Aviation Environmental Policy and Issues of Timescale

by

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Abstract

Every operational, technological, and policy decision affecting aviation represents a potential tradeoff among economic efficiency and impacts to climate, air quality, and community noise. Furthermore, effects in these domains occur over different temporal and spatial scales and with different aleatoric and epistemic uncertainty. Thus, robust, fast, and flexible tools that value these impacts on a common scale such as monetized changes in net welfare are needed along with methodologically sound and appropriate analysis frameworks to inform decision-making. The objectives of this thesis are threefold: 1) to advance the modeling tools used to calculate and value aviation's impact on the environment; 2) to analyze real-world aviation environmental policies and advance policy decision-making support; and 3) to explore the underlying issues of timescales in policy modeling, to develop and make explicit the treatment of these timescales, and thereby to improve policy support best-practices.

In support of the first objective, a model is presented for calculating the health impacts of aviation noise, particularly hypertension, myocardial infarction, and stroke to complement current modeling approaches of the willingness-to-pay for noise abatement. Also, advances are made to an existing simplified climate model for aviation by improving the representation of uncertainty, updating modeling components for both long- and short-lived forcing agents, and developing a module to consistently model the life-cycle impacts of alternative fuels. Finally, a method for modeling the social costs of aviation lead emissions is developed.

In support of the second thesis objective, three policy case studies are presented: aircraft noise certification, residential soundproofing and land acquisition, and general aviation lead emissions. The costs and benefits of different policies are evaluated for each case. Results are calculated with explicit accounting for scientific, modeling, and economic uncertainty and are presented considering a range of policy-maker preferences for near- or long-term benefits. The thesis finds that aircraft certification stringency increases up to -5 dB from prior noise limits are cost-beneficial for all discount rates and for the entire range of scientific and economic assumptions and that a -7 dB stringency is cost beneficial when environmental costs are high or are discounted at a lower rate than market costs. The benefits of these policies are less than $5 billion USD over the lifetime of the policy. Further, this thesis finds that noise impacts on health cause an additional 40%-60% of welfare damages compared to considering annoyance costs alone. Noise insulation projects for homes in the vicinity of an airport are found to be on average cost beneficial only when aircraft related noise levels are above 75dB Day-Night Level, and that residential land acquisition projects are not cost-beneficial when considering environmental benefits alone. Finally, the work estimates the average environmental cost of leaded fuel emissions from general aviation at $1.06 billion USD per annum, with the environmental costs of aviation lead being sensitive to background atmospheric lead concentrations.

To support the third thesis objective, a framework is introduced for explicitly considering appropriate timescales in environmental policy analysis. This thesis identifies a modeling framework consisting of three timescales: the policy influence period, the environmental lifetime, and valuation timescale. Focusing on the policy influence period, this framework is tested using the noise stringency certification policy as a test case. Failure to account for the full policy lifetime leads to an undercounting of environmental benefits. Furthermore, not considering the full timescale of policy costs or the impact of exogenous technological improvement on cost projections can impact the apparent appropriateness of a potential policy. In the case of noise stringency certification, a -7 EPNdB stringency increase appears to have net costs when a static policy time period from 2006 to 2036 is considered, but is cost beneficial over a more appropriate timescale that covers the full costs and benefits of the policy.

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Chapter 1
Introduction

1.1 Background

Aviation operations can have complex and significant impacts on the environment. Aircraft contribute to community and environmental noise pollution, degradation of surface air quality, and long-lasting changes in the radiative balance of the earth's atmosphere, leading to global climate change. Because technology development times can be longer than a decade, aircraft remain in service for twenty or thirty years, and the atmospheric residence time of key aviation emissions can vary from hours to centuries, decisions made regarding aviation today can have an environmental impact long into the future. Poorly informed decisions can result in high costs to human health and welfare, slowed economic growth, and/or limited domestic and international mobility.

Aviation's impact on the environment is significant in scope and magnitude. Aircraft noise causes an estimated $23.8 billion in capitalized damages to global housing property values (He et al. 2014). Air transport noise also effects the value and appropriateness of other land uses; in particular, it is an annoyance in public parks, historic sites, and areas with low background noise and may be detrimental to schools and other learning environments. Global aviation emissions cause about 16,000 premature mortalities from air quality related health impacts, and approximately $21 billion in welfare damages from climate change are attributable to emissions from one year of commercial aviation operations (Yim et al. 2015). Estimates of demand for global air transport predict growth of 5-6% a year over the next several decades, meaning that, without continual improvement in operational or technical environmental efficiency, aviation's environmental footprint will also grow.

Several approaches exist for controlling, mitigating, or abating the impact of aviation on the environment. One approach is to adopt source-based limits on pollutant and environmental nuisance outputs. The International Civil Aviation Organization (ICAO) first published Annex 16: Environmental Protection, Volume I - International Noise Standards in 1971, which set the international standard for aircraft noise. Other source-based international environmental protection initiatives overseen by the ICAO Committee on Aviation Environmental Protection (ICAO-CAEP) include engine certification for NO_X emissions, hydrocarbons, carbon monoxide, and smoke. Recently, ICAO-CAEP has proposed and explored the development of an aircraft CO_2 standard.

Another approach is to set local environmental standards. In the United States, the Environmental Protection Agency (EPA), under the Clean Air Act, sets National Ambient Air Quality Standards for common air pollutants known as "criteria pollutants", including lead and particulate matter. For noise, the Federal Aviation Admission has designated 65dB of weighted-cumulative noise exposure over a 24-hour period (measured by Day-Night Level, DNL) as the critical level above-which residential land-use is not compatible. These regulations apply penalties for non-compliance or non-attainment of the regulatory limit. In the case of air quality, non-attainment areas are required to submit implementation plans and may face sanctions and loss of highway funds. For noise, airports are responsible for mitigating noise impacts in the airport vicinity through providing sound insulation or acquiring the noise-affected property. Other possible approaches to limiting the environmental impact of aviation include market-based approaches to emissions like cap-and-trade (Malina et al. 2012), procedural and operational measures to improve environmental performance (Marais et al. 2012), and subsidies and tax credits for environmentally appropriate technologies.

Regardless of the policy's form, a set of complex questions exists that make decision-making difficult. The importance and size of the aviation industry and the magnitude of the environmental effects mean that policies regulating or affecting the air transport industry have the potential to provide billions of dollars in benefits (or cause billions of dollars in damages) and directly impact tens of millions of people world wide. In addition, three factors further complicate the decision-making process: epistemic uncertainty, trade-offs and feedbacks across different domains, and robust design of decision-making and decision-aiding processes. These factors are not exclusive to the field of aviation and the environment, and they offer opportunities for research contributions with near-term, real-world implications in aviation and other domains.

1.2 Objective

This thesis develops a consistent approach for evaluating the costs and benefits of aviation environmental policy as a tool for aiding and evaluating decision-making. The thesis contributes to the literature in three areas: advancement of state-of-the-art modeling approaches for analyzing aviation's impact on the environment, policy evaluation of real policy problems and communication of these results to decision-makers, and finally introduction of a new framework for approaching the issue of timescales in policy analysis, analyzing the impacts of implicit and explicit timescales in cost-benefit analysis. This final area uses aircraft environmental certification as a case study and draws more general conclusions for policy analysis.

1.3 Thesis Organization

This thesis is composed of six chapters. This section presents a brief description of the structure and content of the remaining chapters.

Chapter 2 provides the motivation for the thesis work. This section explores the context for environmental policy and, in depth, aviation's contribution to noise pollution, air quality degradation, climate change, and local lead concentrations. Furthermore, this section provides a literature review on environmental policy assessment focusing on applications of Cost-Benefit Analysis.

Chapter 3 discusses the modeling approaches used for assessing aviation's impact on the environment. Models for measuring the physical and monetary impacts of aviation noise, air quality degradation, climate change, and lead emissions are presented. A model that assesses the contribution of aircraft noise to physical health damages is developed, and the benefits and limitations of this model relative to models that only assess amenity damages such as changes to housing value are discussed. Drawing on previous work, a simplified climate model for aviation is developed and expanded, in particular exploring integrated modeling of climate life-cycle impacts of different fuels. Finally, a model for assessing the nation-wide impacts of leaded fuels from general aviation including full-flight emissions is developed.

Chapter 4 focuses on three environmental policy analyses. The first analysis is that of a global aircraft noise certification stringency. The environmental benefits of US operations for 5 policy scenarios and the societal cost-benefit for these policies are calculated under a variety of lenses and discount rates. The second analysis looks at US airport land-use policies to reduce the burden of aviation noise. The same modeling approach from the noise certification stringency policy is used to develop a Willingness-to-Pay for noise abatement relationship of affected properties. In addition, the potential impacts of near-airport noise on incidences of myocardial infarction, hypertension, and stroke are assessed. These results are used to calculate the net cost-benefit of residential insulation and land acquisition policies at US airports. Finally, this chapter examines the contribution of general aviation lead emissions to deleterious human health and welfare impacts in the continental United States.

Chapter 5 explores how accounting for heterogeneous timescales impacts environmental policy analysis. Three timescales relevant for policy analysis are identified: policy influence period, environmental impacts lifetime, and valuation timeline. This chapter explores how these timescales are currently accounted for in the literature to develop formal definitions. The noise certification stringency analysis from 4.1 is reassessed to evaluate how explicit accounting for these timescales impacts the analysis and policy appropriateness.

Chapter 6 provides a summary of this thesis and discusses the implications of its key contributions. This chapter also briefly highlights areas for future work.

1.4 Key Contributions

This research is part of ongoing work for an integrative approach to assessing environmental policy, particularly in commercial aviation. It draws on a rich history of work from the Partnership for AiR Transportation Noise and Emissions Reduction (PARTNER), the Aviation Sustainability Center (ASCENT), and the Laboratory for Aviation and the Environment (LAE).

Listed below are the key contributions of this thesis work in the area of aviation and the environment and policy decision-making:

- Development of a model to monetize the health impacts of aviation noise. The model considers a range of dose-response functions from primary aviation and ground transportation noise literature. The model calculates hypertension and myocardial infarction costs of aircraft noise building on prior studies and is the first model to compute a range of potential costs from noise-related stroke incidences.

- Continued development and expansion of a simplified climate model for aviation policy. Primary changes to the code include updated parameters and better accounting for uncertainty in the carbon model, the radiative forcing model, and the damage function. Key expanded capabilities include decoupling of projected emissions and socio-economic variables in accordance with IPCC best practices and consistent accounting for life-cycle emissions for 16 fuel feedstocks.

- An assessment of an international aircraft noise certification stringency increase. The policy analysis demonstrates an improvement in current decision-making processes by accounting for uncertainty and analyzing the resulting sensitivity of associated costs, environmental benefits, and policy-maker preferences through a cost-benefit framework.

- An assessment of domestic land-use policies for mitigating the effects of aircraft noise. This policy analysis considers the costs of airport programs to provide sound-insulation of residential properties or to purchase residential land and the benefits the associated decrease in noise exposure provides to the impacted persons through both conventional willingness-to-pay valuation and through separate accounting for noise-induced health impacts.

- An assessment of the environmental impact of leaded fuels used in piston-driven aircraft.

- A framework for more explicitly considering appropriate time-scales in environmental policy analysis. First, three timescales fundamental to modeling and valuing costs and benefits in policy are identified. Next, the implicit and explicit treatments of these timescales in policy analysis are investigated. Finally, the impact of appropriate and consistent timescale adoption on policy decision-making is quantified. This framework is tested using the noise stringency certification policy as a test case.

In addition, an analysis of the current and future noise impacts of London hub airports is presented in Appendix A. This analysis uses the noise modeling approach developed in Section 3.1 of this thesis.

Chapter 2

Environmental Impacts and Regulatory Policy

The objective of this thesis is to improve analysis for environmental policy decision-making, focusing more specifically on aviation environmental policy. Section 2.1 provides an overview of environmental externalties to justify the development of environmental policies. Section 2.2 summarizes the relevant environmental externalities in the air transportation sector. Section 2.3 describes the state of regulatory analysis and highlights recent literature contributions to theory and practice. Methods for estimating physical and monetary impacts of aviation environmental impacts are presented in Chapter 3, and example policy analyses are presented in Chapter 4.

2.1 Environmental Externalities

For this thesis, an externality is defined as a change to an agent's or party's welfare function that is produced through the byproducts of unagreed-upon actions of another agent or party leading to an inefficient allocation of resources. This definition encompasses three features that characterize an externality: A party incurs a cost or benefit as the result of some other party's actions that (1) he did not choose to incur; (2) the underlying cause of the cost or benefit is not from a primary deliberate attempt to effect said party's welfare; and (3) the cost or benefit does not simply redistribute income, such as through a change in price or asset value.

While each of these tenets is broadly consistent with the literature (see, for example, Scitovsky 1954, Mishan 1971, Baumol and Oates 1975, Verhoef 2002), there exist some differences in individual definitions that may violate any one of these features and the definition provided is not meant to be rigid. With regards to the first tenet, while Buchanan and Stubblebine (1962) find an externality to be present if any individual's utility function is affected by an external actor's actions regardless of choice or acceptance; they define all such externalities that do not provide any motivation or desire of the affected individual to take action as "irrelevant externalities." Mishan (1971) stresses that the byproduct nature of an externality noted in the second tenet is essential, but he notes that it is missing from the common mathematical definition. However, without the second tenet an externality is indistinguishable from other unpriced interactions such as jealousy, altruism, violence, and goodwill promoting activities (Verhoef 2002). The third tenet ensures the existence of economic inefficiency and not simply a redistribution of welfare (i.e. there exists no theoretically mutually beneficial trade between parties) (Baumol and Oates 1975). Impacts that change only the value of goods or assets are sometimes called pseudo externalities (Schipper et al. 2001) or pecuniary externalities (Holcombe and Sobol 2001).

Environmental externalities exist when the byproducts defined above impact a person or party through the way they affect the natural or built environment. These byproducts can include long-lasting pollution to air, soil, or water reservoirs; modification, removal or deterioration of ecosystems; and accumulation of waste with non-zero disposal costs. Environmental externalities persist because there is often no well-functioning market for environmental goods as it is difficult to assign micro-level property rights (Delucchi 2000) or because there are significant barriers to trade such as high transaction costs or a disagreement on the division of trade gains (Schipper et al. 2001). The existence of environmental externalities is a manifestation of a market failure, where actions external to the market can lead to a more efficient allocation of resources. Thus, if market efficiency is a goal, a policy prescriptive approach to environmental externalities is justified.

2.2 Aviation and the Environment

Air transport gives rise to several environmental byproducts. The following subsections review the current literature on direct aviation environmental externalities with a particular focus on noise, air quality degradation, climate change, and lead pollution. These four environmental domains are the focus of the policy analyses presented in Chapter 4. This section expands on other reviews of aviation environmental externalities (Janic 1999, Schipper et al. 2001, Marais and Waitz 2009).

2.2.1 Noise

Commercial air transportation generates numerous direct, indirect, and induced benefits and can be a driver of economic growth (Button and Yuan, 2013; Green, 2007). However, the expansion of the national airspace system (NAS) and the associated infrastructure has driven the need for assessments of the external costs of aviation. Personal and property damage from accidents, congestion, local air pollution, water degradation, and climate change are all important considerations for the NAS. Aviation noise is one such negative externality that is borne by individuals who are not necessarily the producers, users, or benefiters of the NAS. Noise is a byproduct of aircraft operations both en-route and in the airport vicinity as well as of ground and support equipment. Furthermore, airport use and development can induce greater noise pollution through an associated increase in ground traffic to and from the airport (Gosling, 1997).

Annoyance is one of the most readily apparent effects of unwanted noise. Noise annoyance is used as a broad term to describe a reaction in which individuals, groups, or communities would, if given the possibility, actively try to reduce exposure through mitigation or avoidance (Molino, 1979). Because noise is a subjective experience, translating exposure to a measure of predicted annoyance is non-trivial, and there is a large variation in individual reactions to the same exposure levels (Miedema and Oudshoorn, 2001). Additionally, noise exposure has been linked to changes in human health, including mortality and morbidity pathways. Hansell et al. (2013) found a statistically significant increase in hospital admissions for stroke and coronary heart disease for residents living in areas with higher levels of daytime and nighttime noise. Jarup et al. (2008) and Correia et al. (2013) present large multi-airport studies that link aircraft noise exposure to hypertension and hospital admissions for cardiovascular disease, respectively. Basner et al. (2013) perform a metastudy to determine relative risks of hypertension and myocardial infarction from noise exposure.

It is convenient to measure welfare loss in monetary terms so that disparate impacts can be more easily compared and so that the benefits and tradeoffs of policy options can be assessed. Many studies look at depreciation in housing value as a way to monetize the impacts of noise (Wadud 2009). By investigating the housing market in the vicinity of an airport, one can develop a relationship between the exposure to aircraft noise and observed differences in housing value, while controlling for other determinants of housing value such as neighborhood amenities, community composition, and access to the airport. This relationship can be quantified as a percentage decrease in property value corresponding to a 1dB increase in time and frequency-weighted noise exposure. Housing value depreciation can be treated as a proxy for the willingness-to-pay for noise removal as with compensation equal to the differential between the value of the house under a noise burden and the unaffected house, a person would theoretically be able to move to an equivalent house in a quieter area.[1]

However, housing value may be an incomplete proxy for the overall health and welfare impacts since it likely only accounts for the perception and comprehension of the full extent of negative effects and therefore may not reflect the actual risk or

[1] Because property rights are not traditionally granted to the present value of assets (Holcombe and Sobel 2001), changes in housing values in and of themselves may be only pecuniary and not true externalities. However, two nuances complicate this argument. First, while the housing value change is itself pecuniary, it is representative of the change in welfare perceived by the resident (He 2010), and therefore it can be used as a proxy for the true externality. Second, change in property value may alter the welfare function of residents and neighbors indirectly through changes in status or as a signal of better neighborhood maintenance (Andersson et al. 2010).

burden of long-term health effects (Harrison and Rubinfeld 1978). Furthermore, using housing price differentials implicitly assumes there is a well-functioning and equilibrium housing market and that people and families can move between equivalent houses with marginal transaction costs (Freeman 1979, Gjestland et al. 2014). A Dutch study found evidence that a more appropriate noise damage function would include a residual cost differential accounting for impacts not accounted for in the housing market (Van Praag and Barsma, 2005).

While the preceding paragraphs frame the aviation industry as the generator of the noise externality and residential communities as the bearers of that externality, other frames are possible. McCloskey (1998) points out that if no one lived near airports (and noise did not harm the natural environment), then the related noise would be harmless and it would be counterproductive to regulate it, and therefore the presence of ears is just as much a cause of the externality as aircraft themselves. Thus, the delineation of property rights is particularly important to consider. This delineation may include either the right to produce noise or the right to quiet. US case law, which has placed liability for damages on airport proprietors (Falzone, 1998), has upheld the view that aircraft noise constitutes a taking of property.

With property rights assigned and in the presence of minimal transaction costs, bargaining will lead to the most efficient use of resources and the maximum of net social welfare – regardless of the delineation of property rights (Coase, 1960). However, transaction costs can be real and significant. The number of parties impacted by noise near any individual airport can be in the thousands, making the time, effort, and money required to bargain with each party practically infeasible. Bargaining through multi-party agreements also opens up the possibility of classical collective action problems such as free-ridership or holdouts that would eliminate the chances of an efficient outcome.

Furthermore, the noise impact is not necessarily homogenous across different airports. While it may be socially efficient for one airport and surrounding community to ban the operation of the noisiest of aircraft, less constrained airports may achieve a welfare maximizing solution with no fleet restrictions. The resulting piece-meal approach focused on local optima could greatly impact the structure of the NAS leading to a globally inefficient outcome. Finally, even if the transaction costs of bargaining were minimized, there is no guarantee of a socially equitable outcome. As some people disproportionately bear the costs, aircraft noise may represent a social justice concern (Sobotta et al. 2007).

Aviation noise may affect the environment beyond health and annoyance impacts to residents near the airport. Aircraft overflights are a source of sound intrusion into pristine environments such as national parks and public lands, diminishing park quality and visitor enjoyment (Miller 1999). Aircraft noise leads to annoyance in school and work environments, and may impair reading comprehension (Clark et al. 2013). In addition to its direct impacts on humans, aircraft noise may have negative ecological consequences (Barber et al 2011).

2.2.2 Air Quality

Aviation emissions can have detrimental impacts on human health through induced changes in ambient air quality. Like other fossil fuel combustion sources, aircraft engines emit a variety chemical species that have an impact on health and ecosystems. While emissions are primarily composed of carbon dioxide (CO_2) and water vapor (H_2O), other emitted species include nitrogen oxides (NO_X), sulfur oxides (SO_X), volatile organic compounds (VOCs) and unburned hydrocarbons (UHCs) such as formaldehyde and benzene, carbon monoxide (CO), the extended family of nitrogen compounds (NO_Y), and both volatile and nonvolatile particulate matter (PM) (Masiol and Harrison, 2014). Primary PM includes dust, dirt, soot, smoke, and liquid droplets directly emitted into the air. Particles formed in the atmosphere by the condensation or transformation of emitted gases such as NO_X, SO_2, and UHCs are also considered particulate matter, and are referred to as secondary PM. Fuel venting and spills also introduce hydrocarbons directly into the atmosphere.

PM with an aerodynamic diameter of less than 2.5 μm is designated as $PM_{2.5}$. When $PM_{2.5}$ is inhaled, the particles can become trapped in the lungs or can pass into the blood stream, potentially causing health problems. Exposure to increased PM concentrations has been correlated with adult early mortality, infant mortality, asthma, chronic bronchitis, restricted work days, respiratory hospital admissions, and cardiovascular hospital admissions (EPA 2004). A 2013 study estimates that PM inhalation leads to ~200,000 premature mortalities annually in the US alone (Caiazzo et al. 2013), a factor of four increase over earlier estimates (Mokdad et al. 2004). Brunelle-Yeung et al. (2014) estimates that between 130 and 340 deaths are attributable to aviation $PM_{2.5}$ emissions from ground level to 3000 ft in the US in 2005, with the majority of mortality impacts coming from secondary PM formation. A study by Barrett et al. (2010) shows that including full flight emissions (those above 3000 ft) leads to an estimate of around 10,000 global mortalities from aircraft emissions, ~80% of which are due to cruise emissions. Kapadia et al. (2015) find between 1000 and 6000 global mortalities from aircraft-attributable PM each year.

Aircraft emissions also contribute to surface level ozone formation, which can lead to further health and mortality impacts. However, the contribution of aviation-induced ozone concentrations to US mortalities is < 5% of that of aviation-induced $PM_{2.5}$ (Caiazzo et al. 2013). Furthermore, air quality degradation can lead to reduced visibility and enjoyment of the natural environment. Delucchi et al. (2002) estimate that visibility changes contribute 15-35% of the total damages from air quality impacts.

2.2.3 Climate Change

The changing climate exerts pressure on many aspects of the earth's natural systems. Temperature changes can lead to extreme changes in both natural and managed systems. Hydrology and water resources, marine and terrestrial biosystems, cryosphere, human health, agriculture and land use are all domains that are impacted by changing temperature (IPCC 2014).

Aviation operations have an impact on the global climate through direct emissions of greenhouse gases and induced changes in cloudiness (Mahashabde et al. 2011), changes in atmospheric greenhouse gas concentrations from direct land use change and production of aviation fuels (Stratton et al. 2011a), changes in atmospheric greenhouse gas concentrations from indirect land use change and displacement associated with fuel production (Plevin et al. 2009), changes in planetary albedo from land conversion associated with fuel production (Caiazzo et al. 2014), and changes in surface and cloud albedo from black carbon emissions (Bond et al. 2013).

Aviation emissions from combustion have impacts on climate change that are diffuse and heterogeneous. CO_2 is a chemically stable well-mixed greenhouse gas; its effects are globally distributed and can persist in the atmosphere for centuries. Aviation NO_X emissions alter the chemistry of the atmosphere leading to changes in ozone (O_3) and methane (CH_4) concentrations resulting in northern hemisphere-concentrated, short-lived increases in O_3 and decadal more globally homogenous decreases in CH_4 and O_3 (D.S. Lee et al. 2009, Holmes et al. 2011). Short-lived aviation emissions include sulfates, soot, water vapor, contrails and induced-cloudiness can range from local to continental scale (Dessons et al. 2014). Aviation CO_2 accounts for about 2.5% of total anthropogenic CO_2 burden, but aviation contributes to 4.9% (2-14%) of the current anthropogenic radiative forcing due to short-lived climate forcers (D.S. Lee et al. 2009). Additional (positive and negative) climate forcing may be caused by stratospheric water vapor and nitrate formation from aircraft activity at cruise, but these impacts are currently highly uncertain (Brasseur et al., *in preparation*).

The production of aviation fuels is also associated with CO_2 emissions from extraction, processing, and transportation. Using the model developed in Section 3.3 of this thesis, these life-cycle emissions from conventional fuel contribute an additional 11% of climate damages from aviation. Bio-derived fuels may provide a sustainable alternative to conventional jet fuels. While these fuels also have emissions associated with production processes, the CO_2 from the combustion of these fuels does not represent additional carbon to the natural earth-atmosphere carbon cycle. However, the climate impact of these fuels must account for changes in carbon pools resulting from converting land from other uses to fuel production. In addition to production and non-CO_2 combustion emissions, alternative fuels may influence the environment through biogeophysical effects such as changes in local evapotranspiration and planetary surface albedo. Research indicates that these biogeophysical effects may globally be of the same order of magnitude as biogeochemical effects from emissions and may locally be of greater importance than the biogeochemical effects (Caiazzo et al. 2014, Bright 2015).

Because aviation's emissions are temporally and spatially heterogenous and occur over different time scales, consistent metrics are required to compare the individual species and compute the net climate impact. The Global Warming Potential (GWP) compares the integrated radiative efficiency of a greenhouse-forcing agent to that of CO_2 and is one of the most prevalent metrics for quantifying the impacts of climate change. The Intergovernmental Panel on Climate Change (IPCC) has used the GWP since the inception of its scientific assessments (IPCC 1990), and it is the primary metric of the Kyoto Protocol. However, the GWP is widely contested, as summarized by Shine et al. (2005) and Dorbian et al. (2011). Despite the implication of its name, the GWP does not indicate the impact on climate system warming or cooling that a temperature metric would give. Furthermore, because of the atmospheric lifetime of CO_2, the metric is highly sensitive to the time horizon chosen. An analysis by Tanaka et al. (2009) shows that GWPs alone do not give a good indication of expected impact even using a "best fit" time horizon. Although they introduce significant additional uncertainty, metrics that directly assess temperature change and human welfare impacts are becoming more commonly used. The strengths and weaknesses of various metrics are described in the literature (e.g. Dorbian et al., 2011; Azar and Johansson, 2012; Deuber et al., 2013; Aamaas et al. 2013).

2.2.4 General Aviation and Lead Emissions

Lead is a persistent toxic pollutant that impacts human health and welfare through inhalation and ingestion pathways. Lead can be inhaled from direct emissions to ambient air. Lead is emitted as organometallic particles and lead salts to the atmosphere from a variety of anthropogenic stationary and mobile sources. Stationary sources include utility boilers for coal power plants, iron foundries, waste incineration, and primary lead smelting (EPA 2006a). While lead as a source for anti-knock in motor vehicles was the largest source of domestic anthropogenic lead emissions in the 1980s and early 1990s, this use of lead was phased out by 1995 in the United States (EPA 2006a, EPA 2010a). However, lead is still used as a fuel additive for some agricultural machinery, certain high performance racecars, and for piston-engine aircraft. By 2008, piston-driven aircraft emissions accounted for about half of all US anthropogenic lead emissions, and were the single largest source of lead emissions to the air (Feinberg and Turner, 2013). Natural sources of lead include sea spray, crustal weathering, wild forest fires, and volcanoes with total emissions averaging an estimated 19,000 metric tons of lead per year (Nriagu and Pacyna, 1988). However, much of this "natural lead" in the atmosphere may be from resuspended soil containing elevated Pb levels from prior anthropogenic emissions, including prior aviation emissions (Harris and Davidson, 2005). Human exposure to lead can also occur through ingestion of contaminated soil or lead paint, lead from drinking water distribution systems, and through organic lead absorption through the skin.

Lead accumulates in human bone and soft tissues, with soft tissue accumulation being more prevalent in infants, and leads to a variety of deleterious health impacts including disruption of neurological, renal, reproductive, and physical development systems. Cognitive and neurodevelopmental effects of lead include decrements in IQ tests, lower performance on standardized testing, and decreased graduation rates (Needleman 2004). Other cognitive effects include an increase an attention-deficit behavior (Tuthill 1996), memory loss (Jett et al., 1997), and poor language performance (Campbell et al. 2000).

In addition to direct impacts on human health from primary emissions, lead can also contribute to increased pollutant concentrations in vegetation and aquatic systems. Anthropogenic lead can have an impact on terrestrial and aquatic ecosystems as evidenced by lead toxicity in horses (Palacios et al. 2002) and fish (Hagner 2002) among others. Lead may have an indirect cooling effect on climate through cloud whitening and albedo changes (Czizco et al. 2009). Previous studies on the environmental impact of lead that include aviation have failed to account for lead emissions from cruise phases of flight (Carr et al. 2011) or have focused only on local impacts in the vicinity of the airport (Feinberg and Turner, 2013).

2.2.5 Other Aviation Environmental Externalities

The air transport industry impacts the environment in domains beyond those of noise, air quality, climate change, and toxic species. Ground operations at airports can affect local watershed quality. Aircraft deicing, fuel spills, herbicides to manage airside grounds, and surface runoff from ground transport can all impact the quality of waterways, rivers, and streams surrounding the airport. Deicing procedures lead to the discharging of 21 million gallons of aircraft deicing fluids into surface waters each year in the US alone (EPA 2002). An overview of the environmental impact of deicing is provided in Marais and Waitz (2009).

Aviation operations and airport expansion can also have a negative impact on wildlife and the local ecology. Airside and landside operations require large tracts of land, making siting of airports difficult. Significant airport expansion projects can require building on green field land or reclaiming wetlands. The resulting expansion can restrict or restructure water flow or lead to urbanization of previously wild or rural areas (Wyatt 2011). The impact of operations on waterfowl and bird migration can be especially problematic both for the environment and for the safety of airport operations themselves (Navjot 2002).

While these environmental impacts are important to consider in the context of future aviation decisions and especially airport expansion, it is assumed that these domains are decoupled from aircraft noise and emission stringency regulation. Thus, to first order, policy decisions impacting wildlife or local water quality are agnostic to policy decisions on emission or noise reductions considered in Chapter 4. Finally, there exist indirect and secondary environmental externalities associated with aviation. Indirect effects include upstream impacts of aircraft production and use of materials and downstream impacts of aircraft scrapping and recycling (Schipper et al. 2001). Secondary effects result from the coupled relationship of the environment with other externalities, such as congestion. For instance, congestion at an airport leads to inefficient operations at that airport and elsewhere in the air transportation network, increasing noise exposure and emissions (Schlenker and Walker 2011). In turn, community reactions to these environmental externalities are barriers to airport expansion, thereby further increasing congestion.

2.3 Regulatory Analysis

Where an environmental externality occurs, there exists an opportunity for regulation to induce a positive societal change. A normative definition of positive change (although one that is neither necessary nor sufficient) is Pareto efficiency, where at least one party would prefer the change relative to the status quo and no parties would prefer the status quo. In practice, achieving Pareto-efficient policies is difficult as it is rarely possible to guarantee that no parties would prefer the status quo. However, if under a certain policy the parties that prefer the change ("the winners") gain more than those who prefer the status quo ("the losers") would lose, there exists a theoretical trade the winners could make to compensate the losers such that both the winners and losers are both better off than under the status quo. This potential Pareto-efficiency is known as the Kaldor-Hicks criterion.

Cost-benefit analysis (CBA) is an economic analysis tool that characterizes policy performance under the Kaldor-Hicks criterion (Pearce 1998)[2]. CBA typically consists of monetizing the costs and benefits of a policy so as to allow for a direct comparison of the two. The favorable effects (i.e. positive impacts on social welfare) of policy actions are defined as benefits whereas the opportunities foregone because of a new regulatory policy are defined as economic costs (Revesz and Stavins 2007). The goal of CBA is to estimate the net benefit of a policy implementation, where the net benefit is the benefits of the regulation minus the costs.

CBA is not uncontroversial and has been attacked on philosophical, legal, and procedural grounds. For a review of critiques see Posner (2000) and Adler and Posner (1999). However, despite these critiques, CBA in practice is in ascendency in public policy (Sunstein 2005). Federal agencies within the United States are required by directives and executive orders from the Office of Management and Budget to evaluate costs and benefits of regulatory measures, but other economic evaluation approaches are often used in cases where sufficient data is not available to quantify costs and/or benefits (OMB 2003).

While International and European governing bodies have recommended, at least provisionally, the use of Cost-Benefit Analysis (see Mahashabde 2009 for a review), they have more traditionally used the Precautionary Principle and Cost-Effectiveness Analysis (CEA) (Sunstein 2005). CEA consists of selecting the policy that costs the least for the same expected results. CEA is most useful for evaluating policies with very similar expected benefits or when knowledge is limited about a possible catastrophic harm, and the primary role of the regulation is absolute prevention of that harm. CEA can be misleading, however. It does not necessarily reveal what level of control is reasonable, nor can it be used to directly compare situations with different benefit streams, and it often provides no indication of the trade-offs (monetary or otherwise) associated with the proposed policy.

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[2] Adler and Posner (1999) identify four conventional defenses of CBA (Pareto efficiency, Kaldor-Hicks Criterion, utilitarianism, and a lack of better alternatives) before providing a reconceptualization of CBA that does not require justification under Kaldor-Hicks.

In international aviation policy, the primary regulatory body is the International Civil Aviation Organization – Committee for Aviation Environmental Protection (ICAO-CAEP). Four areas of reference guide ICAO-CAEP's work and decision-making framework: environmental benefit, technological feasibility, economic reasonableness, and consideration of interdependencies. However, ICAO-CAEP has historically only modeled the effect of policy stringency and implementation with respect to environmental benefit and economic reasonableness (ICAO 2007). To wit, ICAO-CAEP has historically employed Cost-Effectiveness Analysis (CEA) in its decision-making process. In 2010, the first integrated Cost-Benefit Analysis of an aviation environmental policy was presented as part of a reconsideration of aircraft engine NO_X stringency levels (Mahashabde et al. 2011) considering environmental interdependencies and considering technology readiness for policy compliance.

The policy and environmental analyses presented in Chapter 4 focus on the Cost-Benefit framework described above. The policy analyses cover issues at three different governance scales: local (near-airport land use), national (trans-boundary lead), and global (international certification).

One structural weakness of Cost-Benefit Analysis is that it only indicates the efficiency of a proposed policy and not its equity. Benefits from environmental regulations are heterogeneous over space and time, and thus some people may disproportionately bear the costs or be excluded from the benefits of a policy. This can be especially true in aviation where noise is perceived instantaneously and concentrated near an airport while climate change impacts are diffuse and may not be perceived for decades. As shown in Figure 2-1, the aggregate damages from a year of operations indicate that climate change dominates total monetized environmental damages, while noise damages per person are double that of climate change within 5km of an airport (Wolfe et al. 2014).

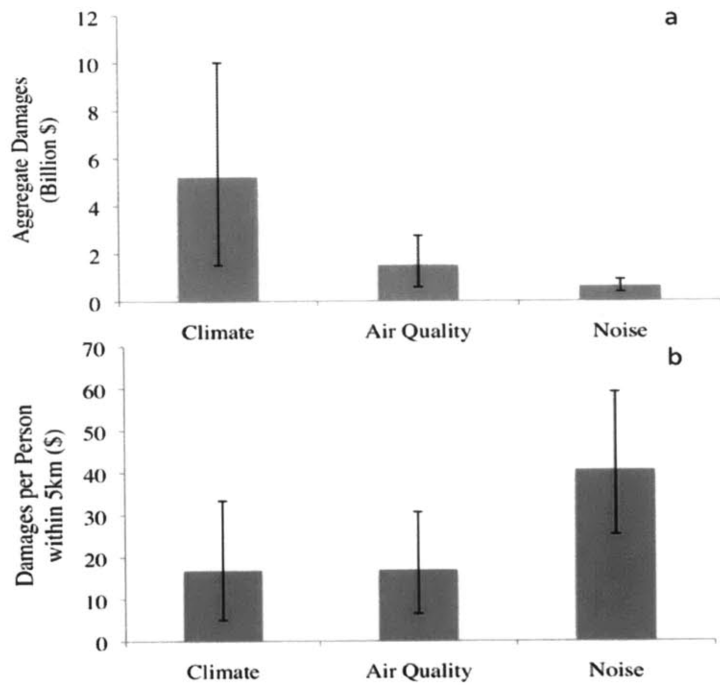


Figure 2-1 Environmental damages from one year of aviation operations on (a) a national aggregate and (b) per person basis within 5 km of an airport. (Wolfe et al. 2014)

Wolfe (2012) provides background on the procedural and ethical issues of spatially heterogeneous costs and benefits. Of similar concern is the distribution of costs and benefits over time (Goulder and Stavins 2002, Sunstein 2005). A framework for conceptualizing the modeling, inclusion and explicit accounting for environmental impacts over time is presented in Chapter 5.

Chapter 3
Modeling Aviation's Impact on the Environment

Chapter 2 provided an overview of select aviation environmental impacts, emphasizing the diverse and disparate nature of environmental domains and the potential need for regulatory policy to control or mitigate these impacts. Because decisions impacting one environmental domain may have countervailing (or co-beneficial) impacts in another domain and because environmental concerns must be considered in the context of their cost and their impact on economic growth, there is a need for comprehensive analyses that quantify environmental impacts and address tradeoffs and co-benefits. This chapter presents analytical methods and tools used to assess aviation's impact on noise [Section 3.1], air quality [Section 3.2] climate change [Section 3.3] and surface lead concentrations [Section 3.4].

Three of the tools presented in the following sections were developed as part of the Aviation environmental Portfolio Management Tool (APMT). The focus of APMT is the economic analysis and environmental impact assessment of future operations, policies, and goals within the Federal Aviation Administration – National Aeronautics and Space Association – Transport Canada aviation environmental tools suite. The impacts module of APMT (APMT-Impacts; APMT-I) assesses the physical and economic effects of aviation and their associated uncertainties using Monte Carlo methods. APMT-Impacts has three sub-modules: Noise, Air Quality, and Climate.

3.1 Noise Impacts Model

Environmental noise can be a pervasive nuisance that causes an array of deleterious public health and wellness effects. To account for the full range of noise impacts while reducing the risk of double-counting damages from interrelated impact pathways, noise impacts are separated into perceivable effects (such as day and nighttime annoyance) and long-term non-auditory health impacts. The perceivable effects of noise are modeled by calculating the willingness-to-pay for noise abatement as described in Section 3.1.1, while the non-auditory health impacts are calculated by determining costs of illness and a valuation of premature mortalities as described in Section 3.1.2.

3.1.1 Willingness-to-Pay Model

Monetized damages from perceivable and attributable aircraft noise are calculated using the APMT-Impacts Noise Module (He et al. 2014). The APMT-Impacts Noise Module overlays noise contours and population data and then applies a monetization formula based on willingness-to-pay for noise abatement. This monetization is derived from a meta-study of residential housing hedonic pricing surveys that correlates willingness-to-pay per decibel Day Night Level (dB DNL) of noise reduced and citywide per capita income levels. Inputs to the code include contours of the spatial extent of noise exposure from aviation, population data, and city-level per capita income. Influential and uncertain parameters in the code include the modeling uncertainty of the noise contours, the background (non-aviation) ambient noise level, the level at which noise annoyance becomes significant, and the willingness-to-pay regression model parameters. A functional diagram of the noise code is shown in Figure 3-1.

