cruise condition.

Aerosol precursors form through the combustor dilution zone, turbine, and exhaust nozzle of gas turbine engines. The intra-engine environment is more important to the production of condensable volatile PM in the area near an aircraft than emissions processing in the engine plume. However, due to an inefficient combination of thermodynamic and kinetic factors, there is overall little opportunity for the production of SO3, the most likely of the precursors to result in volatile PM. Comparing combustor εSO3 to conversion magnitudes through the post-combustor gas path suggests that SO3 production in older technology engines would tend to be located in the combustor whereas for more recent cycle designs, the turbine and exhaust nozzle have a more prominent role. Since the combustor is likely the dominant source of precursors for most power conditions, further research should focus on a more detailed investigation of the combustor. Best estimates for aerosol precursor production from in-service commercial engines are summarized in figure A8.1 The figure shows the conversion efficiencies for each of the 8 cycles examined at each of the 5 operating modes simulated (table A8.1). The values shown result from a Monte Carlo simulation sampling from distributions based on the results of this analysis. Uncertainties in the specification of conversion efficiencies are specified by a random multiplier, chosen from a uniform distribution bound by factors of 2 above and below the central value (Whitefield et al. 2001).

Figure A8.1 Intra-engine and near-field plume conversion efficiencies

SO3, HONO, and NO_2 as a function of technology and operating mode (Lukachko et al. 2007)



The mean results are consistent with measurements of sulfur and nitrogen precursors. Although HONO and NO_2 oxidation can be on the order of 10%, particularly at low power conditions, we would not expect nitrate contributions to particulate mass until well after the plume mixes with the atmosphere. In contrast to NOy species, SOx chemistry is active over the entire operational range of aircraft currently in the fleet. The trends suggest that mean εSO3 is limited to the range 2.8% to 6.5%. This reflects technological differences within the fleet, the variation in oxidative activity with operating mode, and modeling uncertainty. Note an additional 1-2% conversion to SO3, and up to 1% for HONO (and NOy) may be realised in the plume. Since fuel flow increases with power setting, the SO3, NO_2, and HONO emissions rates (e.g. kg/s) will be higher at take-off and climb than that suggested by the conversion efficiencies in figure A8.1. Subsequently, for the landing take-off cycle, higher levels of sulfate in the near-field plume can be expected along the departure portion of a flight profile as opposed to landing.

Sulfate production is initiated by gas phase oxidation of SO2 that begins in the post-combustion intra-engine flow. Precursor emissions of trace sulfur oxides are formed within the engine over time-scales on the order of 10 ms. These emissions contribute to the generation of volatile fine PM formed in the engine plume at longer time-scales (10 ms to < 1 s) compared to soot formation in the combustor. Experimental data characterizing sulfur species in engine exhaust consist primarily of concentrations inferred from in situ plume measurements at altitude [Anderson et al. 1998a, 1998b; Miake-Lye et al. 1998; Schumann et al. 2002] and a relatively fewer direct measurements at and downstream of the engine exit plane [Arnold et al. 1998; Curtius et al. 2002; Katragkou et al. 2004], mostly at higher power conditions for older in-service commercial and military aircraft. Inferred concentrations from measurements indicate an apparently broad range of SO2 to H2SO4 oxidation; however, a detailed analysis of the instrument responses and age of the sampled air has refined the estimates of oxidative conversion to 0.5-5% of the fuel sulfur [Kärcher et al. 2000]. In-flight trends and ground measurements suggest oxidation efficiency is dependent on engine technology and operating point (Schumann et al. 2002; Katragkou et al. 2004).

Although production of HNO3 occurs at rates about an order of magnitude lower than for H2SO4 at exhaust and ambient conditions (Miake-Lye et al. 1994), HNO3 can play a role in plume PM processing (Gleitsmann and Zellner 1999). We thus also examine intra-engine production of NO_2, the chemical precursor to nitric acid. Measurements of NO and NO_2 are routine in engine development and certification, but there are few measurements of their oxidation products, HONO and HNO3. Model estimates of N(IV) species (Lukachko et al. 1998; Tremmel and Schumann 1999) and measurements of HNO3 and HONO (Arnold et al. 1992; Fahey et al. 1995b; Whitefield et al. 2002; Miller et al. 2003) indicate that conversion of NO to HONO and NO_2 to HNO3 at the engine exit plane amounts to a few percent or less at higher power conditions. Intra-engine conversion of NO to NO_2 has been estimated from measurements at much higher levels, up to ~25% (Haschberger and Lindermeir 1997; Schulte et al. 1997).

Current sampling programs are now examining volatile organic particulates in the exhaust plume. Indications of their presence were reported several years ago [Yu et al. 1999, Schumann et al. 2002]. Recent ground measurements confirm that condensible organics exist in aircraft engine exhaust and that a portion of PMv is attributable to organics [Wey et al. 2006; Knighton et al. 2007; Lobo et al. 2007; Yelvington et al. 2007]. Given the emerging nature of measurements that resolve organic speciation in the gaseous and condensed phases, empirical data does not currently provide a basis to define a parametric specification for organic PMv emissions.

Compared to PMnv and sulfate PMv, heterogeneous microphysical processes have a more significant role in determining condensed organic mass. The question of speciation is important for organics given the significant range of vapor pressure at altitude and the identification of a number of condensible species in the exhaust. There is a diversity of organic species, many of which have only recently been identified by measurements. More organics in the exhaust, however, may not be correlated with an increase in aerosol mass. This relationship depends to some extent on the exhaust concentrations of low vapor pressure species, such as polyaromatic hydrocarbons (PAHs), oxygenated hydrocarbon species, and engine lubrication oil. Thermodynamic conditions in the exhaust plume severely limit hydrocarbon condensation.

Appendix

9. Noise characterization

Because myriad characteristics contribute to noise response, it is difficult to propose a single metric that describes a community's reaction to changes in the noise environment. Correlations have developed primarily around integrative measures that sum total sound levels over a given period. The most widespread measure of adverse reactions to living in noisy environments is annoyance, a generalized and subjective descriptor that underpins current noise policy and by definition overlaps with other, more precise descriptions of noise impact, such as sleep disturbance or speech interference. In the context of exposure-response relationships, the day-night noise level (DNL), calculated as the A-weighted sound level (i.e. accounting for unequal loudness perception across different frequencies) averaged over a 24-hour period, has been the central noise metric since its adoption in response to the 1979 Aviation Safety and Noise Abatement Act (ASNA) (see also American National Standards Institute (ANSI) standard S3.23-1980 and again in S12.40-1990)(EPA 1974. FAA 1985; FICAN 1992; FAA order 1050.1E).[1]

Use of the DNL metric originates in its correlation with annoyance — $W(E_{annoy}) \propto e^{\mathrm{DNL}}$ [e->exp(-)] (cf. section 2.3.2 for other noise endpoints not evaluated in this analysis). With the estimation of costs in mind —assessments of noise damages use DNL (or a proportional equivalent) almost exclusively as the dependent variable representing noise levels—we are compelled to use less than comprehensively descriptive metrics. To emphasize, the choice of metrics is not exclusive to individual evaluation steps in MAIPA; metrics must be linkable over the entire estimation process. Thus, even if an effect metric that more precisely and accurately tracks a particular environmental change is on hand, it cannot be used here unless there is a process to attribute economic value to its change. Many of the ongoing arguments surrounding the application of a richer set of noise metrics are moot for this study, but this should not be perceived to preclude their use for other decision-making protocols (cf. Fields 1993 in WHO 1999

[1] The DNL integration includes a 10 dB penalty added for nighttime events, assuming that night operations are twice as annoying as those occurring at other times of the day because of the potential for sleep disturbance and because background noise is lower.

FICUN, Albee, supp. Metrics).[2] Less established metrics may provide a more precise interpretation for specific endpoints (NRC 1981, FAA 1985, WHO 1999); reducing annoyance into more specific metrics of health and welfare impacts is an area of active research. A more complete synopsis of noise metrics and their applicability to environmental analyses can be found in (FAA Measures of Sound doc, Smith text, FICON 1992, 1050E.1, ANSI S3.23-1980, ISO 1982).