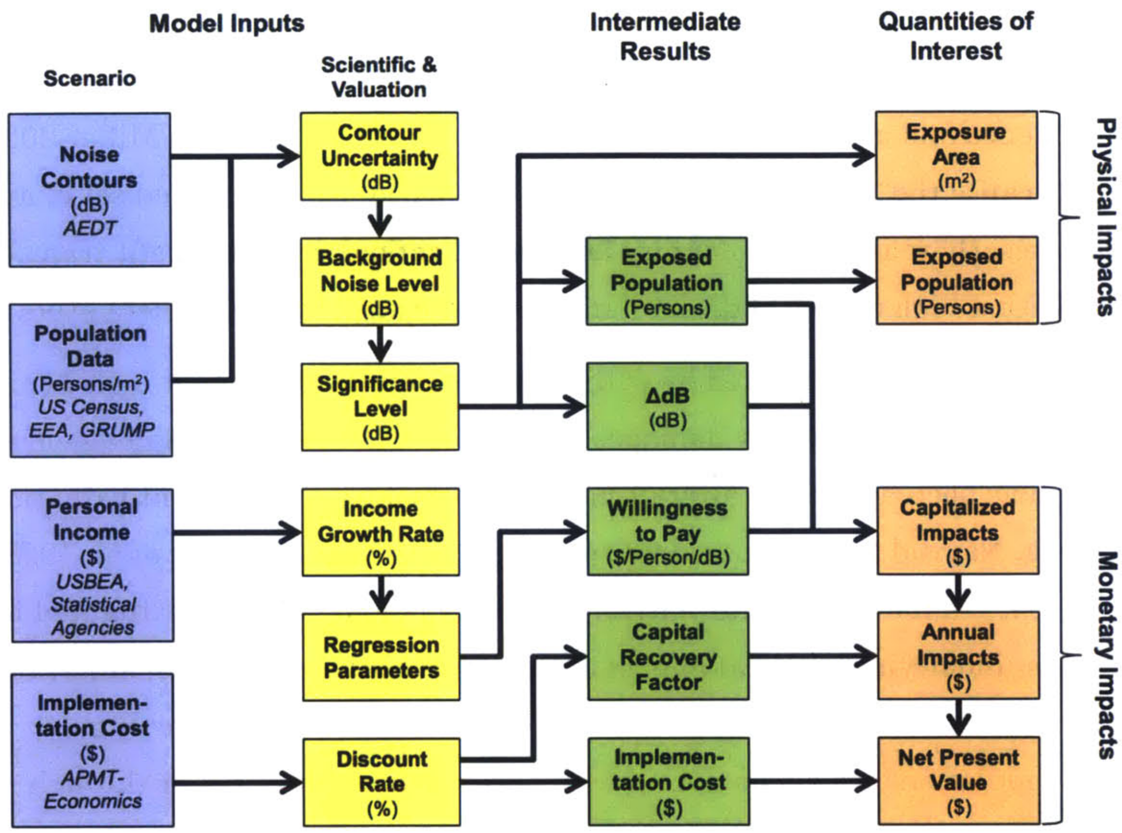


Figure 3-1: APMT-Impacts Noise Module Schematic (adapted from He 2014)

Sensitivity analyses performed in He (2010) show code robustness and comparable results to an alternative valuation model described in Kish (2008). The APMT-Impacts Noise Model methodology has some limitations. The income-based model does not monetize the impact of aviation on noise on areas with low background noise levels, such as national parks (Gramann 1999, Lim et al. 2008). These areas may be susceptible to damage from overhead flights, and are considered critical research areas (Eagan et al. 2011). Furthermore, traditional noise damage indices may not be applicable for noise contours above 75 dB DNL, leading to underestimation of damages at very near airport locations (Feitelson et al., 1996). The dB DNL noise metric applies a 10 dB noise penalty to nighttime noise to account for increased annoyance and decreased background noise levels at night. Other metrics, such as the Day-Evening Night level (DENL), which includes an additional 5 dB penalty for evening noise, have also been used to measure aviation's noise burden and may result in different damages. In particular, cumulative metrics like DNL and DENL may provide a poor relationship between noise incidence and annoyance from sleep awakening (Anderson and Miller 2007). Finally, because the APMT-Noise model was developed using a limited set of airport noise studies, there is the opportunity for generalization error in benefit transfer to airports with a high degree of dissimilarity from the airports in the meta-study.

Noise damages as calculated by willingness-to-pay for abatement models are also referred to as the amenity damages from noise. Other amenity damage models for noise have been applied to aircraft noise developed from contingent valuation studies (e.g. Navrud 2002) and hedonic pricing studies (e.g. Schipper et al. 1998, Nelson 2004, Wadud 2009). Alternatively, amenity damages can be estimated by valuing the underlying physical impact of the externality: in this case, annoyance. The value of noise annoyance can be calculated by first estimating the highly annoyed population based on a noise-annoyance relationship curve or through direct survey of the affected population. For each annoyed person, the relative disamenity of a year of annoyance is then compared to a year in standard health using a disability weight; for example, the World Health Organization considers the disability weight of annoyance to be 0.02 [0.01 – 0.12] (WHO, 2011). A comparison of different amenity models in the context of a noise analysis is provided in Appendix A.

3.1.2 Health Impacts Model

While the exact relationship between aircraft noise exposure and physical health impacts is still uncertain, a growing body of literature suggests that incidences of health endpoints, particularly cardiovascular and cerebrovascular impacts, can be expressed through an exposure-response curve (Basner et al. 2013). This section presents a method for calculating the costs associated with three health endpoints: hypertension (HYT), myocardial infarction (MI) and stroke.

Incidents of MI and HYT attributable to aviation noise are estimated by developing a relationship between exposure to noise and an increased risk in incidence. The relative risk for MI and HYT at each dB DNL noise level is taken from a meta-study by Basner et al. (2013). The meta-analysis considers six aircraft noise studies relating relative risk of HYT and MI to noise exposure in DNL and develops linear relationships between noise and increased relative risk. Relationships are also developed for ground transport noise in average equivalent noise level (L_{aeq}). The baseline incidence rate for those endpoints for American's aged 20 and over by age and gender (Go et al. 2013) were used to calculate the expected noise-attributable incidence rate for each age, dB level, and gender combination. Once attributable incidents are calculated, damages can be estimated using a Value of a Statistical Life (VSL) and Cost of Illness approach. For HYT, there is limited evidence that disease incidence has an effect on expected future earnings so damages are calculated using annual expected medical contact, hospital, and drug costs (Cropper and Krupnick 1990). The CPI was used to adjust expenditures to a consistent base year of 2010, resulting in annual medical expenses of $776 per incidence. Average life expectancy by age and gender (Arias 2014) and a discount rate of 3% were used to determine net present value of total noise damages.

For MI, the bulk national fatality rate due to MI (Myerson 2009) was used to separate morbidity and mortality impacts. Mortality impacts are valued using two methods frequently used in policy analysis: a Value of Statistical Life (VSL) of $7.93M (2010 USD) as recommended by the US EPA (EPA 2006b) and a Value of Life Years Lost from Bickel and Friederich (2005) using Purchasing Price Parity (PPP), CPI, and a 3% discount rate over the expected remaining life expectancy (Arias 2012) to determine net costs in a consistent baseline year. Non-fatal MI costs are calculated as the sum of opportunity costs and direct medical costs by age at a 3% discount rate using the approach and values outlined in the EPA's Regulatory Impact Analysis for the Federal Implementation Plans to Reduce Interstate Transport of Fine Particulate Matter and Ozone in 27 States (EPA 2011a).

For stroke, Hansell et al. (2013) find a relative risk of hospital admissions of 1.24 (1.08 to 1.34) for people living in the highest noise level (>63 dB) compared to the lowest noise levels (<51 dB). For comparison, Floud et al. (2013) find a slightly higher odds ratio (1.25 for every 10 dB) for 'heart disease and stroke' based on a study of six EU cities. However Floud et al. (2013) include changes in average nighttime noise only and control for people who had been in the same residence for more than 20 years. They find positive but smaller odds ratios for day-time air traffic noise exposure and air traffic noise exposure not controlled for residence length (1.11 and 1.12 per 10 dB respectively), but these results lack statistical significance. Correia et al. (2014) find increased hospital admission rates for all cardiovascular endpoints including cerebrovascular (stroke) events, of 3.5% per 10 dB noise at US airports, but this analysis is limited to the population older than 65.

Because of the different metrics and study parameters used in these analyses, direct comparison across studies is not possible. The Hansell et al. study is used as a midrange estimate of the relative risk of stroke from noise exposure and is applied as a linear relative risk of 1.24 for every 15 dB of noise over 50 dB DNL. The relationship between noise exposure, and in particular aircraft noise, and stroke incidence is uncertain with still other studies reporting no significant risk relationship (Huss et al. 2010, Kolstad et al. 2013). Uncertainty in the stroke concentration response function should be examined, and one representation of uncertainty is described in Section 4.2.3.

The baseline incidence rate and the one-year mortality rate for all-cause strokes by age and gender are taken from Go et al. (2013). Lifetime stroke costs by age of first onset are taken as the sum of direct (hospitalization and rehabilitation) and indirect ischemic stroke costs from Taylor et al. (1996). In addition to opportunity costs from lost wages due to morbidity and mortality, indirect costs from stroke used here include the nonmarket value of household services. The total health cost per person at each dB level is taken as the sum of the person-weighted average health costs of MI, HYT, and stroke.

The health valuation model can be applied to non-US airports by substituting underlying incidence and fatality statistics for the country of interest. However, benefit transfer issues may arise if the baseline health and environmental conditions around the airport of interest differ significantly from the regions included in the dose-response studies or meta-studies. Appendix A provides an analysis of the noise impacts of the London hub airport at present and in 2030. Appendix B examines alternative valuation approaches and additional health endpoints, such as dementia as a secondary effect of hypertension (Harding et al. 2013).

3.2 Air Quality Impacts Model

The Air Quality Module within APMT-I estimates the health impacts of primary and secondary particulate matter from aircraft for emissions produced during the landing-takeoff cycle. Ozone-related health impacts are not considered here as they are estimated to be small relative to PM-related impacts (less than ± 10%) (Ratliff et al. 2009, Watkiss et al. 2005). APMT-I quantifies PM-related health impacts in terms of incidences of premature adult mortality, infant mortality, chronic bronchitis, respiratory and cardiovascular hospital admissions, emergency room visits for asthma and minor restricted activity days (MRADs) and their associated costs. Mahashabde et al. (2011) provides detailed information on the modeling methodology for the APMT-I Air Quality Module, while Ashok (2011) provides updates that reflect the module used in this study. Pollutant and precursor emissions indices, health incidence concentration response functions, and the Value of a Statistical Life (VSL) are treated as uncertain parameters with distributions for the low, mid, and high lenses adopted from Mahashabde et al. (2011).

There are three important unaccounted for effects in the APMT-Impacts Air Quality Module version used in this thesis: the impact of cruise emissions on surface air quality, the impact of changing background concentrations over time, and the impact of modeling scale and resolution.

The current APMT-I Air Quality methodology accounts for only the effects of LTO emissions up to 3000 ft. While this is consistent with other aviation air quality modeling tools and impact studies, there is growing evidence that aircraft cruise emissions contribute significantly to air quality impacts (Barrett et al. 2010, Lee et al. 2013). Koo et al. (2013) estimate that US operations contribute up to 70% more mortalities than accounted for by LTO emissions.

In addition, while emissions from the aviation sector are expected to grow over time, anthropogenic emissions from other sectors are expected to decline in the future. Woody et al. (2011) find a shift in the burden of air quality damages from aviation when using projections of changing background scenarios taken from the IPCC and with reference to domestic air quality control and mitigation strategies. Studies by Ashok (2011) indicate that the APMT-I Air Quality methodology used in this study currently under-represents impacts in 2036 by a factor of 1.6. An updated response surface model (RSM) that directly accounts for background scenario changes has been developed, but was not used for the studies in this thesis (Ashok et al., 2013).

Finally, the choice of model resolution can have an impact on estimates of societal damage from aviation-attributable particulate matter. The APMT-I Air Quality module models air quality impacts on a 36km x 36km grid with a domain that covers the continental United States. This resolution may not capture the severity of air quality impacts at locations very near to the airport, where particulate matter concentrations are higher. Findings by Barrett et al. (2010) find that the APMT-I Air Quality resolution may underestimate physical impacts by 10%. A study by Thompson et al. (2014) using a different air quality model also found that scale effected PM health impacts by ±10%.

The APMT-I Air Quality Module does not model the health impacts of changes in surface-level ozone from aviation or the welfare impacts from reduction of visibility. The response surface model fixes meteorological and climate to present day conditions, so climate-air quality interactions are unaccounted for in model projections. Further, uncertainty in model background concentrations of pollutants and pollutant-precursors such as ammonia are not accounted for in the air quality model.

3.3 Climate Change Impacts Model

3.3.1 Model Description

The APMT-Impacts Climate Module estimates the societal impact of aviation CO_2 and non-CO_2 emissions for both physical and monetary metrics. The APMT-I Climate Module adopts the impulse response modeling approach based on the work by Hasselmann et al. (1997), Sausen and Schumann (2000), and Shine et al. (2005). The module determines the climate system response by superimposing a time series of yearly impulse response curves onto prescribed background anthropogenic emissions. The aviation impacts are calculated by taking the difference of total emissions and total emissions less aviation. The temporal resolution of the APMT-I Climate Module is one year while the spatial resolution is at the global mean level. The effects modeled include long-lived CO_2, the intermediate-lived impact of NO_X on methane (NO_X-CH_4) and its associated primary mode interaction on ozone (NO_X-O_3 long), the short-lived effects of NO_X on ozone (NO_X-O_3 short), the production of aviation induced cloudiness, sulfates, soot, and H_2O. Long- and intermediate-lived radiative forcing impacts associated with yearly pulses of CO_2 and NO_X emissions decay according to their atmospheric lifetimes while the RF from short-lived effects including the warming NO_X-O_3 short effect is assumed to last only during the year of emissions.

A detailed description of past versions of the APMT-I Climate Module can be found in Marais et al. (2010), Mahashabde et al. (2011) and Wolfe (2012). Version 23 of the APMT-Impacts Climate code has three areas of improvement over past models: structural and formatting changes that aid usability and increase speed and flexibility, parameter updates to better reflect current literature and more fully account for uncertainty, and improved functionality in determining the life-cycle costs of conventional and alternative fuels. A brief description of each and the model updates over previous versions are described in the following sections.

3.3.2 Inputs and Model Parameters

The APMT-Impacts Climate module Version 23 has nine functional choices for analysis users: which analysis scenarios to consider (the model inputs); model parameter values and their uncertainty distributions; the number of Monte Carlo runs to perform; the societal damage function; how many years into the future to project radiative forcing, temperature, and monetary impacts; the discount rate; the analysis base year (the year back to which future damage streams are discounted or from which temperature change is calculated); the valuation year (the year of the baseline currency value); and the fleet average fuel mix by fuel type and by year. Version 23 of the model aggregates all analysis choices and makes them explicit for the user.

Inputs for the APMT-Impacts Climate code are single-year impulses or multi-year time-series of full-flight aviation fuel burn, CO_2, and NO_X emissions. Each multi-year time-series of emissions generated from a consistent set of assumptions is called an emissions scenario. Multiple scenarios that occur over the same time period under different policy levers or different potential growth projections are considered an analysis. For instance, an aviation CO_2 policy analysis may consider the impact of aviation's emissions on climate change over the next 40 years and would consist of 10 different scenarios that range from a business-as-usual (or "baseline") projection to a stringent global cap-and-trade policy projection. If scenario emissions are only specified for certain years (e.g. once every decade), the APMT-Impacts Climate code linearly interpolates emissions inventories for the remaining years.

First, the climate module calculates atmospheric concentrations from the input emissions scenarios and background anthropogenic emission projections. A 0.5% uniform uncertainty and 10% uniform uncertainty are applied to input scenario CO_2

and NO_X projections respectively to account for the convergence and resolution of the models that generate these emissions (Mahashabde et al. 2011). The background anthropogenic emissions are selected from the Intergovernmental Panel on Climate Change (IPCC) Representative Concentration Pathways (RCP) (Meinhausen et al. 2011, Van Vuuren et al. 2011). These four pathways were developed to provide consistent background emissions to integrated assessment modelers and explicitly span the range of expected radiative forcing values for 2100 from the literature. Emissions are converted to atmospheric concentrations using a simple impulse response of the form of Equation 3.1.

$$C = a_0 + \sum_{i=1}^{3} a_i \cdot e^{\frac{-t}{\tau_i}} \qquad 3.1$$

The α's in Equation 3.1 represent fractions of airborne carbon while the τ's indicate the timescales of major carbon sink processes over the next thousand years. As such, a_0 is the fraction of carbon that is still present in the atmosphere after 1000 years. Parameters for the impulse response function are derived from a recent model intercomparison (Joos et al. 2013) where:

$a_0 = 0.21787$ $a_1 = 0.22896$ $a_2 = 0.28454$ $a_3 = 0.26863$

$t_1 = 381.33$ $t_2 = 34.785$ $t_3 = 4.1237$

Previous versions of the climate module have used a variety of parameterizations of the impulse-response function (Hasselmann et al. 1997, Hooss et al. 2001, IPCC 2007), and these versions are functionally retained in APMT-Impacts Climate to allow for easy comparison among models and to investigate sensitivity of results to the choice of carbon model.

Second, the code computes the resulting normalized radiative forcing, RF_{CO2}^* at time t' associated with CO_2 concentration at time t' by assuming a logarithmic dependence of radiative forcing on atmospheric CO_2 concentration:

$$RF^*_{CO_2}(t') = \log_2\left(\frac{X_{CO_2(present)} + \Delta X_{CO_2}(t')}{X_{CO_2(1750)}}\right) \qquad 3.2$$

The simplified expression for radiative forcing in Equation 3.2 is found to have good agreement for CO_2 for a wide range of radiative transfer models (Myhre et al. 1998), but differs from complex model projections by up to 5% (Prather et al. 2012, Aamaas et al. 2013). A uniform 5% uncertainty is applied to the resulting RF of Equation 3.2 to account for this uncertainty.

Radiative forcing from short-lived forcers not including NO_X-induced impacts are calculated by scaling the associated speciated radiative forcing in a given year by the change in projected fuel burn as described in Mahashabde (2009). Default parameter values for short-lived forcers are taken from the most recent comprehensive assessment in the literature (D.S. Lee et al. 2009) and are consistent with the values used in Wolfe (2012). In addition, a recent multi-year program initiated by the United States Federal Aviation Administration (US FAA) has presented new model estimates of short-lived forcer radiative forcing (Brasseur et al., 2015). Users are able to choose between the two sets of parameter distributions in the APMT-Impacts Climate module Version 23. Radiative forcing for NO_X related impacts are scaled using a transient response model that accounts for the approximately decadal atmospheric lifetime of induced methane impacts as described in Mahashabde (2009).

Temperature change is related to the time series of induced radiative forcing impacts. APMT-Impacts Climate calculates total anthropogenic, total aviation, and speciated aviation-induced temperature change using a two-box dynamic energy balance model that simulates oceanic heat transfer. Uncertain parameters include the rate of deep-ocean upwelling, the mixing depth, and the rate of convective heat transfer. This temperature response model is described in detail in Wolfe (2012). The climate sensitivity, the expected temperature change for a doubling of atmospheric CO_2, is expanded to a triangular distribution from 1.5 K to 4.5 K with a mode at 3K in alignment with the most recent IPCC Assessment Report (IPCC 2013). This is consistent with recent estimates of climate sensitivity that are lower and more constrained than previous assessments (Skeie 2013).

Next, the health, welfare, and ecological impacts are modeled using damage functions in terms of percentage change of global welfare. APMT-Impacts Climate employs the general analytical framework of the damage function from the latest version of the Dynamic Integrated model of Climate and the Economy (DICE-2013R) to estimate aviation-specific climate damages (Nordhaus 2013). The DICE-2013R model is an integrated assessment model that couples economic growth with environmental constraints to assess optimal growth trajectories in the future and impacts of potential policy measures. While the damage function approach in this model is functionally similar to the damage function of previous DICE models (Nordhaus 2008), its parameterization is derived from a meta-analysis of climate economic studies (Tol 2009). This new formulation includes a multiplier of 1.25 on market damages to account for non-market and non-monetized impacts.

Global gross domestic product (GDP) is used as a proxy for monetized global welfare. Projections of GDP are provided through the five global reference shared socio-economic pathways (SSPs) (O'Neill et al. 2013). The SSPs are a set of socio-economic projections that cover a range of possible socio-political futures along a two-axis scale representing increasing barriers to mitigation and increasing challenges to adaptation. For example, SSP1 models a future with a strong global focus on sustainability and is, therefore, close to the axes intersection in the lower left corner. SSP5 assumes regional fragmentation, heavy competition, and regional economic disparity and is in the upper right corner of possible futures. The impact of SSP on baseline mean social cost of carbon (SCC) is calculated using APMT-Impacts Climate and shown in Figure 3-2.

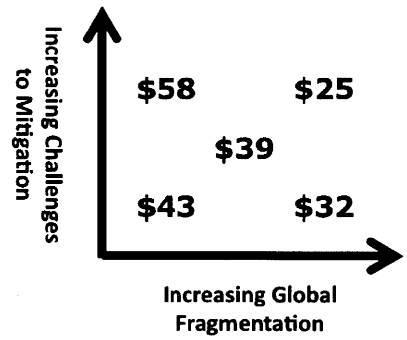


Figure 3-2 SSP influence on mean social cost of carbon (2007 USD)

The combined use of SSPs and RCPs allows integrated assessment modelers greater flexibility in developing consistent scenarios that span the range of possible combined economic and climate outcomes. Disaggregating economic projections from forcing and overall emissions projections is useful as research has indicated that little correlation exists between individual drivers (e.g. population growth, regional economic assumptions) and forcing levels for policy projections (Van Vuuren 2012). The disaggregation also allows integrated-assessment modelers to separately examine the sensitivity of result metrics to economic and scientific assumptions. However, the increased degrees of modeling freedom may add to computational and analysis time, and precaution is necessary as not all SSP and RCP combinations lead to consistent scenarios (Kreigler 2012).

The primary tool for conducting uncertainty analysis in the APMT-Impacts Climate module is Monte Carlo simulation. The model is run thousands of times, each run being a draw of model parameters from the specified uncertainty distributions in the user specified lenses. The resulting distribution is then used to determine the statistical properties such as the mean and the variance of the output metrics of interest. Previous versions of the APMT-Impacts Climate code used pseudo-random drawings using a random number generator. Version 23 of the APMT-Impacts module uses Sobol' sequences, a deterministic set of low-discrepancy number sequences that attempt to sample a space as uniformly as possible (Sobol' 2001).

3.3.3 Fuel Choice and Lifecycle Impacts

Middle distillate (MD) fuels make up approximately 36% of global liquid fuel consumption (Staples et al. 2014). Because ethanol blending increases the vapor pressure of MD fuels and biodiesel is inappropriate for aviation due to its high freezing point and thermal instability, renewable drop-in fuels that are chemically equivalent to conventional MD fuels are of particular interest for issues of both climate sustainability and energy independence (Hileman et al. 2009, Staples et al. 2014). Incorporating the well-to-tank emissions and the combustion effects of alternative fuels in the APMT-Impacts Climate module in a consistent manner allows for a rigorous investigation of either the net climate impact of fuel choice for a given emissions scenario or the net impact of an emission scenario given a fuel choice.

The APMT-Impacts Climate module Version 23 includes 16 fuel feedstocks from four different fuel types: conventional petroleum fuels, pyrolysis fuels, Fischer-Tropsch Fuels, and HEFA fuels as shown in Table 3-1. Non-combustion emission intensities are taken as gCO_2e/MJ from Stratton et al. (2011a). Non-combustion emissions on a per-fuel burn basis are computed by multiplying the emission intensity by the energy density of the fuel. The resulting emissions are accounted for in the climate module as impulses of CO_2 emitted in the same year that the fuel combustion occurs.

Table 3-1 Fuel options in APMT-Impacts Climate v23

Conventional Fuels	**Pyrolysis Fuels**	**Fischer – Tropsch Fuels**	**HEFA Fuels**
1. Jet-A	3. Pyrolysis Jet-A	5. Natural Gas to FT	11. Soy to HEFA
2. Ultra-Low Sulfur Jet	4. Ultra-Low Sulfur Pyrolysis	6. Flared Gas to FT	12. Palm to HEFA
		7. Landfill Gas to FT	13. Rapeseed to HEFA
		8. Biomass to FT	14. Jatropha to HEFA
		9. Coal to FT	15. Camelina to HEFA
		10. Coal/Bio to FT	16. Algae to HEFA

The APMT-I Climate module also accounts for the difference in combustion emissions that result from the choice of fuel. For Jet-A, fuel sulfur content is assumed to average 600 ppm, while for Ultra-Low Sulfur (ULS) fuels, a fuel sulfur content of 15ppm is assumed (Barrett et al. 2012). The sulfur content of FT and HEFA fuels are both assumed to be negligible. For FT and HEFA fuels, the impact of the fuel choice on non-CO_2 combustion emissions is taken from Stratton et al. (2011b). For water vapor and soot, the magnitude of the attributable radiative forcing directly scales with fuel burn, so the non-CO_2 combustion emissions impacts are applied as scaling factors on the expected radiative forcing. For NO_X-induced impacts, the alternative fuel impacts are applied by scaling the full-flight NO_X inputs. The impact of alternative fuels on contrail formation, induced cloudiness, and indirect cloudiness is currently indeterminate. Thus, aviation induced cloudiness radiative forcing is treated identically for all fuel choices.

Figure 3-3 shows the mean temporal evolution of temperature impacts by component species for 30 years of aviation operations with conventional jet fuel. The black dot-dash line indicates non-combustion emissions. Combustion CO_2, aviation induced cloudiness, and the short-term ozone response from NO_X emissions dominates warming during the aviation emissions scenario while combustion CO_2

and non-combustion emissions dominate the long-tailed response. The temperature response from sulfates and the long-term NO_X impacts on methane and ozone are cooling but of a lower magnitude than the effects from the dominant warming emissions.

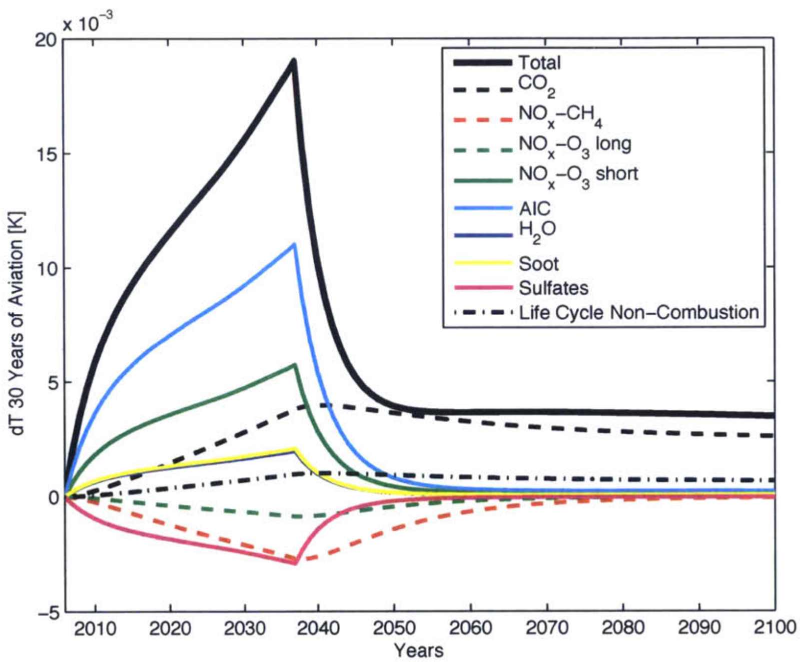


Figure 3-3 Mean life-cycle speciated temperature change for 30 years of aviation with conventional jet fuel

Figure 3-4 shows the difference between the temperature projection from conventional jet fuel and the temperature projection from identical operations with a fleet-wide 50/50 HEFA blend. A switch to the HEFA alternative fuel leads to a small (<0.002 K) short-term warming due to the resulting reduction in sulfates.[3] However, in the long-term response, conventional fuel produces a greater climate warming than a camelina blend primarily because of differences in non-combustion emissions.

[3] A recent study has found that reducing the sulfur content of jet fuel may result in a net cooling impact from trade-offs in nitrate formation (Unger 2011). However, another study has found that the indirect cooling from sulfate-induced cloud brightening is of greater magnitude than the direct cooling effect from sulfates (Gettelman and Chen 2013). This indirect-cooling effect would only exacerbate the short-term warming in a switch to HEFA fuels. These secondary impacts are an area of necessary future research and are not accounted for in the current model.

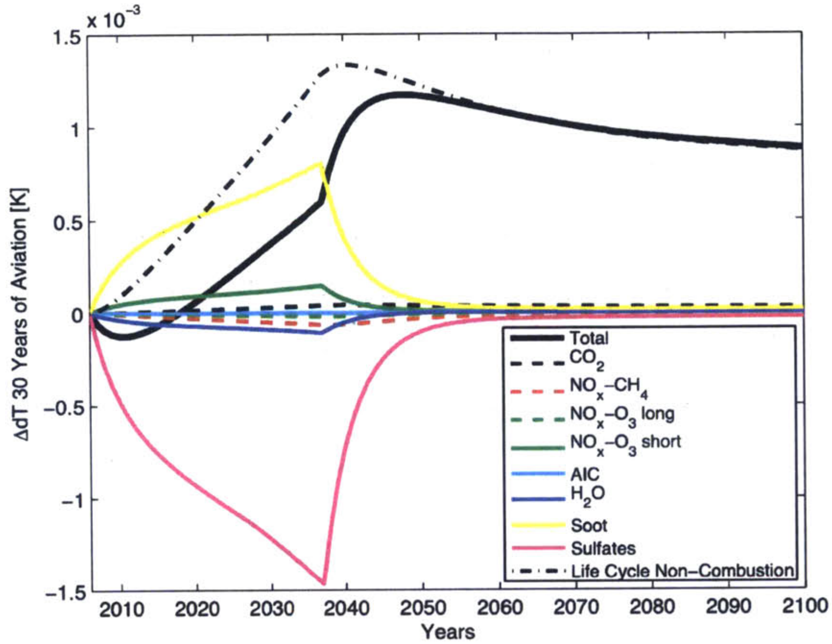


Figure 3-4 Life-cycle speciated temperature change differential between conventional jet fuel and a 50/50 camelina HEFA blend for 30 years of aviation. Negative values indicate where conventional jet would be cooler relative to camelina and positive values indicate where camelina fuel would provide cooling relative to conventional jet.

3.3.4 Model Summary and Limitations

Thousands of unique combinations of inputs and model parameters that may be of interest in assessing different policy options are possible. In order to extract meaningful insights about the possible costs and benefits of a policy, it is therefore expedient for the analysis options to be synthesized into a set of pre-defined combinations of inputs and assumptions that are developed through an iterative process with policymakers, modelers, and stakeholders. These combinations can be thought of as describing a particular point of view or perspective, and are thus designated as lenses. When chosen wisely and viewed as a whole, the lenses can be seen as representing the range of perspectives of the relevant environmental impact domains.

Limitations of the APMT-I Climate Module include the use of a global spatial scale that does not capture regional variations in short-lived aviation climate effects, the lack of consideration of feedbacks in the climate system, and independent treatment of aviation effects that does not account for interactions among some of the different physical and chemical mechanisms. The APMT-I Climate Module currently does not consider the climate impacts of nitrate emissions or the indirect effect of aerosol emissions on cloud or surface albedo. For alternative fuels, the climate module does not currently consider the impacts of planetary albedo differences from land use change, indirect land use change impacts, or the impact of alternative fuels on cloud formation and properties.

3.4 Aviation Lead Impacts Model

This section describes the development of a modeling approach for calculating the societal costs of piston-driven aviation lead emissions. An overview of lead's impact on human health and development and the environment was provided in Section 2.2.4. The model developed in this section focuses on IQ reductions from lead exposure as the primary driver of monetized environmental damages. Section 3.4.1 provides the motivation for an aviation-specific approach. This section provides a review of the aviation lead literature and highlights opportunities for model contributions. Section 3.4.2 describes the approach for developing the emissions inventories and for modeling emissions from LTO and full-flight lead emissions. The relationship between increases in atmospheric concentrations of lead and IQ losses is presented in Section 3.4.3. The modeling approach for calculating economic damages is described in Section 3.4.4. Finally, Section 3.4.5 discusses some of the limitations of the current model, highlighting opportunities for future work.

3.4.1 Model Motivation and Literature Review

Lead is a persistent, bioaccumulative and toxic pollutant that has negative health and environmental effects; in particular, lead ingestion and inhalation impacts neurodevelopment in children. Lead emissions from general aviation (GA) piston-driven aircraft are attributable to the addition of tetraethyl lead (TEL) for the formation of hundred octane low lead (100LL) avgas. The lead additive prevents engine knock, improves effective fuel octane, and prevents valve seat recession. As discussed in Section 2.2.4, general aviation now accounts for the largest source of lead emissions in the United States.

Recent research has progressed the understanding of the immediate and long-term effects of aviation lead emissions. Miranda et al. (2011) studied blood lead levels in children in the proximity of airports in North Carolina and found a correlation between distance of residence from the airport and blood lead concentrations. Zahran et al. (2014) expanded the analysis of Miranda et al. (2011) to over 1 million children in the proximity of 448 Michigan airports, and again find statistically significant increases in blood lead level, while controlling for seasonality of soil lead re-emissions. Carr et al. (2011) developed a local dispersion modeling technique to model lead concentrations in the immediate vicinity of GA airports using a case study of the Santa Monica airport finding that landing and takeoff operations lead to higher concentrations of lead up to 450m from the airport boundary. Feinberg and Turner (2013) also use a local dispersion model to predict lead concentrations near a GA airport further examining the impact of meteorological and diurnal emissions distribution assumptions.

While these studies help explain the impact of leaded emissions on the environment, they leave three opportunities for modeling improvement. First, these models do not consider the impact of full-flight emissions from general aviation. For the 2008 National Emissions Inventory, the United States Environmental Protection Agency redeveloped a methodology for calculating lead emissions inventories from General Aviation aircraft (EPA 2010b). The EPA assumes that over 50% of aviation emissions occur outside the LTO phase of flight, but these emissions are not accounted for in point- and area-source dispersion modeling at the airport. Further, these important emissions may not be distinguishable from background emissions in monitor data. Therefore, modeling long-range dispersion from GA aircraft is necessary to understand the potential impact of these emissions.

Second, modeling total lead concentrations is important for understanding air quality with regard to federal limits and standards, but it does not indicate the magnitude of environmental costs from lead emissions. Miranda et al. (2011) and Zahran et al. (2014) both estimate blood lead levels in children, but only Zahran et al. (2014) estimate the societal damages from these elevated lead levels. In the model described in this thesis, the social cost of future IQ-related employment losses are developed from childhood exposure to lead. Foregone wages are a common monetary endpoint in lead analysis (e.g. Zahran et al. 2014). Additional neurodevelopment health impacts, such as attention, impulse control, hyperactivity, and conduct disorders, are not monetized in this model (Burns et al. 1999; Chandramouli et al. 2009; Wright et al. 2008). Thus, this modeling approach produces economic impact estimates that are a lower bound on the societal impacts of lead exposure.

Finally, labor and earnings losses from cohort-wide IQ reductions may have an impact on total economic productivity. Studies of the environmental benefits of transportation and energy generation policies have demonstrated the importance of modeling economy-wide damages (Thompson et al. 2014; Rausch et al. 2010). The dynamic economy-wide impact of lead-related IQ loss has not been quantified for lead exposure. GA emissions may be of particular importance in determining US-wide economic impacts. Because there is no known safe threshold concentration for lead and research has indicated that significant neurologic damage occurs even at very low exposure levels (Lanphear et al. 2005; Bellinger 2008), and because some models suggest that the rate of IQ loss is greatest per unit blood lead at lower concentrations (Lanphear et al. 2005; Gould 2009; Crump et al. 2013), these previously unaccounted for emissions may lead to significant unaccounted for economic damages.

Dispersion of cruise emissions may lead to small surface-level atmospheric concentration increases at a distance from the typically modeled lead sources (e.g. an airport). The EPA assumes that 53% of aviation lead emissions are emitted outside of the landing and takeoff cycle and states that "it is reasonable, then, to generally expect that lead emitted outside the LTO cycle during itinerant operations [...] will be more widely dispersed and at greater distances from the airport" (EPA 2010b).

Thus, this work looks to consistently model the continental US-wide impacts of aviation lead emissions. The model is the first to examine the contribution of full-flight GA emissions to atmospheric lead concentrations. Further, the model is the first to compute the magnitude of aviation emissions related IQ loss from atmospheric concentration changes, and is the first model to quantify dynamic economy-wide losses from lead-related damages. The model is subject to scientific uncertainty from the dispersion, concentration-response, and monetization modeling approaches as well as structural uncertainty and potential bias from the model and input assumptions. Therefore, the model described in this chapter as well as the analysis provided in Section 4.3 should not be interpreted as a definitive assessment of General Aviation lead emissions but as a benchmark for understanding the social and environmental impact of full-flight lead emissions in the United States.

3.4.2 Lead Inventory and Emissions Modeling

Lead is emitted primarily as lead salts during all phases of aviation flight (Biggins and Harrison, 1979) with between 0.3-3% of lead emitted as undecomposed TEL (Seyferth 2003). In addition a small percentage of lead is emitted to the atmosphere as evaporated losses during refueling (Carr et al. 2011). The evolution of lead particle emissions and lead particles from aviation is shown in Figure 3-5.

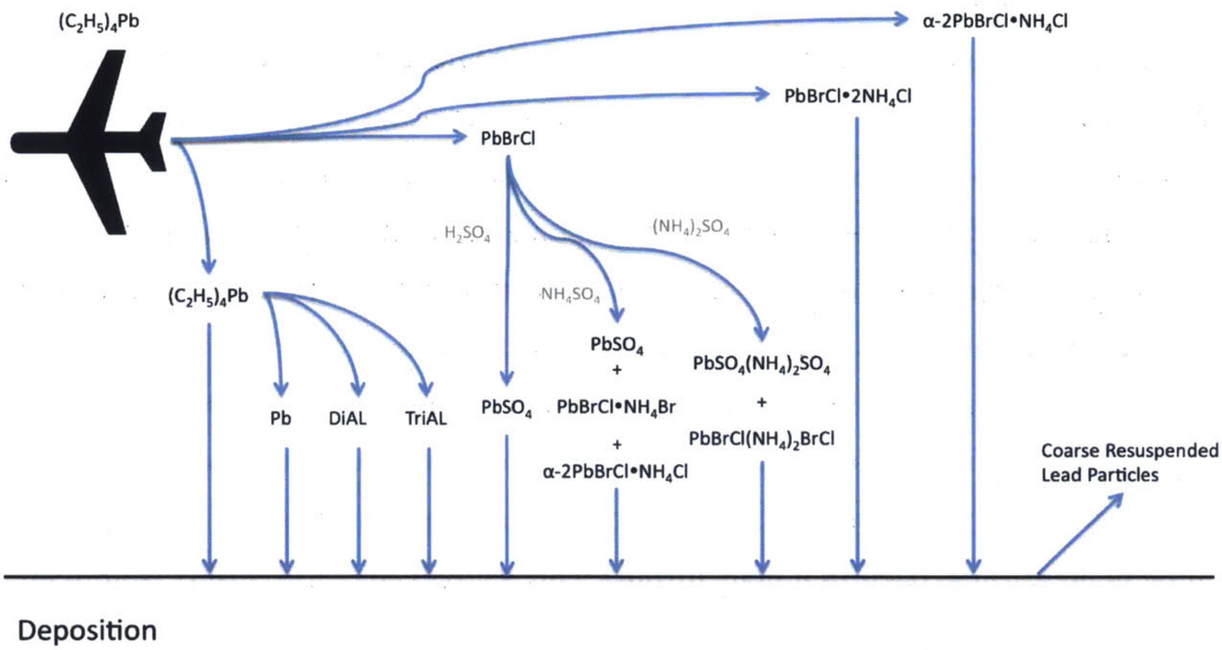


Figure 3-5 General aviation lead emissions pathway

The impact of aviation lead on atmospheric concentrations of lead is modeled for the year 2008. The total consumption of leaded aviation gasoline (avgas) in the United States in 2008 was 248 million gallons (FAA 2013). The most common formulation of avgas supplied in the US is "100 Low Lead" (100LL), which has a maximum lead concentration of 2.12 gPb/gal. For airport landing and take-off emissions, the model follows the methodology of the EPA guidance for calculating piston-engine aircraft airport inventories that was first developed for the 2008 NEI (EPA 2010b). The inventory assumes that 5% of lead is retained in the engine, engine oil, or the exhaust system (EPA 2010b). While some lead may be lost through fuel evaporation, leakage, fuel checks, and in the on-airport transport process, these losses are assumed to be negligible (Huntzicker et al. 1975). Limiting the domain of the analysis to the continental United States, this results in total aviation lead emissions of 539 short tons of lead in 2008.

Landing and take-off (LTO) emissions inventories are provided by the EPA National Emissions Inventory for nearly 20,000 airports and airfields resulting in 257 short tons of lead emitted in LTO. 2.6% of these emissions occur outside of the continental United States and are not accounted for due to the geographic domain of the air quality model. A national seasonal distribution to the GA operations that peaks in May (9.8% of operations) and reaches a minimum in January (6.8% of operations) is applied in accordance with a detailed study of the spatial and seasonal patterns of general aviation (Wang and Horn 1985). This seasonal pattern is functionally similar to the site-specific GA pattern used in a lead study at Santa Monica Airport (Carr et al. 2011). However, regional seasonality may be greater in some areas, in particular the Great Lakes region (Wang and Horn 1985). A one-peak diurnal profile of operations with operations beginning in the morning, peaking at midday, and ending in the evening that approximates the temporal profile of operations used in a near-airport lead study (Carr et al. 2011) is applied for all airports. A sensitivity study on lead dispersion found that annual concentration levels were not sensitive to choice of diurnal profile (Feinberg and Turner 2013).

The remaining lead is emitted during the cruise phase of flight. GA flights occur for a variety of purposes including flight instruction, personal or business use, patrol and firefighting, and charter use. While each of these uses has a unique operational pattern, most GA flights occur within a one hour flight-radius of the originating airport. Thus, for the latitudinal and longitudinal distribution of emissions from GA flights, cruise emissions are apportioned across a state in accordance with the percentage of operations that originate in that state according to the methodology of the EPA NEI guidance (EPA 2010b). The altitudinal distribution of non-LTO phases of GA flights is heterogeneous and can occur both above and below the mixing height depending on the flight purpose. While most GA aircraft are unpressurized, some pressurized planes will achieve average operating altitudes above 10,000 ft. A study of 71 GA aircraft that cover a range of aircraft type, primary-use purpose, and operational characteristics is used to develop a triangular characteristic altitudinal distribution of emissions with a mode of 3000ft and a peak of 13,000ft (Locke et al. 1993). These operational characteristics are in line with altitude profiles of GA planes from December 2007 and June 2008 radar data, which had a modal peak in the 1200-3000 ft altitudinal range and decreasing frequency of flights with increasing altitudinal band (Kochenderfer et al. 2008). The spatial distribution of the average hourly lead emissions is shown in Figure 3-6.

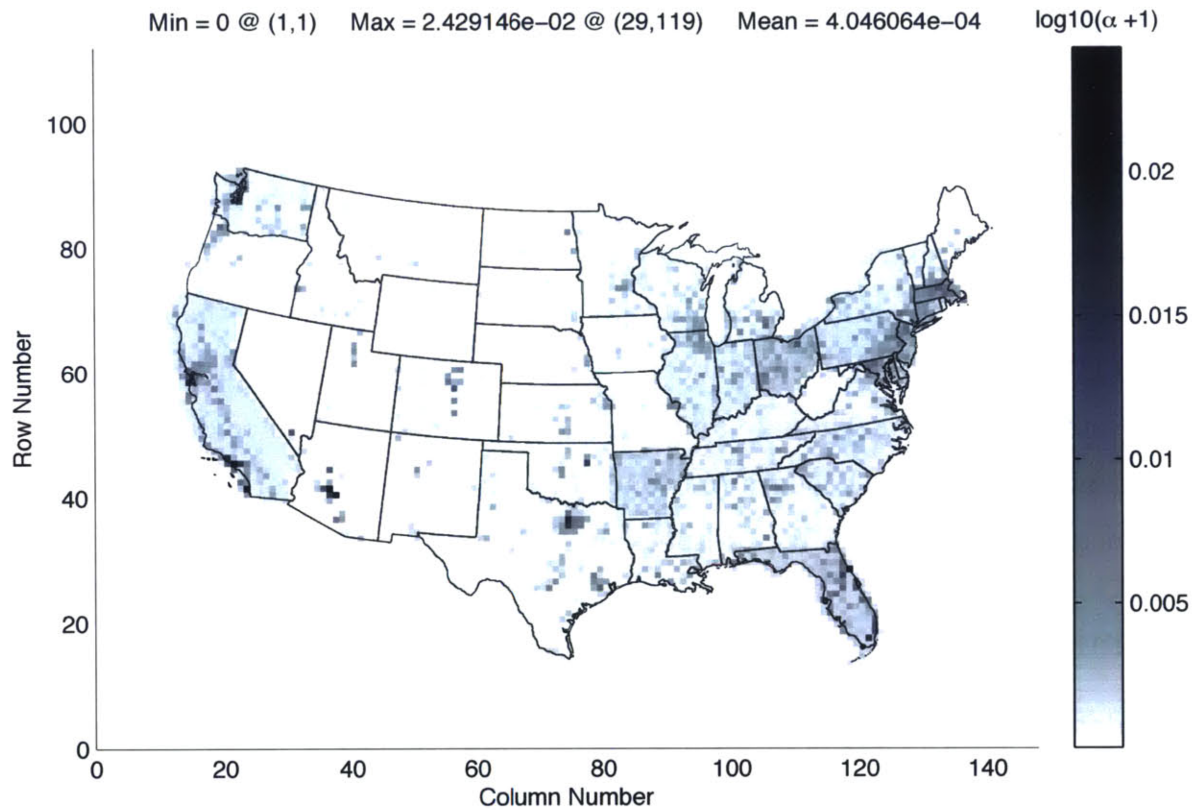


Figure 3-6 Modeled spatial distribution of the log of the aviation lead emissions inventory [α] (gPb/s)

Background emissions for all emissions species were developed from the U.S. EPA National Emission Inventory for 2005 (Woody et al. 2011). Lead emissions from this inventory were scaled by 55% to account for the removal of 2005 aviation emissions, which were generated using the old EPA inventory methodology (EPA 2010b) and not distributed in a spatially consistent manner. While 2005 background emissions were used as a surrogate for 2008 background emissions, total lead emissions from non-aviation sources are expected to have changed by less than 2.5% from 2005 to 2008. Meteorological inputs are provided using the Weather Research and Forecasting (WRF) v3.3.1 model (Skamarock and Klemp 2008) for the year 2005. Initial and boundary conditions are obtained from three-dimensional tropospheric chemistry simulations from the Goddard Earth Observing System of the NASA Global Modeling Assimilation Offices (GEOS-Chem) (Bey et al. 2001, Lam and Fu 2010). The Community Multiscale Air Quality (CMAQ) modeling system (Byun and Schere 2006) v4.7.1 is used to model aviation emission-attributable lead concentrations in the continental United States. CMAQ is a high-resolution regional air quality model used by the EPA to support regulatory impact assessment. CMAQ has been developed for multi-pollutant and air toxic assessment. Aerosol phased hazardous pollutants are tracked using the multi-pollutant CMAQ model and, while chemically inert, undergo microphysical processes and deposition. The fate and transport of metals and air toxics have been modeled and compared to background concentrations from monitor data for several species including lead (Hutzell and Luecken 2008).

3.4.3 Emissions-to-IQ Loss Pathway

Population exposure to emissions is determined by overlaying annual average surface concentrations on population projections stratified by age group provided by Woods and Poole and previously used in aviation environmental analyses (Levy et al. 2012). Lead in ambient air can contribute to several exposure pathways, including direct inhalation, and—once the lead is deposited to the surface—ingestion with indoor or outdoor dust, soil, water, and food. Young children's exposure to ambient lead is predominantly through the ingestion pathway, with lead-based paint ingestion representing up to 70% of US childhood lead exposure in the 2000's (Cornelis et al. 2006; Levin et al. 2008). Historical anthropogenic emissions, and existing stock of lead-containing paints and other products can also contribute to the burden of lead in dust and soil. Because of these multiple pathways, the relationship between recent ambient lead (PbA) and blood lead (PbB) concentrations can be difficult to determine. Several studies use historical data to develop regression models that estimate the impact of changes in PbA on children's

PbB, controlling for factors that could be predictors for non-recent air pathways, like geographic location, home age, and race/ethnicity (Richmond-Bryant et al. 2014; Zahran et al. 2013; Ranft et al. 2008; Tripathi et al. 2001; Hilts 2003; Hayes et al. 1994; Schwartz and Pitcher 1989; Brunekreef 1984).

Based on these studies, the model developed here considers two functional forms for the PbA (µg/m^3) to PbB (µg/dL) relationship. The first relates ln(PbA) to ln(PbB) (ln-ln): $\ln(PbB) = \beta \cdot \ln(PbA) + \gamma$. The ln-ln model results in larger changes in PbB per change in PbA at lower PbA concentrations. The second linearly relates untransformed PbA and PbB (linear): $PbB = \beta \cdot PbA$. Table 3-2 summarizes the ambient concentration and blood lead level relationships used in this model and their sources. For the linear models, slope values are consistent with ranges developed from case studies using the mechanistic Integrated Exposure Uptake Biokinetic (IEUBK) model of the PbA-PbB relationship (EPA 2007).

Table 3-2 Parameterization of PbA (μg/m3) to PbB (μg/dL) relationship

Model Form	Source	*β*	Notes
Ln-ln	Richmond-Bryant et al. (2014) A	0.076	Based on NHANES III (1988-1994) national data and annual average TSP from the EPA Air Quality System, for children 1-5 yrs.
	Richmond-Bryant et al. (2014) B	0.140	Based on NHANES 9908 (1999-2008) national data and annual average TSP from the EPA Air Quality System, for children 1-5 yrs.
	Brunekreef (1984) A	0.3485	Meta-analysis of 18 studies spanning 1970-1982, for a variety of countries, and children's ages. Model based on data for all children.
	Brunekreef (1984) B	0.2159	Meta-analysis of 18 studies spanning 1974-1982, for various countries, and children's ages. Model based on data for children with "low" PbB (<20 μg/dL).
	Hayes (1994)	0.24	Based on PbB data from Chicago, IL (1968-1988), with quarterly average PbA data from the IL EPA, for children 0.5-6 yrs.
Linear	Hilts (2003)	7	Based on PbB data from Trail, British Columbia (1991-2000), with quarterly average PbA data from the smelter company, for children 0.5-6 yrs.
	Schwartz and Pitcher (1989)	8.6	Based on PbB data from Chicago, IL (1976-1980), with PbA estimated from gasoline usage, for children 0-5 yrs.
	Tripathi et al. (2001)	3.6	Based on PbB data from Mumbai, India (1984-1996), with 24-hr PbA data collected by the authors at residences of the study population, for children 6-10 yrs.

The relationship between blood lead level and neurological development disruption is uncertain. Research indicates that concurrent blood lead level measured during childhood is the best predictor of IQ changes when controlling for other environmental variables (Crump et al. 2013; Budtz-Jørgenson et al. 2013). Four concentration response functions identified by the EPA are used to model the resulting IQ decrements from changes in children's concurrent PbB (EPA 2007; EPA 2011c). These models are based on the pooled dataset from the Lanphear et al. (2005) meta-analysis of seven longitudinal cohort epidemiological studies, adjusted for the errors identified in the independent re-analysis performed by Crump et al. (2013). The four concentration-response functions take different functional forms (log-linear with threshold, log-linear with no threshold, dual linear with hinge at 5 μg/dL, and dual linear with hinge at 3.75 μg/dL[4]) to capture uncertainty in the PbB-IQ relationship and are shown in Table 3-3 (EPA 2007; EPA 2011c).

There is no clear exposure threshold below which lead is not expected to have a negative impact on IQ (Budtz-Jørgensen et al. 2013). Thus, blood lead levels of 0.1 – 1 may be interpreted as a Benchmark Dose (BMD), a dose that leads to a specific known loss, rather than a dose that produces no damages (Budtz-Jørgensen et al. 2013). Thus, while the log-linear threshold CRF is presented for consistency with the EPA (EPA 2007; EPA 2011c), instances where this threshold leads to 0 expected damages are excluded from statistical analysis of the results.

Table 3-3 Blood lead to IQ loss relationship

Model Form	Equation
Log-linear with threshold	*For concurrent PbB > 1 μg/dL* $\Delta IQ = \ln(PbB) \cdot 2.65$ *For concurrent PbB ≤ 1 μg/dL* $\Delta IQ = 0$
Log-linear with linearization at low exposure	*For concurrent PbB > 1 μg/dL* $\Delta IQ = \ln(PbB) \cdot 2.65 + 2.65$ *For concurrent PbB ≤ 1 μg/dL* $\Delta IQ = PbB \cdot 2.65$
Dual linear with hinge at 5 μg/dL ("Dual linear - 10")	*For concurrent PbB > 5 μg/dL* $\Delta IQ = (PbB - 5) \cdot 0.13 + 5 \cdot 0.77$ *For concurrent PbB ≤ 5 μg/dL* $\Delta IQ = PbB \cdot 0.77$
Dual linear with hinge at 3.75 μg/dL ("Dual linear - 7.5")	*For concurrent PbB > 3.75 μg/dL* $\Delta IQ = (PbB - 3.75) \cdot 0.15 + 2.53 \cdot 3.75$ *For concurrent PbB ≤ 3.75 μg/dL* $\Delta IQ = PbB \cdot 2.53$

3.4.4 Economic Modeling

Following previous studies estimating the economic impacts of lead, this model focuses only on the earnings and productivity losses associated with IQ loss due to

[4] The EPA originally used peak blood concentration relationships from Lanphear et al. (2005) in examining low concentration IQ deficits, which had hinges at 10 and 7.5 μg/dL. However, the reanalysis by Crump et al. (2013) and other analyses use concurrent blood lead, a convention that is now standard. However, even though the relationships using concurrent blood lead have hinges at 5 and 3.75 μg/dL, they are often still referred to as "dual linear – 10" and "dual linear – 7.5".

children's exposure to lead as it one of the most well-understood damage pathways and it is expected to be the most costly (Schwartz 1994; Salkever 1995; Grosse et al. 2002; Gould 2009). These estimates are therefore lower bounds on the societal impacts of lead exposure, and other potential damage pathways are discussed as model limitations in Section 3.4.5. The economic impacts of IQ loss associated with ambient lead exposure are estimated using two methods: a static estimate of the net present value (NPV) of earnings losses for one cohort of 1 year olds, and a dynamic computable general equilibrium estimate that uses earnings losses to individual cohorts as an input to estimate the economy-wide losses associated with IQ loss. Following a 1-year cohort is a useful modeling simplification as it provides an indication of the annual environmental costs of aviation-attributable lead emissions as IQ loss correlates best with concurrent blood lead level. However, because IQ damage is also correlated to childhood average blood lead concentration and because several same-aged cohorts of children will be exposed to aviation lead for each year of operations, this assumption again provides a lower bound of lead impacts.

Estimates of the percentage change in lifetime earnings associated with an IQ point reduction are taken from both the environmental health and labor economics literature (Salkever 1995; Zax and Rees 2002; Grosse et al. 2002; Heckman, Stixrud, and Urzua 2006). These estimates take into account both the direct impacts of IQ on wage, and indirect effects of IQ on schooling. As a base case, the model uses an estimate based on Zax and Rees (2002) of 1.1% loss of lifetime earnings per IQ point. To provide an indication of parameter uncertainty, the model also considers relationships from Heckman, Stixrud, and Urzua (2006) and the relationship from Salkever (1995) of 2.37% per IQ point, which has been used in previous EPA analyses, though this value is high compared to results from labor economics studies (Grosse 2007). The model calculates the NPV of lifetime earnings for a cohort of 1 year olds using earnings data, stratified by age group, from the US Department of Labor's Bureau of Labor Statistics. Following Grosse et al. (2002), productivity is assumed to increase by 1% per annum and future earnings are discounted with a constant 3% discount rate.

To estimate the impacts of children's IQ-related earnings loss on the US economy as a whole, the US Regional Energy and Environmental Policy (USREP) model is expanded to calculate economic damages from labor losses due to lead pollution. USREP is a recursive-dynamic general equilibrium model of the US economy, described in Rausch et al. (2010; 2011). USREP represents utility-maximizing households and profit-maximizing firms as rational economic agents, and finds the optimal, equilibrium condition of the economy (expressed through commodity prices). Market interactions (production and consumption) are based on microeconomic theory, and therefore depend on the relative prices of different goods, services, and availability of production factors like labor and capital. In USREP, the available labor pool results from a choice between labor and leisure at the household level. USREP uses 2006 as a base year, and solves recursively for equilibrium economic conditions at 5-year intervals starting in 2010. Production and consumption are modeled as nested constant elasticity of substitution functions.

USREP has been used to explore the economy-wide effects of climate, energy, and air quality policies over time (Rausch et al. 2010; Rausch et al. 2011; Saari et al. 2014; Caron, Rausch, and Winchester 2014). Medical services, and labor and leisure are diverted to the pollution health sector to produce good health. In the case of IQ loss, only IQ's effect on total lifetime earnings is considered—the labor input to the pollution health sector. Economy-wide estimates of the costs of IQ-related earnings loss include not only the direct earnings losses to cohorts of children, but the compounding effect of these losses over time on the productivity of the economy as a whole. These losses are expressed in terms of changes to consumer welfare (the amount required to compensate households for the health impacts), which includes consumption and changes to leisure.

3.4.5 Model Limitations and Caveats

Each sub-model of the aviation lead impact model has important limitations that may influence the usefulness of results in some contexts. The lead inventory is limited by the sources provided in the EPA National Emissions inventory. Recent research suggests that forest fires and especially legacy lead re-emissions from soil are increasingly important sources of lead to the atmosphere (Odigie and Flegal 2014; Harris and Davidson 2005). These sources, like aviation, were an insignificant source of airborne lead during the peak of leaded gasoline, but now may be a principal source of emissions in certain regions. Due to the logarithmic relationships between atmospheric lead burden and eventual IQ loss, failure to account for these emissions may upwardly bias the damages attributed to aviation. Further, leaded paint and paint dust is still expected to be the largest contributor to childhood exposure to lead, with exposure risk being spatially and demographically heterogeneous (Leven et al. 2008). This heterogeneity is not currently accounted for in the model.

The spatial resolution of CMAQ incorporated in this model is 36km × 36km. This resolution may lead to an overestimation of lead concentration further from an airport boundary and an underestimation of lead concentration nearer the airport. Zahran et al. (2014) found the risk of elevated blood lead levels in children to be substantially greater at <2km from an airport than at >2km from the airport. Modeling LTO emissions, Carr et al. (2011) found elevated lead concentrations up to 900m from the airport. However, Carr et al. (2011) consider the background concentration of lead to be 10 ng/m^3, greater than the 0.5 ng/m^3 EPA recommended value for pristine background concentration, and do not consider aviation's contribution to the elevated background level.

Further, CMAQ makes possible the modeling of full flight emission contribution to lead concentrations and the consideration of lead concentration deltas that are smaller than air quality monitor resolution. The dry deposition velocities for lead particles between 0.01 and 0.4 μm stokes diameter range from 0.003 – 0.5 cm/s, and the bulk lead particle dry deposition velocity has been estimated at 0.1 – 0.5 cm/s (EPA 1977, EPA 2006a). For full flight emissions, deposition rates and horizontal transport will depend upon factors beyond dry deposition velocity including boundary layer resistance, coefficient of drag, and meteorology. However, dry deposition velocity gives an indication of the importance of modeling dispersion. For example, for a lead particle emitted at 3000 ft, the dry deposition velocity would predict an atmospheric residence time on the order of 50 hours. Thus, in the presence of 10-30 km/hr wind speeds, the particle could travel between 10-1500km, placing lead emission concentrations on a regional scale. The CMAQ domain is also limited to the continental United States, and therefore does not account for over 8 tons of yearly lead emissions in Alaska.

The modeled near-airport surface concentrations are limited by the EPA methodology for compiling the aircraft-attributable lead emissions. Feinberg and Turner (2008) and Carr et al. (2011) find near-field and on-site concentrations to be highly sensitive to the amount of run-up emissions. A modeling approach that combines more detailed airport-by-airport operational data, higher-resolution modeling near-field dispersion of run-up, surface, and LTO emissions, and more spatially coarse modeling of full-flight emissions for lead concentrations, such as the approach in Wolfe et al. (2014) or Yim et al. (2015) for PM concentrations, represents an opportunity for future work.

Earnings reductions related to IQ loss are only one effect from lead exposure. High lead levels can lead to damages to the nervous, circulatory, endocrine, and renal systems, which may contribute to health costs and foregone wages (Bernard 2003). At high blood lead levels, the Centers for Disease Control and Prevention (CDC) prescribes medical intervention for heavy metal poisoning that can include oral or intravenous chelation. Based on cost analyses by Kemper et al. (1998), Gould estimates \$11-\$53 billion dollars as a conservative estimate of medical treatment costs from total lead hazards, about 6%-20% of total lead damages. While aviation emissions may contribute to elevated blood lead levels, there is limited evidence that aviation emissions alone contribute to blood lead levels in excess of 10μg/m^3, the threshold for which CDC intervention medical costs have been fully assessed in literature. However, there is no blood lead level threshold below which adverse health effects have not been observed (Bernard 2003).

Childhood exposure to lead has also been linked to criminal activity. The environmental hypothesis for social crime rates suggests, first, that childhood exposure to lead increases the likelihood of possessing low behavior and cognition self-control and that, second, low-self control is an important predictor of adolescent and adult criminal behavior (Needleman et al. 2002; Stretesky and Lynch, 2004; Mielke and Zahran 2012). Mielke and Zahran (2012) relate aggravated assault to 22-year lagged emissions of lead, suggesting that abating lead in the present may provide long-term benefits. Nevin (2007) linked lead-exposure to increases in burglaries, robberies, aggravated assaults, and murders, which suggests a total direct cost of lead-linked crimes in the US in 2006 of $1.8 billion (Gould 2009). Indirect costs, including treatment for psychological and physical damages may attribute to an additional $11.6 billion in damages.

Finally, unlike the APMT-Impact models described in Sections 3.1, 3.2, and 3.3, the GA lead model only examines the impact of one environmental stressor: aviation-attributable lead emissions. GA operations will also potentially contribute to non-negligible concentrations of surface level PM and ozone that degrade local air quality, and emissions of CO_2 and other short-lived forcers from GA aircraft may contribute to increased climate warming. Further, aviation-lead-related cloud brightening may contribute to a negative radiative forcing, which would be in counter to the climate warming from other chemical species. In a traditional cost-benefit analysis, this cooling could be an unintended benefit that is also not calculated by this model. Finally, the effects of GA noise are also not considered. While these shortcomings do not negate the findings of the lead pathway model, they do suggest that all findings as to the environmental costs of GA emissions be properly contextualized.

3.5 Chapter Summary

Byproducts of aviation effect human health, welfare, and the environment in diverse ways. Aircraft noise can be a burden on local communities, causing high levels of annoyance and activity and sleep disruption. Further, aircraft noise exposure can raise the risk for premature mortality and morbidity through incidences of heart attack and stroke and prevalence of hypertension. Aircraft emissions during the LTO phase as well as cruise emissions that are transported to the surface impact local air quality, which can lead to premature mortality and decreased visibility. Climate forcing attributable to aviation emissions will affect global agriculture, ecology, security, and health. Finally, leaded fuel used in piston-driven aircraft presents a persistent hazard that can disrupt neurological development in children. This chapter described models of varying complexity used to quantify the environmental impacts of aviation noise and emissions. In particular, the contributions of this chapter are a model for quantifying the health impacts of aircraft noise in Section 3.1.2, continued development of an aviation specific climate model expanded to model the potential adoption of alternative fuels in Section 3.3.3, and the preliminary development of a consistent method for modeling the economic costs of general aviation lead emissions in Section 3.4.

Designing impact models represents a tradeoff across several factors including accuracy, robustness, speed, flexibility, scientific and political acceptance, and applicability to a range of objectives. Much as air transport decision-makers must balance safety, economics, and the environment, modelers must balance these performance criteria in designing applicable models. The impact models described in this chapter are primarily designed for policy analysis and baseline impact assessment. To that end, the next chapter presents three case studies applying the models developed here.

Chapter 4

Aviation Environmental Policy Assessment

A complex set of questions exists at the nexus of technology and policy. The developments that facilitate transportation, energy production, and economic expansion can also have unintended impacts on human health, the environment, and safety among others. As explored in Chapter 2, these externalities are diverse, homogenous in time and space, and uncertain. Evaluating policies to mitigate and control the unwanted byproducts of technological progress is difficult. How should society balance environmental protection and economic development? How should a policy be evaluated if, beyond its primary benefits, it leads to tradeoffs and disbenefits for other environmental objectives?

Aviation and the environment provides a case study for exploring policy evaluation. Noise, air pollution, climate change, economic growth, efficiency, and safety are interconnected, and decisions in any one domain may impact others. This chapter provides three examples of aviation environmental analysis. The tools developed in Chapter 3 are used to quantify and assess the environmental impacts of aviation. These impacts are then weighed against the costs, feasibility, and political appropriateness of the policy. Section 4.1 analyzes a proposed policy to increase the stringency of international aircraft noise certification limits. Section 4.2 assesses current land-use policies at US airports to control and mitigate residential noise exposure. Section 4.3 examines the impact of lead emissions from general aviation and air taxi.

4.1 Aircraft Noise Certification Stringency

Research presented in this section includes work performed in support of informing the US position on aircraft noise policy at the 9th Meeting of the International Civil Aviation Organization – Committee on Environmental Protection. The work from that information paper was conducted in collaboration with Aleksandra Mozdzanowska, Maryalice Locke, and Ian A. Waitz. Results presented in this section do not necessarily reflect the opinions of the US FAA.

The main objective of this section is to provide a direct assessment of environmental impacts with explicit consideration of uncertainties and tradeoffs and to compare these to estimated implementation costs for an increase in stringency of international aircraft noise certification. The impacts are assessed under three different lenses to cover the range of economic and scientific uncertainties as well as for discount rates ranging from 2-9%. The results show that stringency increases up to -7 EPNdB provide environmental benefits in the domains of noise, air quality, and climate change. More moderate stringency increases (-3 and -5 EPNdB) result in a net societal benefit for all lenses and discount rates considered. A -7 EPNdB policy is cost-beneficial when environmental impacts are discounted at 5% or lower but presents a net cost when recurring costs and environmental benefits are discounted at market discount rates. For the projection scenarios provided by the Federal Aviation Administration (FAA), more severe stringency increases (-9 and -11 EPNdB) force fleet composition changes that result in countervailing environmental trends and net social costs.

4.1.1 Policy Background

The International Civil Aviation Organization (ICAO) published Annex 16: Environmental Protection, Volume I - International Noise Standards in 1971, which set the international standard for aircraft noise. Over the past 40 years, ICAO has increased the stringency of those regulations prior to this study. Despite these increases, noise is still a significant environmental concern, causing an estimated $23.8 billion in capitalized damages to global housing property values (He et al., 2014). International aircraft noise limits were last set in 2001 as part of the ICAO Committee for Aviation Environmental Protection (ICAO-CAEP) CAEP/5 cycle. As part of CAEP/7, a formal review of noise certification limits was initiated.

Subsequently, proposed stringency options were developed for consideration at CAEP/9 in February 2013.

Cost-effectiveness has historically been the dominant approach for environmental policy analysis in ICAO-CAEP (CAEP 2004). However, cost-effectiveness has several weaknesses in that it does not reveal what level of environmental control is economically or socially reasonable, it cannot compare policies with different benefit streams, and it often does not account for co-benefits when they occur in different environmental domains. A cost-benefit analysis of a global aviation environmental stringency was first performed using older models of the tools applied in this work to determine the costs and benefits of an engine NO_X emissions standard as part of the 8th CAEP cycle (Mahashabde et al., 2011).

Other studies have assessed the costs and benefits of aircraft noise policies. Morrison et al. (1999) performed a cost-benefit analysis of the United States Airport Noise and Capacity Act mandatory phase-out of noisier aircraft and find a net $5 billion social cost. However, this analysis of an older noise policy does not explicitly model fleet changes, instead assuming a constant capital stock and a parameterized fleet replacement model. Furthermore, that study considers no environmental co-benefits for air quality and climate change. Lijesen et al. (2010) performed a bottom-up cost-benefit analysis for optimal aircraft noise reductions while considering several policy pathways for one Dutch airport. The work in this section formed the basis of a US Information Paper and was used to inform the discussion of the ninth meeting of the ICAO-CAEP in February 2013.

4.1.2 Methodology

This section provides an overview of the methodology used for the analysis. Policy stringency scenarios and a baseline business-as-usual cases were specified by ICAO-CAEP; for each scenario fleeting projections were made using growth and retirement curves. From these and assumed future demand, operational projections were made. Emissions and noise contours were modeled by the Volpe National

Transportation Systems Center using the Aviation Environmental Design Tool (AEDT) based on these projections. The physical and monetary impacts of these emissions on human health and welfare due to noise pollution, surface air quality degradation, and climate change were modeled using the Aviation environmental Portfolio Management Tool for Impacts (APMT-I) model described in Chapter 3. These monetized environmental changes were then considered against the estimated recurring technology and operating costs of the proposed policy.

Policy Scenarios and Forecasts

Noise certification is based on precision microphone measurements of aircraft noise generation on takeoff and landing. Measurement sites are designated in three locations: 1) approach, 2000 meters before the runway threshold along the runway centerline; 2) sideline, 450 meters from the runway centerline where noise is greatest on takeoff; and 3) takeoff, 6500 meters after break release along the runway center line. Noise is measured in Effective Perceived Noise Level (EPNL or EPNdB), a weighted measurement that accounts for duration and tonal qualities of the noise, and limits are set for each individual measurement site as well as for the cumulative noise across all three sites. ICAO-CAEP considered five potential policy options for increasing noise certification stringency: -3 (Stringency 1), -5 (Stringency 2), -7 (Stringency 3), -9 (Stringency 4), and -11 (Stringency 5) cumulative EPNdB reductions relative to Chapter 4 noise limits as well as a baseline no change case. Each scenario was analyzed until 2036, with an implementation year of 2020 (CAEP, 2010a).

The ICAO Growth and Replacement (G&R) database formed the basis of available aircraft for developing future fleets. The database included current in-production aircraft and project aircraft (the latter being aircraft models that have been designed and tested sufficiently to provide noise and emissions estimates, but have not yet entered full production) (CAEP 2013). Commercial aircraft were split into nine seat class (SC) assignments by seat capacity where SC1 consists of 20-50 seat aircraft and SC9 consists of 601-650 seat aircraft. Business jets were modeled separately, and fleet replacements were modeled as being consistent across seat classes. Passenger and freighter traffic projections were taken from CAEP/8 forecasts and are shown in Figure 4-1.

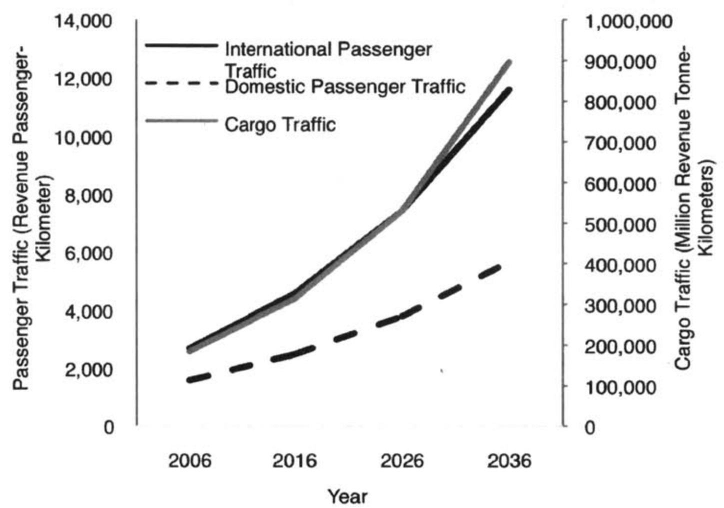


Figure 4-1 Passenger and Freight Projections 2006-2036

Total global business jet operations and fleet size are expected to increase by an average of 4.3% over the 30-year period (CAEP, 2013a). Specific attention was given to conserving projected seat demand provided by the Forecasting and Economic Analysis Support Group (FESG) CAEP/8 forecasts. This assumption required adjusting the number of operations flown when quieter aircraft replaced non-compliant aircraft that had a different capacity.

Noise and Emissions Inputs

Emissions were modeled by AEDT and were divided into landing and takeoff emissions (LTO) for air quality and greenhouse gas (full flight) emissions for climate change (Roof et al., 2007). The AEDT modeling was performed by the FAA and their contractors using inputs and assumptions provided by ICAO/CAEP. AEDT inputs to the APMT-Impacts Air Quality Module include fuel burn, emissions of NO_X, SO_X and non-volatile PM below 3000 feet for the landing and takeoff flight segments. While the relationship between SO_X emissions and fuel burn is well understood, there remains some uncertainty about the emissions index (EI) ratio with the AEDT default EI of 1.16 g/kg-fuel, AEDT guidance for use recommendations of 1.1762 g/kg-fuel and upper-bound estimates of 1.2 g/kg-fuel. Here, 100% conversion of fuel sulfur to SO_2 is conservatively assumed with no emissions as S_{VI}. This assumption leads to an approximately 1% increase in total damages over the typical AEDT guidance for use recommendation.

Stringencies 1-3 all have LTO fuel burn improvements across the lifetime of the policy. From Stringency 3 to Stringency 4, the number of compliant aircraft in higher fleet seat classes becomes significantly reduced, with this subset of aircraft having lower seat capacities than the unconstrained fleet average. Therefore, to maintain FESG seat demands, operations must be increased at these seat classes by upwards of 7% for Stringency 4 and Stringency 5 resulting in total fuel burn increases of up to 1.31% in 2036. All stringency options lead to a reduction in LTO NO_X emissions relative to the baseline.

Inputs for the APMT-Impacts Climate Module include full flight fuel burn, CO_2, and NO_X emissions. Fuel burn and NO_X emissions relative to baseline business-as-usual assumptions are given in Table 4-1. CO_2 emissions scale directly with fuel burn with an EI of 3155g/kg-fuel and are not presented here. AEDT results for full flight emissions are provided for North America, and US emissions have been scaled from these results assuming that US operations account for 93% of North American operations.

Table 4-1 Changes in Full Flight and LTO Fuel Burn and Emissions

Scenario	LTO Fuel Burn and Emissions (% Change from Baseline)				Full Flight Fuel Burn and Emissions (% Change from Baseline)			
	Fuel burn 2026	Fuel burn 2036	NO_X 2026	NO_X 2036	Fuel burn 2026	Fuel burn 2036	NO_X 2026	NO_X 2036
Stringency 1 (-3dB)	0.00	(0.02)	(0.01)	(0.05)	0.01	(0.01)	(0.01)	(0.04)
Stringency 2 (-5dB)	(0.12)	(0.25)	(0.21)	(0.39)	(0.11)	(0.24)	(0.16)	(0.31)
Stringency 3 (-7dB)	(0.06)	(0.13)	(0.46)	(0.88)	(0.17)	(0.27)	(0.40)	(0.72)
Stringency 4 (-9dB)	0.31	0.66	(0.21)	(0.29)	0.05	0.23	0.15	0.37
Stringency 5 (-11dB)	0.66	1.31	(0.61)	(1.16)	(0.15)	(0.12)	(0.63)	(0.94)

Stringency 1 has a fuel burn detriment for full mission flights in 2026, but it provides a minor benefit of less than 0.01% by 2036. The difference between the Baseline and Stringency 1 is always within the AEDT data resolution limits of 0.05%. Stringencies 2 and 3 show greater fuel burn benefit with increasing stringency, while Stringency 4 once again displays the fuel burn detriments from increased operations of some inefficient aircraft in certain seat classes as described above. While Stringency 5 is also impacted by this increase in operations, a relatively fuel inefficient aircraft in seat class 5 also becomes noncompliant. This results in Stringency 5 full flight emissions presenting a fuel burn benefit of 0.12% relative to the baseline in 2036. Stringency 3 remains the most beneficial to fuel burn reduction across the lifetime of the policy with a 0.27% annual reduction by 2036.

Noise contours for 99 US airports expressed in average day-night noise level (DNL) are modeled by AEDT in 5 dB DNL increments from 55dB to 75+ dB. These US airports are a part of 185 AEDT Shell-1 airports worldwide that account for 91% of total global noise exposure (102 of the Shell-1 airports are located in North America). Absolute reductions in area exposure are largest at the 55 dB DNL level and range from 707 km^2 reduction for Stringency 1 to 7021 km^2 reduction for Stringency 5. For this analysis, AEDT also provided estimates of population impacted by noise at each airport at each noise level using US Census population estimates at the Census Block Group level and 200m x 200m resolution.

Operator Costs

Recurring operator costs were calculated using the Aviation environmental Portfolio Management Tool Economics (APMT-Economics) model by the FAA and their contractors using inputs and assumptions provided by ICAO-CAEP, and broken into three functional cost categories: fuel costs, capital costs, and other direct operating costs. All stringency options have been assessed for a policy implementation year of 2020 using the core set of assumptions described in Appendix C of the CAEP/9 economic assessment of noise stringencies paper (ICAO 2013), specifically; fuel prices of $3 per U.S. gallon from 2020 to 2036; no pass through of increased costs to fares in the Stringency cases and hence no seat demand changes between Baseline and Stringency cases; and market-driven (relative cost) market share across each aircraft type. Non-recurring costs and loss of fleet value are not included in the primary policy analysis, as they represent pecuniary externalities (see section 2.1). However, they are included in the cost sensitivity analysis in part because they inform how stakeholders (e.g. manufacturers, aircraft operators) may react to a proposed policy.

Impacts Modeling

Physical and monetary impacts of aviation noise and emissions are modeled using the APMT-Impacts (APMT-I) modules of the aviation environmental tools suite. Detailed descriptions of the APMT-I models are provided in Chapter 3. The health impacts of aviation noise are not considered in this analysis because at the time the analysis was done, the methods for modeling the health impacts had not been developed. The APMT-Impacts Climate v22 and the APMT-Impacts Air Quality RSMv2 were used in this analysis.

Policy evaluation with APMT-I provides information on the environmental benefits and economic costs resulting from the implementation of a policy relative to a baseline scenario. In conveying this information to decision-makers, the uncertainties in the quantified impacts and the key assumptions about inputs and model parameters are provided with the results. The APMT-I noise stringency analysis presented is limited to US-related impacts in order to align with the geographic scope of the APMT-I air quality model and to ensure that the economic costs and environmental benefits are compared in a consistent manner. Noise damages consider only the impacts from changes in housing value and not health costs.

Impacts can be represented in physical or monetary terms, with the computation of monetary metrics introducing additional influential parameters relative to the evaluation of physical effects. In order to extract meaningful insights about the possible costs and benefits of a policy, it is expedient for the analysis options to be synthesized into a set of pre-defined combinations of inputs and assumptions that are developed through an iterative process with the policymakers. These combinations, which describe a particular point of view or perspective, are designated as lenses. For this analysis, low range (low), midrange (mid) and high range (high) lenses are adopted to explore the range of potential environmental impacts. An illustrative lens is also developed that accounts for potential undercounting of air quality impacts from changes in future background concentrations, model resolution, and cruise emissions as described in Section 3.2.

The only parameter not grouped in the lens assumptions was the discount rate. This was done so that the full range of discount rates could be applied to each result regardless of the lens selected for analysis. Further, while the probabilistic parameters modeled in the APMT-Impacts modules reflect the range of scientific, economic, and modeling uncertainty, choice of discount rate can be seen as a choice of policy-maker preference. A high discount rate emphasizes near term impacts where a low discount rate more fully considers longer-term costs and benefits. For all lenses, zero population growth and income growth were assumed, in accordance with the CAEP practices for policy analysis.

4.1.3 Results and Discussion

The goal of the policy analysis presented in this section is to compare the environmental benefits and economic costs of the range of noise stringency options relative to the baseline no stringency case. The impacts analysis is conducted using Monte Carlo methods, the results represent the mean of several thousand Monte Carlo runs, and error bars in figures represent the 10-90th percentiles of the runs unless otherwise noted.

Figure 4-2 shows the impacts of the five stringencies on physical metrics across the three environmental domains considered in the analysis. Noise impacts are measured as cumulative people-years impacted at the 55dB level and higher, air quality is measured in total premature mortalities, and climate is measured in integrated global mean surface temperature change out to a 100-year time horizon. The physical metrics show that all policies have a larger effect on noise impacts than other environmental effects, and that these impacts increase disproportionately with increasing stringency. While the noise benefits from Stringency 3 to Stringency 4 more than double, there is a reverse in the emissions trend for both air quality and climate.

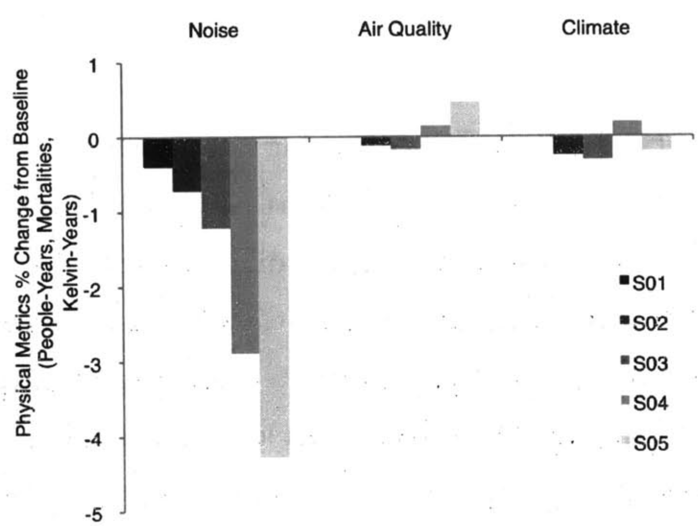


Figure 4-2 Physical impacts percent change from the Baseline

When comparing physical damages, the policies have a larger impact on noise than on air quality or climate. This is sensible as the policy's primary aim is to change noise performance. However, it is important to examine impacts on a monetized scale since the societal cost of an impact can be non-linear with respect to physical impacts and because different environmental impacts have different baseline expected costs. First, results for environmental benefits at the midrange lens are presented with costs and benefits both monetized under a 3 percent discount rate as presented in Table 4-2.

Table 4-2 Cost-Benefit Analysis, Midrange Lens, 3% Discount Rate

Stringency	Noise (2006 US $Billion)	Air Quality (2006 US $Billion)	Climate (2006 US $Billion)	Recurring Cost (2006 US $Billion)	Net Cost Benefit (2006 US $Billion)
	mean (10-90%)	mean (10-90%)	mean (10-90%)	mean	mean (10-90%)
Stringency 1 (-3 dB)	-0.11 (-0.14 - -0.08)	0.00 (-0.01 - 0.00)	0.00 (-0.01 - 0.00)	-0.27	-0.39 (-0.42 - 0.35)
Stringency 2 (-5 dB)	-0.19 (-0.25 - -0.14)	-0.05 (-0.07 - -0.02)	-0.34 (-0.57 - -0.14)	-1.35	-1.93 (-2.24 - -1.65)
Stringency 3 (-7 dB)	-0.33 (-0.42 - -0.23)	-0.07 (-0.11 - -0.03)	-0.44 (-0.73 - -0.19)	2.43	1.60 (1.17 - 1.97)
Stringency 4 (-9 dB)	-0.79 (-1.02 - -0.56)	0.07 (0.01 - 0.13)	0.27 (0.12 - 0.45)	9.72	9.26 (8.82 - 9.74)
Stringency 5 (-11 dB)	-1.21 (-1.56 - -0.88)	0.18 (0.05 - 0.36)	-0.30 (0.50 - 0.13)	15.66	14.3 (13.6- 15.0)

Despite having a much smaller percentage change in physical metrics relative to noise, climate impacts have the largest monetized benefit for Stringencies 2 and 3 of $0.34 and $0.44 Billion USD (in 2006 dollars) respectively due to the nonlinear relationship between temperature change and societal costs. The fuel burn increase from increased operations in Stringency 4 results in an air quality dis-benefit of $0.07 Billion and a climate dis-benefit of $0.27 Billion, which are both offset by the noise benefit. Stringency 5, with dominating noise benefits, has the largest total environmental impact with a mean benefit of $1.3 Billion USD followed by

Stringency 3. However, under this lens and discount rate only Stringencies 1 and 2 provide net societal benefits, with Stringencies 3, 4, and 5 dominated by high recurring costs.