9.1. Choice of a noise metric

Before examining the consequences of the exposure trends on damages, we turn to an important assumption in their evaluation—the selection of a lower threshold that defines the absence of aircraft-related noise effects. In this thesis, effects extend as far as noise has a discernible impact on the health or well-being of populations around airports. This study uses an uncertain range $P_{unif}^{quiet}[50:55]$ for this threshold, referencing regulatory context and the effects literature. This section provides a rationale for this choice.

Noise impacts on airport-local communities are manifest strongly as quality of life issues.[3] Risk-based evaluations explicitly embrace quality of life factors in their definition of health, and as reviewed in the next section, emerging methods that seek to directly assess quality of life, interpreted as happiness or other qualitative measures, hold promise as alternatives for damage estimation. Risk-based evaluations suggest 50-55 DNL as a goal for the protection of public health and welfare. EPA (1973, 1974) conducted its first comprehensive assessment in response to a mandate in the Noise Control Act of 1972 and identified noise levels of 55 DNL and 45 DNL to minimize interference with outdoor and indoor activities, respectively, for all populations without regard to cost or feasibility and including a discretionary 5 dBA margin of safety. A more recent review considering a broader range of health endpoints by the World Health Organization (WHO 1999) suggests an outdoor target of 50-55 dBA

[2] Characteristics determining annoyance can include descriptors beyond physical noise levels and controlling factors can be similar to those controlling perception of risk. Consider the effective perceived noise level (EPNL), which is the event metric underlying certification standards set under the 1972 Noise Control Act (NCA) and subsequent amendments (REF). While the DNL metric is based on dBA, a loudness weighting, EPNL uses the tone-corrected, perceived noise level (PNL), a subjective measure of relative noisiness; this is an annoyance weighting. The essential connection between PNL and correlations between A-weighted sound level and magnitude of annoyance is displayed in the additive relationship between DNL and the cumulative measure it replaced, the Noise Exposure Forecast (NEF), which uses EPNL as the single event metric. Thus, there is a basic connection between MAIPA and certification, even though we do not use certification data in the evaluation of noise effects.

[3] Typical noise levels in near-airport communities are not high enough to cause acute hearing loss, but chronic exposure poses a long-term health risk for auditory effects.

integrated over 16 hours, and a target of 30-35 dBA integrated over 8-16 hours for indoor activity. The WHO guidelines are qualitatively different from the perspective of EPA (1974) in that they are presented in the context of sustainability. Considering annoyance in particular, the exposure-response correlation developed by Schultz (1978) and updated by various authors since (cf. Fidell 1990, Miedema and Vos 1998) enjoys acceptance as the most well-defined description of community response to aviation noise, commonly with citation of the percent of the population highly annoyed (%HA) as a function of DNL (NRC 1977, EPA 1982, FAA 1985); %HA is not significantly different from zero in the 50-55 DNL range. Annoyance analyses also find a significantly non-zero percentage of the population that is moderately-annoyed at levels where $\%HA \cong 0$. FAA references the number of people living within the 65 DNL contour as its primary measure of noise impact and progress in its mitigation, reflecting an interpretation of the noise – annoyance relationship. This does not, however, define the extent of welfare effects resulting from aviation noise, although this is often used as a definition of quiet. The 65 DNL contour is a land use guideline, the threshold of acceptability for activities with consideration of cost and feasibility (FICUN 1980); 14 CFR Part 150, established in response to the Aircraft Safety and Noise Abatement Act of 1979, describes FAA policy on fund disbursement for remedial noise mitigation. This policy continues to be debated—airport-local communities in Cleveland, Minneapolis, and Orlando, for example, have developed local actions that look to mitigate out to 60 or 55 DNL based on their own internal benefit-cost assessments (Albee Wyle Supplemental analysis 2002). FAA policy recognizes the need to attend to noise levels less than 65 DNL if actions alter noise levels significantly (FAA order 1050.1E, proposed noise policy 00).[4]

Noise impacts can be measured in a microeconomic sense by declines in utility, under which people adjust their consumption patterns to accommodate the shift. The corollary to this is that there is some amount people would be willing to pay to return to their original level of utility. Statistical investigations of price impacts on housing around airports resulting from aviation noise—a component of welfare loss which is addressed in more detail in the next section—find a significantly positive depreciation at levels between 55-60 DNL. However, uncertainty in DNL contour estimates at these levels (the primary source of uncertainty in MAIPA damage estimates) and the presence of other noise sources that may mask the

[4] An additional argument can be made that because certain land areas around airports are restricted from development because of incompatible noise levels, macroeconomic effects result, and regional economies are also impacted, increasing the number of people impacted to populations not necessarily near the airport.

impact of aircraft noise confound these results. Most airports are located near urban or suburban areas where the aviation noise signature is decreasing relative to other sources. EPA (1974) suggests the median outdoor exposure to noise in urban areas is 59 dB DNL with a range of 58 to 72 dB; corresponding ranges for suburban and wilderness areas are 48 to 57 dB and 20 to 30 dB, respectively (cf. EPA 1982). FICAN (1992) concludes that noise analyses do not provide a basis for determining the contribution of ambient levels to annoyance, including auditory masking for which there is no clear method to determine the extension of such a concept to long time scale integrations of noise level such as DNL. Aircraft noise, which varies with time, is not always fully masked, as some portion of the signature may exceed the ambient and thus the effect of the background may play as strong a role in comparison to constant level noise (cf. Gjestland et al. 1990 and Fields 1998 re: WHO 1999). However, auditory masking plays a role in determining whether aircraft contribution can be discerned, and suburban and urban noise levels are considered in selecting the noise effect threshold. FICON (1992) recommends that a 5 dBA difference in contours resulting from a change in airport operations warrants further analysis between the 55-60 DNL contours, 3 dBA for the 60-65 DNL contours, and 1.5 dBA above 65 DNL, each change resulting in a similar change in annoyance; these recommendations partly reference the ±3-5 dBA threshold for which an individual can discern a difference in noise level.

Appendix

10. Linear response models of the climate

The identification problem with respect to AOGCM simulations is more complex than that solved to determine the carbon-cycle impulse response; signal-to-noise ratios generally limit the number of significant modes. To evaluate the magnitude of cold start errors in climate modeling, Hasselmann et al. (1993) presents the model in equation A10.1 derived from the response of the ECHAM1/LSG coupled AOGCM to radiative forcing resulting from the IPCC IS92a emissions scenario.[1] In this case, one mode was found to simulate the response to a rms error of 0.06 K.

(A10.1) $$[T\text{-Ha93}] \quad g(t) = \sum_{i=1}^{5} \alpha_i e^{t/\tau_i} \quad \begin{bmatrix} \alpha_i \\ \tau_i \end{bmatrix} = \begin{bmatrix} 0.0679 \\ 36.8 \end{bmatrix} \begin{matrix} K/yr \\ \text{years} \end{matrix}$$

Additional models have been proposed based on different climate model experiments. Equation A10.2 shows the impulse response function from Hasselmann et al. (1997) based on a long timescale, 800-yr calculation of the response to a small step function input. Both this function and that shown in equation A10.2 are calibrated to an equilibrium $2 \cdot X_{CO_2}$ response (climate sensitivity, λ) of 2.5 K. The additional modes used in the fit as compared to Hasselmann et al. (1993) are included to capture the long-term response.

(A10.2) $$[T\text{-Ha97}] \quad g(t) = \sum_{i=1}^{5} \alpha_i e^{t/\tau_i} \quad \begin{bmatrix} \alpha_i \\ \tau_i \end{bmatrix} = \begin{bmatrix} 0.00383 & 0.0632 & 0.576 \\ 138.6 & 12 & 2.1 \end{bmatrix} \begin{matrix} K/yr \\ \text{years} \end{matrix}$$

Equation A10.3 shows the impulse response function from Hooss et al. (2001) which is derived from a stronger 850-yr $4 \cdot X_{CO_2}$ perturbation determined using the ECHAM3/LSG AOGCM (cf. DKRZ 1993; Voss et al. 1998; Voss and Mikolajewicz 2001) computation where the CO2 concentration rises exponentially to the 120th year and keeps constant thereafter. The model is calibrated to λ = 2.39 K.

[1] ECHAM is based on the spectral numerical weather forecasting model of the European Center for Medium Range Weather Forecasts Modified by Max Planck Institute (Hamburg), with added physical parameterizations to adjust for climate analysis (e.g. radiation, cloud formation, land-surface processes). ECHAM1 was coupled to Hamburg Large Scale Geostrophic (LSG) ocean general circulation model. Circulation fields derived from LSG underlie the derivation of impulse response functions using HAMMOC.

(A10.3) [T-Ho01] $g(t)=\sum_{i=1}^{5}\alpha_i e^{t/\tau_i}$ $\begin{bmatrix}\alpha_i\\ \tau_i\end{bmatrix}=\begin{bmatrix}0.000725 & 0.0592\\ 400 & 12\end{bmatrix}$ $\begin{matrix}K/yr\\ \text{years}\end{matrix}$

Cumulative radiative forcing and temperature change computed using the impulse response functions of equations 4.11-4.13 agree with AOGCM computations of temperature change resulting from global aviation CO2 contribution as reported in the IPCC special report on aviation and the global atmosphere (IPCC 1999, cf. Marais et al. 2008 and Sausen and Schumann 2000). Note these linear response models indicate instantaneous surface temperature change at t = 0 and thus do not capture the transient increase in Ts.

As with the linearized carbon cycle models presented in the previous section, coefficient and characteristic timescale uncertainties have not been evaluated for MAIPA. Uncertainty associated with climate sensitivity is characterized using the analysis of Forest (2002) which is based on calculations performed with the MIT AOGCM (Sokolov and Stone 1998). For ensemble of 1000 runs, the distribution derived from this analysis has a median λ = 2.38 K and percentiles $F(\lambda)$ as shown in equation A10.4.

(A10.4) $$\begin{bmatrix}F(\lambda)\\ \lambda\end{bmatrix}=\begin{bmatrix}0.025 & 0.05 & 0.25 & 0.5 & 0.75 & 0.95 & 0.975\\ 1.3 & 1.4 & 1.95 & 2.38 & 2.96 & 4.2 & 4.7\end{bmatrix}K$$

The uncertainty is implemented as a scaling on the temperature response predicted by equations A10.1-A10.4 relative to the climate sensitivity of the underlying AOGCM.[2] The shape of this distribution is characteristic of AOGCM results (cf. IPCC FAR 2007).

For the purposes of this study, we use the estimates for RF^{ref}_{short} in (Schumann 2003) (cf. Sausen et al. 2005), which update the IPCC special report (1999) based on a review of recent literature. The reported ranges are representative of the 67% confidence interval; for MAIPA, they are extended to represent a 3σ confidence interval. Equation A10.5 gives these distributions for RF^{ref}_{short}. The RF values in equation 4.16 result from the whole of global aviation emissions. To maintain relevance to US emissions, RF^{ref}_{short} are

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[2] The sum of coefficients α is such that RF* = 1 at the equilibrium T_s response to $2\cdot X_{CO_2}$, *e.g.*: $\sum\left[\left(\alpha_i\Big/\sum_{i=1}^{I}\alpha_i\cdot\lambda\right)\Big/\tau_i\right]=1$

(Marais et al. 2008) explore the effect of uncertainty in λ differently, using Punif=[1.5,4.5] K in an energy balance model proposed by Shine et al. (2005).

scaled by a factor α to account for the portion attributable to the inventories Q using the ratio shown in equation A10.5, where $Q^{global}_{CO_2}$ is specified as reported in (Sausen and Schumann 2000).

$$
\begin{array}{ll}
\underline{\left|\ \mathrm{RF}^{ref}_{short} \text{ in } m\mathrm{W/m^2} \text{ for 1992 emissions}\right.} & \underline{\left|\lambda^*\right.} \\
P_{trig}\left(\mathrm{RF}^{ref}_{\mathrm{SO_4^{2-}}}\right)=\left[\ -9\ \ -3\ \ -1\ \right] & P_{unif}\left(\lambda^*_{\mathrm{SO_4^{2-}}}\right)=\left[\ 0.6\ \ 1.1\ \right] \\
P_{trig}\left(\mathrm{RF}^{ref}_{\mathrm{C}}\right)=\left[\ 1\ \ 3\ \ 9\ \right] & P_{unif}\left(\lambda^*_{\mathrm{C}}\right)=\left[\ 0.4\ \ 0.8\ \right] \\
P_{unif}\left(\mathrm{RF}^{ref}_{\mathrm{clouds}}\right)=\left[\ 6\ \ 80\ \right] & P_{unif}\left(\lambda^*_{\mathrm{clouds}}\right)=\left[\ 0.6\ \ 0.7\ \right] \\
P_{trig}\left(\mathrm{RF}^{ref}_{\mathrm{H_2O}}\right)=\left[\ 0\ \ 5\ \ 18\ \right] & \lambda^*_{\mathrm{H_2O}}=1 \\
P_{trig}\left(\mathrm{RF}^{ref}_{\mathrm{O_3}=f(\mathrm{NO_x})}\right)=\left[\ 12\ \ 18\ \ 27\ \right] & P_{unif}\left(\lambda^*_{\mathrm{O_3}=f(\mathrm{NO_x})}\right)=\left[\ 0.45\ \ 2.5\ \right] \\
\mathrm{RF}^{ref}_{\mathrm{CH_4}=f(\mathrm{NO_x})} \propto \mathrm{RF}^{ref}_{\mathrm{O_3}=f(\mathrm{NO_x})} \text{ via Eq.(2.17)} & \lambda^*_{\mathrm{CH_4}}=1.1
\end{array}
\tag{A10.5}
$$

Equation A10.5 also shows the distributions applied in MAIPA for the climate sensitivity ratio $\lambda^* = \lambda_{short}/\lambda_{CO_2}$. The remainder of this section discusses the basis for the specifications in equation 4.16. Section 4.2.1 discusses ozone and methane effects related to nitrogen oxide emissions and section 4.2.2 reviews specifications for aerosol and cloud effects related to water, soot, and sulfur emissions. Values for $\mathrm{RF}^{ref}_{short}$ are specified independently from λ_{short}; without computations to describe this dependency that explicitly consider the aviation case, there is currently no mechanistic basis to accurately capture the coupling in MAIPA.

Appendix

11. Upper atmospheric effects of nitrogen oxide emissions

There are several sources of NO_x at altitude, including vertically transported surface emissions from biomass burning, industrial sources, vehicular traffic, and soil microbes, and in the free troposphere, lightning and subsidence of NO_x from the stratosphere. Aviation is unique among anthropogenic sources in that emissions occur at altitude. Significant chemical and dynamical processes differentiate the troposphere and stratosphere. Aircraft emissions injected into the troposphere generally lead to the production of O3 through the processes that lead to ozone formation at ground level and related air quality impacts. Despite deficiencies in understanding source magnitudes, it is understood that NO_x emissions at altitude have a more potent effect on O3 than equivalent emissions at ground level, primarily due to the relatively clean background concentrations in the upper atmosphere, but also because of longer residence times and the large radiative efficiency near the tropopause as compared to the surface, the latter resulting from low temperatures and thus low reemission to space.[1]