Under both the physical and monetary metrics, the environmental impact of the policy is noticeably small; the policies induce a <1% change in welfare effects from emissions and between 0% and 4.5% change in effects from noise. There are two factors that potentially contribute to the small changes. The first is that although the policies are modeled for 30 years, but only 15 of these years occur after implementation. Thus, if aircraft service lifetimes are on the order of 30 years, none of the fleet will have turned over for the first half of the analysis, and less than half the fleet will have turned over for the second half of the analysis. While the effects of emissions beyond the current analysis timescale will be highly discounted under most discount rates, by only considering emissions through 2036, the analysis may be failing to consider additional environmental benefits. The role of timescales in policy analysis is the focus of Chapter 5.

Second, the proposed regulations do not require phase-out of currently operating non-compliant aircraft and only apply to new aircraft certification. Because the noise certification stringency has been increased every ten to twenty years, a cumulative noise level tightening of 2-6% each time, aircraft designers and manufacturers would have reason to expect an increase in noise stringency of up to -7 EPNdB (Stringencies 1, 2 and 3) at CAEP/9. Thus, these small and moderate increases may have little impact on fleet composition, noise, and emissions with very few of the current fleet being non-compliant (let alone future aircraft).

4.1.4 Lens Analysis

The sensitivity analysis presented here focuses on variability in results depending on selection of inputs and model parameters explored using the lens concept. The sensitivity to lenses for a sample stringency (Stringency 3) minus baseline at a 3% discount rate is shown in Figure 4-3. Monetized benefits from climate change are highly sensitive to lens assumptions. This sensitivity is explained by the asymmetric distribution on climate sensitivity and the squared relationship between temperature change and damages. Thus, while a midrange of scientific and economic expectation indicates moderate climate benefits from Stringency 3, the high lens shows that those benefits could be significantly greater.

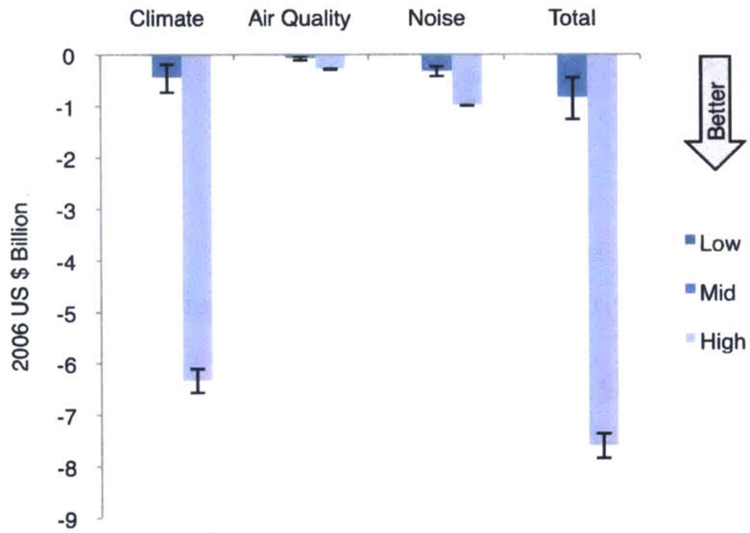


Figure 4-3 Sample stringency (Stringency 3) environmental impact lens sensitivity.

Figure 4-4 shows the sensitivity of cost-benefit results to lens assumptions. Stringency 2 remains the most cost-beneficial policy under all lenses with net benefits ranging from about $1 billion 2006 USD to about $7.5 billion USD. Stringency 3 is dominated by costs in the low and mid lenses, but becomes significantly cost-beneficial under the high lens. Stringencies 4 and 5 maintain a net detriment across all lenses, but their relative order changes under the high lens where large climate benefits make Stringency 5 less detrimental than Stringency 4.

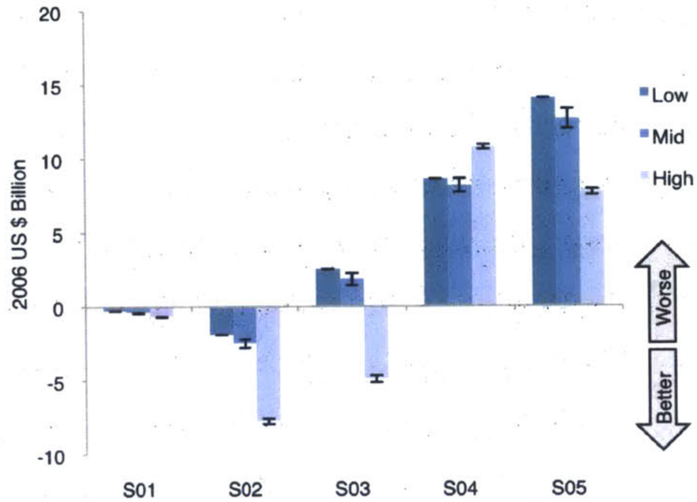


Figure 4-4 Stringency - Baseline cost benefit lens sensitivity.

Because APMT-I Air Quality contains several known omissions as described in Section 3.2, an additional illustrative lens is used to examine the sensitivity of the results to those assumptions. Specifically, this lens considers the impacts of modeling resolution, cruise emissions, and changing background on air quality and the impact of modeling domain on noise damages. Other assumptions, such as changes in population over the policy lifetime or the effect of noise and emissions on non-market impacts such as public space enjoyment and visibility are not examined in this lens.

Figure 4-5 shows the sensitivity of the cost benefit analysis to first order approximations to account for these current modeling limitations. As shown in the figure, the cost benefit results are not sensitive to these assumptions. For example, for Stringency 3 at a 3% discount rate the illustrative lens has an air quality benefit that is 2.9 times greater than that of the midrange lens. However, air quality accounts for less than 10% of total environmental impacts and less than 3% of the recurring costs. While accounting for these limitations may have compounding effects under high lens or lower discount rates, this sensitivity analysis shows that the net cost benefit result for CAEP/9 Noise Stringency is more sensitive to choice of discount rate or choice of environmental lens than it is to these modeling assumptions.

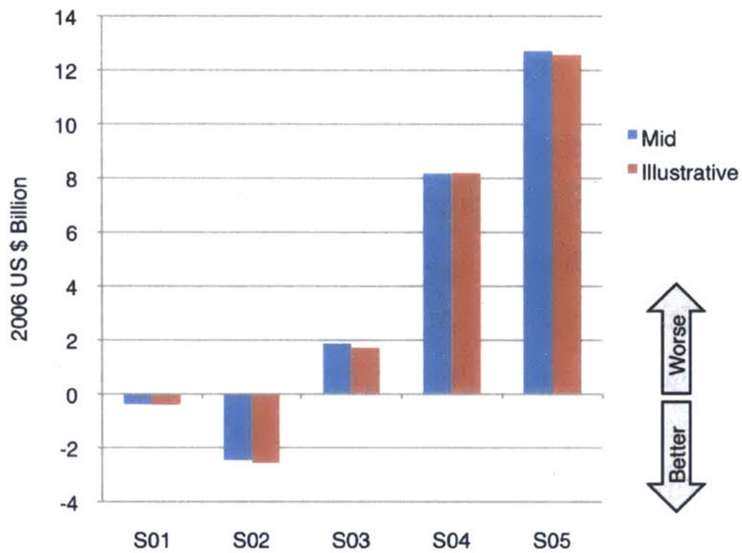


Figure 4-5 Illustrative lens sensitivity

4.1.5 Discount Rate Sensitivity

The discount rate is the method by which future damages are monetized in comparison to current year costs. The discount rate accounts for the positive rate of time preference and the change in marginal utility of consumption over time. Societal impatience, uncertainty in future outcomes, and opportunity costs all influence the pure rate of time preference, whereas changes in the average rate of consumption (such as an expectation of greater wealth in the future) and concerns of interpersonal inequity influence the marginal utility of consumption over time. Proponents of exponential discounting, applying a constant discount rate over time, cite a Kaldor-Hicks partial Pareto improvement criterion: if aggregate welfare is higher under one policy, then a compensation mechanism can be implemented to transfer benefits between parties, including parties in different time horizons (Goulder and Stavins 2002). Under this assumption, it would appropriate to assess a policy by applying a market-based discount rate across all costs and benefits. Opponents of market discount rates cite intergenerational inequity due to the long lifetimes of environmental damages, high cost or infeasibility of benefit transfer, uncertainty, and the inability to account for irreversible events such as species extinction as some of the justifications for non-market rates (Ackerman and Heinzerling 2002). A review of discounting is provided in Section 5.3.

The costs and benefits of the noise stringency policy are assessed for a range of discount rates in keeping with US and international best practice. Figure 4-6 illustrates the effect of the discount rate across the environmental impacts for a sample stringency (Stringency 3) analyzed for a midrange lens. Climate, which has the longest-lived impacts, is most affected by discount rate with monetized impacts decreasing below those of noise at discount rates of 5% and higher. For Stringency 3 environmental benefits range from $1.2 billion USD (2006) at 2% to just under $200 million USD (2006) at 9%.

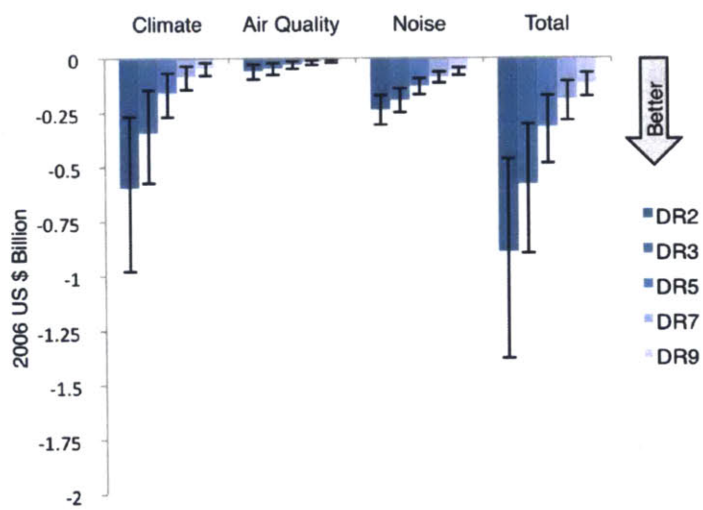


Figure 4-6 Effect of discount rate on environmental benefits for a sample stringency

The cost and benefit trends across a range of discount rates are examined where recurring costs and environmental benefits are computed with the same discounting assumptions. Figure 4-7a shows that choice of discount rate does not impact the relative ranking of stringency options, and that Stringencies 1 and 2 remain cost beneficial at all three discount rates, while Stringencies 3,4, and 5 all have net societal costs. However, choice of discount rate can have a large impact on the net cost as seen in the nearly $7 billion USD difference in net costs between a 3 and 7% discount rate on Stringency 5.

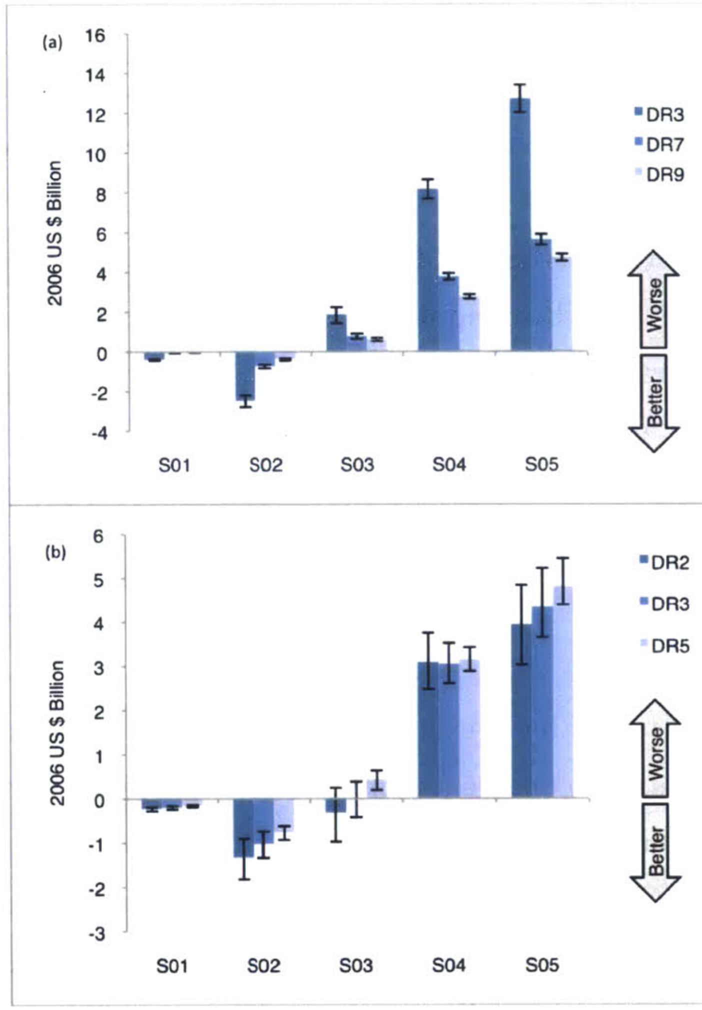


Figure 4-7 Cost-Benefit sensitivity to discount rate (a) with a single discount rate (b) with operational costs discounted at 9% and environmental impacts discounted at a lower rate

Because of the long lifetime and potential irreversibility of environmental impacts, decisions in the present have the possibility of affecting future generations with no opportunity for benefit transfer. In the United States, the OMB recommends discount rates of between 2.5 and 7% for regulatory analysis but allows for the use of a lower discount rate for intergenerational impacts (US OMB, 2003). In past ICAO-CAEP analyses of costs of environmental stringencies, discount rates of 9% and 13% have been used to discount future recurring costs (CAEP, 2001), while environmental benefits have been estimated using discount rates of 2-5% (CAEP, 2010b). The cost-benefit analysis of proposed stringencies is examined where the environmental benefits are discounted under the 2, 3, and 5% discount rates, while costs are discounted under a 9% discount rate. Results are shown in Figure 4-7b. With costs computed at a 9% discount rate, Stringency 3 becomes cost beneficial at 2 and 3% discount rates on environmental benefits. Stringency 4 and Stringency 5 maintain a net societal dis-benefit for all discount rates. While non-aligned discount rates have been applied in other US regulatory analyses, their use is controversial. Several economists recommend using declining discount rates as an alternative (Arrow et al., 2014), while others note that reframing the argument over future resource prices would lead to a similar outcome while eliminating the apparent temporal inconsistency (Horowitz, 1996).

4.1.6 Conclusions

This section reports on an environmental impact assessment and quantification of modeling uncertainties to enable a more comprehensive evaluation of CAEP/9 noise stringency options. The APMT-Impacts modules were employed to conduct the cost benefit analysis (CBA). The environmental benefits and economic costs associated with the CAEP/9 noise stringency options were analyzed for the US domain from 2006 to 2036 with policy implementation in 2020. Operating costs were taken as economic costs and were calculated by APMT-Economics. All stringency options lead to a noise benefit ranging from 0.5 to 5% improvement in the cumulative people-years noise exposure over the next 30 years.

An increase in aircraft noise certification stringency of up to -7 EPNdB applied in 2020 can provide a net benefit to society. The inclusion of tradeoffs across the environmental domains of air quality, climate, and noise are important and can impact the utility of a policy option. These policy options promote early adoption of new more efficient technologies that provide significant environmental and cost co-benefits through reduced fleet wide fuel burn and reduced emissions. However, policies that have a significant impact on fleet composition can introduce countervailing environmental trends. For instance, the modeling approach chosen here indicates that the most stringent policies lead to a decrease in average seat capacity of certain classes of aircraft. This decrease requires an increase in overall flight operations to meet underlying demand, causing climate and air quality dis-benefits.

The estimated policy benefit or cost is sensitive to some analysis parameters, such as choice of environmental discount rate and environmental lens. Under discount rates of 7% or greater and under assumptions that assume the smallest impact on human health and welfare from environmental effects, a -7 EPNdB stringency increase appears to have a net dis-benefit to society. However, when a low discount rate is used on environmental impacts, or a high damage lens is applied, the same stringency can appear cost-beneficial.

The known sensitivity of environmental benefits to choice of discount rate has been cited as a major shortfall of environmental cost benefit analysis (Pindyck 2013). Normative and ethical critiques of various discounting schema have also made it such that any reasonable choice of discount rate can be equally attacked and defended (Frederick, Lowenstein, and Donoghue 2002). As shown in Figure 4-6, the expected climate benefits differ by an order of magnitude between the highest and lowest discount rates considered by policymakers, and opponents of cost benefit analysis have stated that this range makes monetizing climate damages for policy analysis essentially meaningless (Pindyck 2013). However, as shown in this analysis, while the total costs and benefits are sensitive to discount rate, the result of interest (whether the policy provides a net positive cost-benefit and the rank order of the policies considered) is rarely sensitive to discount rate, even when costs and benefits are discounted at different rates. Thus, while choice of discount rate may be contentious, it does not necessarily limit cost benefit analysis as a useful tool in support of policy decision-making. The discount rate sensitivity analysis further illustrates how and when discount rate preference impacts the cost-benefit result. Additional discussion of the discount rate is provided in Section 5.3.

The impact of noise exposure on health was not considered in this analysis as the methodology for considering these impacts had not been developed at the time of the study and the acceptance of health damages from noise had not been developed in the international aviation environmental policy community. Recent estimates of noise-related health impacts indicate that cardiovascular and cerebrovascular impacts contribute an addition 10% - 60% in damages depending on the valuation technique and the health end points considered. Secondary health effects, such as incidents of dementia from increased prevalence of hypertension, could increase damage estimates further.

At the 9th meeting of the International Civil Aviation Organization Committee on Aviation Environmental Protection (ICAO-CAEP), CAEP agreed to recommend an amendment to Annex 16, Volume 1 setting new standards for noise certification, based in part on the analysis performed in this section of the thesis. The amendment introduces increased stringency of a cumulative drop of 7 EPNdB relative to the Chapter 4 standards across all certification points. This policy is closest to Scenario 3 analyzed in this section, the most stringent policy that results in a cost-beneficial result under a subset of assumptions.

4.2 Aviation Noise Land-Use Policies

Research presented in this section was conducted in collaboration with Robert Malina, Steven R. H. Barrett, and Ian A. Waitz.

Aircraft noise affects human health and welfare. One method US airports use to mitigate the impact of noise on nearby residents is through sound insulation and residential land acquisition projects. The average cost of sound insulation projects is \$15,600 per person affected while that of land acquisition is \$48,900 per person affected. The welfare benefits of these measures are estimated using a meta-study of differential housing values to calculate willingness-to-pay for abatement and by using the direct and indirect costs of cardiovascular and cerebrovascular health impacts associated with noise exposure. Only in 15% of projects do the benefits to residents from willingness-to-pay for reduction and reduced risk of mortality and morbidity exceed the costs of sound insulation for residences exposed to 65 dB Day Night Level (DNL) of noise. At an annual income level of \$40,000 or less, benefits from land acquisition policies never exceed project costs. Benefits from reduced risk of mortality and morbidity are 39-41% and 61-64% of the benefits from changes in housing value when valued using Value of Life Years Lost or Value of Statistical Life respectively. Estimates suggest that noise insulation projects are more cost-effective than fleet wide mandatory aircraft retirement.

4.2.1 Policy Background

Several policy approaches and methods are available for controlling the impact of aviation noise. In addition to the global certification stringency promulgated by the ICAO-CAEP discussed in Section 4.1, command-and-control source-based policies include mandatory phase out of noisier aircraft at the national and international level and per-movement limits set at the airport level (Girvin 2009). Globally, various governments and airports have adopted other noise-abatement policies and procedures including quotas, curfews, direct noise charges, preferential runway treatment, and land-use management. Two land-use management policies that have been adopted at a number of US airports are noise insulation and land acquisition.

Previous research on aviation noise policies has not specifically examined the role of land-use management policies. Janic (1999) and Girvin (2009) present a qualitative assessment and comparison of noise mitigation policies including acquisition and insulation, but do not consider the direct assessment of costs or benefits from these land-use change policies. Other studies have examined the costs and/or benefits of alternative noise policy instruments including mandatory retirement of noisier aircraft (Morrison et al. 1999), airport per-movement and cumulative noise constraints (Brueckner and Girvin 2008), and noise taxes and fees (Morrell and Lu 2000; Brueckner and Girvin 2008). Mahashabde et al. (2011) examine the co-benefits and tradeoffs to noise of an emissions-based policy and its impact on net policy costs and benefits. However, a rigorous assessment of land-use policies and their impact on social welfare has not been accomplished. The contribution of this section is to quantitatively assess the costs and benefits of land-use management noise mitigation, specifically housing insulation and property acquisition, as it has been applied in practice at US airports, and to quantitatively compare these costs and benefits to other policy instruments.

Land-use management strategies are investigated at 16 US airports. FAA Airport Improvement Program (AIP) Grant Histories provide the costs of these programs, which are then assessed as a function of the number of people impacted. A Willingness-to-Pay for noise abatement formulation based on a meta-study of hedonic pricing surveys is used to compute the benefits of these improvements from changes in housing values as presented in Section 3.1.1. Utilizing exposure-response relationships from the literature, the costs of aviation noise-induced hypertension and myocardial infarction and stroke are also calculated as described in Section 3.1.2. These health costs are used to estimate a benchmark bounds of the social welfare benefits of land-use policies. Finally, the costs and benefits of land-use management through traditional policy perspectives such as Cost-Benefit Assessment and Cost-Effectiveness are examined to compare the results to other policy instruments.

4.2.2 Land-Use Policy Costs

Under Part 150 of the Federal Aviation Regulations, participating airports are eligible for noise compatibility program grants through Airport Improvement Program (AIP) grants and moneys from Passenger Facility Charges (PFC). There are 11 airports for which grant reporting provides both the money provided by AIP and the number of people impacted by the land-use change from AIP grant reporting between 2000-2012. For five airports, only the number of households impacted is provided, household impact is converted to a per person impact by assuming a US average of 2.6 people per household. Of these 16 projects, ten implemented noise insulation and soundproofing while six implemented primary land acquisition. Where a project applied for grants over several years, the total cost of the project per person is used.

Noise compatibility projects are eligible for 80% federal share of costs at medium and large hub airports and 90-95% federal share of costs at other airports (FAA 2009). It is assumed that the total cost of the airport project reflected full allowable cost sharing from federal moneys. Costs for all projects are converted to 2010 dollars using the Consumer Price Index (CPI). Costs per person for the 16 airport noise projects considered are shown in Figure 4-8.

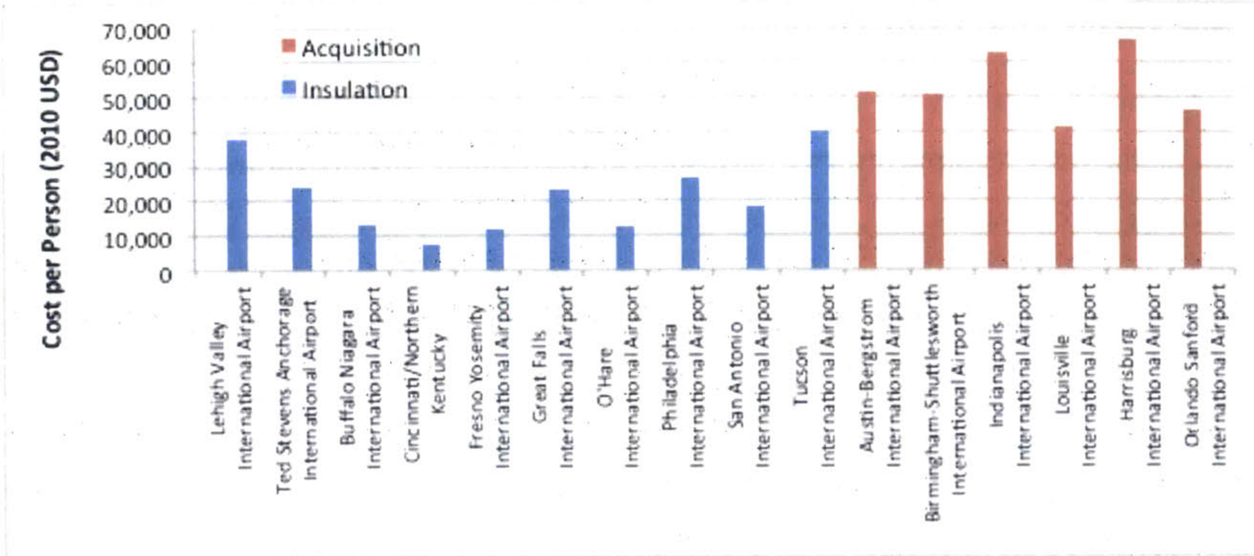


Figure 4-8 Airport specific aviation land-use policy cost per person affected

The average cost for home insulation across all noise projects considered is $15,600 per person affected, and the average cost for property acquisition across all noise projects considered is $48,900 per person affected (2010 USD). For comparison, a GAO analysis of FAA Noise Grant beneficiary goals found that average yearly AIP noise grant expenditures for all noise grants rewarded not broken down by expenditure type or project location ranged from $14,050 to $22,219 per beneficiary, and that 71% of projects occurred at primary commercial airports (GAO 2012). This is equivalent to average total project costs of $16,600 to $26,300 per beneficiary.

4.2.3 Environmental Benefits

The environmental benefits of removing people from noise contours are calculating using both the Willingness-to-Pay and health impacts described in Section 3.1. Because property acquisition effectively removes the noise-afflicted population from all aviation noise, the expected noise level after implementation is assumed to be the background ambient noise level. Noise insulation does not fully mitigate the impacts of aircraft noise on property. Even if the insulation were to fully soundproof the residence, the affected persons would still be limited as to when they could open their windows or to how they could enjoy outdoor space on their property. Van Praag and Baarsma (2005) find that the presence of noise insulation improves residential welfare by 2/3 the amount of eliminating the noise burden and note similar results from Feitelson et al. (1996). This assumption is adopted for this analysis.

Combining direct health costs with Willingness-to-Pay values from hedonic pricing studies may contribute double-counting of those health impacts that residents bear, recognize, and attribute to aviation noise. Not all health costs are borne by the afflicted person, and therefore those costs may not be accounted for by the noise hedonic. Cropper and Krupnick (1990) find that only 23% of total hypertension costs are borne by affected individuals and their families. Furthermore, many residents may not fully understand the potential health impacts of noise exposure. Between 2007 and 2008, 20% of hypertensive adults were unaware of their health status let alone had the ability to attribute their disease to genetic or environmental causes (Egan et al. 2010). Erikkson et al. (2009) find that the relative risk of HYT from noise exposure is strongest with those residents who report annoyance to aircraft noise, suggesting that some of the health costs may be captured by the hedonic. Ecoplan (2011) assume willingness-to-pay for abatement accounts primarily for noise annoyance and sleep awakening impacts while health impacts from cardiovascular endpoints can be considered in addition to these willingness-to-pay values. This approach is applied here, but the risk for double counting exists, and the actual combined costs of annoyance and health impacts would be bounded by the sum of the health and willingness to pay costs and the willingness-to-pay cost alone.

Monte Carlo techniques consistent with Mahashabde et al. (2011) are applied to calculate the uncertainty and variability of the welfare benefits associated with noise reductions. To account for the variability of noise land-use policy costs, distributions of sound insulation and land acquisition per person costs are fit to the noise projects considered in Section 4.2.2. He et al. (2014) analyze the uncertainty in the willingness-to-pay model in detail. Normal distributions are applied to the model coefficients based on the result of the He at al. (2014) bootstrapping procedure. In addition, a uniform distribution is used for the background noise level of 50-55 dB DNL (Navrud 2002).

For health impacts, uniform distribution is applied to the relative risk of HYT based on the 5th-95th percentile odds ratios from an aircraft noise meta-study that ranges from no increased risk to an odds ratio of 1.28 (Babisch and Van Kamp 2009). Babisch and Van Kamp (2009) caveat that the relationship they derive between noise and HYT is only a "best guess", and that more studies are needed to establish a single generalized exposure-response relationship. The impact of the VSL is examined probabilistically using a Weibull distribution in accordance with EPA recommendations (EPA 2006) and the VOLY is assumed to be a uniform distribution from 50%-150% of the mean based on the variation seen in individual studies of the value of a life year (e.g. C.P. Lee et al. 2009). The relative risk of stroke is taken as a normal distribution fit to the 5th-95th percentile relative risks from Hansell et al. (2013).

4.2.4 Results

First, the costs and benefits of the land use policies are calculated considering only Willingness-to-Pay benefits from the He et al. (2014) model based on housing hedonics. Because the Willingness-to-Pay relationship with noise is a function of metropolitan statistical area income level, results are presented for a range of per-capita incomes. Figure 4-9 shows the bulk costs of noise land-use policies compared to potential willingness-to-pay benefits for a range of effective dBs avoided for three Metropolitan Statistical Area average per person income levels: \$20,000 (top) \$40,000 (middle), and \$60,000 (bottom). At an income level of \$40,000, benefits range from \$0 per person at 0 dB removed to \$19,000 at 35 dB removed. 5th and 95th percentiles of the noise damages avoided are given by the dashed lines. At 28 dB DNL effectively avoided in areas with average income levels of \$40,000 a year or at 20 dB DNL effectively avoided in areas with average income levels of \$60,000 a year, the cost of an insulation project is on average covered by the benefits to the residents affected. For income levels up to $40,000, welfare benefits from Willingness-to-Pay never exceed the cost of even the cheapest land acquisition projects for the range of dB levels considered.

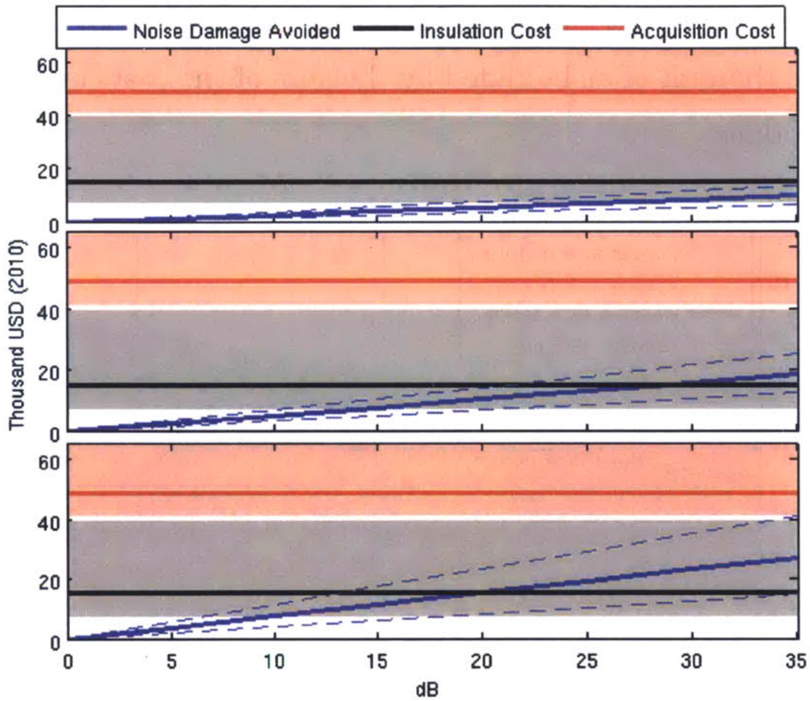


Figure 4-9 Per person housing value benefits of noise reduction compared to policy costs at MSA average income levels of $20,000 (top), $40,000 (middle), and $60,000 (bottom) Dotted lines represent the 5th-95th percentile ranges of 3000 MC runs for the damages avoided. Shaded regions represent the total variability in per person cost across all land-use projects considered.

Figure 4-10 shows the policy cost-benefit differential considering both health and housing benefits when land use policies are implemented to residences affected by 50 dB DNL through 80 dB DNL in a MSA with an average income level of $40,000. Health impacts are calculated using a Cost of Illness Approach with mortalities valued using a Value of Statistical Life as suggested by the US EPA and described in Section 3.1.2. The typical noise levels at which each land-use policy is implemented are delineated in blue. At 65 DNL dB, a noise insulation project is expected to cost $7,000 per person more than benefits accrued to the affected residents through the combined changes in property value and health impacts. By 75 DNL dB, however, the cost of the housing insulation is equivalent to the welfare benefits to the impacted parties. For land acquisition policies, the welfare benefits accrued by affected parties are valued at $30,000 less than the cost to the airport to purchase that land. Because the combination of the Willingness-to-Pay and health costs may double-count some impacts, the actual welfare benefit from these impact pathways can be thought of as bounded by the sum of the costs and the Willingness-to-Pay measure alone.

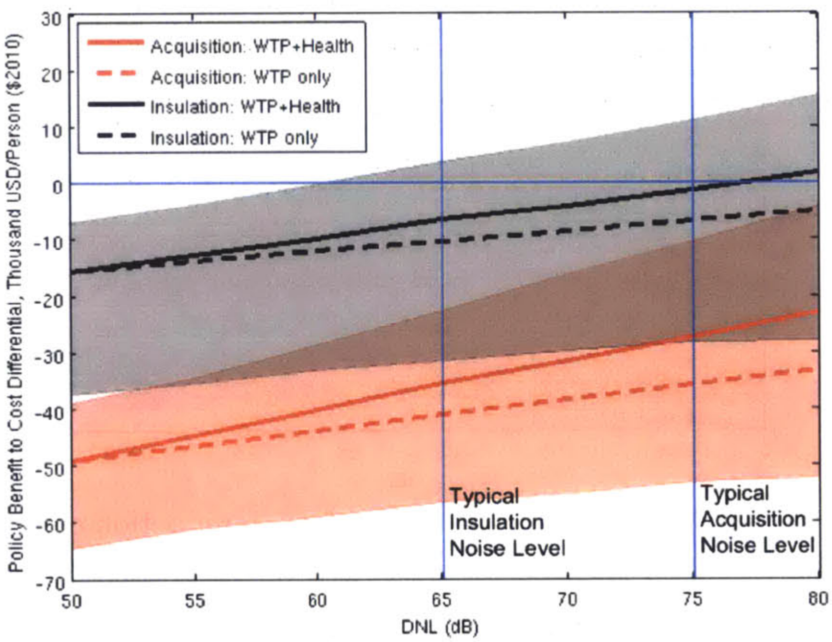


Figure 4-10 Policy impact differential for a range of DNL at a MSA average income level of $40,000 (2010 $). Shaded regions denote 5th-95th percentile ranges of the sensitivity analysis for combined impacts.

For the 65-80 dB DNL range, health impacts amount to 61 - 64% of the housing impact using the VSL approach and 39 - 41% of the housing impact using the VOLY approach. For comparison, a European study found that VOLY- monetized ischaemic health impacts from noise (not including stroke) amount to 10% of the willingness-to-pay costs for the case of road noise (ECOPLAN and INFRAS 2008), while the health impacts not including stroke account for 14 – 16% of costs using the VOLY approach.

At the 95th percentile, noise insulation program benefits from willingness-to-pay and health exceed the program costs for all noise levels above 60 dB DNL.

However, at the 5th percentile, costs exceed benefits by $28,000 per person even at the noisiest level. The spread in the results is driven by the variability in insulation project costs. The costs of residential land acquisition programs never exceed the benefits to residents measured by willingness-to-pay and health impacts for an income level of $40,000 for the range of dB levels considered. The breakdown of policy costs and environmental benefits are shown in Figure 4-11 for three dB and income level combinations. While the average policy costs often outweigh the human health and welfare benefits, they are of the same order of magnitude. Willingness-to-pay for abatement, as related to the housing value hedonic, makes of the largest portion of benefits, followed by stroke and MI. The uncertainty in the health impacts relative risks lead to uncertainty ranges in benefits that range from $0 per person (no relative risk) to impacts that approach the magnitude of willingness-to-pay benefits.

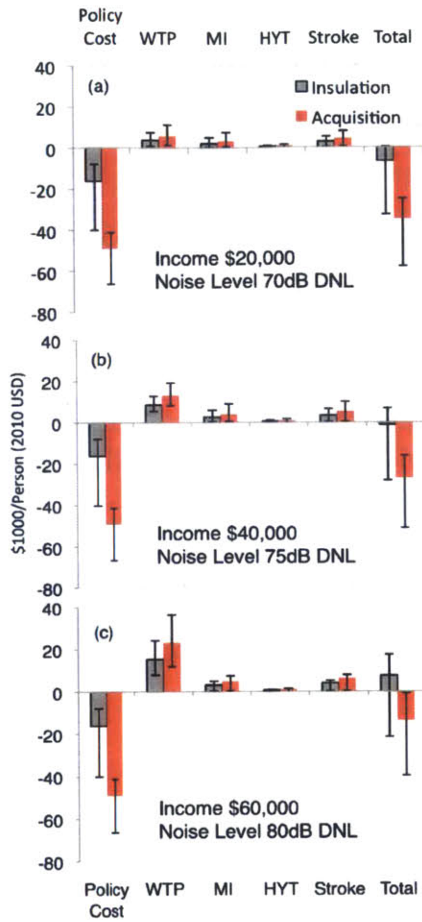


Figure 4-11 Breakdown of policy costs and noise benefits per person in $1000 (2006 USD) for three representative cases: (a) city-level per capita income of $20,000, 70 dB DNL, (b) city-level per capita income of $40,000, 75 dB DNL, (c) city-level per capita income of $60,000, 80 dB DNL.

4.2.5 Discussion and Conclusions

The results in Section 4.2.4 show that housing noise-insulation project welfare benefits exceed costs when implemented at noise levels above 75 dB DNL in metropolitan statistical areas (MSA) with average annual per person income levels of $40,000 or greater. From the study presented in Section 4.1, 2500 people were exposed to noise levels above 75 dB DNL in 2006 as measured at 99 US airports. Further, they show that only the 5th percentile of Willingness-to-Pay benefit estimates exceed policy costs of land acquisition at the highest MSA income levels and dB levels when not considering health costs. However, these results should not be interpreted as indicating that these policies are entirely inappropriate at lower income and noise levels. Aviation noise land-use policies provide other ancillary benefits not accounted for in this analysis. Improved airport-community relationships, a reduction of time and resources spent fielding and addressing noise complaints, improved flexibility in airport expansion, and improved flexibility in operational constraints are all additional potential benefits of effective noise land-use policies. Land use acquisition policies in particular have the added benefit that the acquired land may be used by the airport authority or can be rezoned for a more appropriate use given the noise environment. The environmental welfare cost-benefit results here are only one tool in examining policy appropriateness and must be used in appropriate context.

At noise levels of 75 dB DNL and higher, noise is likely to be the most important environmental community impact. When comparing alternate noise policies, it is helpful to consider all of the co-benefits, and the cost-effectiveness of reducing the environmental burden of concern. Because land acquisition may be the only way to entirely remove an effected population from the noise burden at high noise levels (>75 dB DNL), it may be the only appropriate policy solution.

Replacing a portion of the fleet with quieter aircraft is another effective strategy for reducing the community noise burden close to an airport. One such strategy for promoting the adoption of quieter aircraft is the forced retirement of aircraft that exceed a certain limit on take-off and landing. Morrison et al. (1999) investigated the costs of the accelerated mandatory phase-out of Stage II aircraft at US airports. They estimate that the phase out cost $10B USD (1995 $), equivalent to $14.3B USD (2010 $) and that the policy resulted in a 5 dB DNL noise reduction for 2,001,000 people previously exposed to >65 dB DNL. By using a valuation of housing prices, which is similar to the technique in this analysis, Morrison et al. 1999 estimate the monetary benefit of this reduction at $4.9B USD (1995 $), equivalent to $7B USD (2010 $). Using the valuation method (Willingness-to-Pay only) described in Section 3.1.1 and assuming an average income level of $40,000, the benefits of the policy would be $5.5B USD (2010 $), resulting in a net policy cost of $8.8B USD (2010 $). Thus, the policy has a net cost-effectiveness of $880 per person-dB using the methodology described above. Alternatively, a noise insulation program at the 65 dB DNL level assuming the same income level has a net average cost-effectiveness of $534 per person-dB[5]. However, accelerating fleet-retirement may have co-benefits to air quality, climate change, and energy efficiency not accounted for in the Morrison et al. (1999) analysis (Lee, 2010).