In contrast, pollutants emitted into the middle stratosphere can reduce O3 levels through catalytic chemical cycles directly involving NO_x emissions, but also indirectly through nonlinear interactions among NO_x, hydroxy (HOx), and halogen (e.g. bromine oxides, BrOx, and chlorine oxides, ClOx) catalytic O3 chemistry. Baughcum et al. (ref) estimate that ~20% of aircraft emissions are injected into the stratosphere. The loss of O3 at this altitude is most often associated with the emission of chlorofluorocarbons and other halogen compounds. Thus, one difficulty in discerning the climatic effects of aircraft is that increases in O3 in the middle and upper stratosphere lead to decreased forcing whereas increases in ozone in UT/LS lead to increased forcing. The crossover point between these differing influences is in the lower stratosphere at around 15 km altitude, varying with season and latitude;

[1] Models suggest that $\mathrm{OPE}_{alt} \sim 20 \cdot \mathrm{OPE}_{ground}$ and that $\mathrm{RF}_{alt}^{f(\mathrm{NO}_x)} \sim 20 \cdot \mathrm{RF}_{ground}^{f(\mathrm{NO}_x)}$ (Wahner and Geller 1995; Johnson et al. 1992)

subsonic aircraft fly at altitudes spanning this crossover. The net effect is estimated to be an increase in O3 and thus a positive RF_{O_3} (IPCC 1999).[2]

With this heterogeneous distribution of ozone perturbations and forcings that manifest in different ways depending on location and the manner of impact (e.g. different parts of the spectrum impacted by different mechanisms), climate sensitivity estimated with specific reference to aircraft NO_x emissions is most consistent with application to MAIPA. (Ponater et al. 1999) explicitly calculates λ_{O_3} based on computations focusing on aircraft-induced O3 perturbations, using the ECHAM3-LSG model (the basis for several of the impulse response functions discussed in the previous section). Their results find λ_{O_3} = [1.52 2.48] at the 95% CI with $\bar{\lambda}_{O_3} = 2.01$ compared to a sensitivity for CO2 (and other well-mixed species) of λ_{CO_2} = [0.67 1.10] at the 95% CI with $\bar{\lambda}_{CO_2} = 0.88$, giving $\tilde{\lambda}^*_{O_3} = 2.28$. These results are, however, singular in the literature. To specify the climate sensitivity ratio in equation 4.16, we additionally reference assessments not specific to aviation.

Comparatively, analyses of O3 response in the atmosphere suggest a smaller climate sensitivity. (Hansen et al. 1997) conducted computational experiments in which ozone was increased by 100 Dobson units at various altitudes and latitudes, finding λ_{O_3} = [0.43 2.27] with negative forcing at the lowest altitudes ($\lambda_{O_3} = -2.05$ at <~ 100 m). For these computations, $\lambda^*_{O_s}$ = [0.47 2.46] ($\lambda_{CO_2} = 0.92$ for a $2 \cdot X_{CO_2}$ computation using the climate model described in Hansen 1997b). An updated assessment in (Hansen et al. 2005) suggests a net $\lambda^*_{O_s} = 0.80 \pm 0.16$ from computations examining uniform tropospheric increases in in O3. (Stuber et al. 2001) reports similar results in an analysis examining O3 impacts using the same GCM as in (Ponater et al. 1999), finding a mean $\lambda^*_{O_s}$ = [0.72 1.81]. In equation 4.16, $\lambda^*_{O_3}$ is specified to encompass these results.[3] Note that while the trend of NO_x impact on O3 is explicitly treated, the impact of emitted CO and HC on O3 is not; as in the air quality analysis described in the previous section, this

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[2] Soot and sulfate can enhance the destruction of ozone (Weisenstein et al., 1995; Danilin et al., 1997; Bekki, S., 1997), but these impacts have been assumed to be small relative to the NO_x impact on ozone (IPCC 1999). Recent analysis suggests that the role of heterogeneous chemistry in the troposphere involving volatile particles in the engine plume can increase the rate of NO_x to nitrate conversion, thereby reducing or reversing NO_x effects on ozone (increasing) and methane (decreasing) concentrations. The question of heterogeneous effects on NO_x conversion is not resolved and RF estimates published to date have not included the detailed treatment of heterogeneous chemistry required to capture the conversion rate. Regardless of the role of heterogeneous chemistry, both of these processes will be affected similarly (even if reversed), reducing the importance of NO_x conversion to climate processes.

[3] The (Ponater *et al.* 1999) results indicate that some weight should be given to the upper end of this distribution, but instead of formally implementing a correction in MAIPA, this is left as a point for further investigation.

chemistry modifies ozone chemistry leading to additional formation but with an impact much less than the that of NO_x since CO and HC perturbations are much smaller than the background CO and HC.

Appendix

12. Aerosol and cloud effects of water, soot, and sulfur emissions

There is comparatively less understanding of climate sensitivities to aviation aerosols and cloud effects. Particle emissions perturb the mass and size distribution of the background atmospheric aerosol if scavenged, alter the chemical makeup of the upper atmosphere through heterogeneous chemistry, and may freeze, persisting as contrail particles. Microphysical processes involving emitted and generated particles also lead to changes in contrail optical properties, but have minimal apparent impact on formation tendency (Schumann et al. 2002). However, the incidence of persistent contrails is expected to increase as aircraft engines become more efficient (Schumann 2000; Schumann et al. 2000; Detwiler and Jackson 2002).

Measurements suggest the presence of radiative impacts from aviation contrails and related cirrus (Travis et al. 2002), but the impact of aerosols on clouds is not well understood (Changnon 1981; Liou et al. 1990; Boucher 1999). Scattering of radiation by sulfate and absorption by soot are estimated to result in negative and positive forcings, respectively (IPCC 1999). (Taylor and Penner 1994) find that sensitivity to sulfate is less than to CO2, estimating $\lambda^*_{SO_4^{2-}} = 0.59$; similarly, (Rotstayn and Penner 2001) results estimate $\lambda^*_{SO_4^{2-}} = 0.69$. (Hansen et al. 2005) finds a large range of values— $\lambda^*_{SO_4^{2-}} = [0.6\ 1.1]$ and $\lambda^*_C = [0.4\ \ 0.8]$— using a series of computations varying altitude, spatial distribution, and albedo, that implies great sensitivity to the character of the soot and sulfate. Given the uncertainty in the application of these results to the aviation case, the specifications in equation 4.16 are inclusive.

Particles also play a part in cirrus formation near flight tracks, via contrail evolution into cirrus for example, and as a result of the more extensive spatial coverage that results, aviation-induced cirrus impacts are expected to be larger than contrail impacts (Anderson et al. 1999; Schröder et al. 2000). (IPCC 1999) place $RF_{clouds} = [5{:}60]$ mW/m2 with $\overline{RF}_{clouds} = 0.020$ mW/m2 (based on the results of Minnis et al. 2004) and suggest an uncertain range for the impact of aerosols on cirrus clouds of [0:40] mW/m2. Subsequent analyses suggest a lower contrail forcing; (Marquart and Mayer 2002) estimate a contrail impact of 3.2 mW/m2 and (Marquart et al. 2003) estimate 3.5 mW/m2. Mannstein and Schumann (2003) extrapolate from contrail forcing estimates and observations of cloud cover over Europe to suggest that the cirrus impact related to contrail aging is 10 times the contrail forcing, giving RF_{clouds} = [30 80] mW/m2. The mechanism and magnitude of contrail/cirrus forcings are unsettled questions. For MAIPA, a range that encompasses existing studies is specified in equation 4.16.

At the altitudes of cirrus formation where aircraft are expected to fly at cruise, (Hansen et al. 1997) finds climate sensitivities of λ_{clouds} = [0.59 0.61] through a computation increasing cloud cover by 5% in each of the model's gridded layers. This places λ^*_{clouds} = [0.64 0.66], rounded in MAIPA to specify equation 4.16. Based on the relationship between RF and fuel consumption estimated in (Marquart et al. 2003), the scaling in equation 4.15 is assumed to apply over the period 1991-2003, with the EI ratio at unity. Since water vapor is the main component of cloud impacts, MAIPA attributes cloud and contrail impacts to water emissions; the potential for operational approaches to reduce contrail formation by rerouting around water saturated regions of the atmosphere is included in the damage function as multiplier set to 1 for MAIPA.