Noise taxes and landing fees can be levied to control aviation noise proliferation. Morrell and Lu (2000) provide a detailed summary of how landing fees and taxes are applied at various airports around the world. In theory, noise fees can be charged at a socially optimal rate where the marginal welfare benefit from the induced noise reduction is equal to the cost of the marginal cost of that reduction. Morrison et al. (1999) find that net US welfare benefits from an optimal taxation scheme are small and on the order of $0.28 Billion (2010 USD). Morrison et al. (1999) note that despite this scheme being economically efficient, the relative magnitude of the benefit transfer to homeowners may make such a policy politically unattractive.

[5] Morrison et al. (1999) estimate that housing insulation costs range from $25,000-$52,000 (1995 $) per house based on expert elicitation. This result is equivalent to insulation costs of $13,750-$28,600 (2010 $) per person. The projects in this study have costs ranging from $7,800 - $39,400 per person with a person-weighted average of $15,600 per person.

4.3 General Aviation Lead

Research presented in this section was conducted in collaboration with Akshay Ashok, Amanda Giang, Noelle Selin, and Steven R. H. Barrett.

While leaded fuels for automobiles were phased-out of use in the United States by 1996, lead continues to be used as an anti-knock additive for piston-driven aircraft. Mean and median estimates of annual damages attributable to lifetime lost earnings are $1.06 and $0.60 billion respectively. Economy-wide impacts of IQ-deficits on productivity and labor increase expected damages by 54%. Damages are highly sensitive to background lead concentrations; as emissions decrease from other sources, the damages attributable to aviation are expected to increase holding aviation emissions constant. The monetary impact of General Aviation Lead emissions on the environment is the same order of magnitude as noise, climate change, and air quality degradation from all commercial operations. Command-and-control regulation of leaded fuels, leaded fuel taxes, and cap-and-trade legislation may help limit the environmental damages from aviation lead emissions, but without a drop-in unleaded fuel replacement, policies may introduce some economic inefficiencies.

4.3.1 Policy Background

The potential dangers of lead exposure have been theorized since at least the second century BC, but lead has remained a consistent health threat since that time (Needleman 2004). Lead was used as a wine-additive in ancient Rome and saw continued use as a wine sweetener and preservative through the sixteenth century. In 1696, Eberhard Gockel, a German physicist, linked deadly widespread disease outbreaks known as colica Pictonum to lead-laced wine, and the practice of correcting wine with lead was ended (Eisenger 1982). Understanding of lead poisoning (plumbism) in miners and metalworkers followed, and the cause of acute childhood lead poisoning was established between 1904 and 1914 (Needleman 2004). However, lead exposure risks in the United States increased in the 20th century. As understanding of the long-lived effects of subclinical lead exposure improved, lead was slowly removed from most products that could lead to childhood exposure. Lead was banned from residential housing paint in the US in 1978 (despite having been banned elsewhere as early as 1920), from plumbing in 1986, from solder in food cans in 1995, and from automobile gasoline by 1996 (Kessler 2013). Through this time, the US Environmental Protection Agency (EPA) has increased the stringency of atmospheric lead regulations through the National Ambient Air Quality Standards (NAAQS). The lead NAAQS stringency has tightened from a maximum calendar quarter average total suspended lead concentration of 1.5 μg/m3 to a maximum concentration of 0.15 μg/m3 over a rolling three month average. Despite these regulations, lead has continued to be used as an anti-knock agent in avgas.

In 2006, the environmental nonprofit group Friends of the Earth petitioned the US EPA to make a ruling regulating leaded emissions from GA aircraft under an endangerment finding or to pursue a course of research necessary to make an endangerment decision. In April 2010 the agency issued an Advance Notice of Proposed Rulemaking, which described existing data and planned research and requested comment and further information on the subject (Kessler 2013). In July 2012, the EPA responded to Friends of the Earth stating that it was not able to make an endangerment finding for aviation lead emissions, but was continuing to undergo monitoring and modeling of lead emissions from airports. In 2013, the EPA released findings that two airports had lead levels above the NAAQS Throughout this process, the FAA has announced an intention to certify and make available an unleaded replacement fuel by 2018 (FAA 2011).

Further, it is estimated that eliminating lead from gasoline contributed to significant economic benefits. IQ-related gains for the cohort of American children born in 1998 have been estimated to be between $110 and $319 billion (Needleman 2004). The environmental and economic benefits of eliminating lead from aviation fuel have not been fully quantified. The only study of the economic impacts of leaded avgas estimated that GA emissions annually contribute to $4 billion dollars in damages through IQ-related lost wages (Zahran et al. 2014). That study was limited to impacts in the vicinity of airports flying GA operations, and therefore does not include the potential contribution from full-flight GA emissions. Further, that study considered soil loadings of lead, and therefore may include damages from historical lead emissions in the airport vicinity.

This work is he first comprehensive study to estimate the present costs of yearly emissions of leaded aviation fuel on society. This study uses the leaded emissions and impacts pathway model and inputs described in Section 3.4. Combustion emissions of lead for all non-aviation sources are taken from the 2005 National Emissions Inventory while aviation emissions are developed from the 2008 National Emissions Inventory using the approach detailed in Section 3.4.2. Section 4.3.2 details the results of the lead emissions modeling while Section 4.3.3 develops the annual health costs of aviation lead as measured by the earnings reductions for a one-year birth cohort and by the dynamic effect the associated lost productivity has on the economy. To place the environmental costs in perspective, the benefits of General Aviation and their associated impacts on the economy are briefly discussed in Section 4.3.4. Section 4.3.6 explores the current understanding of alternatives to leaded avgas including alternative fuels and technological retrofitting. Finally, Section 4.3.7 discusses the implications of regulating leaded aviation fuels given the results of the preceding sections and presents study conclusions.

4.3.2 Atmospheric Lead Concentrations

Surface-level atmospheric concentrations are modeled using the approach described in Section 3.4.2. The contribution of aviation emissions to lead concentrations is calculated by first modeling particulate and toxic species concentrations from all emission sources and then by modeling concentrations for all sources minus general aviation. Aviation-induced lead concentrations are estimated as the difference between the two model runs.

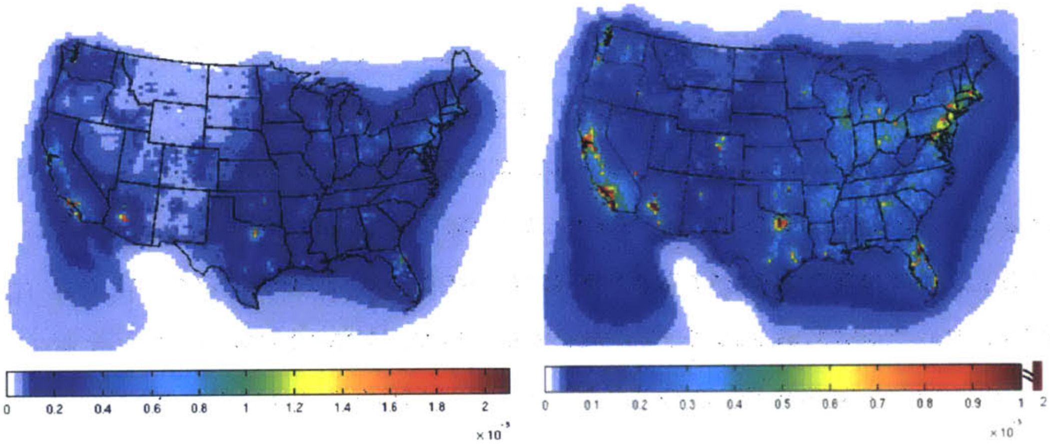


Figure 4-12 Surface atmospheric fine-particulate lead concentrations attributable to aviation in the continental United States (μg/m³). The right panel shows the same data with a finer scale resolution between 0 and 0.001 μg/m³ to highlight the spatial distribution of low concentrations.

Figure 4-12 shows the contribution of aviation emissions to yearly average surface fine-particulate lead concentrations in μg/m³. Model results show that General Aviation contributes to a wide dispersion of low concentrations of fine particulate lead emissions. For comparison, the median national total atmospheric surface lead concentration for the same period is estimated to be 0.011 μg/m³ from the National Health and Nutrition Examination Survey (NHANES) 9908 study (Richmond-Bryant et al. 2014) and fine particulate lead accounted for an average of between 0.0053 μg/m³ and 0.00723 μg/m³ of total atmospheric lead at US monitoring sites in July 2001 and January 2001 respectively (Hutzell and Luecken 2008). The model shows local areas of high aviation lead contributions, particularly the Sand Diego – Los Angeles Corritor, the Washington – Boston Corridor, and the Dallas/Fort Worth area. Further, the results indicate that aviation contributes to surface lead concentration across the entire continental United States. Because these aircraft-attributable concentrations are small (on the order of 0.0005-0.001 μg/m³), these contributions may be indistinguishable from background lead concentrations in monitor data. For example, the EPA estimates pristine atmospheric lead concentration at 0.0005 μg/m³ (Carr et al. 2011) and detection limits and resolution for several monitors are of the same order (Hutzell and Luecken 2008). However, because there is no known threshold for lead impacts on health, these concentrations may contribute to significant health and welfare impacts.

Lead emissions are expected to decrease exponentially as a function of distance from a point or area source. Carr et al. (2011) found aircraft emissions were indistinguishable from background concentrations at monitor stations further than 900m downwind from Santa Monica Airport[6]. Thus, modeled concentrations, as they are averaged over 36km × 36km grid cells, will likely under-estimate peak concentrations. Hutzell and Luecken (2008) found that CMAQ, on average, underestimated trace metal concentration values at monitor sites. For example, lead values had an average mean bias of -48.10% at suburban monitoring stations in January. While modeled lead concentrations more closely matched observations at rural and urban monitoring sites on average, some modeled values were under-predicted by between -100% and -75% at individual monitoring stations. Thus, examining the impact of modeling lead emissions across different scales and the impact of spatial scale on model results is an area of necessary future work.

4.3.3 Health Costs of Aviation Lead

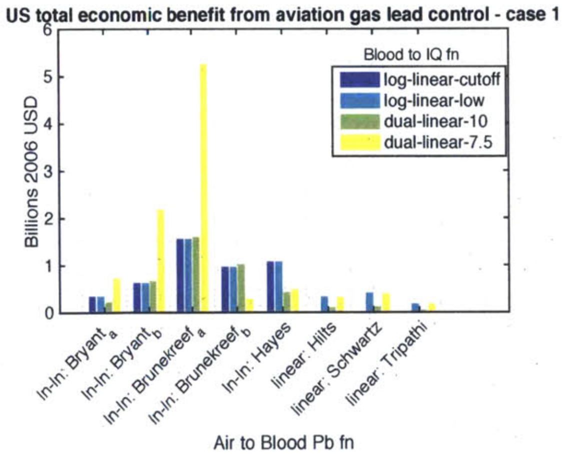
The costs of aviation lead emissions are first modeled by considering only the impacts of atmospheric lead burden on lost earnings through the pathway of reduced IQ scores. The costs are calculated per annum by considering the impacts of lead on the cohort of one-year-old children in 2008. The functions relating atmospheric lead concentration to blood lead levels, blood lead levels to IQ loss, and IQ loss to foregone wages are presented in Sections 3.4.3 and 3.4.4. Eight air-to-blood, four blood-to-IQ, and three IQ-to-losses functions are utilized to develop 96 estimates of the social cost of aviation lead (or the social benefit of eliminating

[6] Importantly, however, Carr et al. (2011) did not estimate aviation's contributions to the local background concentration and did not model cruise emissions.

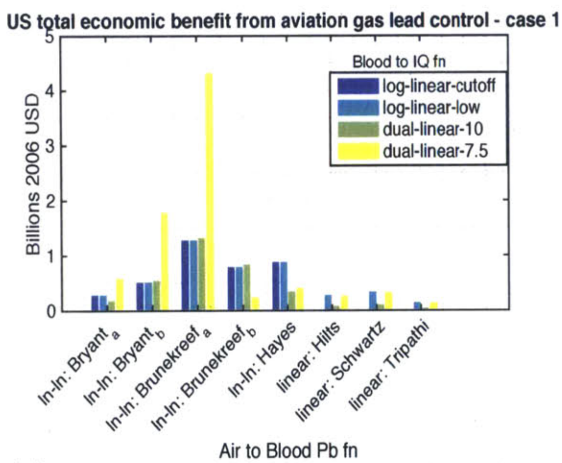
aviation lead emissions). These estimates indicate the range of uncertainty in the social cost of aviation lead from uncertainty in the concentration response function.

Because some of the concentration response functions are non-linear, the contribution of aviation lead to social damages is measured as the difference between all-source lead total lead measured in total suspended particles and all-source lead minus aviation lead. Lead in total suspended particulates (TSP) has decreased in the United States over the past two and half decades. The annual maximum 3-month average lead TSP for the United States has decreased from 1.57 $\mu g/m^3$ in 1980 to 0.13 $\mu g/m^3$ in 2013 based on the average of 12 monitoring sites used in the EPA's Air Trends assessment. The median surface level annual average concentration for the 1-5 year-old population was estimated at 0.037 for 1988-1994 but decreased to 0.011 for 1999-2008 (Richmond-Bryant et al. 2014). Because toxic metal concentrations are expected to vary over small spatial scales (Carr et al. 2011, Luecken, Hutzell, and Gipson 2006) and because lead concentrations have decreased dramatically over a short time period, three cases for background lead concentration are modeled. The first case assumes that the background annual lead TSP in 2005 is 0.011 $\mu g/m^3$, consistent with the NHANES measured average lead TSP for 1 year-olds over the same period. To better understand the impact of decreasing background lead levels on the social benefit from aviation lead control, the second case assumes that the background annual lead TSP 0.4 $\mu g/m^3$. This is the 90th percentile value of the yearly maximum 3-month average lead TSP from the EPA Air Trends study in 2005 and the mean value for 1994. Finally, to understand the effect of regional heterogeneity on social damages, the third case takes background concentrations for each 36km $\times$ 36km grid cell as the fine, coarse, and Aitken modeled contributions from all sources as modeled in CMAQ.

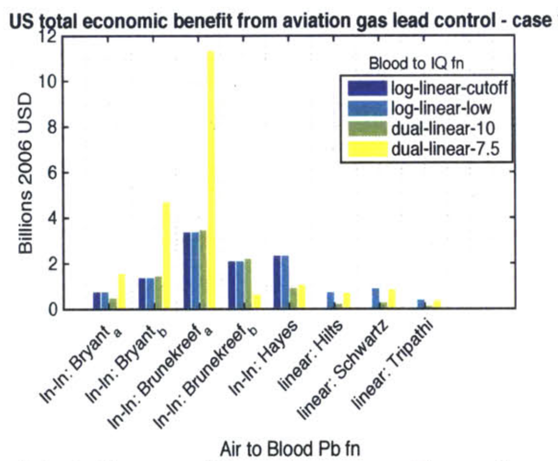
The static societal benefits of controlling all aviation-related lead emissions for Case 1, 0.011 $\mu g/m^3$ background lead TSP, are shown in Figure 4-13. Estimates for annual societal impacts range from less than 0.01 billion USD to 11.3 billion USD. Nine estimates return 0 values, all for the log-linear with cutoff blood-to-IQ function and are excluded from the results statistics as described in Section 3.4.3. The mean and median societal benefit of aircraft lead control are 0.95 and 0.51 billion USD per annum respectively for all estimates and 1.06 and 0.60 billion USD per annum respectively for all non-zero estimates. Case 1 will overestimate local societal damages in locations where total lead concentrations are above the national average and will mostly underestimate local societal damages in locations where total lead concentrations are below the national average. All three linear air-to-blood functions provide the lowest damage estimates. These linear damage functions are expected to provide conservatively lower damage estimates as they include concentration responses developed from studies with larger lead emissions and blood lead levels and lead is expected to have decreasing marginal damages with increasing concentration. For example, Schwartz (1994) found that if meta-analyses on lead damages were limited to studies with blood lead levels < 15 µg/dL, the marginal effect of additional lead would nearly double.



(a) Zax and Rees IQ-to-Income Function



(b) Heckman IQ-to-Income Function



(c) Salkever IQ-to-Income Function

Figure 4-13 Static economic benefit of eliminating lead from avgas under three different monetization functions for an average background concentration of 0.011 μg/m³.

The two additional cases provide insight into the impact of decreasing background lead concentration and regional variability on the societal benefit from controlling lead emissions. Case 2 estimates the impact of aviation lead emissions with background concentrations of 0.40 μg/m³ lead TSP. While this concentration is an order of magnitude higher than that of Case 1, 0.40 μg/m³ was the mean annual maximum 3-month average lead concentration in the EPA's Air Trends analysis in 1999. The mean and median static aviation lead societal cost for Case 2 are \$0.09 and \$0.04 billion USD respectively. Whereas in Case 1, the linear air-to-blood concentration response functions provided the lowest cost estimates, in Case 2 they provide the highest cost estimates as background concentrations are much higher.

The results indicate that lead damages attributable to a single source are highly sensitive to emissions from other sources. Between Case 2 and Case 1, a 96% reduction in background lead emissions equates to a 92% reduction in median expected societal cost. Case 2 suggests that, as emissions from other sources have decreased dramatically, aviation's environmental impact has become more significant. Case 2 also indicates that use of median lead concentration may overestimate benefits from lead reduction in areas of high lead exposure. Areas of Los Angeles county, Tampa, and Chicago are three high-population areas that are designated lead non-attainment areas under the 2008 lead standard where benefits may be overestimated (EPA 2015).

Further, since 2005, deliveries of leaded avgas have increased by 10%, while the US has continued to tighten lead controls on other emitters. In 2013, Doe Run Co.'s smelter in Herculaneum, MO, the last primary lead smelter in the US, closed as sulfur and lead emission stringencies increased. In addition, there is significant regional variation in the background concentration of lead in the US. Thus, Case 3 models the impact of aviation lead emissions where background emissions are the fine particulate concentrations modeled from the 2005 National Emissions Inventory for all sources using the data sources and methodology described in Section 3.4.2. The average (non-population weighted) modeled concentration of fine particulate lead over the continental US in 2005 is 0.0034 $\mu g/m^3$.

The static societal benefits of aviation lead control for all three cases are shown in Figure 4-14. For Case 3, the estimated benefits of reducing lead increase to a median of $5.2 billion USD and a mean of $7.9 billion USD. Case 3 produces an upper value estimate of $51 billion USD, an order of magnitude greater than the median value. As in Case 1, 9 of the 96 cost estimates were $0 values, all for the concentration response function that includes a cut-off value below 1 μg/dL. The results show that the aviation contribution to lead damages is highly sensitive to the average background lead concentration, suggesting that damages from aviation lead have grown significantly as background lead concentrations have decreased and even holding aviation emissions constant, would further increase by an order of magnitude over the next decade if atmospheric lead concentrations continued to decline at the same rate as they have since 1995.

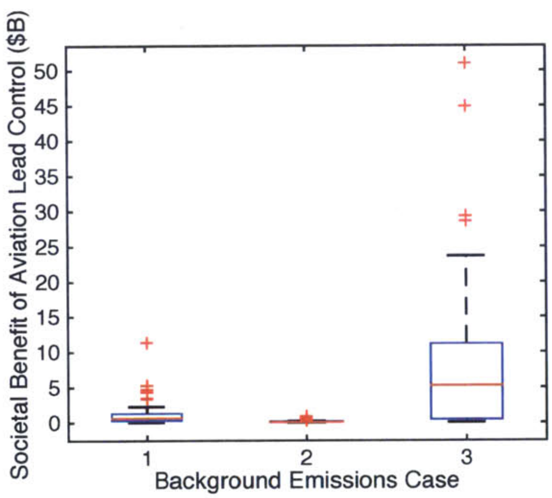


Figure 4-14 US-wide societal benefit of aviation lead control for (1) the baseline background concentration of 0.011 μg/m^3, (2) the low-impact case with a background concentration of 0.40 μg/m3, and (3) for the spatial distribution case considering fine particulate lead only.

The median estimated societal benefits of lead control from aviation are broken down by state for each of the three background cases in Table 4-3. California has the largest benefits from lead control for all three cases. In Case 1, the midrange case, California benefits by $168.81 million per annum from eliminating lead in avgas, over $140 million more than the next highest state. The next highest states for lead damages are Texas, Florida, New Jersey, and Arizona. For comparison, the top five states for total aviation lead emissions are California, Florida, Texas, Arizona, and Washington. Wyoming, South Dakota, Montana, and Vermont have the lowest total expected lead benefits, each at less than $0.36 million per annum in Case 1. The states with the lowest expected benefits are also the states where the estimated damage is most sensitive to background lead estimates as shown in the Case 3 totals.

Table 4-3 State-level aviation lead control benefits (million USD)

State	Case 1	Case 2	Case 3
Alabama	5.24	0.360	45.10
Arizona	27.33	1.793	201.4
Arkansas	3.08	0.2121	30.66
California	168.8	10.69	1300
Colorado	8.11	0.5521	99.50
Connecticut	6.90	0.4710	74.05
Delaware	0.67	0.0462	6.342
Florida	39.61	2.691	479.6
Georgia	12.41	0.8502	118.9
Idaho	1.46	0.0999	22.20
Illinois	20.82	1.427	96.37
Indiana	8.12	0.5579	39.25
Iowa	2.32	0.1602	24.07
Kansas	3.39	0.2332	37.70
Kentucky	4.82	0.3312	34.25
Louisiana	4.81	0.3316	52.12
Maine	1.13	0.0778	14.07
Maryland	16.28	1.107	120.9
Massachusetts	16.40	1.112	168.5
Michigan	14.72	1.008	112.6
Minnesota	6.27	0.4300	41.51
Mississippi	2.52	0.1737	28.04
Missouri	4.45	0.3064	28.80
Montana	0.36	0.0248	6.372
Nebraska	1.30	0.0894	14.16
Nevada	5.58	0.3782	69.78
New Hampshire	2.06	0.1415	26.52
New Jersey	32.15	2.191	212.3
New Mexico	1.83	0.1257	27.79
New York	26.10	1.7851	236.9
North Carolina	10.71	0.7359	118.1
North Dakota	0.25	0.0169	3.810
Ohio	21.23	1.451	127.7
Oklahoma	5.16	0.3527	58.44
Oregon	5.00	0.3422	69.52
Pennsylvania	22.39	1.530	112.2
Rhode Island	2.06	0.1417	25.23
South Carolina	5.60	0.3849	63.46
South Dakota	0.33	0.0227	5.267
Tennessee	7.40	0.5086	59.06
Texas	40.15	2.719	382.0
Utah	2.17	0.1496	31.41
Vermont	0.36	0.0250	5.501
Virginia	10.48	0.7169	75.93
Washington	17.74	1.194	210.9
West Virginia	1.56	0.1078	14.53
Wisconsin	7.37	0.5059	68.77
Wyoming	0.13	0.0092	2.436

4.3.4 Economy-Wide Aviation Lead Costs

The US Regional Energy and Environmental Policy (USREP) model is used to estimate the impacts of children's IQ-related earnings loss on the US economy as a whole. USREP is a recursive-dynamic general equilibrium model of the US economy that models how IQ-losses from a cohort of one-year olds lead to economic losses as goods and services are diverted and productivity is impacted over time. The yearly environmental impacts of aviation lead emissions are calculated by taking then the sum of discounted differentials between the economic output considering a cohort of one-year olds exposed to aviation lead emissions and one where aviation lead emissions are eliminated for that cohort. The median midrange case (background lead concentrations of 0.011 μg/m3, ln-ln PbA to PbB relationship, dual-linear blood to IQ relationship with inflection at 7.5, and IQ-loss to earnings relationship from Salkever (1995)) is used to explore the impact of economy-wide costs. The static earnings loss for the median midrange case at a 3% discount rate was estimated at $602 million per cohort. The USREP results are shown in Figure 4-15 for three discount rates.

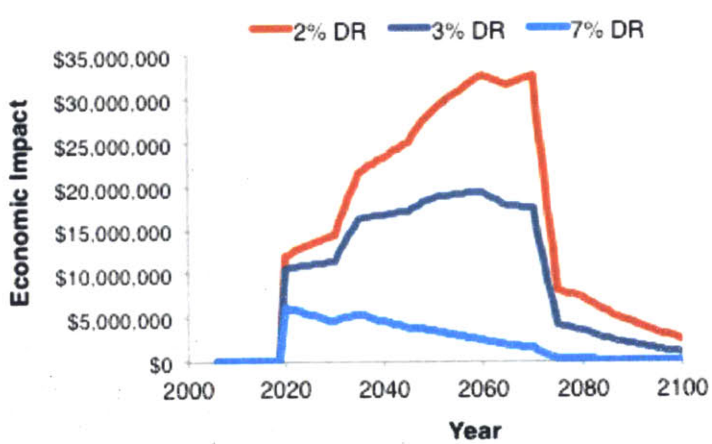


Figure 4-15 Dynamic economy-wide impacts of one year of aviation lead emissions.

The economic impact of lead pollution for one childhood cohort starts 15 years after initial emissions as they start to enter the workforce and peaks 50 years later.

Because impacts are delayed, results are highly sensitive to discount rate. At a 3% discount rate, economy-wide impacts are $926 million, an increase of 54% over the static case. At 2% and 7% discount rates, the economy-wide impacts are $1,460 million and $202 million respectively.

4.3.5 Benefits of General Aviation

The environmental costs of piston-driven GA aircraft must be placed in context of the benefits GA operations provide. General aviation provides several economic benefits and services to the American public. GA activities include operations that are essential for agriculture, fire fighting, law enforcement and security, tourism, and travel to remote or rural areas in addition to hobby flying and personal and business travel. The General Aviation Manufacturers Association (GAMA) estimates that 67% of all GA flights and 40% of piston-driven GA flights are for business purposes (NATA 2009). In 2012, General Aviation's contribution to the United States economy was estimated at $78.5 billion USD (FAA 2014a), while total economic output from direct, indirect, and induced effects of GA has been estimated at $219 billion USD (PwC 2015). However, only a portion of these benefits is attributable to piston-driven aircraft. Single- and twin-engine piston-driven aircraft account for 69% of the aircraft in the US GA fleet, 53% of logged annual GA flight hours, and 3.6% of annual GA aircraft sales revenue (PwC 2015). While isolating the annual economic impact of aircraft using leaded fuel is difficult, assuming piston-driven aircraft account for 60% of General Aviation operations direct output and visitor expenditures and 3.6% of GA manufacturing output provides a first order approximation of $31.8 billion USD in annual direct economic output. If banning leaded fuel made these operations infeasible, in many cases alternative travel modes or product substitutions would replace some of the output. In these cases, the benefits of allowing leaded fuel (or the costs of banning the fuel) would be the difference in consumer and producer surpluses between the case with GA operations and the case without. However, considering the direct economic output as a benefit is useful for placing the environmental costs of lead in context.

4.3.6 Leaded Fuel Alternatives

The FAA is actively pursuing replacements for 100LL leaded avgas for piston-driven aircraft. The FAA propagated a request for fuel producers to submit proposals for unleaded avgas formulations to replace 100LL that closed July 1st, 2014 with intent to make an unleaded fuel alternative available by 2018 (Kessler 2013). The FAA received nine fuel proposals from five fuel producers. The FAA selected four fuels for the first phase of testing with respect to technical feasibility, impact on the existing fleet, the production and distribution infrastructure, their impact on the environment, their toxicology, and the cost of aircraft operations (FAA 2014b). The Phase 1 test program is anticipated to take approximately one year, at which point the FAA will evaluate the fuels for continued participation in Phase 2 test of the test program (FAA 2014b).

Unleaded mogas (or autogas) may be a suitable alternative for between 70% and 83% of the piston-driven fleet when it is not blended with ethanol (Kessler 2013). While mogas is already certified for some piston-driven aircraft, ethanol-free mogas is only available at a small percentage of GA airports, and many older aircraft are not certified to use mogas. Additionally, mogas is on average more than a dollar cheaper per gallon then 100LL fuel (AirNav 2015). While more aircraft could be retrofitted to fly on existing unleaded fuel alternatives likes mogas, General Aviation practitioners are highly sensitive to price (Shetty and Hansman 2012), indicating that forced retrofits would severely restrict GA demand.

4.3.7 Discussion

The midrange mean environmental cost of aviation lead emissions is $1.06 billion per annum considering only static losses from lost wages at a 3% discount rate. Dynamic economy-wide losses are estimated to be 58% higher than static losses. Wolfe et al. (2014) estimate the climate and noise damages attributable to

US airports at are \$5.25 billion and \$0.63[7] billion respectively, while Yim et al. (2015) estimate air quality damages from ozone and $PM_{2.5}$ in North America as \$6.89 billion. Assuming US operations account for 92% of North American air quality damages consistent with ICAO-CAEP methodology (see Section 4.1.2), US air quality damages are approximately \$6.3 billion. Thus, the cost of General Aviation lead emissions are of the same order of magnitude (albeit smaller) than estimated costs of commercial aircraft climate and air quality but exceed the costs from commercial aircraft noise.

For comparison, Zahran et al. (2013) find a first-order estimation of the annual societal costs of General Aviation lead to be \$4.0 billion. The Zahran et al. (2013) estimate used average soil loadings of lead near airports and their expected relationship with blood lead levels. They also model only aircraft lead contributions in Michigan and then extrapolate this result to the entire United States. This estimate falls within the range of damage estimates calculated across the 96 concentrations-to-earnings model combinations tested above. Using only model combinations with the concentration response functions from Zahran et al. (2013), the results from this thesis would be an average societal damage from aviation lead of \$2.6 billion and median damage of \$0.94 billion (continental US only).

The Clean Air Act gives the EPA the authority to regulate emissions that cause or contribute to air pollution that may reasonably be anticipated to endanger public health or welfare. Since 1976, lead has been a criteria pollutant, thereby having been designated as endangering public health and/or welfare, and has been subject to a National Ambient Air Quality Standard (NAAQS). To regulate a source or sector emitting a designated pollutant, the EPA has found that the sector or emitter accounted for at least 1.2 percent of total pollutant inventory, which would qualify General Aviation lead emissions (EPA 2010a). The EPA, therefore, has the authority, if it issues an endangerment finding for General Aviation, to regulate lead emissions from aircraft engines. However, the FAA retains the authority to

[7] This analysis did not consider the health costs of aviation noise.

promulgate fuel standards for fuels used exclusively in aircraft. In this section, three potential policy frameworks for controlling aviation lead are briefly examined: command-and-control emissions standard, a fuel tax, and a cap-and-trade program.

The first policy is a command-and-control emission standard that would require aircraft engines to operate on unleaded aviation fuel by some date. A first-order estimate of the maximum direct benefits of leaded-fuel based operations is $31.8 billion per annum. However, under reduced background concentrations in the future, damages from general aviation lead emissions could exceed economic benefits as modeled in Case 3 above. Since some operations currently using leaded fuel could conceivably be operated using mogas if the aircraft were certified to fly using mogas and it were available at the departure airport, banning leaded avgas would not be expected to lead to a full cost of $31.8 billion. Thus, while a command-and-control ban on leaded aviation fuel (either through a fuel standard or an engine emission standard) is potentially not economically efficient in the near term, it could become economically efficient in the future even without the development of an alternative fuel.

An alternative policy would be to apply the emission standard to newly-manufactured engines. However, because of the long service lifetimes of GA aircraft, this regulation may not effectively limit environmentally costly emissions from in-use aircraft. An emissions tax could potentially be put in place to internalize external environmental costs. As the lead inventory modeled here assumes 241.1 million gallons of avgas consumption, the mean estimated social cost of lead is $4.34 per gallon of fuel (static losses only). For context, assuming a $33 Social Cost of Carbon, General Aviation avgas usage contributes to $0.075 billion USD in climate damages per annum, or approximately $0.30 per gallon of fuel. The first barrier to a lead tax is the magnitude of the marginal social cost of lead, which would nearly double the cost of leaded avgas. Considering the already high cost of fuel and the price sensitivity of General Aviation operators, a fully efficient lead tax would be politically unattractive. Second, because the societal damages attributable to lead are non-linear and highly dependent on background concentration, setting an appropriate tax schedule would be difficult.

In lieu of an immediate cessation of the sale of leaded avgas or a lead tax, airport caps on lead emissions would potentially lead to environmental improvement without creating an undue regulatory burden for essential services. There is precedent for a lead cap-and-trade program. A cap-and-trade on lead in automotive gasoline was in place during the leaded gasoline phasedown from 1982 to 1987 (Hahn and Stavins, 2010). The lead cap-and-trade allowed refineries to earn credits for producing gasoline with lower than required lead to ease the burden of refineries that might have higher compliance costs. The lead cap-and-trade was considered a success with only some cost-effectiveness inefficiencies (Hahn and Stavins, 2010). A cap-and-trade for aviation lead would face several difficulties. Because marginal costs are not uniform in space and lead aerosols are not well-mixed in the atmosphere, setting a market for lead that would ensure health and safety benefits for all populations while still permitting flights for essential services such as access to rural communities is difficult. Allocating a large portion of the permits and auctioning the rest would guarantee allocations for all airports, ensuring equity at the expense of some economic efficiency. Further, using the income from the sale of permits to cross subsidize fuel alternatives (such as mogas or an unleaded aviation fuel when it becomes available) would promote technology development supporting aircraft that fly with unleaded fuels while still allowing older legacy GA aircraft to continue flying as a stopgap measure. A cap-and-trade where the cap is tightened each year could be used to transition to full 0-emission emission standard much as the automotive cap-and-trade was implemented in the 1980's. However, there appears to be little political support for a lead cap-and-trade currently.

4.3.8 Result Caveats

The three cases explored in Section explore the sensitivity of results to variability in background atmospheric concentrations, but they do not consider the sensitivity of results to variability in other sources of lead including leaded paint and soil lead. The impact of aviation lead may be overestimated for populations with significant non-atmospheric sources of lead, but the future impact of aviation lead may be larger than estimated if controls on non-emission sources are tightened or as the available housing stock with leaded paint decreases.

As discussed in Section 3.4.5, there are several potential effects of aviation lead emissions that are not included here that may lead to systematic undervaluing of the impact of aviation lead on human health and welfare. First, this work does not monetize lead's effect on adult hypertension, medical costs of lead treatment, and neurological effects related to antisocial behavior and crime. The analysis also only assesses the atmospheric burden of lead in the year of emission and does not consider cycling of atmospheric lead emissions and impacts from changes in soil concentrations. While historically aviation has represented a small percentage of total anthropogenic lead emissions, aircraft emissions will continue to represent a larger and larger percentage of legacy emissions and may contribute to significant soil concentrations near an airport with a high concentration of GA traffic.

Due to limitations of the model domain, damages from Alaska are not considered despite accounting for a significant portion of leaded emissions. Further, the inventory for background emissions do not consider sources from Canada and Mexico that may transport into the US domain or re-emission of lead particles during the year. The model is limited by spatial resolution, and may underestimate concentrations near the airport and overestimate concentrations further from the airport. Combining high-fidelity chemistry and transport model results with local airport dispersion results may provide additional insight into general aviation lead's environmental costs. Further, research suggests that empirical concentration and deposition models need not be "species"-dependent (i.e. lead-specific) as long as the expected distribution of particle sizes is known (Zhang et al. 2012). Thus, models currently used to study aviation's in other air quality contexts may be appropriate for the study of atmospheric lead concentrations.

4.4 Chapter Summary

This chapter presented three assessments of aviation environmental policies. In the first, an increase in the stringency of aircraft noise regulations was examined. The analysis found that moderate increases in stringency lead to environmental co-benefits and are cost-beneficial under several discount rates and lenses. With increasing stringency, environmental tradeoffs between noise and emissions arise, leading to policies that never appear to be cost-beneficial. The second assessment was that of land-use policies near US airports to mitigate the impacts of noise. This study was the first aircraft environmental assessment to consider both the amenity and health impacts of noise exposure by considering noise-induced hypertension, myocardial infarction, and stroke. The study found that neither soundproofing projects nor land acquisition were on average cost-beneficial. However, the projects had the same order of magnitude costs and benefits even without considering the potential utility of the purchased land. Finally, an assessment of General Aviation lead emissions found that the costs of leaded avgas are highly dependent on background concentration and there is uncertainty in concentration response functions linking lead concentrations to economic damages. However, a midrange estimate for societal costs was developed at $1.06 billion ($5^{th}$ and 95^{th} percentile ranges of $0.1 billion - $11.3 billion). While this social cost does not exceed the potential benefits of piston-driven General Aviation, it is an environmental cost on the same order as noise, air quality, and climate damages from US commercial aviation.

Cost-benefit analysis is a useful framework for assessing aviation environmental policies. The costs of the policy are the technological and compliance costs and foregone economic benefits caused by adopting the regulation where the benefits are the environmental and welfare improvements caused by reducing noise or emissions. However, as developed through the preceding analyses, cost-benefit's utility is as a decision-aiding tool, not as a decision-making procedure. Uncertainty and risk, equity, societal preference, and political feasibility will all impact the efficacy or appropriateness of proposed policies.

Chapter 5
Timescales in Environmental Policy Analysis

Defining timescales over which a policy analysis is conducted and over which the costs and benefits are accrued can have a significant impact on the apparent attractiveness of potential policies. There are three timescales embedded in a policy analysis. The first timescale is the policy influence period, which is the duration over which a policy is assumed to significantly influence personal activity. The second timescale is the environmental impact lifetime, the time over which the impacts of the different environmental effects attributed to the activity persist. The final timescale is the time period over which society values these policy-induced changes, and is often subject to a weighting factor such as a discount rate.

The second of these timescales, the environmental impact lifetime, is controlled by the chemical and physical systems of the environment and is often expressed in terms of atmospheric residence time (for air pollution and climate forcers), biogeochemical cycling equations (for long-lived toxic pollutants), or half-life (for nuclear waste) among others. While accurate and reliable modeling of these phenomena may be difficult, the timescale is conceptually easy to understand. The third timescale, the valuation timescale, is driven by societal and policy-maker preferences between short-term and future costs and benefits as well as investment rates of return. While these issues are complex and uncertain, they are well discussed in the literature (see Section 5.3). Therefore, the work in this chapter focuses on the first timescale, the policy influence period.

The work presented here first identifies and characterizes these three timescales. Second, for each timescale it also identifies an approach to explicitly account for impacts along this timescale in policy analysis. Third, this study uses a sample case, an aircraft noise certification policy, to quantify the impact of this timescale framework on modeled environmental impact and apparent policy appropriateness. Finally, this chapter looks at interactions between the policy timescales and derives conclusions to improve environmental policy modeling.

5.1 Policy Influence Period

In evaluating environmental regulation, it is often the goal to calculate the incremental costs and benefits of the policy action. To that end, practitioners recommend specifying a clear baseline against which the policies costs and benefits will be measured (Arrow et al. 1996, Farrow and Toman 1999). Unfortunately, there is limited specificity as to what a clear baseline entails. Is this baseline a representation of the baseline state of the technology, the state of the industry, or the state of the world? Is it static and present (how the world is now), static and future (how the world will be in the future without imposition of a regulation), or dynamic (how the world changes over time without imposition of a regulation)? This vagueness of prescription leads to inconsistency in practice.

Even where guidance is more specific, the implications are not always clear. The Office of Management and Budget guidance to cost-benefit analysis states that the third step of a regulatory analysis should be to select a time horizon for the analysis (OMB 2003). However, the meaning of this time horizon is ambiguous. For one, it suggests that for analyses with large up-front capital investments the life of the capital is an appropriate time horizon, an assumption that is shown to be non-ideal in Section 5.1.2 and Section 5.1.3. Second, it recommends limiting the analysis to only the time period for which an agency can reasonably predict the future. This suggestion is ambiguous for two reasons. First, it is unclear what level of uncertainty is permissible in a reasonable prediction of the future, and second, the recommendation fails to recognize that policy impacts have costs and benefits that occur over different time periods and their associated levels of uncertainty may also differ over time.

For example, consider a regulation mandating that car manufacturers maintain a minimum fleet-averaged fuel economy. A modeler may claim that the direct impact of the regulation on manufacturer, distributor, and consumer behavior can only be predicted within a certain level of confidence for 30 years. Should the analysis of this regulation be limited, in total, to these 30 years? What if the impact of the emissions on the climate is expected to remain significant for decades or even centuries? What if there are known feedbacks to the used-car market that will not manifest for several years and are expected to be long-lived?

In this section, the policy influence period is defined and delineated as a useful framework for determining incremental changes induced by a policy. The definition introduces a level of specificity in cost-benefit analysis design and provides an improvement over previous analysis guidelines. Section 5.1.1 presents the definition, introduces the relevant terminology, and further develops the context and practical use for delineating amongst different policy-relevant timescales. This section also introduces one notional model for understanding the policy influence period. Section 5.1.2 applies this notional model to an illustrative example to better characterize the policy lifetime for several applications. Finally, Section 5.1.3 applies the results of Section 5.1.2 to a sample case, a reanalysis of the aircraft noise stringency analysis presented in Section 4.1.

5.1.1 Terminology, Definitions, and Justification

The first policy-analysis-relevant timescale is the policy influence period, the period over which a given policy is expected to impact the actions or outputs of the producers or consumers of the relevant system. Mahashabde (2009) notionally explains the policy influence period in the context of aviation by noting that while its reasonable to assume that, for any policy, aviation will continue well into the future, after some time the fleet mix is no longer influenced by the stringency of the policy[8].

While studies have recognized the import of this timescale, it has not been explicitly accounted for in many policy analyses. Mahashabde et al. (2011) delineates the policy influence period and provides a limited justification for selecting a timescale of 30 years for an engine certification stringency noting that it is both consistent with prior best practice and the time period for "development, adoption and significant use of a new technology in the fleet". This convention is repeated in later aviation certification policy analyses including the study presented in Section 4.1. Other environmental cost-benefit studies have focused on single-year snapshot or static results that do not consider capital stock or technology evolution under the regulation (Barrett et al. 2012, EPA 2011b). A Dutch technical report on cost-benefit analysis of public policy applies the representative-year approach for twelve environmental problems ranging from biodiversity loss to coastal management (Pearce and Howarth 2000). However, studies of ex-ante regulatory analysis found that failure to account for industry changes over time (either by assuming an efficient industry response or by failing to account for technology development) was the most significant bias in computing regulatory costs and benefits, often leading to overestimation of industry costs (Hahn and Hird 1991, Harrington, Morgenstern and Nelson 2000). Morrison et al. (1999) apply policy lifetimes to cost and benefit estimates inconsistently. For costs, they consider depreciation rates for the asset value of the fleet and consider how capital replacement costs differ between two different policies into the future to maintain a constant capital stock value. For benefits, they examine only capitalized damages to property for one year.

[8] Mahashabde (2009) identifies two timescales of importance for analysis: the policy influence time period and the impacts time period. The work in this chapter works to formalize these timescales while also identifying a third independent timescale: the valuation timescale, the period over which society and/or decision makers value the impacts of the policy.

A notional diagram of the policy influence period for the case of regulating environmental efficiency of a product or industry over time is shown in Figure 5-1. Environmental efficiency here is defined as a unit production value per unit environmental impact; this metric also appears in the literature as environmental productivity (Huppes and Ishikawa 2008). The inverse of environmental efficiency as defined here is the environmental intensity. The relevant regime of the diagram starts at time $t(0)$, the time at which a theoretical policy may be enacted. Absent that policy, the rate of change in environmental efficiency is unaffected. This "business-as-usual" case is the policy baseline. Under the policy case, the environmental efficiency is improved to a level that would not have occurred until a later time. This differential improvement is here identified as the forcing gap, the length of time forward the policy has forced the environmental efficiency[9].

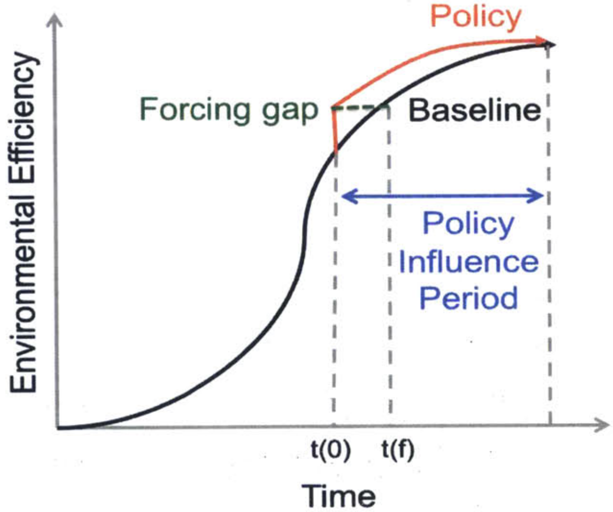


Figure 5-1 Notional policy influence period model diagram

The notional model presented here applies the aggregate industry technology S-curve (Christensen 1992), which models product performance over time, to environmental efficiency. It is meant to be case descriptive and not prescriptive or a fundamental property of environmental progress. It is applicable to many cases of

[9] An alternative formulation would be to assume that Φ_0 is the environmental efficiency at time $t(0)$ and that the proposed policy mandates (or induces) an efficiency of Φ^*. The forcing level would then be the incremental change in environmental efficiency $\Phi^*-\Phi_0$. The time at which the baseline case reaches an efficiency of Φ^* (unforced) is the forcing gap.

command-and-control regulatory stringency as well as market-based and mixed measures, and is consistent with (but not dependent upon) an assumption of autonomous energy efficiency improvement (AEEI)[10]. The Environmental Kuznet Curve (EKC), that environmental degradation is related to per capita income by an inverse u function, has been used to explain several domains of environmental progress. While the literature is mixed on whether the EKC is a fundamental relationship, it has been observed for several environmental problems (Stern 2004). Because the EKC relationship requires net environmental degradation to decrease whereas the above model only requires an improvement in the environmental efficiency, the notional model is consistent for cases of an EKC and for a weaker formulation of the EKC in which total production outpaces environmental efficiency as per capita income increases over time. Furthermore, the model is applicable for developing nations in cases where environmental improvement does not follow an EKC but instead follows (and potentially improves upon) the time-lagged improvement demonstrated in developing nations (Stern 2004). However, it should not be assumed that the model is universally applicable or even applicable to all instances of the cases delineated above.

The relationship between policy pressure and efficiency through innovation raises an important question as to the validity of the above model: is the environmental policy itself the primary driver of technological innovation? If the answer to this question were yes, then there would be an expectation that the policy case would not simply create a forcing gap, it would significantly change the slope of the environmental efficiency in the policy case. There may be no expectation that the two curves would approach one another in the future. A study of 127 manufacturing industries finds that while environmental innovation is an important driver of reductions in environmental degradation (specifically toxic

[10] AEEI is a modeling approach that assumes there is an energy efficiency improvement in each modeled period applied to industrial production or demand. AEEI can also implicitly account for the representation of exogenous assumptions of the costs of substitutes over time. AEEI is a common assumption among several integrated models of energy, the economy and the environment including IMAGE, DICE, G_cubed, GREEN, and Pizer (Grubb, Kohler, and Anderson 2002).

emissions), the proportionate contribution of policy-induced innovation is small (Carrion-Flores and Innes 2010). Thus, while policy and innovation are jointly endogenous, this feedback mechanism is not expected to invalidate the model above as a useful tool for investigating policy timescales.

Clear identification and delineation of the policy influence period is useful for several reasons. First, it provides an indication of the period over which the relevant industry or technology needs to be modeled to ensure that all impacts, costs, and benefits are captured in the analysis. This is especially important in analyses where each successive year of analysis adds computational time and strains resources or where the response of the built-environment is modeled first and the future environmental impact is modeled "off-line". Furthermore, this simplified model may provide insight to policy-makers as to at what timescale they should revisit policy stringency levels. This is important as it can help indicate the necessary level of flexibility for a policy or ex-ante identify when a future impact analysis may be useful as such analyses are costly and resource intensive.

5.1.2 Policy Influence Period: An Illustrative Case

This section uses the notional guidance above to develop a simple practical model for determining the policy influence period of an illustrative case. While the model here is developed using aircraft, it is easily applied to other pollutant emitters. The illustrative case serves two purposes: to determine the relationship between stringency increase and policy influence period and to serve as inputs to the practical case in Section 5.1.3.

Figure 5-2 and Figure 5-3 demonstrate the applicability of the notional model to the case of aircraft certification regulation. The figures show environmental intensity of aircraft over time in two domains: energy use and noise, where energy use can also be considered as a proxy for many emissions species. The progression of both aircraft energy use and noise levels are notably cases where the externality is either internalized or correlates to an internalized variable: energy use is related to the cost of fuel while aircraft noise has been regulated throughout the time period shown in Figure 5-3. The model developed here may be less applicable to cases where this relationship does not hold, especially if there is no external pressure to reduce environmental impact of a given domain. Testing this relationship is an area of potential future work.

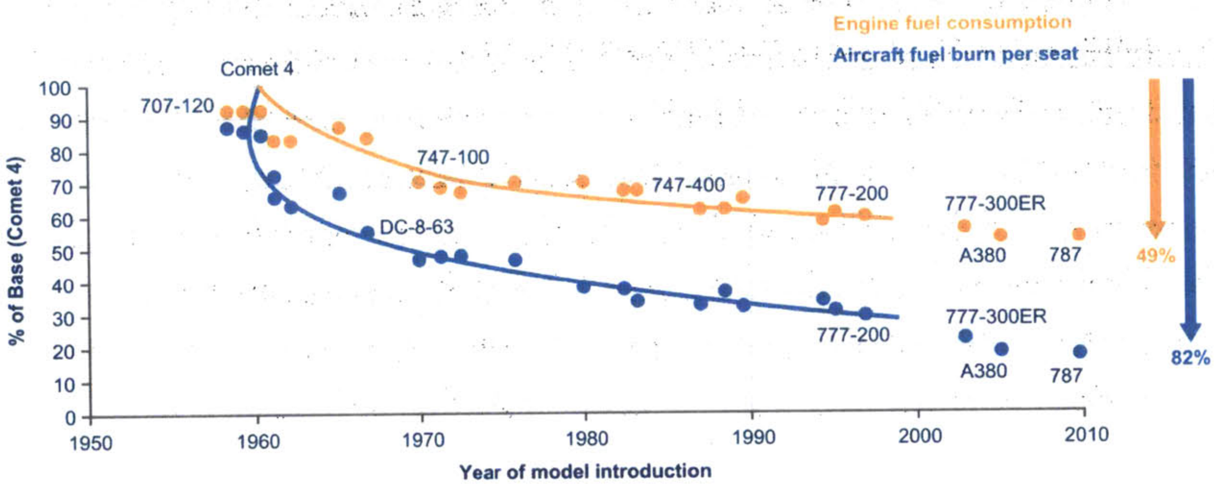


Figure 5-2 Historical and projected progress of aircraft energy use at time of entering service (IATA 2009).

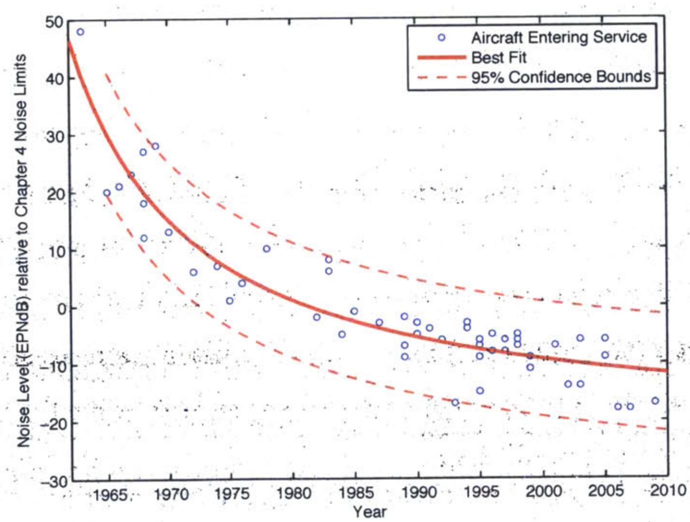


Figure 5-3 Historical progress of aircraft noise level at time of certification over the past 50 years.

For the illustrative case, the impact of aircraft certification emissions stringency is examined. *Emission* here is used generically, but could be seen as applying to noise, CO_2, or NO_X among others. For simplification, the following assumptions are adopted:

Assumption 1: One type of capital investment. It is assumed that there is a fleet of uniform emitters, in this case aircraft. These aircraft have a specified and identical product lifetime.

Assumption 2: Uniform retirement rate. It is assumed that investment decisions are made continuously, and that the stock is significantly large to model retirement rate as uniform. The fleet size, therefore, stays constant over time.

Assumption 3: A proposed policy is environmentally forcing. That is, the policy will spur the adoption or the development and adoption of a technology improvement that would otherwise not occur at the same time. A policy like the subsidizing of early investment in alternative fuels, where the fuel is expected to reach market rates as the technology matures, would be an example of a forcing policy. Two alternate policy formulations exist. First, a policy may be environmentally following, it regulates environmental intensity or environmental efficiency to a level already achievable and adopted by industry. While such a policy may protect against future backsliding, its baseline and policy scenarios would be identical, and is therefore trivial. Second, a policy may be environmentally revolutionary, stringent at such a level that compliance is only possible under a dramatic shift in the production system and not through incremental improvement. Under this case, it is unclear if the model heretofore described would be applicable.

Assumption 4: No mandatory phase out of older investments.

Assumption 5: Technology improvements are discrete.

Assumption 6: The forcing gap is less than the aircraft lifetime.

The following notation is adopted:

S: aircraft stock size (#)

L: aircraft lifetime (y)

L/S: retirement rate $\left(\#/\mathrm{y}\right)$

N: emissions per aircraft

α: emissisions improvement factor

f: forcing gap (y)

Figure 5-4 shows the impact of the policy on environmental intensity of aircraft entering service. The operation of aircraft creates an externality, in this case generic unpriced emissions, that justifies a potential policy response. Under the baseline case, aircraft enter service with an emissions profile of *N* until time t(*f*), at which point aircraft entering service are expected to have an emissions profile of *N*(1-a). Explanations for this change could include exogenous innovation, learning-by-doing, partial-internalization of the externality through other regulation, correlation with internal costs, or cost decreases of less environmentally intense substitutes and are described in Section 5.1.1. The policy case mandates a shift in environmental performance to an emissions profile of *N*(1-a) at time t(0), *f* years before this occurs in the baseline. It is assumed that this mandate shift carries with it some cost. For instance, if the underlying the cause of the (unforced) emissions change is primarily from exogenous innovation, then research and development (R&D) expenditures will likely increase to advance the technology at an increased pace. In another example, if the cause of the unforced change is primarily from the costs of substitutes, then firms will incur higher capital costs in adopting these technologies before it would otherwise be efficient to do so.

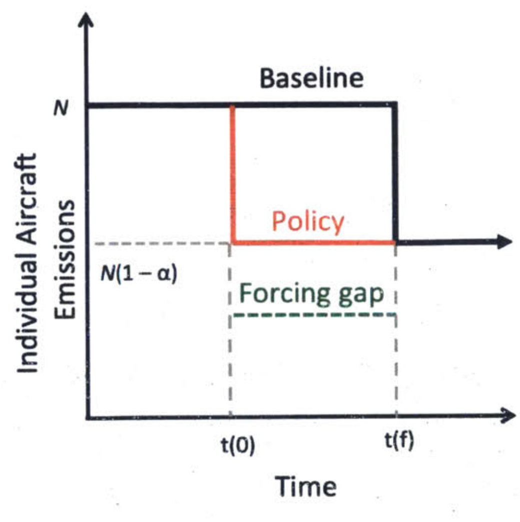


Figure 5-4 Policy impact on environmental intensity for aircraft entering service in the simple case of aircraft emissions.

While Figure 5-4 demonstrates the impact of the policy at the time of adoption of the aircraft, it does not fully explain how the policy impacts the fleet and how decisions made while $t(0) \leq t < t(f)$ impact fleet composition at later times. Following Assumption 2, Equations 5.1 and 5.2 give the fleet-wide emissions E under baseline and policy scenarios.

$$\begin{aligned} E_b(t) &= SN & 0 \leq t < f \\ &= SN\left(1 + \frac{\alpha f - \alpha t}{L}\right) & f \leq t < f + L \\ &= SN(1-\alpha) & f + L \leq t \end{aligned} \qquad 5.1$$

$$\begin{aligned} E_p(t) &= SN\left(1 - \frac{\alpha t}{L}\right) & 0 \leq t < L \\ &= SN(1-\alpha) & L \leq t \end{aligned} \qquad 5.2$$

In the baseline, emissions are constant until time f, when the aircraft entering service switch to the low-emissions type. Aircraft retire at a uniform rate of L/S, resulting in linearly decreasing emissions until the entire fleet has been replaced with low-emission aircraft. While, the rate of change in emissions occurs at the same rate in the policy case, it does so starting at an earlier time. At time L, the last high-emissions aircraft, purchased at time $-\varepsilon$, is retired and constant emissions at a new level is achieved. The fleet-wide emissions evolution is shown in Figure 5-5.

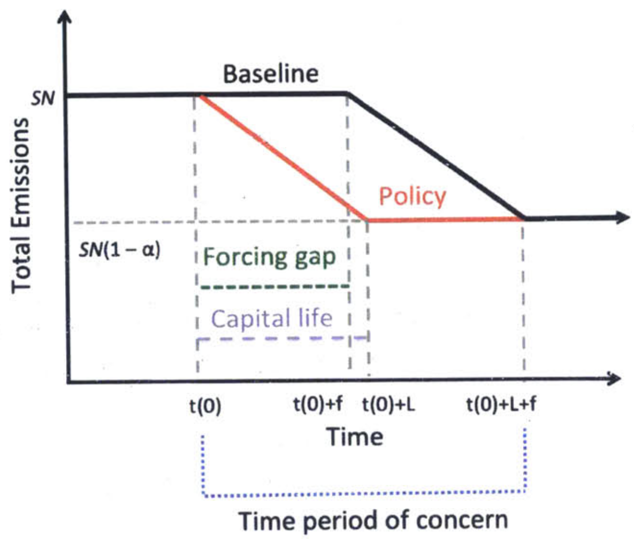


Figure 5-5 Temporal evolution of fleet-wide emissions under Baseline and Policy conditions.

The *emissions benefit B* is defined as the net difference between the two scenarios. The emissions benefit is not to be confused with the monetized benefits or the difference in environmental impact between the baseline and policy as environmental effects may lag emissions changes in time and may be non-linear. However, the emissions benefit may be a useful metric as the denominator of a Cost-Effectiveness Analysis. The emissions benefit is derived in Equation 5.3.

$$\begin{aligned}
B &= \int E_b(t) - E_p(t)\,dt \\
&= \frac{SN\alpha t^2}{2L} && 0 \le t < f \\
&= SN\alpha\left(\frac{2ft - f^2}{2L}\right) && f \le t < L \\
&= SN\alpha\left(\frac{2Lt - (t-f)^2 - L^2}{2L}\right) && L \le t < f + L \\
&= SN\alpha f && f + L \le t
\end{aligned} \tag{5.3}$$

This case study develops and highlights some important implications. First, when modeling the built environment system, the time period of concern lasts beyond the forcing gap. Thus, while by time f, all planes entering the fleet are of the low-emission type regardless of policy purchases that were made during the time $0 \leq t < f$ impact the fleet characteristics relative to the baseline beyond this point. Furthermore, the time period of concern lasts beyond the capital lifetime even though under the policy all aircraft have been replaced by time L. The relevant policy timescale here is in fact $0 \leq t < f + L$. This provides a useful benchmark for analysis design suggesting that the policy and associated baseline should be modeled for at least this time period.

Following this result, the model indicates that the OMB recommendation that the *time horizon* of the analysis should be for the lifetime of the upfront capital is insufficient. Foremost, the conception of one time horizon for an analysis is itself problematic as it fails to account for the long lifetime of environmental damage. Further, even considering the guidance as applying only to the policy influence period, the guidance may not be optimal. Following Equation 5.3, at the capital life L, only $1-f/(2L)$ percent of emissions benefit E are accounted for, regardless of stock size or initial emissions rate. While total emissions benefit will change, this finding holds even in the case of industry growth where stock size S increases consistently over time. Where analysis resources are constrained, modeling the policy timescale from 0 to L may be appropriate when $f/2L$ approaches 0.

5.1.3 Policy Influence Period: An Applied Case

This section applies the policy influence period framework to a reanalysis of the CAEP/9 aircraft noise stringency cost-benefit analysis presented in Section 4.1. Consistent with prior CAEP best practices, potential changes in aircraft noise certification stringency were analyzed over a 30-year time period. Estimates for the economic or service lifetimes of commercial aircraft range from 10 to 45 years (Baldwin and Krugman 1988, OECD 2001, Dray 2011), indicating that this analysis time horizon aligns with US regulatory guidance (OMB 2013). However, as demonstrated in Section 5.1.2, this may not be sufficient for capturing the fleet and emissions response of the airspace system. Here, the CAEP/9 analysis is reassessed using an explicit representation of the policy influence period. The effects of this modeling improvement are quantified, seeking to answer the questions (1) What impact does improving the policy influence period have on quantified environmental benefits?; (2) What impact does improving the policy influence period have on apparent policy appropriateness?; and (3) How can modelers account for the uncertainty in the policy influence period?

Policy Influence Period Estimation

From Section 5.1.2, the period of concern for the policy influence period lasts from the implementation of the policy until a time equal to the forcing gap and the capital life. Because the sample policy does not require phase-out of already certified non-compliant aircraft (grandfathering), the forcing gap is the time until under business-as-usual technology and operational projections, no aircraft would be entering the fleet at the policy-mandated noise level. Two methods are used to ex-ante estimate the forcing gap. A best-fit curve is applied to aircraft certification data taken at the time of the aircraft type entering to estimate the rate of aircraft noise improvement over time (ICAO 2010). This best-fit curve is shown in Figure 5-3. In the first estimation method, the expected last date of product delivery is determined for each aircraft type by estimating its model production life. Grimme (2008) models the production ranges for 17 jet aircraft types, finding an average production life of 27 years, an assumption adopted here. The forcing gap is the time between the start of the proposed policy and the expected delivery date of the last policy non-compliant aircraft. In the second estimation method, the forcing gap as the difference between the year of technology achieving the Chapter 4 noise limits and the year of technology achieving the mandated policy limits as predicted by a best-fit curve to the historic noise certification data. The average of the two methods is adopted for this analysis rounding up to the nearest year. The results of the two forcing gap estimation methods are shown in Table 5-1.

Using the first estimation method gives a negative forcing gap for the first stringency. This would indicate that all aircraft entering production at the time of policy implementation are already compliant with the proposed limits, resulting in no difference between the policy and the baseline. While AEDT modeled differences between the two scenarios, noise and emissions differences from the baseline were mostly within the model resolution limits, thereby suggesting a nominally small forcing gap. The aircraft economic lifetime is assumed to be 25 years, consistent with the median of literature estimates (Baldwin and Krugman 1988, OECD 2010, Dray 2011, Jiang 2013).

Table 5-1 Forcing gap estimates for CAEP/9 noise policies

Policy	***f*(1)** (years)	***f*(2)** (years)	***f*** **(mean)** (years)
Stringency 1	-4	3	0
Stringency 2	1	7	4
Stringency 3	7	11	9
Stringency 4	15	18	17
Stringency 5	27	25	26

Because f is greater than L in the case of Stringency 5, the illustrative model may not be appropriate. The forcing gap estimation suggests that the aircraft fleet will turnover in its entirety while the policy is still influencing purchaser and consumer decisions. Using the terminology of Section 5.1.1, the impact of the policy at this stringency may not be environmentally forcing and may require manufacturers and operators to undergo changes that go beyond continual marginal improvements. Evidence of these effects can be seen in the modeled fleet projections and operations for 2036 described in Section 4.1.2. The number of available compliant aircraft in the fleet Growth and Replacement model becomes small enough that several aircraft have no available substitutes resulting in large shifts in the fleet composition and number of operations as stringency is tightened. Further, because the policy impacts aircraft entering the fleet over two product development lifecycles, it is difficult to justify the simplified modeling assumptions used below to estimate emissions and costs for later years of this policy.

Environmental Benefits

Noise and emissions inputs for the years 2012-2036 are taken from AEDT2a model results as provided by the John A. Volpe National Transportation Systems Center and are identical to the modeled inputs from the original analysis. The baseline year is changed from 2006 to 2012, and all prior years of input are discarded so as only to account for emissions that would occur after policy adoption (but includes the time before policy implementation). Traffic, fleeting, and emissions models for future year scenarios are computationally and temporally intensive and involve coordination across several modeling groups. Thus, emissions projections for years beyond 2036 are developed using simplified scenarios. Adopting a simplified framework for projections allows this study to explore a range of different assumptions and to explore the significance and sensitivity of different modeling choices. As the aim of the study presented in this section is to determine whether the results of a policy cost-benefit analysis are sensitive to policy influence period modeling assumptions, the simplified scenario approach is appropriate. Further, if assumptions for the entire policy influence period are shown to have an impact on results, then the simplified assumptions presented here may provide useful and computationally efficient techniques for approaching policy analysis. Testing the efficiency, consistency and accuracy of these projection assumption techniques is the subject of future work.

Baseline emissions are linearly extrapolated into the future at a constant rate of growth to account for countervailing trends in increasing operations and increasing energy efficiency. For each policy case, emissions in the year t_0+f+L are assumed to converge to the baseline projection, and emissions for years between the last year modeled by AEDT2a and the year of convergence are linearly interpolated. The baseline noise footprint for years after 2036 is estimated by first calculating the 2026-2036 improvement in the average noise impact per person per aircraft operation at each of the 99 airports and assuming an identical percentage improvement over each consecutive 10 year period. Operations are linearly extrapolated consistent with the growth assumed in the emissions modeling. Policy noise footprints are extended using the same methodology used to extend the emissions scenarios with the baseline and policy converging in year t_0+f+L.

Air quality benefits are calculated using the APMT-Impacts Air Quality model with RSMv2 and the climate benefits are calculated using the APMT-Impacts Climate Code model v23. The models are run with emissions scenarios for both the 2012-2036 case consistent with previous CAEP methodology[11] designated as the *truncated* case and for the complete policy timescale described above designated as the *full* case. For the noise domain, the APMT-Impacts noise module determines capitalized housing value loss and then converts total cost of capital into yearly annuities using a capital recovery factor. The capital recovery factor is a function of the discount rate and the investment life, typically 30 years for houses. Because some but not all policy scenarios have policy influence periods that go beyond the 30-year investment lifetime assumed in the capital recovery factor, it is difficult to directly compare the calculated net present value of damages across the scenarios with this method. Therefore, a simple willingness-to-pay (WTP) per dB of noise reduction per person per annum damage function as given by Equation 5.4 is adopted as a reduced order model for this comparison.

$$W = 1.17B - 46.6 \qquad B \geq 45 \tag{5.4}$$

Where W is willingness-to-pay (WTP) per dB per person per annum (2006 USD) and B is the noise contour level in DNL dB(A). This model is developed from a review of appraisal values for noise damage from Nellthorp et al. (2007) converted to USD through PPP. The simple non-income differentiated approach consistently produces policy minus baseline noise results 40-60% greater than that of the APMT-Impacts noise module for CAEP/9 inputs. This may in part be because the WTP values in Nellthorp et al. (2007) are primarily from European valuation studies, and WTP for noise reduction may be higher in Europe (He 2010).

[11] In the original CAEP/9 analysis, the truncated case was run with emissions from 2006-2036. However, since costs are discounted back to the year of analysis (2012) and all scenarios have identical emissions until the implementation year (2020), not modeling the years 2006-2012 will not impact the results.

Results of the environmental benefits comparison for the CAEP/9 reanalysis are shown in Figure 5-6. Results presented are policy minus baseline environmental benefits in NPV, and therefore negative results represent an improvement over the baseline. The traditional *truncated* method underestimates policy benefits by up to 39%. Furthermore, while the rank order of the environmental benefits of the policies is unchanged, the impact of accounting for the full policy timescale is not the same across all policies. Under the truncated method, Policy 4 (S04) provides a 25% better total environmental performance than Policy 2 (S02), but, modeled over the full policy influence period, Policy 4 performs only 4% better. This occurs because countervailing air quality and climate effects offset the noise benefits in Policy 4. As explained in Section 4.1.2, stringent noise policies (Policies 4 and 5) cause several commonly used aircraft to become non-compliant, some with few or no direct available substitutions. These policies require airlines to shift some traffic to differently sized aircraft, which introduces tradeoffs on fuel efficiency and number of operations necessary to meet demand. The truncated policy influence period accounts for less than half of the total climate disbenefit and 62% of the noise benefit for Policy 4. Because there is no such countervailing impact in the Policy 2 case, the total environmental benefit for Policy 2 is closer to the total environmental benefit of Policy 4 when considering the full policy influence period.

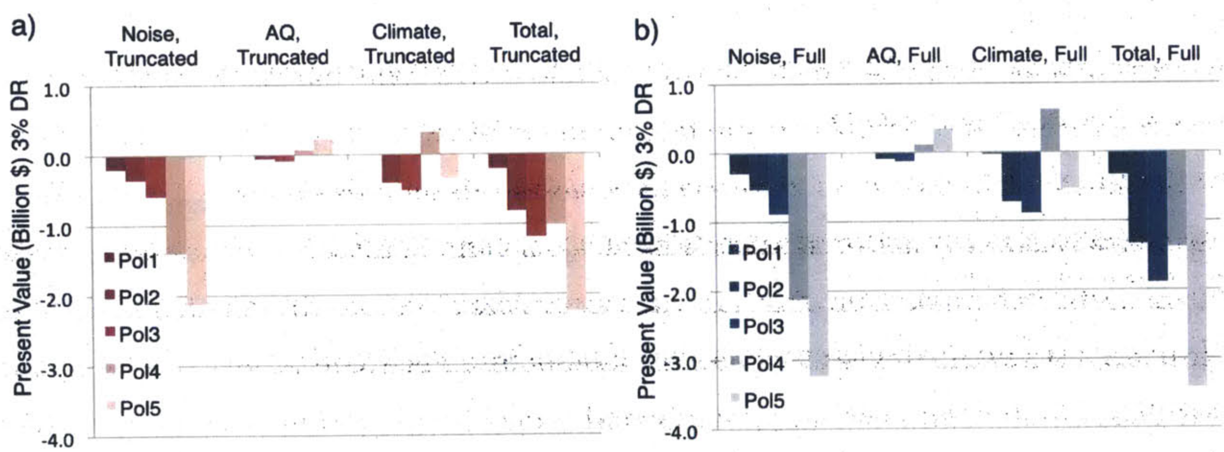


Figure 5-6 CAEP/9 re-analysis policy minus baseline environmental benefits, midrange lens, 3% discount rate, mean results only.

Industry Costs

Two sample cases are developed to projects costs through the policy influence period: C_1, recurring costs under a simple extrapolation and C_2, recurring costs with changing technology costs over time. In C_1, fuel costs and other direct operating costs are extrapolated consistent with the operations projections from the environmental modeling. Overestimating policy costs in future years is a common concern of regulatory analysis, most often occurring from failing to account for changes in technology costs (Harrington, Morgenstern, and Nelson 2000). In the original CAEP/9 study, costs for noise reduction technologies were calculated from the 1994 Aircraft Noise Design Effects (ANDES) study and assumed to be constant from 1994 through the entire policy time period (ICAO 2014). For C_2, this assumption is accepted for the time period modeled previously, but an experience curve is applied to undiscounted capital costs for all years after 2036 that assumes a 20% decrease in technology costs for each doubling of production consistent with observed trends in aviation technology costs (Yeh and Rubin 2012). Fuel costs are identical to C_1. These projections are not intended to represent most-likely cost estimates, but are used to investigate the impact of policy timescale for two reasonable distributions of costs over time. The impact of policy time scale model on cost is shown in Figure 5-7.

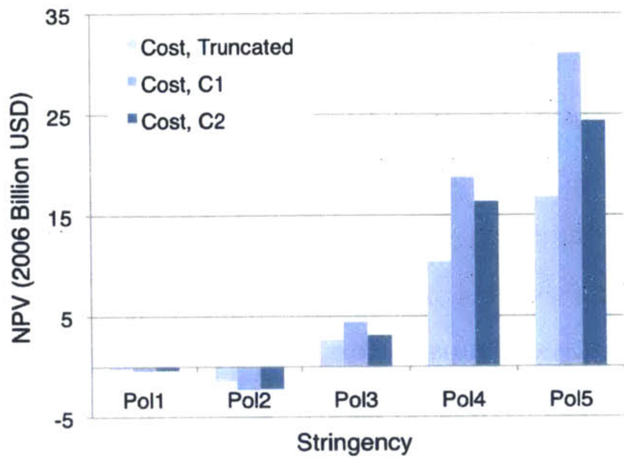


Figure 5-7 CAEP/9 Costs under constant technology costs and with an experience curve.

Cost-Benefit Analysis

The net cost benefit of the CAEP/9 reanalysis is shown in Figure 5-8. The impact of accounting for the policy timescale is non-uniform across policies and dependent upon technology cost assumptions. While the rank-order of the policies do not change, Policy 3 changes from having net cost in the truncated analysis to being cost-neutral if noise-abatement technology costs decrease from year 2036 onward when considering the full policy influence period.

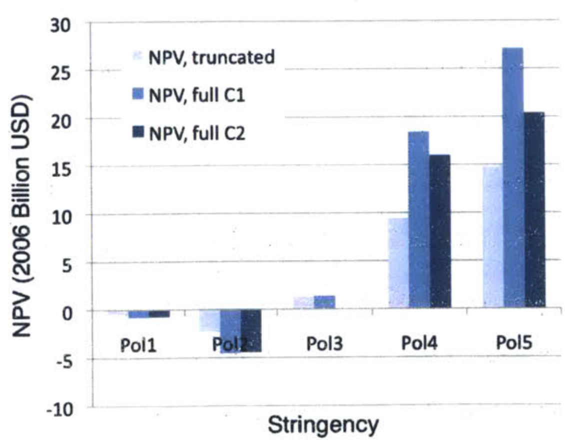


Figure 5-8 CAEP/9 Cost-Benefit re-analysis, midrange lens, 3% discount rate

Further, in the original CAEP/9 Cost-Benefit study, the policies were analyzed for a range of discount rates and policy lenses. Under different combinations of lenses and discount rates, the policy may appear to provide a different magnitude of costs in benefits. Under no combination of lenses and discount rates were Policy 4 and Policy 5 (-9 and -11 dB respectively) cost-beneficial. The performance of Policy 3, however, depends upon how one values future benefits relative current costs (discount rate) and how one considers the risk associated with the highest and lowest estimates of environmental damages (lenses). Under the midrange lens, the performance of Policy 3 is shown for a range of discount rates for both the economic and environmental costs for the truncated case in Table 5-2 and the full policy influence period under C_2 cost assumptions in Table 5-3. As shown in the tables, Policy 3 is cost-beneficial for a wider selection of discount rates when considering the full costs and benefits of the stringency.

Table 5-2 Policy 3 cost-benefit performance for a range of discount rates for the original truncated analysis. Red cells indicate a net cost (C), yellow cells indicate a net break-even (0), and green cells indicate a net benefit (B). Capital letters indicate costs or benefits greater than $3B.

		Environmental Cost DR				
		2	3	5	7	9
Economic Cost DR	3	c	c	N/A	N/A	N/A
	7	0	c	C	C	N/A
	9	b	0	c	c	C

Table 5-3 Policy 3 cost-benefit performance for a range of discount rates for the full policy-influence period analysis. Red cells indicate a net cost (C), yellow cells indicate a net break-even (0), and green cells indicate a net benefit (B). Capital letters indicate costs or benefits greater than $3B.

		Environmental Cost DR				
		2	3	5	7	9
Economic Cost DR	3	b	0	N/A	N/A	N/A
	7	b	b	c	C	N/A
	9	B	b	0	c	C

Finally, under the original truncated analysis assumptions, Policy 3 was never more cost-beneficial than Policy 2. Under the C_2 cost assumptions for the full policy-influence period analysis, Policy 3 outperforms Policy 2 for the high lens assumption at the lowest environmental discount rate (2%). Thus, the policy influence period modeling is influential for the costs, benefits, and the net policy performance of the analysis in this case. Policy 3 appears to be a societally beneficial policy for a greater range of viewpoints than it did in the initial analysis, which would make it more easily defensible in a decision-making context.

5.1.4 Policy Influence Period Conclusions and Generalizations

In this section, I have proposed a formal definition and delineation of the policy influence period: the time period over which a policy or decision impacts the physical behavior or composition of a system. When considering the appropriateness of different policy options, the analyst should attempt to consider the behavior of the system over the entire policy influence period. The policy influence period was then examined using a simplified illustrative example for a specific set of assumptions. This analysis showed that, under these assumptions, the policy influence period is important for determining the effects of a decision or policy, thereby indicating that failure to account for this time scale explicitly may impact apparent policy appropriateness. To test this, a policy that had been analyzed for only 30 years was reanalyzed using simplified assumptions to extend the analysis for the entire policy influence period. In this case, the extension of the time period was seen to change the evaluation of the policies considered, which may have implications on the decision making process.

Three important conclusions of the policy influence period developed in this section are summarized below.

First, the policy influence period is a distinct time period. Previous guidance on modeling environmental policy has urged modelers and analysts to consider the full costs and benefits of proposed policies, but has been unspecific as to how to delineate over which time horizon is important for analysis. Building on the two time period framework proposed by Mahashabde (2009), a three time period framework is developed to aid modelers, analysts and decision-makers in evaluating environmental policy. The policy influence period is the first of these time periods; it is the time period over which the policy decision governs the composition and behavior of the system. For example, in aviation environmental certification policy, it would be the duration over which the policy influences the composition of the fleet.

Next, the policy influence period is needed in policy analysis to produce valid baseline and policy projections. In assessing environmental policy, the quantity of interest is the difference in societal benefit of possible future scenarios. Typically, this is done by considering one possible future as the baseline (often a "do nothing" or "do minimum") scenario and delineating the other futures as policy or stringency options. To consider the difference between these scenarios, the system must be modeled for an appropriate length of time. (In the case of aviation environmental policy, the system might be global operations and emissions). This time period is the policy influence period, and it is a function of the system and the stringency of the proposed policies. As shown in 5.1.2, this period is not necessarily the lifetime of initial capital investment, and failure to consider the full policy influence period would mean that not all benefits (or costs) of the policy are being captured.

Finally, in practice, modeling the entire policy influence period is important to understand the quantity of interest for decision-makers. The illustrative case shown in Section 5.1.2, showed that failure to account for the entire policy influence period results in undercounting the benefits and costs of a policy. In that section, equations were developed to indicate the magnitude of emissions unaccounted for in an example policy. However, when the quantity of interest (QOI) is net benefits, the impact of the policy influence period on the QOI may not be consistent across different policies and is not determinable ex ante. This behavior can occur because of system nonlinearities, such as emissions having a non-linear impact on damages as is the case with climate; system trade-offs and co-benefits, such as fleet turnover acceleration from a noise policy providing emissions co-benefits under some regulations but not under others; or if the time projection of costs evolves at a different rate than the time projection of environmental benefits, as can be the case when learning-effects lead to reduced technology costs in the future. Thus, it is important to model decisions for the entire policy influence period to correctly account for all costs and benefits.

An omniscient analyst with no time or computational constraints would be able to use these conclusions to model the full costs and benefits of proposed policies. However, in the real world, modeling the policy influence period is fraught with uncertainty and limited resources. Thus, improving policy analysis is constrained by the ability to model the system. Further, because the impact of modeling the entire policy influence period instead of a truncated timescale is not apparent ex ante, it is important to explore the policy influence period with care. As such, the following points of guidance are developed.

1. **Develop a policy-consistent method for determining the policy influence period.**

Two parameters necessary to understand the policy influence period are the forcing gap (the time over which the policy dictates investment decisions) and the lifetime of capital investments. The forcing gap can be determined by considering the past behavior of the system. In the CAEP/9 study presented in 5.1.3, the advancement of aircraft noise performance in the past was used to estimate the forcing gap of the policy. As in this example, using past projections of performance and efficiency can be one useful metric of predicting forcing gap. Other possibilities include examining rates of technology lifetimes, projections of technology change rates, or the influence of external policies. Examples of methods to estimate these parameters for sample policy domains are presented in Table 5-4.

Table 5-4 Potential forcing gap and capital life parameters for various environmental policies

Policy Domain	Forcing Gap (f)	Capital Lifetime (L)
Aircraft Emissions Certification	Historical rate of improvement in emission efficiency; time period between last policy stringency increase	Operational lifetime of commercial aircraft
Airport Noise Land Use Planning[1]	Rate of change in noise contour extent	Housing lifetime or housing occupancy duration
General Aviation Lead Emissions	Rate of past GA technology turnover	Operational lifetime of a piston-driven aircraft
Power Plant Emissions	Plant-type specific historical rate of emission efficiency; grid-wide historical emission efficiency; rate of policy change; rate of change of technology alternative costs	Plant service life
Fire Suppression	Historical technology turnover rate, regulatory timeline	Existence lifetime of the fire suppression system; service life of the protected building or investment
Mercury Emissions from Small Scale Gold Mining	Global rate of change in mining economy size with respect to economic development, country-specific rate of legislation, technology transfer rates	Employment duration, capital life of mining equipment; expected existence life of gold-bearing mineral vein

[1] In the land-use study presented in 4.2, the capital lifetime is reflected in the Capital Recovery Factor used to estimate Willingness-to-Pay for noise abatement. The analysis did not consider the forcing gap. If an airport's noise footprint were expected to shrink in the future, the analysis may overestimate benefits where residences insulated today are not exposed to noise 5 years into the future. In this case, the framework developed here would present a methodological improvement.

The forcing gap and the capital lifetime may not be the only relevant parameters for understanding the policy lifetime, and care should be taken to ensure that the framework developed here is applicable for the policy analysis. This heuristic for estimating the policy influence period may, in particular, poorly serve policies that induce revolutionary changes. Further, where policies lead to investments and decisions that are irreversible or that have feedbacks that influence future decisions in other domains, modelers should pay attention to the impact these effects may have on the policy influence period.

2. **Where capital lifetimes are long relative to the policy stringency, assuming the capital lifetime represents the policy influence period may be sufficient.**

Cost-benefit guidance, such as the regulatory guidance promulgated by the US Federal Government, suggests modeling the policy over the length of initial investments. As developed in Equation 5.3, under the assumptions of an idealized case, the percent of physical emissions or externality outputs not captured under this assumption is a function of ratio of the forcing gap (f) to the capital lifetime (L). This means that where $f << L$, the modeler gains little additional benefit for each additional year beyond L that is modeled. For example, historical aircraft noise certification levels suggested that Stringency 1 in the CAEP/9 Noise Analysis would be technology forcing for at most 3 years, but aircraft service lifetimes are an order of magnitude longer. As shown in the reanalysis in Section 5.1.3, expanding the modeled policy influence period for this stringency resulted in changes in environmental benefits of less than 5%. Thus, in these cases, especially where modeling additional years is time or computationally expensive or introduces significant uncertainty, modeling the policy only for the capital lifetime may be appropriate.