Apart from its role in clouds, water vapor is also a radiatively active gas; but while it is a major quantitative component of aircraft emissions, these emissions are much smaller than the flux of the hydrological cycle in the troposphere and thus, water vapor affects a relatively small positive forcing—approximately the same as for soot. Emissions into the stratosphere may lead to a larger impact due to the relatively dry conditions and longer lifetimes, but this effect has not been fully assessed. Water emissions also contribute to an enhanced catalytic HOx destructiveness and interfere with NO_x and halogen catalysis. However, this occurs to a lesser extent than the similar effect of NO_x emissions described above because of the small H2O perturbation to the background level (Schumann, 1996). For water vapor, $\lambda^*_{H_2O}$ = 1 is assumed in the absence of additional understanding.

Appendix

13. Tropospheric ozone production

The formation of tropospheric ozone strongly depends on ambient NO_x concentrations. In the troposphere, NO_2 is converted to O3 through a fast, photochemically-initiated catalytic cycle. This cycle leads to a steady O3 concentration balanced as a result of the interaction of NO and NO_2 with oxygen in the presence of sunlight. In this chemistry, NO_2 photodissociates to NO, producing an oxygen atom which combines with ambient oxygen to form ozone. The ozone is then recycled through reaction with NO into NO_2, leading to what is commonly referred to as the photostationary state. In the lower troposphere, carbon monoxide and hydrocarbons are plentiful enough to spur the formation of more atomic oxygen through further recycling of NO_x, producing O3 without the subsequent destruction implied by the photostationary state. In the presence of CO and HC oxidation via atmospheric OH, NO is converted to NO_2 by reaction with peroxy radicals and then converted back to NO in the presence of sunlight. This chemistry releases atomic oxygen which reacts with ambient O2 to form O3. The oxidation of NO by HO2 leads to the production of OH which contributes to further CO and HC oxidation.

The ratio of reactive organic gases (ROG) to NO_x is a useful parameter for describing the evolution of emitted precursors (cf. NRC 1991), but not a sufficient marker of O3 sensitivity (Lu et al. 1998). For example, investigating four episodic scenarios—along the east coast from Boston to Miami, in the northeast, in the Los Angeles air basin, and in the Chicago-Lake Michigan air basin—Milford et al (1994) find that downwind of source areas, sensitivity to ROG decreases with distance. While NO_x levels are plentiful and ROG/NO_x is still low, maximum ozone production is limited by the availability of ROG. In this condition—termed ROG-sensitive or NO_x-saturated—the addition of CO and HC enhances the ozone production efficiency of NO_x emissions; additional NO_x has the opposite effect. Milford et al (1989) show that aging urban plumes evolve through dilution and chemistry towards a condition where O3 sensitivity to NO_x is positive and changes in CO and HC emissions are much less influential—termed NO_x-limited or NO_x-sensitive—due to the relatively higher rate of NO_x consumption compared to ROG consumption; at the extreme, O3 production varies directly with NO_x input.

There are numerous sensitivities in this chemistry, including availability of solar radiation for photolysis and air temperature, as well as the availability of precursors. Consistent with this picture of lower tropospheric ozone, rural and suburban areas, where ROG sources are relatively abundant (e.g isoprene from forests in the eastern US) tend towards the NO_x-limited end of this spectrum while urban areas, where NO_x sources are numerous, are typically NO_x-saturated (Sillman et al. 1990; Lu et al. 1989; Sillman 1999).

13.1. Plume ozone production

In the earliest of these measurement campaigns, Trainer et al. (1993) report a series of seven sampling efforts in rural areas of the eastern U.S. and Canada in which measured O3, NO_x, and NOy are correlated to estimate the OPE as dO3/d(NOz), where NOz = NO_x-NOy, essentially a measure of HNO3 and thus the extent of NO_x oxidation. They set a requirement, generally followed by subsequent analyses, that only samples where plume oxidation is >= 60% (i.e. the plume is photochemically aged) are applied to the correlation, thus avoiding any region of NO_x titration. Trainer et al. (1993) find consistent values for dO3/d(NOz) across four sites with differences in emissions factors, background concentrations, relative rates of O3 formation and NO_x oxidation, and transport process, reflecting a regional homogeneity along the interior of the eastern U.S. For one site (Scotia, PA), the OPE ~= 8.5 against day averages (R^2 = 0.99). Using a similar method, Olszyna et al. (1994) examine another rural site (Giles County, TN) estimating OPE~=10, integrating between a plume age of 1-NO_x/NOy = 0.2 to the extent of ozone production at 1-NO_x/NOy = 0.7 (cf. Kleinman et al. 1994 for additional rural measurements). Subsequent studies using this assessment approach estimate OPE~=7 for the Birmingham urban plume (Trainer et al. 1995) and OPE = 4-10 for the Atlanta urban plume (Imhoff et al. 1995). These initial reports tend towards higher OPE than subsequent analyses. This is largely a result of missing loss processes; in using NOz as the correlating species for ozone formation, only the photochemical portion of the processes that influence the rate of ozone change are accounted. If a longer-lived species is correlated instead, such as CO, with additional information as to source NO_x/CO emissions ratio, lower OPE estimates are obtained. Using this approach, Trainer et al. (2000) reassess the Scotia PA result and find instead an OPE = 1.5-2.8 compared to the previously reported OPE = 8.5 (cf. Trainer et al. 1993).

A large portion of the work to date on measurement-based estimates of OPE originates with the Southern Oxidant Study (SOS) of photochemistry in the southeastern US (Hübler et al., 1998; Meagher et al., 1998;

Goldan et al., 2000). Nunnermaker et al. (1998) estimate OPE from 2.5-4 from measurements of the Nashville urban plume (which includes several large power plant contributions). In a companion modeling effort, Sillman et al. (1998) estimate that using a O3-NOz correlation would result in a 50-60% higher value for OPE. Nunnermaker et al. (1998) further observe the variation of OPE across the plume, finding a factor of 2 decrease from boundary to center.

Again in Tennessee, Ryerson et al. (1998) specifically measured power plant plumes at ages up to 10 hours using two methods to estimate net OPE that account for loss rates of soluble NOz species: first pinning loss rates to longer-lived species co-emitted in the plume (a mass balance approach) and second estimating NO_x lifetimes based on concentration measurements, using SO2 as a tracer of the plume. They report OPE = 1-3 using mass balance approach, a lower limit, and 2-7 using the concentration approach, an upper limit. Ryerson et al. (1998) also note the dependence of O3 on mdot NO_x, finding a decreasing trend in OPE with increasing NO_x emission rate (cf. Milford et al. 1989, 1994); in a subsequent analysis of power plant plumes in the area, Ryerson et al. (2001) infer from their measurements that, for similar meteorological and geographic conditions, a factor of 8 reduction in NO_x emissions from these sources would result in a in factor of 2.3 difference in net ozone production, another reflection of the nonlinearity in the ozone chemistry and an indication of a larger OPE for the smaller sources.

Ryerson et al. (2001) also conclude that power plant plumes in the east, where biogenic sources of ROG are prevalent, will tend to have higher OPE than similar power plants in midwest agricultural areas (i.e. where biogenic sources dominate, the chemistry tends towards NO_x-limitation). Additional measurement and model analyses of urban plumes appearing since the SOS are consistent in their estimates of OPE, ranging from 1-7 for major cities in the eastern and western U.S. (cf. Kleinman et al. (2000) NYC urban plume with OPE=2.2-4.2; Kleinman et al. (2002) Phoenix urban plume with OPE=1.5-7; Kleinman et al. (2002); Griffin et al. (2004) CA South Coast air basin modeled OPE=4.7-6.4; Frost et al. (2006) eastern U.S. power plants modeled OPE=1-5).