3. Develop scenarios to test different policy influence period projections.

An epigram attributed to Niels Bohr is "Prediction is difficult, especially about the future." This is especially true in modeling the expected outcomes of complex interdependent decisions. Use of scenarios to project the future provides two benefits: it allows modelers to test initial assumptions and it provides insight into the impact of a policy under different future conditions. Simplified models can be used to explore the relationships between the policy, investment decisions and behaviors, and system performance, such as was done with the Illustrative Case in Section 5.1.2. The simplified model can then be used to understand the important drivers of the policy influence period to determine over what time horizon with a more complex or higher fidelity model. The use of the Representative Concentration Pathways (RCPs) and Shared Socioeconomic Pathways (SSPs) in the climate model developed in Section 3.3 also represents a scenario approach to modeling.

In several policy contexts, there may be system feedbacks and nonlinearities that depend on the stringency of policy. In these cases, the policy influence period may extend beyond the forcing gap and capital lifetime of the primary entity being regulated. In these cases, while the illustrative example developed in Section 5.1.2 may not be directly applicable, the approach may still be adopted to understand the length of the policy influence period. Simple system dynamics models that examine the strength and effect of feedback loops on decisions and investment decisions can provide insight into the appropriate time horizon for the policy influence period. Systems dynamics models have been used to explore environmental and business policy effects over time in fields as diverse as climate policy (Sterman 2011), forest management (Collins et al. 2013, and water dynamics (Simonovic 2002).

4. Explore the sensitivity of results to different influential variables.

The magnitude of impacts from a given policy is subject to scientific and economic uncertainty both aleatoric and epistemic. Further, several system assumptions may not be time invariant. Therefore, while some assumptions may be appropriate for static analyses, they may not be valid for the length of the policy influence period. Technology costs and substitution costs may be particularly dynamic over time. For example, in the sample policy case, assumptions of noise technology costs were assumed to be constant overtime. However, the rate of technology improvement and of associated costs, when modeled over time, was influential in determining the net cost or benefit of different noise policies. This finding is consistent with regulatory policy ex post assessments that have found that technology costs have been consistently overestimated (Harrington, Morgenstern, and Nelson 2000).

Only a subset of uncertain parameters may be important. In the CAEP/9 analysis, variable were grouped into lenses representing meaningful combinations of parameter choices to represent an array of policy viewpoints. Additional lenses, such as the illustrative air quality lens, were also analyzed to understand the sensitivity of results to model assumptions and limitations. This approach can be extended to other policy analyses. Furthermore, the work presented in this section suggests applying the lens approach to policy costs and not just environmental impacts as a methodological improvement. This finding is consistent with work by He (2014) that found that implementation costs for the CAEP/9 noise analysis were the largest driver of output variability.

5. **Summarize results to highlight important influential factors and parameters.**

Even after adopting the lens approach to organize uncertain parameters, there exist a large number of influential modeling options. For example, in considering the CAEP/9 Noise analysis in Section 4.1, the study examined five possible policy stringencies, across five discount rates, under three environmental lenses and one illustrative lens. The reanalysis in Section 5.1.3 added to this two different policy influence period models and two cost projection assumptions. This approach produces over 3000 unique policy cost-benefit results. If policy-makers are interested in additional metrics such as distributional equity, the amount of data can become untenable for decision-making. Thus, it is important to summarize results in a way that highlights resulting trends and their driving factors.

In conclusion, Section 5.1 developed and examined the idea of the policy influence period, a timescale in policy analysis over which the policy choice is driving behavior of the system. By drawing attention to this timescale, modelers will be better able to understand over what time period they need to model technical systems and will be able to capture the full extent of policy costs and benefits. Finally, this section demonstrated that adopting an explicit policy influence period can influence the apparent appropriateness of a policy. This section looked at the policy influence period in isolation, however there may be important cross-linkages between the policy influence period and other policy timescales. Policy timescale interactions are explored in Section 5.4.

5.2 Environmental Impacts Lifetime

The environmental impacts lifetime is the timescale over which the effects attributable to policy-regulated activities persist in the environment. In the case of aircraft-induced air pollution, the policy influence period determines when and how aircraft emit harmful pollutants and precursors whereas the environmental impacts lifetime determines how long these pollutants last in the atmosphere and impact human health and welfare. For climate change, the policy influence period is the timescale over which greenhouse gas and climate forcer emission levels are effected whereas the environmental impacts lifetime is the timescale over which these climate forcers influence radiative forcing and temperature change.

The environmental impacts lifetime is governed by earth-system processes and can range from short ephemeral timescales (aircraft noise would cease to annoy a local community mere minutes after the last flight) to decadal (CH_4 effects on atmospheric radiative forcing), centurial (CO_2 impacts on climate) and even millennial timescales (radioactive waste decay). The earth-system processes important for the environmental impacts lifetime may be modeled as several sub-system level processes. In the case of climate change, the atmospheric residence time of CO_2 is a function of exchanges across several earth, ocean, and atmospheric carbon pools; the effect of CO_2 concentration on global surface temperature is then a function of heat exchange between then atmosphere and ocean.

While the environmental impacts lifetime may be more readily understandable than the policy influence period, it is not necessarily easier to model. Projections of the effects of environmental stressors may reflect significant aleatoric and epistemic uncertainty, resulting in a wide range of potential impacts. Further, some environmental pollutants may have complex biogeochemical cycles resulting in spatially and temporally heterogeneous impacts over long timescales. Understanding the environmental impact lifetime associated with a policy is important for considering the full extent of environmental costs and benefits. For example, a weakness of the leaded fuel policy assessment presented in Section 4.3 is that it does not consider how current lead emissions will become legacy soil concentrations in the future, and may further contribute to health and welfare costs.

5.3 Valuation Timescale

The valuation timescale is the time over which society values the impact of a policy. Many decisions entail consequences that are heterogeneous in time. Taking an extravagant vacation may yield short-term enjoyment and rest but it may require long working hours and forgoing future opportunities. Lying on the beach in the sun may produce a desirable tan in the near-term, but it may produce undesirable wrinkles, blemishes and a chance of skin cancer in the future. When people are fully informed about the costs and benefits of their options, decisions are made based on how far into the future costs and benefits occur and how they are valued over time. Decisions made in the public sphere must also weigh when costs and benefits occur and face the additional challenge that preferences may vary significantly from person to person.

Explicitly highlighting the valuation timescale, how current and future benefits are measured against each other, can ensure consistency in environmental policy analysis. The valuation timescale consists of three facets: the time horizon, the future utility gained or lost from an action, and the weight given to future utility. The time horizon refers to the total length of time over which costs and benefits of a decision are allowed to accrue or to the end date of the same period (Klos, Weber, and Weber 2005). Finite time horizons are easily understood in the context of private-sphere decisions: a young professor will consider a study's ability to garner citations over a 7-year timespan to better her chances at tenure; a new parent will make saving and investment decisions based on his ability to pay for a college education in 17 years and retirement in 30 years; and men and women will make decisions on intimacy and exclusivity based on how long they expect a relationship to last (Waite and Joyner 2005). Benefits and costs beyond this time horizon may not be considered (or are considered secondarily) in determining a course of action.

Public decisions must balance near-term objectives with long-lived impacts as policies may effect multiple generations. For example, in considering groundwater management, policymakers must not only ensure immediate reliability and access to water for their constituents but also must consider the effect their decisions have on the long-term sustainability of their community and the hydrological cycle. For the latter consideration, an infinite or essentially infinite time-horizon may be appropriate.

Time horizons are embedded in environmental policy analysis and can be influential in determining apparent policy appropriateness. For life-cycle assessments of alternative fuels, changes in land-use related carbon stocks are often amortized over the expected lifetime of the fuel production plant (Caiazzo et al. 2014). The Global Warming Potential (GWP) and Global Temperature Potential (GTP) are two metrics used to compare the climate impact of different greenhouse gases that have embedded time horizons. As discussed in Section 2.2.3, the GWP measures the climate impact of a given emission by considering its integrated radiative forcing over a time horizon (typically 50, 100, or 500 years) relative to the radiative forcing for an equivalent emission of CO_2. The GTP measures the climate impact of a given emission by considering its expected temperature change at the end of time horizon (typically 20, 50, or 100 years). The choice of time horizon embedded in the analysis can impact whether a policy or technology choice provides maximum welfare or utility or even whether a decision yields a net benefit or cost (Tanaka et al. 2009).

The second facet of the valuation timescale is the expected amount of future utility from an action. The impact of a given policy choice will depend on when it is implemented. The damage function of the climate model from Section 3.3 demonstrates this concept. The loss in welfare from a 1-degree temperature change is related to the total potential welfare generated at the same time absent the change in temperature. Because global productivity is expected to increase in the future under most projections, a 1-degree increase in temperature is estimated to create more disutility in the future than it does today. Further, greenhouse gas emissions will impact radiative forcing and temperature change differently depending on background gas concentrations, meaning that timing of emissions will introduce further heterogeneities in outcome utility (Edwards and Trancik 2014). Conversely, as welfare increases in the future, the marginal return of one monetary unit decreases, meaning that benefits of the same magnitude confer less utility in the future. In addition to the change in utility function over time, the expected amount of future utility will be influenced by uncertainty and opportunity costs of the decision (Frederick 1999). Klos, Weber and Weber (2005) explain the impact of uncertainty on valuation time period by considering an investment decision that gives an independent 50% chance to win $200 dollars or lose $100 dollars each year. The probability of losing money over ten years is about 18%, but that risk decreases to less than 1% over 50 years. Thus, utility over time may depend on uncertainty and risk. Opportunity costs refer to the investment opportunities foregone when policy resources are investable and finite.

The final facet of the valuation timescale is the weight given to future utility. People have a demonstrated preference for, all things being equal, near-term rewards. Thus weight is often given to utility gained earlier as a result of decision[12]. Further, when decisions are long lived, policy-makers must consider how to weight the utility of future generations. Discounting utility of future generations is a matter of how much the current generation cares (or should care) about future generations and how much decision-makers have a political or ethical mandate to represent future generations as well as their current electorate.

A time horizon can be incorporated directly into a policy analysis by selecting a finite time period to monetize or assess costs and benefits or, for an infinite time horizon, modeling costs and benefits until a majority have been assessed and a clear distinction can be made of which decision yields the greatest net benefit.

[12] There is debate whether pure time preference should be reflected in public decision-making at all. Some economists believe that time preference is irrational – a function of either a weakness of will or a cognitive bias (Feldstein 1964). Correcting this irrationality by not accounting for preference in discount rate would be akin to the justification for seatbelt laws in that people cognitively do not account for low-probability high impact events. Ramsay (1928) goes so far as to say that weighing future utility differently to account for time preference is, "a practice which is ethically indefensible and arises merely from the weakness of the imagination."

Opportunity costs, uncertainty, and the amount of future utility as well as the weight given to future utility are accounted for by use of a discount rate. The selection of an appropriate discount rate is a matter of preference and point-of-view based on ethical, normative, and positive arguments (Sunstein and Weisbach 2009; Weimer and Greenberg 2004). Far from being settled, choice of discount rate is still a popular and occasionally heated debate (Nordhaus 2007; Stern and Taylor 2008; Gollier and Weitzman 2010). It is not the objective of this work to decide an optimal discount rate or to discuss the ethics of discounting[13]. However, this section as well as the discount rate analyses in Section 4.1.5 provide useful framework for applying discount rates in practice.

First, assessing policies under a range of discount rates will indicate how sensitive results are to choice of valuation timescale. Second, by communicating the rationale for discounting and the implications of a range of discount rates, the analysis will suggest how different stakeholders consider the costs and benefits of each policy. For example, a policy that has a net benefit under low discount rates but a net cost under high discount rates may find support amongst constituent communities that strongly consider intergenerational equity but may be opposed by industries with a focus on near-term returns. Third, because discount rate is ultimately partially a matter of policy-maker preference, use of a range of discount rates in the analysis will provide more value by making the results applicable to more people.

[13] The discounted utility model explicitly assumes that the overall value of a stream of consequences is equal to the sum of discounted costs and benefits. While this may appear reasonable on its face, it may lead to decisions that are not well supported. For instance, the discounted utility model would consider a tremendously long but nearly flat roller coaster and a short but incredibly steep roller coaster to be indistinguishable. The model suggests that an individual would be indifferent between being a manic-depressive and having a more normally stable mood (Frederick 1999). Likewise, without a corrective factor for distributional equity, a discounted utility model would find that creating a raffle in which everyone was forced to play with the result that the Commissioner of the NFL wins $13 billion dollars today while everyone else lost $10 in a year would be societally beneficial. This work does not propose to test the implications or ethics of the discounted utility model, but it does at least raise the consideration that it may result in suboptimal decisions and at the very least would be a bad model to use indiscriminately to design an amusement park.

5.4 Timescale Interactions

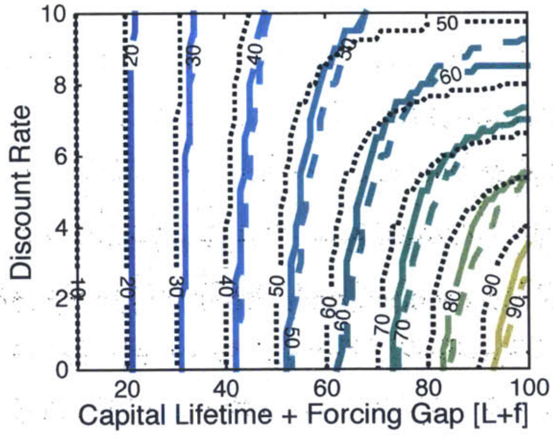
This chapter has defined and described three time periods important in policy in an attempt to identify influential drivers of system behavior and to improve analysis. This work, to this point, has mostly examined these timescales independently. However, the timescales may interact in ways that effect what the primary drivers of the analysis are. For a simple example, consider a case where a decision-making body has a very short valuation timescale (i.e. it is only concerned with the immediate result of a policy). In this case, a modeler would not need to model the entire policy influence period or the environmental impacts lifetime, because these results would occur beyond the valuation time period, only the immediate costs and benefits of the policy would need to be modeled. Modeling the entire policy influence period and considering the environmental impact lifetime correctly would still lead analysts to the (same) appropriate result, but correctly identifying the timescales that are important could save resources and make the analysis faster, simpler, and less costly.

5.4.1 Valuation Timescale Interaction

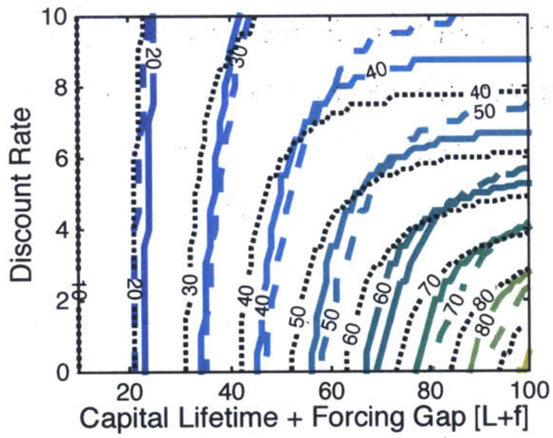
The illustrative model of policy presented in Section 5.1.2 is again used to understand the interaction between the policy influence period, the environmental impact lifetime, and the valuation timescale. In the illustrative model, a policy that dictates some improvement in environmental efficiency is enacted at time t_0. It is assumed that, without such a policy, the regulated emitter would have reach this mandated environmental efficiency in f years, where f is the called the forcing gap of the policy. The analysis in Section 5.1.2 showed that, assuming no system feedbacks, the policy would need to be modeled for $f+L$ years, where L is the relevant capital lifetime. However, it may not be necessary to capture all analysis years if the emissions in those years are vanishingly small. Emissions at the tale end of the policy influence period may be of relative unimportance especially when a discount rate is applied. Thus, while emissions may continue until t_0+f+L, the cost of forecasting later years may outweigh the additional information provided by those forecasts.

Figure 5-9 shows the interaction between the policy influence period and the valuation timescale when a constant discount rate is applied to future emissions. Contour lines reflect the number of years of the policy influence period necessary to model (a) 99% (b) 95% and (c) 90% of the total relevant difference in emissions between the policy and the baseline, B (Equation 5.3). The solid lines are for policies with a forcing gap of 10 years and the dashed lines are for policies with a forcing gap of 20 years. For the range of discount rates and policy influence period times considered, there is little difference between the behavior of policies with different forcing gaps at the 99% capture rate. For low discount rates (0-2%) and for short policy influence periods (10-40 years), 99% capture occurs over the first ~98% of the policy influence period. For these cases, the interaction of the timescales is minimal. However, interactions become more significant at higher discount rates and longer policy influence periods. At a 7% discount rate, modeling the policy impacts over 60 years will capture 99% of all relevant emissions even for policies expected to be influential for over 120 years.

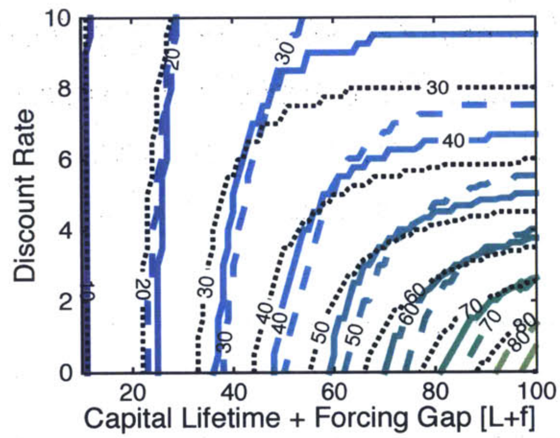
If the quantity of interest for decision makers is relaxed to 95% of total emissions, further gains in projection efficiency can be gained. For a policy that influences system behavior for 100 years, one would have to model the policy for 83 years to capture 99% of damages but would need only to model 59 years to capture 95% of the damages. Results are relatively consistent across different forcing gap time lengths. For larger forcing gaps, fewer years of modeling are necessary to achieve 90 – 95% of the policy benefit at low discount rates, but more years are necessary at high discount rates for the same total policy influence period.



(a) 99% Emissions Captured



(b) 95% Emissions Captured



(c) 90% Emissions Captured

Figure 5-9 Years necessary to model the policy influence period to capture a necessary quantity of emissions for $f = 10$ (solid colored lines, $f = 20$ (dashed colored lines), and a constant function of length $L+f$ (black dotted lines)

Finally, the relevant emissions profiles used here were modeled as the emissions difference between the policy and baseline using the illustrative model of a uniform fleet of emitters with no phase-out. The dotted black lines show the impact of a discount rate acting on a constant function for the length of the policy influence period ($B = h(t) \mid t_0 < t < t_0+L+f$). While the interactions diverge at high discount rates and long policy influence periods, the contour behavior is consistent at discount rates less than 5. This property is useful. It suggests that modelers could use the geometric series for the appropriate discount rate alone as a heuristic to estimate the length of policy influence period to be modeled to efficiently capture an upper bound of the modeling timescale for the policy influence period.

5.4.2 Environmental Impacts Lifetime Interactions

As described in Section 5.2, emissions, releases, detritus that occur as the result of an externality can effect the environment and human health for long into the future. In the cases where long-lived pollutants are of interest, policy-makers must consider modeling choices for both the policy impact period and the environmental impacts lifetime. The policy benefits, where they system behavior is affected for the policy influence period ($L+f$) and emissions impact the environment for the environmental impact lifetime (E), can be represented as:

$$\sum_{i=1}^{L+f} \sum_{j=i}^{L+f+E} b_i g(j-i+1) \qquad 5.5$$

where b_i is the emissions benefit for each year (from 5.3) of the policy influence period i and $g(t)$ is the evolution of each unit of emission overtime with $g(\mathrm{t})$ being nonzero $0 < \mathrm{t} < E$.

This benefits formulation as written makes the assumption that g and b are independent. This is not necessarily the case, and in many cases, may be a poor assumption as $g(\mathrm{t})$ can be dependent on b in both magnitude and extent. For instance, the residence time of an emission of a given atmospheric species may be dependent on the background atmospheric concentration of that species or the magnitude of damages from each additional environmental stressor may be non-linear. Likewise, b may also depend on $g(\mathrm{t})$, where environmental conditions lead to feedbacks that influence the behavior of actors in the system. For example, health impacts from environmental degradation may hurt the economy through the labor sector, which may harm the rate of technological progress. Nevertheless, the assumption if independence is useful when appropriate, such as in using a linear impulse response function to model CO_2 concentrations from small emission deltas.

A further benefit to this formulation is that if g and b are independent, then the effect of truncating the policy influence period is independent from the effect of truncating modeling the environmental impacts lifetime. This can be shown by calculating the percentage of benefits unaccounted for by failing to account for either timescale in its entirety. For example, the percentage of benefits unaccounted for by truncating the policy influence period is shown in Equation 5.6 where T is any arbitrary time before the full policy influence period.

$$\%B = \frac{\sum_{i=1}^{T} \sum_{j=i}^{l+f+E} b_i g(j-i+1)}{\sum_{i=1}^{l+f} \sum_{j=i}^{l+f+E} b_i g(j-i+1)} \quad \text{for} \quad u = j - i \qquad 5.6$$

$$j = u + i$$

$$\%B = \frac{\sum_{i=1}^{T} b_i \sum_{u=0}^{E} g(u+1)}{\sum_{i=1}^{l+f} b_i \sum_{u=0}^{E} g(u+1)}$$

$$\%B = \frac{\sum_{i=1}^{T} b_i}{\sum_{i=1}^{l+f} b_i}$$

Equation 5.6 is useful in that it shows that if the policy influence period and the environmental lifetime are independent, then the impact of modeling choices for each lifetime are also independent. This finding indicates that the conclusions and takeaways of Section 5.1.4 are applicable regardless of the lifetime of the environmental degradation.

5.5 Chapter Summary

This chapter presented a framework for considering environmental policy as being governed by three timescales: the policy influence period, the environmental impact lifetime, and the valuation timescale. The policy influence period is the length of time over which specific policies affect decision-making and capital stock composition. For many environmental policies, this timescale lasts beyond the lifetime of initial capital investments because the policy will impact investment decisions for several years. Failure to consider the entire policy influence period may result in significantly undercounting the costs and benefits of a proposed policy, which may make a beneficial policy seem cost-prohibitive or may indicate that a sub-optimal policy is welfare maximizing. This chapter presented both an illustrative model and policy case study to examine how costs and benefits may be undercounted and demonstrate the importance of considering the entire policy influence period. For policies that mandate marginal improvements in environmental efficiency, several generalizations were developed that would aid in modeling the policy influence period for a variety of policy domains.

The environmental impact lifetime and the valuation timescale are also important in modeling environmental policies. The environmental impact lifetime is the lifetime of the stressor on the environment, which may include long-range feedbacks and cycling. The valuation timescale is the time period over which individuals or societies care about environmental damages. The valuation timescale is often considered by discounting future damages relative to more immediate damages based on positivist or ethicist principles including the positive rate of time-preference, the market rate-of-return and opportunity costs, uncertainty and intergenerational equity.

This chapter's primary focus was the policy influence period. The policy influence period is the time over which a decision has an impact on capital stock composition and stakeholder actions, and in policy analysis, it is the period of time over which one should model the behavior of the domain of interest. The chapter demonstrated that the policy influence period, a timescale that is often considered only implicitly in the analysis, has important implications as to the apparent cost-benefit of policy options. The chapter showed that careful ex ante investigation of the policy forcing gap and the lifetime of capital investment can improve policy analysis, and that understanding the interaction the policy influence period has with the valuation timescale through the discount rate can provide insight and save analysis resources.

Chapter 6

Conclusions and Future Work

The primary focus of this work was to further the state and understanding of aviation environmental policy analysis. This thesis provided improved impact modeling tools, particularly in the domains of aviation noise, climate change, and modeling of General Aviation lead emissions. Next, this thesis demonstrated aviation environmental policy analysis for three distinct cases: an international aircraft noise stringency, land-use policies in the vicinity of commercial airports to mitigate the impacts of aviation noise, and the control of lead emissions from piston-driven aircraft. Finally, this thesis addressed issues of timescale, identifying three timescales for environmental policy analysis and highlighting how treating these timescales appropriately is important for estimating the full costs and benefits of different policies. This chapter presents concluding thoughts summarizing the work presented and identifies opportunities for future work.

6.1 Summary and Conclusions

Assessments of the full environmental costs and benefits of transportation projects are constrained by limitations of the tools utilized to model environmental impacts. Transportation noise causes several welfare impacts that are recognizable and attributable to the noise environment. For example, people exposed to high levels of aircraft noise (above 75 dB DNL), are often highly annoyed by noise such that they consider it the greatest deleterious component of their local environment. The perceivable and attributable impacts of noise associated with the environment will then impact the amount people are willing to pay for housing. A survey of hedonic pricing studies can then be used to develop a willingness-to-pay for noise abatement: the amenity cost of noise exposure. However, recent research indicates that noise exposure may be detrimental to long-term health, and, in particular, can lead to high blood pressure, heart attack, and stroke. This thesis developed a model to estimate the health costs of noise considering impacts on mortality, lost wages, and direct medical expenses. The model utilizes local baseline health statistics and a meta-analysis of studies relating noise exposure to increased relative risk or odds ratios of health incidents to project the health impacts and cost of aircraft-related noise exposure.

In addition, this thesis addressed shortcomings in reduced-order climate impacts modeling from aviation emissions. Building on the model of Marais et al. (2008), Mahashabde (2009), and Wolfe (2012), this thesis further developed an impulse-response function based tool to project damages related to climate-forcing aircraft emissions. Improvements to the tool include updating short-lived climate forcer (SLCF) parameters and improving SLCF uncertainty modeling, incorporating new projections of future anthropogenic background emissions, and expanding the tool to be applicable for seventeen fuel feedstocks and incorporating lifecycle emissions. The alternative fuel component of the tool is important because it allows the effect of fuel choice on radiative forcing, temperature, and societal climate damages to be estimated as a function of time.

The next component of this thesis was utilizing these models and other assessment tools to perform aviation environmental policy analysis. A cost-benefit analysis of an aircraft noise certification stringency increase demonstrated the importance of considering environmental trade-offs in policy analysis. Five stringency scenarios were considered ranging from -3 EPNdB to -11 EPNdB increases from current certification limits. The analysis showed that moderate increases in certification stringency lead to cost-beneficial improvements in air quality, climate, and noise performance, but as the policy became more stringent, climate, air quality, trade-offs arose and the environmental benefits did not outweigh the operational costs associated with the policy implementation. A -7 EPNdB change in certification stringency was cost-beneficial under certain assumptions including evaluating future environmental benefits with a low discount rate or considering worst-case reasonable assumptions for the impacts of noise but had a net cost under other assumptions. The analysis highlighted the importance of modeling uncertainty and communicating results that represent the spectrum of stakeholder and policy-maker preference.

A second policy analysis examined the costs and benefits of land-use policies near US commercial airports. Airports undertake land acquisition and residential sound-proofing projects to mitigate the impact of aircraft noise on local communities. This study finds that the health impacts of aviation noise are 61 - 64% of the annoyance or amenity impact of noise when mortalities are monetized using the DOT VSL approach and 39 - 41% of the housing impact using a Value of Life Years Lost approach. Even when the health costs of noise are considered, insulation projects are on average only cost-beneficial when the aviation noise levels are above 75dB DNL (or above 65dB DNL in areas with median income levels above $60,000), and that, on average, land-acquisition projects are not cost-beneficial at noise levels less than 85db DNL. However, this work suggests that the costs of the projects are often the same order of magnitude as the benefits. Further, if a right to quiet below 65 dB DNL is established, there may be no policy that is capable of mitigating the high levels of noise found near airports at lower cost.

An assessment of the costs of leaded fuels for piston-driven aircraft estimated that mean and median annual damages attributable to lifetime lost earnings are $1.06 and $0.60 billion respectively. Economy-wide impacts of IQ-deficits on productivity and labor increase expected damages by 54%. Damages are sensitive to background lead concentrations; as anthropogenic emissions have decreased over the past two decades, the damages attributable to aviation lead have increased by nearly an order of magnitude. The monetary impact of General Aviation Lead emissions on the environment is $4.30 per gallon of fuel burned, 11 times the per unit fuel estimated climate costs. Command-and-control regulation of leaded fuels, leaded fuel taxes, and cap-and-trade legislation may help limit the environmental damages from aviation lead emissions, but without a drop-in unleaded fuel replacement, policies may introduce economic inefficiencies.

A separate component of this thesis focused on understanding the role of different timescales in environmental policy and how these timescales can be accounted for in policy analysis. This thesis identified three timescales important for policy analysis: the policy influence period, the environmental lifetime, and the valuation timescale. The policy influence period is the period over which a policy or regulation effects investment or operational decisions and/or the composition of related capital stock. Current guidance as to how to account for the policy influence period is often incomplete or inadequate, suggesting policies only need to be modeled for the lifetime of initial capital investment. However, policies may influence investment decisions for an extended period of time, and the capital lifetime of downstream purchases will further effect net costs and benefits. Using the aircraft noise certification stringency analysis as a sample case, it was demonstrated that failure to explicitly model the entire policy influence period can result in undercounting of policy costs and benefits and can impact the apparent appropriateness of a decision.

This thesis identified five methodological considerations for improving modeling the policy influence period in policy analysis. First, develop a policy-consistent method for determining the policy influence period; two properties of a policy, the forcing gap and the lifetime of capital stock, can be used to estimate the timescale of the policy influence period. Second, where the forcing gap is sufficiently small compared to the lifetime of initial capital investment, the lifetime of capital is an appropriate length of time to model the policy influence period. Further, developing scenarios for epistemic unknowns such as future economic conditions and technology developments and utilizing uncertainty assessments for uncertain model parameters is important to understand the reliability and risk associated with different policy choices. Finally, relevant assumptions for the policy influence period and their associated results should be summarized and communicated to highlight influential modeling choices. Following this guidance will insure that modeling choices for the policy influence period are made explicitly, thereby limiting the potential for undercounting policy costs and benefits.

6.2 Future Work

Four broad areas of future work have been identified through the research presented in this thesis: improvement of reduced-order environmental assessment tools, development of atmospheric lead modeling across several spatial scales, explicit methodological consideration of implicit decisions made in policy analysis over time, and a more rigorous approach to communicating policy analysis.

The climate, air quality, and climate tools discussed in this thesis are the culmination of research and development work from multiple stakeholders. However, as scientific understanding of the impacts associated with aircraft noise and emissions continues to improve and change, these models will require further development. Further research on the amenity and health costs of noise should be pursued to advance the usability of the code in the US and abroad. In particular, studies of the impacts of noise in developing countries where aviation is expected to grow significantly over the next few decades and many people live in the vicinity of a major airport will help limit benefit transfer errors in the code. Additional studies on noise impacts on health will improve the tool's ability to estimate impacts across the range of scientific uncertainty. For the climate model, better constraining the uncertainty associated with contrails and aviation induced cloudiness as well as better understanding of the relationship between changing fleet composition and expected impacts will improve projections of aviation's impact on the global climate.

The work presented in this thesis has identified that the marginal damage of a unit of lead in the atmosphere is highly sensitive to background concentration.

Because average background concentrations of lead have decreased by over an order of magnitude since the 1980's and emissions from other sources are expected to continue to decrease, an up-to-date and complete inventory of lead sources to the atmosphere is necessary to understand aviation's contribution to lead-related environmental damages. Further, as a mobile source emitting lead at altitude, aircraft may contribute to background lead concentrations that are not distinguishable by air quality monitors. A modeling approach should be developed that is appropriate for accurately modeling lead concentrations at several spatial scales to better understand the combined impact of near-airport and more distant aviation-induced lead concentrations.

Identifying the timescales that impact policy analysis helps to explicate modeling choices that have an impact on the final result. Modelers may need to make many assumptions to make a complex analysis problem tractable given time and resource constraints. Policy effects that occur over different time lengths may be especially susceptible to simplifying assumptions given their associated epistemic and aleatoric uncertainties. Over time, these assumptions can be embedded in a modeling methodology, and their impacts on the assessment of a policy are unquantified. The approach take here to distinguish the policy impact period can be generalized to other time-dependent assumptions. Recent research has identified how timescales become embedded in decision-making metrics (Dorbian et al. 2011, Azar and Johansson 2012, Edwards and Trancik 2014). In aviation environmental policy, the impact of assumptions for alternative fuel plant lifetimes, length of home ownership near an airport, and CO_2 equivalent values for fuel processing may all influence the apparent cost-benefit of a project or policy.

Finally, research in decision-making analysis is only useful when it is communicated to and understood by those tasked with making decisions or to those providing oversight. Thus, continued effort must be placed in communicating results and incorporating methodological changes into environmental policy assessment. Experience thus far in implementing uncertainty assessment, environmental lenses, and covering a range of discount rates in analyses has indicated that the level of detail and information may need to be further distilled to improve communication, especially among stakeholders who are unfamiliar with the modeling tools or the modeling process. Incorporating additional concepts like reliability and risk may provide a better understanding of the impact of a policy, but they may not add value to the policy making process if their implications are not understood by decision-makers.

Appendix A
London Airport Noise Analysis

Research presented in this section was conducted in collaboration with John Kramer and Steven R. H. Barrett.

A.1 Introduction

London's Heathrow Airport (LHR) is a global aviation hub, which today is approaching operating at its full air traffic movement (ATM) capacity. London's Gatwick Airport (LGW) is the second busiest airport in the United Kingdom by passenger movement. Together these airports serve over 75% of passengers of the London airport system. Environmental externalities resulting from aircraft and ground support noise and emissions impact human health and welfare in the airport vicinity. Of these externalities, noise is often the greatest environmental concern for communities living near airports (Durmaz, 2011). Wolfe et al. (2014) calculate that the annual impacts of aircraft noise on residential property values in the US amounts to $100-400 per person near an airport fence line. On a global basis, He et al. (2014) calculate that the capitalized property damages from aircraft noise in 2005 were $23.8 billion.

In this Appendix, the impact of aircraft noise on the London population resulting from operations at Heathrow and Gatwick airports is assessed. Readily perceivable noise impacts such as annoyance are modeled and monetized using a model of willingness to pay for noise abatement. Incidences and associated costs of noise related hypertension, myocardial infarction, and stroke around LHR and

LGW are also assessed. A prior study has assessed the air quality impacts on public health from the London airport system (Yim et al., 2013).

A report commissioned by Heathrow Airport Limited concluded that the lack of capacity at LHR is costing the UK economy £14 billion per year and that this could rise to £26 billion per year by 2030 in the absence of expansion (HAL, 2012). The issue of hub airport capacity is a major public policy issue in the UK, with options having been considered including not increasing capacity, expanding LHR with a third runway, expanding LGW with an additional runway capable of accommodating independent operations or building a replacement hub airport in the Thames Estuary (THA).[14] Historically, aircraft noise has been one of the most important factors in the UK government decision on hub airport expansion (Nwaneri, 1970). This Appendix assesses the societal impacts of aircraft noise in future years under different expansion scenarios, presenting projections of noise impacts in the year 2030 for LHR, LGW, and a proposed airport in the Thames estuary.

A.2 Methods

A.2.1 Noise Modeling

The Integrated Noise Model (INM) version 7.0d is used to model the extent and magnitude of aircraft noise near London's hub airport. INM, which is publically available, has been in use since 1978, and is currently used in more than 65 countries (FAA, 2008). INM uses noise-power-distance (NPD) data to estimate surface-level noise while accounting for operational settings such as operation mode and thrust level as well as environmental factors such as acoustic directivity and local geography. Aircraft noise at the residential level is measured using the A-weighted continuous equivalent sound level (Leq in dBA) for consistency with other analyses of UK airports. A simulation run for this analysis using INM and 2010

[14] See the Airports Commission website https://www.gov.uk/government/organisations/airports-commission

operational data at Heathrow found that the use of A-weighted Day Night Level (DNL dBA) results in a 3.7% higher noise level than Leq dBA. Where impact dose-response relationships are developed from studies using DNL dBA, noise exposure levels computed in Leq are increased by 3.7%.

LHR is located at 51.47 °N, 0.45 °W and LGW is located at 51.15 °N, 0.19 °W. Existing runways at LHR and LGW are modeled based on airport diagrams (HAL, 2013). LHR and LGW flight paths are modeled per runway corresponding to the noise preferential routes (NPRs). Historical operational data for LHR and LGW are taken by noise class and aircraft type for 2010 (Lee, 2011; Lee, 2012b). In 2010 there were approximately 460,000 ATMs at Heathrow airport and 241,000 ATMs at Gatwick airport. Where available, existing INM aircraft representations were applied. If existing aircraft were not available in the INM database the closest matching current aircraft was used. Finally, aircraft are grouped and assigned to arrival and departure flight paths for each runway. For all LHR scenarios an 83:17 modal split of westerly (runways 27L and 27R) and easterly (runways 09L and 09R) operations is used to match LHR in 2010 (Lee et al., 2011). For LGW scenarios a 73:27 modal split of westerly and easterly operations is used to match LGW in 2010 (Lee et al., 2012b).

While INM is widely used internationally, the UK Civil Aviation Authority (CAA) through its Environmental Research and Consultancy Department (ERCD) has developed and maintained the Aircraft Noise Contour model (ANCON) for its own UK airport noise assessments. The INM-estimated Leq > 57 dBA noise contour area is 4.6% (10.9 km^2) larger than the ANCON estimate using the 2011 the ERCD report's flight data (Lee et al. 2012a). This implies a discrepancy between models of less than 5% and validates the INM for forecasting future UK aviation noise contours. A similar comparison performed by AMEC (AMEC, 2014) found that INM produces smaller noise contours than ANCON for lower noise levels (<57 dBA Leq) and larger contours for higher noise levels, but that contour areas agreed within 10%. Uncertainty in noise dispersion models has been estimated as +/- 1.46 dBA

(95% confidence, total noise uncertainty, ANCON) and +/-2 dBA (triangular distribution on contour uncertainty, INM) (White, 2005; Mahashabde 2011).

A.2.2 Population Data

An update of 2001 UK census data provided by CACI is used for the calculation of population noise exposure in 2010. The CACI population database contains data referenced at the postcode level with each postcode having a single coordinate located at the postcode's centroid. The CACI data has been used by the CAA for the ERCD (Environmental Research and Consultancy Department) reports for noise exposure contours for all airports in the UK, including Heathrow and Gatwick. Population noise exposure is computed using GIS software MapInfo, by exporting output of the INM model and overlaying the population grid with the noise contours for each scenario. Population points are then matched for each contour and summed within the software for each scenario.

A.2.3 Impacts Modelling

Willingness to pay for noise avoidance is developed through revealed preferences and taken as a proxy for readily perceivable damages from aircraft noise such as annoyance and sleep awakenings. The willingness to pay model is developed by He et al. (2014) as described in Section 3.1.1, here applied to non-US airports. The willingness to pay per (WTP) dBA (DNL) of excess noise is given by $WTP = \alpha + \beta \times Income + \gamma \times non\text{-}US\text{-}dummy \times Income$ where α and β are the probabilistically defined model intercept and coefficient. An additional term, γ, is included for non-US airports, where studies have indicated a higher willingness to pay for abatement. The non-US-dummy term is equal to 0 if the airport of concern is in the United States and 1 otherwise. Metropolitan per capita income level for London in 2010 is taken from the UK Office of National Statistics. Annual damages are developed from the total capitalized noise damages by assuming a 30 year capital lifetime and a discount rate of 3%, consistent with prior assessments of noise costs (Mahashabde et al., 2011)

The costs associated with three health endpoints, hypertension (HYT, i.e. high blood pressure), myocardial infarction (MI, i.e. heart attack) and stroke, are also calculated. Section 3.1.2 describes the methodology for assessing the health impacts of noise and the justification. In the study presented here, this methodology is adapted for a UK specific airport.

The relative risk for MI and HYT for each five dB DNL band from a meta-study by Basner et al. (2013) and the baseline incidence rate for those endpoints for UK residents by age and gender (Townsend et al., 2012) were used to calculate the expected impact incidence rate for each age, dB level, and gender combination. For HYT, there is limited evidence that disease incidence has an effect on expected future earnings, and, as such, damages are calculated using annual expected medical contact, hospital, and drug costs (Cropper and Krupnick, 1990) adjusted to 2010 British Pounds (£) using the Consumer Price Index (CPI) and Purchasing Price Parity (PPP). Average life expectancy by age and gender for the United Kingdom and a discount rate of 3% were used to determine net present value of total noise damages. For MI, national fatality rate due to MI (Townsend, et al. 2012) for each gender and age group was used to separate morbidity and mortality impacts. Mortality impacts were valued using a Value of Disability Adjusted Life Years of £60,000 using a 3% discount rate and the over the expected remaining life expectancy. The £60,000 value for a life-year lost is consistent with the value used to monetize disability adjusted life years lost by the UK Interdepartmental Group on Costs and Benefits Noise Subject Group (IGCB[N]) second report (IGCBN, 2012). Non-fatal MI costs were calculated as the sum of opportunity costs and direct medical costs by age at a 3% discount rate using the approach and values outlined in the EPA's Regulatory Impact Analysis for the Federal Implementation Plans to Reduce Interstate Transport of Fine Particulate Matter and Ozone in 27 States (EPA, 2011) using PPP and CPI for a consistent base year of monetization.