An aircraft plume is definitively NO_x-saturated early in its evolution. Upon emission, aircraft CO and HC emissions constitute the ROG available to the ozone chemistry (ROG/NO_x = O(1), NO_x-exit >> NO_x-ambient, and ROG-exit <= ROG-ambient). With no initial O3 content in the plume, entrained lower tropospheric ozone can be destroyed rapidly via NO reaction with O3—termed NO titration—leading to a depression in ozone concentrations relative to the ambient average. Similar depressions can be found near streets in downtown areas and in power plant plumes, the latter can be of a scale causing the depression to persist for O(10) km.

Atmospheric mixing affects ozone chemistry by changing the ROG/NO_x ratio as well as the absolute concentrations of ROG and NO_x. Eventually, NO titration is superseded by ozone production, spurred by the photolysis and/or ROG-initiated generation of a radical pool. This evolution pushes the ozone chemistry towards the NO_x-limited condition; Milford et al (1994) demonstrate that for a given ROG/NO_x ratio, a deficit of NO_x is encountered faster for low absolute emissions levels. It is for this reason that the emphasis of emissions control varies with location; VOC controls in urban areas successfully reduce O3 peaks, but in rural areas, biogenic sources contribute to a large background ROG concentration where NO_x controls may be the more effective strategy.

Competing influences determine the OPE realized in aircraft plumes; aircraft engines are relatively small sources compared to power plant or urban plumes—higher OPE values characterize smaller NO_x sources and the diameter of a stack compared to that of an engine exhaust nozzle is ~ O(10)—but are emitted predominantly in urban areas where the availability of anthropogenic NO_x is high and biogenic VOC sources are low, both of which may depress sensitivity to NO_x inputs. There is also the potential difference in that ground emissions evolve against the background of a larger regional plume (e.g from a city) where aircraft emit at higher tropospheric altitudes, perhaps interacting less with ground emissions and moving through an atmosphere that has a lower ROG/NO_x ratio than on the airport property.

13.2. Tropospheric and plume ozone production

The formation of tropospheric ozone strongly depends on ambient NO_x concentrations. In the troposphere, NO_2 is converted to O3 in through a fast, photochemically-initiated catalytic cycle. This cycle leads to a steady O3 concentration balanced as a result of the interaction of NO and NO_2 with oxygen in the presence of sunlight. In this chemistry, NO_2 photodissociates to NO, producing an oxygen atom which combines with ambient oxygen to form ozone. The ozone is then recycled through reaction with NO into NO_2, leading to what is commonly referred to as the photostationary state. In the lower troposphere, carbon monoxide and hydrocarbons are plentiful enough to spur the formation of more atomic oxygen through further recycling of NO_x, producing O3 without the subsequent destruction implied by the photostationary state. In the presence of CO and HC oxidation via atmospheric OH, NO is converted to NO_2 by reaction with peroxy radicals and then converted back to NO in the presence of sunlight. This chemistry releases atomic oxygen which reacts with ambient O2 to form O3. The oxidation of NO by HO2 leads to the production of OH which contributes to further CO and HC oxidation.

The ratio of reactive organic gases (ROG) to NO_x is a useful parameter for describing the evolution of emitted precursors (cf. NRC 1991), but not a sufficient marker of O3 sensitivity (Lu et al. 1998). For example, investigating four episodic scenarios—along the east coast from Boston to Miami, in the northeast, in the Los Angeles air basin, and in the Chicago-Lake Michigan air basin—Milford et al (1994) find that downwind of source areas, sensitivity to ROG decreases with distance. While NO_x levels are plentiful and ROG/NO_x is still low, maximum ozone production is limited by the availability of ROG. In this condition—termed ROG-sensitive or NO_x-saturated—the addition of CO and HC enhances the ozone production efficiency of NO_x emissions; additional NO_x has the opposite effect. Milford et al (1989) show that aging urban plumes evolve through dilution and chemistry towards a condition where O3 sensitivity to NO_x is positive and changes in CO and HC emissions are much less influential—termed NO_x-limited or NO_x-sensitive—due to the relatively higher rate of NO_x consumption compared to ROG consumption; at the extreme, O3 production varies directly with NO_x input.

There are numerous sensitivities in this chemistry, including availability of solar radiation for photolysis and air temperature, as well as the availability of precursors. Consistent with this picture of lower tropospheric ozone, rural and suburban areas, where ROG sources are relatively abundant (e.g isoprene from forests in the eastern US) tend towards the NO_x-limited end of this spectrum while urban areas, where NO_x sources are numerous, are typically NO_x-saturated (Sillman et al. 1990; Lu et al. 1989; Sillman 1999).

Aircraft comprise smaller NO_x sources and the diameter of a stack compared to that of an engine exhaust nozzle is ~ O(10)—but are emitted predominantly in urban areas where the availability of anthropogenic NO_x is high and biogenic VOC sources are low, both of which may depress sensitivity to NO_x inputs. There is also the potential difference in that ground emissions evolve against the background of a larger regional plume (e.g from a city) where aircraft emit at higher tropospheric altitudes, perhaps interacting less with ground emissions and moving through an atmosphere that has a lower ROG/NO_x ratio than on the airport property.

13.2.1. Indications of NO_x processing age from precursor measurements

Modeling and measurement investigations of aircraft exhaust plume chemistry indicate that formation of NO_2 is substantially underway in the near-field (t~1s). The constituency of aircraft NO_x emissions leans predominantly to NO upon exhaust; median estimates of intra-engine conversion efficiencies place the ratio of NO_2/NOy between 5-40% over the range of engine cycles applied to this study, the low end relevant to high power conditions and the high for low power (Lukachko et al 2007). For comparison, Herndon et al. (2004) measure conversion efficiencies in an aged plume (t~100s) of 28-35% over a power range from idle to takeoff at Kennedy International Airport; a companion modeling exercise shows no evidence of NO titration at this age. Wormhoudt et al. (2007) measure an upper-end conversion efficiency of 80% at low powers near the engine exit for a CFM56 engine installed on a B757, indicating that ozone production is underway within the airport boundaries. Field studies have recently focused on the identification of aircraft source signatures outside the airport boundary. Herndon et al (2004, 2005, 2006) report measurable NO_x, VOC, and PM plumes at the airport boundary, and Westerdahl (2007) find aircraft emission signatures up to 1 km from the airport at monitoring locations around Los Angeles. Craslaw et al. (2006), using statistical methods that take advantage of the larger buoyancy of aircraft plumes compared to other nearby combustion sources, find a NO_x source signature at distances up to 3 km outside London Heathrow airport, accounting for a maximum 15% of the total contribution. These studies indicate that aircraft emissions retain a local influence that is differentiable from other sources.

Appendix

14. Concentration-response functions

Concentration-response functions in MAIPA address premature and sudden mortality, chronic respiratory illness (e.g. chronic bronchitis), hospital admissions and emergency room visits for respiratory (e.g. asthma, pneumonia) and cardiovascular diseases (e.g. chronic obstructive pulmonary disease, congestive heart failure, dysrhythmia, ischemic heart disease), and minor symptomatic illness as well as reduced activity that may be associated with illness. These categories aggregate the more specific health endpoints addressed by individual concentration-response functions; category assignments are listed in tables A14.1-A14.6.

Table A14.1 Selecting concentration-response functions for air quality benefits analyses

reproduced from EPA 1999 Table D-1

Consideration	Comments
Peer reviewed research process	Peer reviewed research is preferred to research that has not undergone the peer review
Study type	Among studies that consider chronic exposure (e.g., over a year or longer) prospective cohort studies are preferred over cross-sectional studies (a.k.a. "ecological studies") because they control for important confounding variables that cannot be controlled for in cross-sectional studies. If the chronic effects of a pollutant are considered more important than its acute effects, prospective cohort studies may also be preferable to longitudinal time series studies because the latter type of study is typically designed to detect the effects of short-term (e.g. daily) exposures, rather than chronic exposures.
Study period	Studies examining a relatively longer period of time (and therefore having more data) are preferred, because they have greater statistical power to detect effects. More recent studies are also preferred because of possible changes in pollution mixes, medical care, and life style over time.
Study population	Studies examining a relatively large sample are preferred. Studies of narrow population groups are generally disfavored, although this does not exclude the possibility of studying populations that are potentially more sensitive to pollutants (e.g., asthmatics, children, elderly). However, there are tradeoffs to comprehensiveness of study population. Selecting a C-R function from a study that considered all ages will avoid omitting the benefits associated with any population age category. However, if the age distribution of a study population from an "all population" study is different from the age distribution in the assessment population, and if pollutant effects vary by age, then bias can be introduced into the benefits analysis.
Study location	U.S. studies are more desirable than non-U.S. studies because of potential differences in pollution characteristics, exposure patterns, medical care system, and life style.
Pollutants included in model	Models with more pollutants are generally preferred to models with fewer pollutants, though careful attention must be paid to potential collinearity between pollutants. Because PM has been acknowledged to be an important and pervasive pollutant, models that include some measure of PM are highly preferred to those that do not.
Measure of PM	PM2.5 and PM10 are preferred to other measures of particulate matter, such as total suspended particulate matter (TSP), coefficient of haze (COH), or black smoke (BS) based on evidence that PM2.5 and PM10 are more directly correlated with adverse health effects than are these more general measures of PM.
Economically valuable	Some health effects, such as forced expiratory volume and other technical valuable health measurements of lung functioning, are difficult to value in monetary terms. These health effects effects are not quantified in this analysis.
Non-overlapping endpoints	Although the benefits associated with each individual health endpoint may be analyzed separately, care must be exercised in selecting health endpoints to include in the overall benefits analysis because of the possibility of double counting of benefits. Including emergency room visits in a benefits analysis that already considers hospital admissions, for example, will result in double counting of some benefits if the category "hospital admissions" includes emergency room visits.

Table A14.2 Concentration-response functions for carbon monoxide morbidity

source	effect	MAIPA pool	ages	model	yo / alpha	beta	sigma	averaging
Burnett et al. (1999)	asthma hospitalization	respiratory hospitalization	all	log-linear	4.75E-06	0.0332	0.00861	daily mean
Sheppard 1989	asthma hospitalization	respiratory hospitalization	<65	log-linear	4.52E-06	0.0528	0.0185	daily mean
Burnett et al. (1999)	lung disease hospitalization	respiratory hospitalization	all	log-linear	5.76E-06	0.025	0.0165	daily mean
Moolgavkar et al. (1997)	chronic lung disease hospitalization	respiratory hospitalization	65+	log-linear	3.75E-05	0.0573	0.0329	daily mean
Schwartz 1999	all cardiovascular hospitalization	cardiovascular hospitalization	65+	log-linear	2.23E-04	0.0127	0.00255	daily 1-hr max
Schwartz 1997	all cardiovascular hospitalization	cardiovascular hospitalization	65+	log-linear	2.23E-04	0.0139	0.00715	daily 1-hr max
Burnett et al. (1999)	dysrhythmia hospitalization	cardiovascular hospitalization	all	log-linear	6.46E-06	0.0573	0.0229	daily mean
Burnett et al. (1999)	congestive heart failure hospitalization	cardiovascular hospitalization	all	log-linear	9.33E-06	0.034	0.0163	daily mean
Schwartz and Morris 1995	congestive heart failure hospitalization	cardiovascular hospitalization	65+	log-linear	5.82E-05	0.017	0.00468	daily 1-hr max
Schwartz and Morris 1995	ischemic heart disease hospitalization	cardiovascular hospitalization	65+	log-linear	9.96E-05	0.000467	0.000435	daily 1-hr max
Notes	% NAAQS CO linear (1): 35 ppm linear (1)-hr average, log-linear (2): 9 ppm 24-hr average unit equivalencies assumed: % 8hr CO for [daily.mean,daily.1hr.max]							

Table A14.3 Concentration-response functions for nitrogen dioxide morbidity

source	effect	MAIPA pool	ages	model	yo / alpha	beta	sigma	averaging
Burnett et al. (1997b)	all respiratory hospitalization	respiratory hospitalization	all	log-linear	2.58E-05	0.00378	0.00221	daily 12-hr mean
Burnett et al. (1999)	respiratory infection hospitalization	respiratory hospitalization	all	log-linear	1.56E-05	0.00172	0.000521	daily mean
Moolgavkar et al. (1997)	pneumonia hospitalization	respiratory hospitalization	65+	log-linear	5.30E-05	0.00169	0.00125	daily mean
Burnett et al. (1999)	congestive heart failure hospitalization	cardiovascular hospitalization	all	log-linear	9.33E-06	0.00264	0.000769	daily mean
Burnett et al. (1999)	ischemic heart disease hospitalization	cardiovascular hospitalization	all	log-linear	2.23E-05	0.00318	0.000521	daily mean
Hasselblad et al. 1992	respiratory symptoms	respiratory illness	5-9	log-linear	-5.36E-01	0.0275	0.0132	annual mean
Notes	% NAAQS NOx linear (1): 0.053 ppm linear (1)-yr average unit equivalencies assumed: % NO2 arith mean for [daily.12hr.mean,daily.mean,ann.mean] % logistic (3) pop equivalencies assumed: % for Hasselblad92, 5-9 for 6-7 % 4 for Hasselblad92, alpha & gender coef (-0.0295 for males) obtained via pers comm by Abt Assoc., eqn based on Mella et al. (1980) for summation of incidences, males first with gender coefficient, gamma, = -0.0295, then females with gamma = 0							

Table A14.4 Concentration-response functions for sulfur dioxide morbidity

source	effect	MAIPA pool	ages	model	yo / alpha	beta	sigma	averaging
Burnett et al. (1997b)	all respiratory hospitalization	respiratory hospitalization	all	log-linear	2.58E-05	0.00446	0.00293	daily 1-hr max
Moolgavkar et al. (1997)	pneumonia hospitalization	respiratory hospitalization	65+	log-linear	5.30E-05	0.00143	0.0029	daily mean
Burnett et al. (1999)	ischemic heart disease hospitalization	cardiovascular hospitalization	all	log-linear	2.23E-05	0.00177	0.000854	daily mean
Linn et al. (1987, 1988, 1999); Roger et al. (1995)	respiratory symptoms	respiratory illness	all	log-linear	-5.65	0.00589	0.00247	peak 5-min concentration in an hour
Notes	% NAAQS SO2 linear (1): 0.14 ppm 24-hr average, log-linear (2): 0.03 ppm linear (1)-yr average unit equivalencies assumed: 24hr SO2 for [daily.1hr.max,daily.mean,peak.5min.in.hr] % for chest tightness etc., results of four chamber studies were combined % to develop the cr function; for calculation, 1.7% are exercising % asthmatics, one-third of whom are moderate asthmatics, two-thirds of % whom are mild asthmatics (see Adams et al 1995 Table 57, and US EPA % 1997 pD-39; assuming this represents a daily incidence gamma = [0,1.10]; % gamma, illness coeff for use in eqn type 5 popfrac = [0.67,0.33]; % pop frac of mild and moderate asthmatics							

Table A14.5 Concentration-response functions for ozone mortality and morbidity

source	effect	MAIPA pool	ages	model	yo / alpha	beta	sigma	averaging
Fairley (2003)	short-term mortality	mortality	all	log-linear	0.0079	0.002828	0.002668	daily 8-hr max
Samet et al. (1997)	short-term mortality	mortality	all	log-linear	0.0079	0.000936	0.000312	daily mean
Ito and Thurston (1996)	short-term mortality	mortality	all	log-linear	0.0079	0.000634	0.000251	daily 1-hr max
Kinney et al. (1995)	short-term mortality	mortality	all	log-linear	0.0079	0	0.000214	daily 1-hr max
Moolgavkar et al. (1995)	short-term mortality	mortality	all	log-linear	0.0079	0.000611	0.000216	daily mean
McDonnell et al. (1999)	chronic asthma	chronic respiratory disease	<65	log-linear	0.00219	0.0277	0.0135	annual 8-hr mean
Burnett et al. (1997)	all respiratory hospitalization	respiratory hospitalization	all	log-linear	2.58E-05	0.004985	0.001093	daily mean
Thurston et al. (1994)	all respiratory hospitalization	respiratory hospitalization	all	linear	0.00E+00	1.68E-08	9.71E-09	daily mean
Schwartz (1995)	all respiratory hospitalization	respiratory hospitalization	65+	log-linear	1.19E-04	0.002652	0.001398	daily mean
Schwartz (1995)	all respiratory hospitalization	respiratory hospitalization	65+	log-linear	1.19E-04	0.007147	0.002565	daily 12-hr mean
Burnett et al. (1999)	asthma hospitalization	respiratory hospitalization	all	log-linear	4.75E-06	0.002497	0.000718	daily mean
Burnett et al. (1999)	chronic lung disease hospitalization	respiratory hospitalization	all	log-linear	5.76E-06	0.003027	0.001105	daily mean
Moolgavkar et al. (1997)	chronic lung disease hospitalization	respiratory hospitalization	65+	log-linear	3.75E-05	0.002743	0.001699	daily mean
Schwartz (1994) Minneapolis	chronic lung disease hospitalization	respiratory hospitalization	65+	log-linear	3.05E-05	0.00549	0.00205	daily mean
Burnett et al. (1999)	pneumonia hospitalization	respiratory hospitalization	all	log-linear	1.56E-05	0.001977	0.00052	daily mean
Moolgavkar et al. (1997)	pneumonia hospitalization	respiratory hospitalization	65+	log-linear	5.30E-05	0.00397	0.00103	daily mean
Schwartz (1994) Detroit	pneumonia hospitalization	respiratory hospitalization	65+	log-linear	5.30E-05	0.003977	0.01865	daily mean
Schwartz (1994) Detroit	pneumonia hospitalization	respiratory hospitalization	65+	log-linear	5.18E-05	0.00521	0.0013	daily mean

source	effect	MAIPA pool	ages	model	yo / alpha	beta	sigma	averaging
Burnett et al. (1997)	all cardiovascular hospitalization	cardiovascular hospitalization	all	log-linear	3.81E-05	0.005313	0.00142	daily 8-hr mean
Burnett et al. (1999)	dysrhythmia hospitalization	cardiovascular hospitalization	all	log-linear	6.46E-06	0.001685	0.001034	daily mean
Cody et al. (1992)	asthma emergency room visits	chronic respiratory emergency	all	linear	0.00E+00	0.0203	0.00717	daily 5-hr mean
Weisel et al. (1995)	asthma emergency room visits	chronic respiratory emergency	all	linear	0.00E+00	0.0443	0.00723	daily 5-hr mean
Stieb et al (1996)	asthma emergency room visits	chronic respiratory emergency	20-65	linear	0.00E+00	0.0035	0.0018	daily 1-hr max
Krupnick (1990)	any of 19 respiratory symptoms	respiratory illness	<65	linear	0.00E+00	0.000137	6.97E-05	daily 1-hr max
Whittemore and Korn (1980)	self-reported asthma attacks	respiratory illness	20-65	logistic	2.70E-02	0.00184	0.000714	daily 1-hr max
Ostro and Rothschild (1989)	respiratory or non-respiratory symptoms resulting in a minor restricted activity day	restricted activity	18-65	log-linear	2.14E-02	0.0022	0.000658	2-wk mean 1-hr max
Notes	Fairley (2003) is a reanalysis of Fairley (1999) MAIPA assumptions: 1-hr mean substituted for [daily.1hr.max,log-linear (2)-wk.mean.1hr.max] 8-hr mean substituted for [daily.mean,ann.8hr.mean,daily.12hr.mean,daily.5hr.mean] EPA (1999) assumptions carried through to MAIPA: McDonnell99 study applies to non-asthmatic males 27+, here used male population 30+ with multiplier in popmlt to account for nonasthmatics of (linear (1)-0.0561) where factor of 0.0561 taken from Whittemore and Korn study, = population of asthmatics of all ages, assumed to apply to males 27+ in same proportion as entire population assuming Thurston et al. study is change in daily incidence for emergency room visit studies, Cody92, Weisel95, Stieb96, 63% of estimate used to avoid double-counting hospital emissions for asthma as suggested by EPA 99; for Cody92 and Weisel95, baseline population in northern NJ = 4,436,976; for Steib, baseline population in Saint John, New Brunswick = 125,000							

Table A14.6 Concentration-response functions for particulate matter (PM2.5) mortality and morbidity

source	effect	MAIPA pool	ages	model	yo / alpha	beta	sigma	averaging
Krewski et al. 2000	long-term mortality	mortality	30+	log-linear	0.0084	0.005348	0.001464	annual median
Krewski et al. 2000	long-term mortality	mortality	25+	log-linear	0.0084	0.013272	0.00407	annual mean
Pope et al. 2002	long-term mortality	mortality	30+	log-linear	0.0084	0.006015	0.002257	annual mean
Fairley 2003	short-term mortality	mortality	all	log-linear	0.0079	0.003404	0.0013	annual mean
Ito 2003	short-term mortality	mortality	all	log-linear	0.0079	0.00074	0.000752	annual mean
Klemm and Mason 2003	short-term mortality	mortality	all	log-linear	0.0079	0.001193	0.000202	annual mean
Moolgavkar 2003	short-term mortality	mortality	all	log-linear	0.0079	0.000588	0.0003	annual mean
Schwartz 2003	short-term mortality	mortality	all	log-linear	0.0079	0.00137	0.0002	annual mean
Abbey et al. 1995	chronic bronchitis	chronic respiratory disease	27+	log-linear	0.00378	0.09132	0.0068	annual mean
Dockery et al. 1996	acute bronchitis	respiratory illness	8-12	logistic	0.044	0.0272	0.0171	daily mean
Thurston et al. 1994	all respiratory hospitalization	respiratory hospitalization	all	linear	0.00E+00	1.81E-08	1.79E-08	daily mean
Sheppard et al. 2003	asthma hospitalization	respiratory hospitalization	<65	log-linear	4.52E-06	0.003324	0.001045	daily mean
Burnett et al. (1999)	pneumonia hospitalization	respiratory hospitalization	all	log-linear	1.56E-05	0.003279	0.000735	daily mean
Burnett et al. (1999)	dysrhythmia hospitalization	cardiovascular hospitalization	all	log-linear	6.46E-06	0.001356	0.00091	daily mean
Schwartz and Neas 2000	lower respiratory symptoms	respiratory illness	7-14	logistic	0.0012	0.016976	0.00668	daily mean
Ostro et al. 1991	asthma exacerbation moderate or worse	respiratory illness	all	linear	0.00E+00	0.0006	0.0003	daily mean
Ostro 1987	restricted activity days	restricted activity	18-64	log-linear	0.0177	0.0177	0.00475	daily mean
Ostro and Rothschild	minor restricted activity days	restricted activity	18-64	log-linear	0.02137	0.00741	0.0007	daily mean
Ostro 1987	work-loss days	restricted activity	18-64	log-linear	0.00648	0.0046	0.00036	daily mean
Notes	unit equivalencies assumed: % 24hr PM2.5 for [daily.mean.PM2.5] % wtd ann mean PM2.5 for [ann.medi.PM2.5,ann.mean.PM2.5] pop equivalencies assumed: % for Dockery96, 10-14 for 8-12 % for Schwartz94, 10-14 for 7-14 % 5 for Pope95, yo = county-level annual non-accidental deaths of persons ages % 30+ per person; given yo derived from CDC mortality rate data NEW: all causes death yo updated % 6 for Dockery93, yo = county-level annual non-accidental deaths of persons ages % 25+ per person; given yo derived from CDC mortality rate data NEW: all causes death yo updated							

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