For stroke, while evidence of an increased relative risk with exposure to high levels of noise exists, there is no consensus on the magnitude, and differences in study methodologies make a direct quantitative meta-analysis inappropriate. Hansell et al. (2013) find a relative risk of hospital admissions of 1.24 (1.08 to 1.34) for people living in the highest noise level (>63 dB) compared to the lowest noise levels (<51 dB) in London for daytime noise. For comparison, Floud et al. (2013) finds an odds ratio of 1.25 for every 10 dB for 'heart disease and stroke' based on a study of six European cities. However that study includes changes in average night-time noise only and controls for people who had been in the same residence for more than 20 years. Floud et al. (2013) do find positive but smaller odds ratios for day-time air traffic noise exposure and air traffic noise exposure not controlled for residence length (1.11 and 1.12 per 10 dB respectively), but these results lack statistical significance at the 0.05 level. In yet another study, Correia et al. (2013) find increased hospital admission rates for all cardiovascular endpoints including cerebrovascular events, of 3.5% per 10 dB noise at US airports, but that analysis is limited to the population older than 65. The relationship between noise exposure, and in particular aircraft noise, and stroke incidence is uncertain with still other studies reporting no significant risk relationship (Huss et al. 2010, Kolstad et al. 2013). For this study, a linear relative risk of 1.12 is applied for every 15 dB of noise over 50 dB. This relative risk dose-response relationship is consistent with Hansell et al. (2013) assuming a background noise level of 50 dB and highest noise levels in excess of 80 dB. This relative risk is also within the confidence interval of the Floud et al. (2013) study but produces fewer stroke incidences than the mean relative risk from that study when not controlling for residence length. The baseline incidence rate and the one-year mortality rate for all-cause strokes by age and gender are taken from the England National Stroke Audit (Hippisley-Cox et al., 2004).

Lifetime stroke costs are similarly uncertain and vary considerably in the literature. Adjusting studies to a common base year and currency (2010 £), the average cost per stroke ranges from £18,000 (Payne et al., 2002) to £300,000 (Cadilhac et al., 2010), assuming a Value of a Disability Adjusted Life Year of

£60,000. The percentage of total costs attributable to indirect costs also varies from under 10% (Oleson et al., 2012) to over 50% (Taylor et al., 1996) with yet other studies finding the largest percentage of costs attributable to social services (Ghatnekar et al., 2004). Valuation of mortality in these studies vary significantly, in some cases not being considered as part of the societal economic burden and elsewhere calculated using a value of human capital approach (Taylor et al., 1996, Youman et al., 2002, Palmer et al., 2005). For this study, the cost of stroke is estimated by age of first onset as the sum of direct (hospitalization and rehabilitation) and indirect ischemic stroke costs from Taylor et al. (1996), equivalent to £119,000 (in 2010 £).

Other schemes to calculate and monetize the impacts of noise exist. Exposure-response relationships between noise and health impacts are still uncertain and the subject of ongoing research. As such, any relationship between health and noise exposure should be considered a preliminary or best-guess estimate. Such estimates are still useful for policy analysis to understand the scope and magnitude of damages. For example, a dose-response relationship between noise and MI can be extrapolated from underlying noise-health studies as a linear or polynomial function (Basner et al. 2013, Babisch 2008). Different methodologies are used to monetize the health and welfare impacts of environmental externalities including cost-of-illness, general equilibrium economic models, and value of quality-adjusted life years approaches among others. Even within one methodology, monetization values may differ based on what impacts to include. For example, in this analysis, noise impacts from noise-induced strokes are measured using a direct dose-response function and published values of direct and indirect stroke costs from literature. An alternative is to estimate stroke incidences as a secondary effect from the (less uncertain relationship) between noise and hypertension and to value these impacts using a quality-adjusted life year's lost approach (Harding et al. 2013). Other impact modelling approaches are examined in Appendix B.

A.2.4 Future Noise Impacts

The impact of noise on populations around London airports in future years is dependent on the future operational characteristics of the in-operation airports, the technological and operational improvements in aircraft operation, and the patterns of population growth and movement in the airport vicinity. For London, hub operations are capacity constrained, and the impact of capacity improvements will impact the total noise burden. The following expansion scenarios are defined with noise impacts modeled for year 2030:

- S1: No expansion of LHR or LGW;
- S2: Expansion of LHR by extension of the northern LHR runway;
- S3: Expansion of LHR with a third runway to the northwest;
- S4: Expansion of LGW with a second independent runway and continued operation of LHR; and
- S5: a four-runway Thames Estuary Airport and closure of LHR.

The location and layout of the expansion projects are shown in Figure A–1. Scenario S5 follows the representative proposal for a Thames Estuary hub airport (Forster + Partners 2012). Despite garnering some political support, the Thames Estuary hub airport was not shortlisted by the UK Airports Commission as a near-term viable expansion project (Airports Commission, 2014). However, analyzing the potential noise impacts of such a proposal remains valuable given that the final decision will be taken by the UK government. An airport in the Thames estuary has been proposed before, such as the Foulness Island proposal of the Roskill Comission (Nwaneri 1970). If London Airports remain congestible, there is a strong possibility that a similar proposal will gain traction in the future. Thus, this analysis (1) provides context for existing policy decisions, (2) quantifies and monetizes potential benefits that have heretofore been analyzed only qualitatively and (3) provides an indication of the scope of benefits for future proposals.

Figure A–1 Airport expansion scenario diagrams. a) S2: Expansion of LHR by extension of the northern LHR runway; b) S3: Expansion of LHR with a third runway to the northwest; c) S4: Expansion of LGW with a second independent runway and continued operation of LHR; d) S5: a four-runway Thames Estuary Airport and closure of LHR

Scenario S1 is consistent with a do-nothing or do-minimum scenario and represents how noise would be expected to progress in absence of any significant physical capacity development implementation. Scenarios S2, S3, and S4 are consistent with the three shortlisted expansion schemes investigated by the UK Airports Commission (2014). Scenario S2, northern runway expansion, is modeled based on runway diagrams from the July 2013 proposal, while Scenario S3 assumes an additional LHR runway consistent with the Heathrow Northwest Runway proposition supported by Heathrow Airport Ltd. (HAL 2013). Three additional schemes for the construction of an additional runway at LHR based on the northern runway and southwest runway proposals submitted by HAL (HAL, 2013) and the pre-2010 runway expansion submitted by BAA (Hillingdon, 2010) are also considered. Scenario S4, expansion of LGW with a second independent runway, is modeled on diagrams from Gatwick's July 2013 expansion proposal for option 3, a southern independent mixed mode runway (GAL, 2013). Scenario S5 considers the scheme in which a new hub airport is constructed in the Thames Estuary (here referred to as THA). THA is assumed to be at 51.45 °N, 0.68 °E consistent with Yim et al. (2013) and Foster + Partners (2013), where runways are 4 km long and 46 m wide with a 2 km centerline separation between the two pairings of runways. A Thames Estuary airport is no longer on the shortlist for consideration in current decision-making process for London hub airport capacity expansion. However, the noise impacts of this proposal are modeled to better understand the potential environmental footprint of such an airport as previous analyses have shown it to have potential significant air quality benefits (Yim et al. 2013).

For projections of future noise impacts, the LHR third runway and LGW second runway flight paths are linked to current NPR flight paths based on ERCD reports from 2010 (Lee et al. 2011, Lee et al. 2012b). Additionally, for Heathrow's proposals the landing glideslope is increased to 3.2 degrees, consistent with HAL's plans for noise reduction (HAL, 2013). At THA flight paths are not noise-optimized and are assumed straight with distribution of ±90° after flying two runway lengths. This assumption implies that noise impacts at THA may be reduced relative to estimates with the development of appropriate NPRs given specific airspace proposals. However, the level of detail of currently available for THA plans does not support assessment of more detailed specific flight paths, which may be subject to airspace redesign given potential conflicts with other European airspace. For THA no assumption was made for modal split and a 50:50 split was used.

Forecasts for air travel demand in the UK were obtained from the UK Department for Transport (DfT, 2011). The DfT provided the number of air traffic movements (ATMs) by aircraft type at each UK airport in each scenario in 2030; the DfT has stated that although their aggregate forecasts have been thoroughly validated, their forecasts on an airport-by-airport and aircraft-by-aircraft basis do not possess the same level of confidence. In S1, the constrained scenario, Heathrow undergoes no expansion and remains a two-runway airport with 486,039 annual ATMs in 20130. With unconstrained growth – which applies to S2 (northern runway expansion) and S3 (an additional NW runway) – LHR has 715,204 annual ATMs, representing a growth of 55% relative to 2010. To account for the displacement of operations to LGW in S4, an expanded Gatwick with continued operation of Heathrow, the difference between LHR S1 (constrained LHR) ATMs and LHR S2 (unconstrained LHR) ATMs were added to Gatwick's constrained 2030 ATMs to create an unconstrained and expanded Gatwick scenario (S4) with 980,774 total ATMs: 486,039 due to a still-constrained Heathrow and 494,735 due to an expanded Gatwick. These projections are approximate as historically this distribution is not generally observed at multi-airport systems since there remains a higher airline and passenger demand to directly fly to the main hub airport (de Neufville, 1995).

Population exposure from future operations is measured using the 2010 UK Census update provided by CACI as is done with the 2010 exposure estimates. Holding population constant while addressing future noise projections highlights the impact of operational changes across years reduces the uncertainty from population trends when comparing across scenarios, and it is consistent with environmental impact modelling approaches used by the International Civil Aviation Organization – Committee for Aviation Environmental Protection (Mahashabde et al., 2011) and the methodology of external consultancies addressing expansion options for the UK hub airport (HAL, 2013). However, the total cost of the noise impacts in future years will be greater if population growth occurs in noise-exposed areas. The sensitivity of the results to future population growth is, therefore, explored in this analysis.

All expansion scenarios assume that residential areas intersecting the expanded airport are depopulated. The displacement of 4100 residents is estimated for S2, and the displacement of 5512 residents is estimated for S3. For S4, only 300 residents are displaced (~100 dwellings) since the land has been safeguarded since 2003 (GAL, 2013). For THA (S5), a displacement of 4161 persons is estimated, broadly consistent with other estimates of a loss of 2,000 dwellings (Johnson, 2013). In scenario S5, the populations around the Isle of Grain would likely increase more than NTEM 6.2 forecasts for 2030 because of the new hub airport THA and associated activities. For example, today there are approximately 76,600 employed at LHR, of whom a significant fraction may live in the LHR vicinity (Optimal Economics, 2011). However, the growth in population in the area could be supported outside of the Leq >57 dBA area with appropriate planning measures and noise preferential routes.

Aircraft noise levels have reduced significantly since the 1970s, with anticipated continued improvements. The Advisory Council for Aviation Research and Innovation in Europe (ACARE) goal for 2020 is a 50% reduction in perceived noise relative to 2010 levels (ACARE, 2008), while the European Commission's equivalent goal for 2050 is a 65% reduction (European Commission, 2011). In this study, a range of expected noise reductions are estimated from technology improvements informed by industry and government targets, and engineering judgment.

For new aircraft types introduced by 2030, the effect of noise reduction technology is explored through three different marker cases. For a "high" noise reduction case in 2030, it is assumed that the European Commission's goal for 2050 of a 65% perceived noise reduction will have been achieved. The "low" estimate places 2030 aircraft noise performance at a 10% improvement from 2010. The "moderate" noise reduction factor is based on the mean of the range of these forecasts and engineering judgment assuming a 32% noise reduction by 2030. As aircraft have an in-service life of ~30 years, and types are produced for more than a decade, each of these noise reduction factors are only applied to new aircraft types from 2010 to 2030 (and not current aircraft types for which specific noise data exist and that will still be flying in 2030). In terms of comparative analyses between hub airport options in 2030, the specific assumption for noise reduction technology has limited importance provided that the scenarios are compared pairwise, e.g. S1-high vs. S3-high (and not S1-high vs. S3-low). This suggests that the analysis is robust in terms of relative changes when comparing in a given year, but that comparing 2030 to 2010 is subject to the (uncertain) progress made in aircraft noise reduction.

A.3 Results

A.3.1 2010 Noise Exposure

The threshold for significant noise impacts in the vicinity of London airports is set at 57 dBA (Leq), consistent with the UK Government's guidance on the onset of significant community annoyance. However, populations may be impacted outside of the 57 dBA contour, especially in areas of low background noise. The geographic extent of aircraft noise above 57 dBA as measured in Leq for LHR and LGW is shown in Figure A-2. For 2010, the Leq > 57 dBA noise contour area around Heathrow (LHR) is 271 km^2 and that the Leq > 57 dBA noise contour area round Gatwick (LGW) is 98.1 km^2.

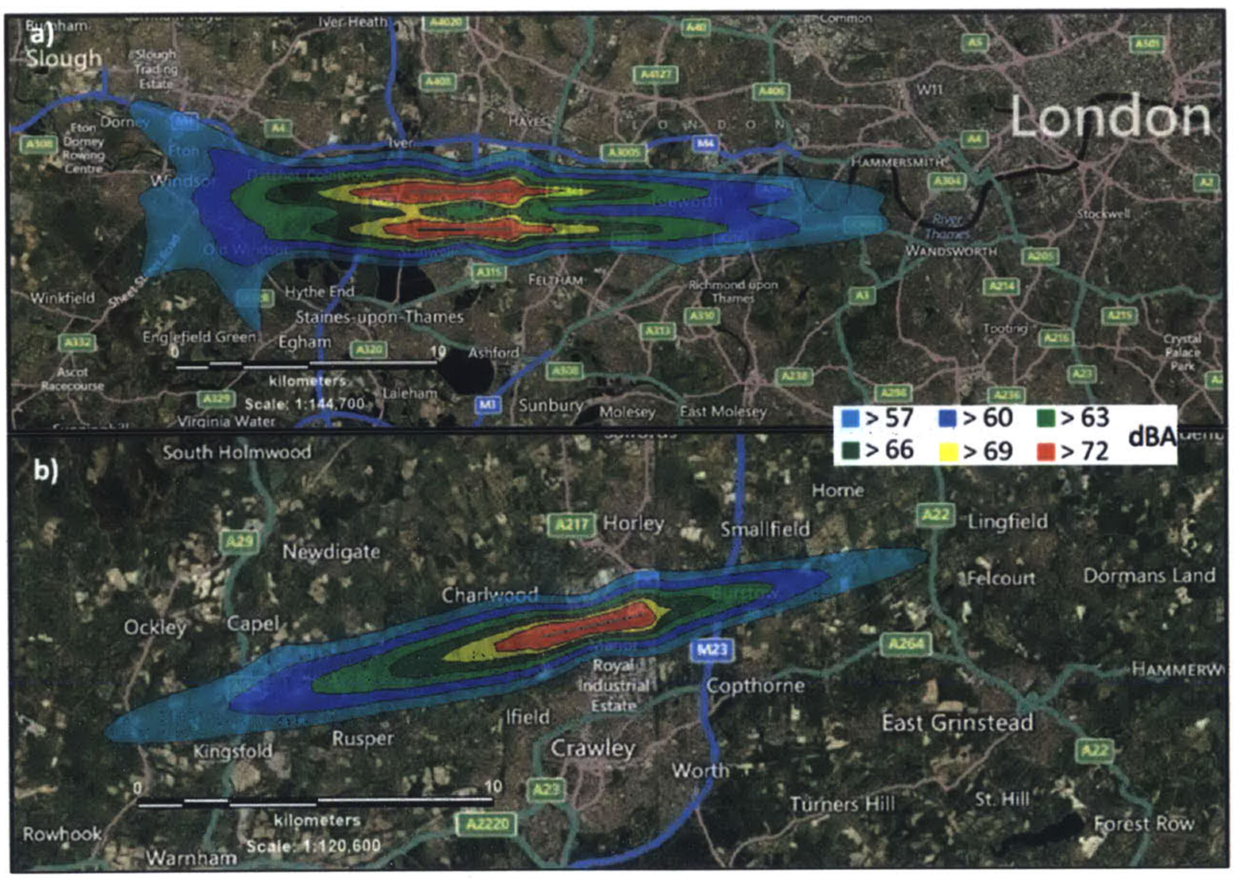


Figure A–2 Noise exposure contours for a) LHR in 2010 and b) LGW in 2010

In 2010, 252,500 people are exposed to aircraft noise from operations at LHR. Over 40% of this population (106,300 people) is exposed to between 57 and 60 dBA and less than 1% of this population (1800 people) is exposed to noise levels greater than 72 dBA. At LGW, only 3300 people are exposed to aircraft noise >57 dBA (Leq), and no persons are exposed to aircraft noise above 72 dBA. Population exposure estimates for 2010 are 9.1% higher than ERCD estimates for the same period (Lee et al. 2012b). Population exposed estimates from this study are smaller than those of the ERCD at low dB levels, while population exposed estimates at higher dB levels are greater. For example, at LHR this study estimates that 27,000 fewer residents are impacted by noise between 57 and 60 dBA but 1600 additional residents are impacted by noise in excess of 72 dBA.

A.3.2 2010 Noise Impacts

The health and welfare impacts from aircraft noise at LHR and LGW for 2010 are summarized in Table A–1. Impacts reflect average expected annual damages from operations in 2010. Noise associated with operations at LHR and LGW contributed to £81.2 million in negative impacts in 2010, with 70% of these damages being associated with willingness to pay for abatement developed from property value depreciation, which here is taken as a proxy for perceivable and attributable effects like annoyance and sleep awakenings. Incidences of 56 myocardial infarctions (MI) are attributable to aircraft noise from LHR and 1 MI is attributable to aircraft noise from LGW. The modeled overall case fatality rate for MI across both airports is 29%, which leads to an estimated 17 annual fatalities from aircraft noise-relate MI. The overall all-cause 30-day case fatality rates for first infarctions in England in 2010 were 32.4% and 30.3% for men and women respectively (Smolina et al. 2012).

Table A–1 2010 Noise Impacts for LHR and LGW

Airport	LHR	LGW
Willingness to Pay (million £)	56.1	0.62
Myocardial Infarction (Incidences)	56	1
Myocardial Infarction Valuation (million £)	16.4	0.20
Hypertension Valuation (million £)	4.4	0.05
Stroke Valuation (million £)	3.4	0.04
Total Noise Damages (million £)	80.3	0.87

All values in Table A–1 are subject to scientific and economic uncertainty as well as variation in social-preference. For instance, willingness to pay for noise abatement damages are influenced by the noise model uncertainty, population uncertainty, the assumed background noise level, the choice of willingness to pay model, variation in the human response to annoyance, and the capital recovery factor. Mahashabde et al. (2011) developed the concept of environmental lenses to account for epistemic and aleatoric uncertainty in noise damages. Each lens uses a different combination of the most influential model parameters to represents one view or perspective of environmental damages. Applying the 'low' and 'high' lens assumptions for noise damages from Mahashabde et al. (2011) give total willingness to pay for abatement values of £10.8 million and £156 million respectively. This indicates a magnitude of uncertainty of 15:1, from 13% to 190% of the average expected willingness to pay values. For comparison, the UK Airport Commission's additional airport capacity noise analysis produced low and high estimates for valuation of annoyance and sleep-disturbance that indicated a combined magnitude of uncertainty of 12:1, from 50% to 600% of their baseline values (Airports Commission, 2014). Estimates and comparisons of yearly damages under different valuation schemes, such as that proposed by the UK IGCB[N], are provided in the SI. Similarly, different preferences or assumptions of capital lifetime or discount rate can affect the noise impact magnitude. For example, the difference in willingness to pay for abatement values at 2% and 7% discount rates is a factor of 2.

An analysis of present day air quality impacts in the United Kingdom found that an average of 77 early deaths in the UK were attributable to operations from London airports (Yim et al., 2013), accounting for between £31.5 million and £315 million yearly damages to the economy. In addition to LHR and LGW, that study included emissions from London City (LCY), Luton (LTN), and Stansted (STN) airports. The results of this noise analysis indicate that while air quality impacts from aviation may contribute to more yearly mortalities than noise, total damages from aircraft noise are of a similar magnitude to the impacts of air quality from London airports. For further comparison, a study by the Health and Safety Laboratory (Harding et al. 2011) found that road traffic and rail noise contribute to £1 B and £43 million in annual damages across the entire UK.

A.3.3 2030 Noise Impacts

The health and welfare impacts from aircraft noise in 2030 are summarized in Table A–2. Values in parentheses represent the damages associated with the high and low technology cases described in Section 2.4.3. Under the expansion scenarios, noise projections at all operating airports change. For example, expanding operations at LHR through extension of an existing runway (S2) will lead to a change in expected carriers, operations, and aircraft operating at LGW relative to the constrained case where no expansion occurs at either LHR or LGW (S1). Therefore results focus on total noise impacts at both operating airports for each scenario and not on detailed scenarios at individual airport, because such results would not represent consistent UK aviation scenarios. This is consistent with the methodology for computing future impacts adopted by Yim et al. (2013).

Table A–2 2030 scenario noise impacts. Numbers in parentheses represent the high- and low-technology noise assumption impact estimates.

Scenario	S1: Constrained	S2: LHR Extended Runway	S3: LHR 3rd Runway	S4: 2nd LGW Independent Runway	S5: THA, LHR Closure
Exposed Population (Thousands)	246 (213 - 263)	283 (245 - 316)	343 (282 - 375)	254 (218 - 273)	8.8 (6.6 - 10.1)
Willingness to Pay (million £)	53.2 (48.3 - 58.1)	64.8 (53.8 - 73.0)	72.5 (58.3 - 81.1)	54.8 (49.2 - 59.9)	1.62 (1.19- 1.91)
MI (Incidences)	53 (41 - 53)	64 (49 - 64)	74 (54 - 73)	55 (42 -54)	2 (1 - 2)
MI Valuation (million £)	15.8 (13.5 - 17.1)	18.7 (15.9 - 20.9)	21.8 (17.8 - 24.1)	16.3 (13.8 - 17.7)	0.53 (0.39 - 0.61)
Hypertension Valuation (million £)	4.22 (3.57 - 4.60)	5.13 (4.27 - 5.78)	5.77 (4.64 - 6.44)	4.35 (3.64 - 4.75)	0.13 (0.10 - 0.15)
Stroke Valuation (million £)	3.26 (2.77 - 3.54)	3.91 (3.29 - 3.85)	4.47 (3.13 - 4.39)	3.36 (3.62 - 4.97)	0.10 (0.08 - 0.12)
Total Noise Damages (million £)	76.5 (68.2 - 83.4)	92.5 (77.3 - 104)	104.6 (84.4 - 117)	78.8 (69.5 - 86.0)	2.38 (1.76 - 2.79)

All damage values are subject to scientific and economic uncertainty associated with modelling decisions and parameters as described in Section A.3.2 and from additional uncertainty related to projections of operations and population. Interpreting results of one scenario relative to another (e.g. S3 relative to S1) or future impacts relative to the present (e.g. S3 relative to 2010 impacts) reduces the uncertainty arising from the choices made in the impacts model (Mahashabde et al. 2011). Thus, while gross results are presented for each scenario, the quantity of interest for policy- or decision-makers may be the difference between a set of projection scenarios and a designated baseline scenario.

With no expansion at either airport by 2030 (S1), there is an 7.6% decrease in noise contour area and a 3.9% decrease in the population exposed at LHR and a 4.5% decrease in noise contour area and a 6.3% decrease in the population exposed at LGW relative to 2010 assuming the baseline moderate noise reduction technology. Furthermore, without expansion, the total exposed population has the potential to decrease by 11.8% relative to 2010 if technological improvements to aircraft noise exceed industry goals (S1-high). If noise technology improvement is lower than expected (S1-low), then the population affected by significant noise will increase by 2.9%. Taking the three cases broadly as a whole, the extra 26,000 ATMs per year at LHR in 2030 are approximately "cancelled out" by fleet evolution and expected improvements in aircraft noise technology. Total expected noise damages under mid-range noise technology and fleet evolution assumptions are £76.5 million.

Noise exposure will likely increase with either version of Heathrow expansion due to a 55% increase in ATMs from 2010 and additional residential areas over which new flight paths pass. For the extended runway scheme (S2), the moderate projection puts the total Leq > 57 dBA noise contour area growth at 6.0% and an exposed population increase of 10.7% in 2030 compared to 2010 with 27,700 additional residents around LHR exposed to Leq > 57 dBA. If aircraft meet the European Commission's 2050 goals for noise reduction by 2030, there will be a 4.1% decrease in the total population affected by significant noise, but if future aircraft noise technologies perform poorly, the total noise-exposed population from both airports will increase by 23.4% relative to 2010. Under moderate technology assumptions, total annual MIs increase to 62 and total noise damages increase to £92.5 million with 70% of these damages associated with willingness to pay for abatement.

For the additional NW runway at LHR proposal (S3), the total damages from noise across both airports is expected to increase relative to 2010 for all noise technology scenarios examined. The total population exposed to Leq > 57 dBA increases by 34.2% relative to 2010 for moderate technology. The resulting total noise damages are £104.6 million and incidents of MI increase to 74 per annum, resulting in 21 expected fatalities. Even assuming noise technology exceeds 2050 goals, noise damages exceed £84.4 million, an increase of nearly 4% over 2010 noise damages.

For the case of a second independent runway at LGW and continued constrained LHR operations (S4), the area exposed to noise Leq > 57 dBA is 431 km^2 for the mid-range technology case, putting it between the two Heathrow expansion scenarios (S2 and S3) in terms of total area noise expansion. However, because of population density differences between communities near LGW and LHR, only 254,000 residents are exposed to noise Leq > 57. Thus, the total noise impacts of £78.8 million are less than the totals of the LHR expansion scenarios, S2 and S3. Of these damages, 69% are due to readily perceivable impacts as measured by the willingness to pay for abatement.

In the final scenario (S5), LHR is replaced by a hub airport in the Thames Estuary by 2030. THA alone would be expected to have a noise footprint of between 278.5 and 360.9 km^2, with a midrange estimate of 321.9 km^2. Despite, impacting a larger area than LHR in 2010 or in 2030 under no expansion, it is expected that the population impacted by noise would be significantly lower than at Heathrow due to landing and takeoff flight paths over water and areas of lower population density. Total population exposed across the two active airports drops to 8,800 resulting in £2.38 million in noise damages for midrange technology and fleet evolution assumptions. The total damages in 2030 in this scenario are only 2.9% of the noise damages in 2010. In addition, expected MI drop to 2 per year, resulting in fewer mortalities than 1 per year.

Because an airport can provide direct and indirect employment opportunities in the surrounding communities, the demographics surrounding the THA site could be expected to change once operations commence. As an upper bound on the potential noise impacts considering these shifts, it is informative to assess the impact of future noise considering the surrounding communities with an increased population density equivalent to the population density surrounding Heathrow, i.e. 1,111 people/km^2. Even with this higher population density, the number of people impacted by noise Leq > 57 dB around THA is 57% lower than the number of similarly exposed people for the Heathrow expansion scenario (S3).

A.3.4 Alternative Heathrow Expansion Scenarios

Other Heathrow airport expansion proposals have been considered in addition to the scenarios examined in Section A.3.3. In this section, the projected impacts associated with three of these scenarios are assessed. A proposal for an additional northern runway [S3_A1] and a proposal for a runway to the southwest of the existing airfield [S3_A2] were submitted by Heathrow Airport Limited as alternatives to the northwest runway [S3] scenario (HAL, 2013). In addition, the noise impacts from an older proposal for an additional runway at LHR submitted before 2010 [S3_A3] are also examined (Hillingdon London, 2008). All three additional proposals perform similarly (+/- 5%) in total magnitude of damages to the NW runway option shortlisted by the airports commission (AC, 2014b). However, S3_A2, the SW runway scenario, performs the best of the additional LHR runway scenarios, with total damages 2.8% lower than that of shortlisted NW runway proposal S3.

Table A–3 Noise impacts for alternative proposals where LHR is expanded with a 3rd runway and LGW continues operating in its 2010 configuration. Numbers represent the mid-range best estimate for noise technology improvements, while values in parameters are for the high- and low- technology improvement scenarios respectively.

Scenario	S3_A1: LHR 3rd Runway N	S3_A2: LHR 3rd Runway SW	S3_A3: LHR 3rd Runway BAA Proposal
Exposed Population (Thousands)	357 (292 - 397)	325 (281 - 346)	362 (298 - 397)
Willingness to Pay (MM £)	75.3 (59.8 - 85.5)	70.7 (58.1 - 78.0)	75.9 (60.7 - 85.0)
MI (Incidences)	76 (62- 87)	71 (59 - 77)	78 (63 - 87)
MI Valuation (MM £)	22.7 (18.4 - 25.5)	21.0 (17.7 - 22.7)	23.0 (18.7 - 25.4)
Hypertension Valuation (MM £)	5.98 (4.76 - 6.79)	5.61 (4.62 - 6.17)	6.03 (4.84 - 6.75)
Stroke Valuation (MM £)	4.64 (3.72 - 5.25)	4.33 (3.60 - 4.73)	4.68 (3.78 - 5.22)
Total Noise Damages (MM £)	108 (86.7 - 123)	102 (84.0 - 112)	110 (88.0 - 122)

A.3.5 Alternative Population Assumptions

In Section A.3.3, total population exposure and the magnitude of damages from noise in 2030 were calculated using 2010 population data. Holding population density constant isolates the impact of operational changes on noise damages. However, as populations change, the absolute value of the total damages are expected to change across all scenarios. Population exposure and total damages from noise (sum of willingness to pay, myocardial infarction, stroke, and hypertension) for the five airport expansion scenarios are calculated using CACI Inc. provided population projections for 2030 and are given in Table A–4.

Table A–4 Impact of population growth assumptions on the total noise exposure and impact estimates of 5 different airport region expansion scenarios.

Scenario	Exposure [people] (2010 pop)	Exposure [people] (2030 pop)	% Difference	Impact [£ MM] (2010 pop)	Impact [£ MM] (2030 pop)	% Difference
S1: Constrained	246,000	320,000	+ 30%	76.5	101	+ 32%
S2: LHR extension	283,000	365,000	+ 29%	92.5	121	+ 31%
S3: LHR NW Runway	343,000	446,000	+ 30%	105	137	+ 30%
S4: LGW Runway	254,000	331,000	+ 30%	78.8	104	+ 31%
S5: THA, Closure of LHR	8,800	10,500	+ 19%	2.38	2.81	+ 18%

Using updated 2030 population projections, there is a consistent 30% increase (+/- 2%) in the population exposed and noise damage estimates in 2030 for all scenarios where Heathrow and Gatwick are operational and only an 18% increase in impact for the scenario where the Thames Hub Airport and Gatwick are operational and Heathrow is closed (S5) relative to the 2030 impact estimates while keeping population held constant. As a result, the rank order of the expansion scenarios by noise damage remains unchanged and the THA airport looks increasingly more attractive than the other options when considering only noise impacts.

There are several pitfalls in applying population projections to environmental impact assessments. For example, the noise analyses of the do-nothing and shortlisted expansion scenarios for the Airports Commission use CACI Inc. provided population projections for 2030, 2040, 2050 to determine future population exposure and noise impact damages (AC, 2014b). These projections include population growth in the areas exposed to the highest noise levels, which is unlikely given the current noise burden. Furthermore, with proper zoning and land-use planning for noise-sensitive buildings and residents, excess damages from population expansion can be avoided in areas where significant noise levels are expected in the future. Furthermore, uncertainty in the population forecast is unaccounted for in deterministic projections. In addition to uncertainty in estimates of immigration, birthrate, and mortality inherent in national population projections (Keilman, 2007), the geographic and demographic distribution of the future population is also important for the noise analysis.

A.3.6 Discussion of Assumptions and Limitations

The uncertainty in future aircraft noise performance is captured by modelling technology factors, which provides a range of potential noise magnitudes depending on how the industry's noise reduction trends. These noise reduction factors are only applied to new aircraft types from 2010 to 2030 and assume that no currently operating aircraft will be modified with new engines, "hush kits", etc., but the range of noise reduction factors applied should also capture the effect of this on aggregate.

Flight paths for the expanded Heathrow and Gatwick cases are assumed to follow current noise preferential routes and the uncertainty of the exact coordinates of these paths may lead to an increase in the Leq > 57 dBA area. Alternatively, reductions may be possible through improved NPRs. In addition the uncertainty and variability from the noise dispersion model (INM) may lead to larger or smaller noise contours. A study of noise model uncertainty found that for a typical long-term study, the uncertainty in the noise code alone is ±1.48 dB (White 2005).

The runway usages (modal split) for 2030 at LHR and LGW are not knowable, and have been assumed to be the same as today. If any change tended towards long-term (20 year) average of 73:27 split, noise exposure would increase at an expanded Heathrow since more flights would be operating to the east over more densely populated areas. For THA, landings would be more likely to occur over water, which would further reduce noise exposure.

In this study UK standard metric for significant noise annoyance is used, Leq. However, other noise metrics such as the Average Individual Exposure (AIE) that weigh ATMs heavily into their calculation of contours could change results. For example, Leq averages noise out over a 16 hour period so that 45 flights at 76 dBA would approximately be equivalent to 450 flights at 66 dBA – a more annoying reality for many people (Kroesen et al. 2010). This may tend to underestimate annoyance in the future, with more frequent but quieter flights. The metric used in this study and the model assumptions further assume similar background noise characteristics and community attitudes toward noise in the areas around each airport location. While these characteristics are expected to be broadly comparable between LHR and LGW, the introduction of aircraft noise to the communities near THA may provide greater noise annoyance or community resistance. Furthermore, people may be annoyed by lower noise levels than 57 dBA. This study does not consider noise impacts on schools, working environments or on enjoyment of public spaces such as parks.

The comparison across different expansion scenarios does not consider equity weighting of different expansion options, which may shift the rank-order of apparent acceptability of each of the scenarios (Nwaneri, 1970). For example, that Heathrow-adjacent communities already bear a disproportionate burden of noise has been an argument for supporting alternative expansion proposals (Johnson, 2013).

The present analysis does not consider a disutility cost associated with the displacement of persons from airport expansion. The impact and cost of displacing residents should be considered when examining the full costs and benefits of different expansion options. Finally, constraints on the London hub airport may impact the airtraffic, and therefore the noise effects, of non-London UK airports. A recent study has shown that congestion spill-over from LHR can impact air traffic in Birmingham and Manchester (Gudmunsson et al., 2014).

A.4 Conclusions

This study has assessed the impact of significant aircraft noise from operations at LHR and LGW airports in 2010. Aircraft noise from LGW and LHR is associated with 57 myocardial infarctions each year leading to an estimated 17 premature mortalities. Total costs of noise in 2010 are £81.2 million a year. Aircraft noise from London airports contributes to human health and welfare damages that are a similar order of magnitude to air quality damages from aircraft emissions from the same operations. The impacts of noise in London airports exceed the total UK noise impacts from rail.

Further, this study assessed the change in noise impacts expected in 2030 under 5 possible future schemes. If Heathrow remains operating in its current configuration, the total impact of noise will remain approximately unchanged, with the increase in ATMs counter balancing the improving aircraft noise technology entering the fleet. Both Heathrow expansion options result in a likelihood of increased noise exposure relative to 2010. Extending the northern runway results in noise damages of £92.5 million and constructing an additional runway to the northwest results in noise damages of £104.6 million.

For the Gatwick expansion option that was shortlisted in December 2013 by the Airports Commission, there is an expected breakeven in overall airport noise exposure in the UK relative to 2010, with total noise performance being only 3% worse than the no expansion (S1) scenario in 2030. However, the > 57 dBA noise contour area increases from 57% to 135% and population affected increases from 177% to 331.7% locally around Gatwick depending on the performance of noise reduction technologies. Thus, while Gatwick expansion performs better than any Heathrow expansion (S2) with regards to net UK population aircraft noise exposure, the noise-exposed population of Crawley and neighboring towns increases from 2,750 in 2010 to around 10,000 in 2030.

For the alternate option of a replacement Thames Estuary Hub Airport, all technology assumptions lead to an order of magnitude reduction in the population exposed to significant noise levels. In order for the number of people affected by noise to exceed that in the Heathrow expansion case in 2030, the population in the surrounding area would have to increase to a density of almost 4000 people per km^2. This would be equivalent to a city of the size of Bristol materialising around the new hub airport with a total population of approximately 428,000 (Office for National Statistics, 2011). Reductions have the potential to be significantly greater with appropriate residential land-use planning and NPRs that enforce aircraft taking off and landing over the Thames.

Appendix B

Alternative Noise Valuation Approaches

Methods for monetizing the societal impacts of aircraft noise were developed and described in Section 3.1 and applied in Sections 4.1 and 4.2 and Appendix A. The valuation methodology calculated perceivable and attributable damages (e.g. annoyance) through a willingness to pay for abatement relationship developed from a meta-study of hedonic pricing literature. The methodology also calculated the costs of myocardial infarction, hypertension, and stroke related to noise exposure. These endpoints reflect one methodological approach to computing the impact of aircraft noise. This Appendix highlights several influential modelling choices and parameters in this methodology, identifies alternative approaches and assumptions for these choices from the impacts assessment literature, calculates the sensitivity of the damage impacts results to these alternatives where applicable, and finally discusses the strengths and weaknesses of these approaches.

The sensitivity of a noise analysis to the impacts and monetization model parameters will be dependent on the distribution of impacted people across different noise levels. As an illustrative case, the results of the 2010 noise damages from operations at LGW and LHR calculated in A.3.2 are used to test the sensitivity of the model parameters. Table B–1 summarizes the difference in total monetary impact of noise at LGW and LHR in 2010 for these model choices and parameters. For each parameter analyzed, all other parameters and model choices are held constant with the baseline modelling assumptions.

Table B – 1 Total noise damages for LHR and LGW in 2010 sensitivity to alternative modelling approaches.

Model Choice	Original Parameter/Model	Alternative Parameter/Model	Total Impact Difference	% Change
Noise Model	Integrated Noise Model	ANCON	– £14.8 million	- 18%
Amenity Damages Model	Income-based approach	Annoyance quality of life	+ £15.9 million	+ 20%
MI Dose Response	Linear model	Polynomial model	- £13.9 million	- 17%
MI Fatality Rate	Longitudinal study by age/gender	2001 survival rate (72.4%)	+ £23.7 million	+ 29%
MI Valuation Approach	Value of Life Years [VOLY]	Value of a Statistical Life [VSL]	+ £75.6 million	+ 93%
Stroke Dose Response	Linear (direct stroke)	Secondary effect of hypertension	- £3.4 million	- 4.2%
Stroke Valuation	Direct, indirect costs	Disability adjusted life years	+ £6.9 million	+ 8.5%
Hypertension endpoint	Direct costs	Dementia	+ £3.4 million	+ 4.2%
Discount Rate	3%	2%	- £8.0 million	- 9.9%

Noise Model

The noise exposure around London airports using the Integrated Noise Model. The UK's ANCON noise model has also been used to assess noise exposure around London (Lee et al. 2012). The two modelling approaches agree within 5% in total area noise exposure above 57 dBA (Leq). However, because ANCON predicts fewer people exposed to higher noise levels (Leq > 69 dBA), the total noise damages are more than 5% lower than those predicted with INM. Using ANCON modeled population exposure results in a total impact £14.7 million less than the impact predicted with INM, a difference of 18%. Both models are widely used in the literature, and are subject to uncertainty on the order of +/- 2 dBA (White 2005, Mahashabde et al. 2011). Thus, both approaches are appropriate for modelling current and future noise exposure.

Amenity Damages Model

Perceivable and attributable damages from aircraft noise (i.e. annoyance) are modeled using a relationship developed from a meta-study of hedonic pricing surveys. Other models of willingness to pay for noise abatement have been developed from hedonic pricing studies and used for calculating damages from aircraft noise. For example, a study of housing prices in Birmingham has been used to quantify noise damages in transportation assessments in the United Kingdom (Nellthorp et al. 2007). Using this model, the estimate of total noise damages at LHR and LGW decreases by 22% relative to using the income-based approach adopted in Appendix A. Further, because the income-based approach uses willingness to pay values developed specifically for aircraft noise and accounts for differences in valuation and preference between metropolitan areas through the income parameter, it is expected the income-based approach to be more appropriate than the Birmingham model. Stated preference techniques, such as direct surveys of noise-affected residents, are another methodology for valuing amenity damage from noised based on willingness to pay for abatement (or willingness to accept damages).

Alternatively, amenity damages can be estimated by valuing the underlying physical impact of the externality: in this case, annoyance. The World Health Organization (WHO) developed a methodology for calculating the monetary impact of noise annoyance, which proposes first calculating the number of people likely to be highly annoyed from a dose-response relationship and then estimating the relative disamenity of a year of annoyance compared to a year in standard health using a disability weight (WHO 2011). The UK Government recently adopted this methodology in its transport assessment tools (Defra 2014). Adopting the UK Defra approach for amenity impacts produces total 2010 London airport noise damages that are £15.9 million greater than those computed by the income-based approach, a difference of 20%. In addition to producing damage values of similar magnitude, the two approaches have similar uncertainty ranges, with the range between low and high values being 1:15 and 1:12 for the income-based approach and the disability weight approach respectively.

Both approaches have advantages and disadvantages. Hedonic pricing methods assume a liquid and unconstrained housing market, are limited by their choices of explanatory variables, and may introduce benefit transfer bias if the affected populations in the test case differ substantially from the populations from which the model was built. The disability weight approach introduces uncertainty in the value of the disability weight but also in the annoyance dose-response relationship and in the value of a quality-adjusted life year. However, this uncertainty is not accounted for in the current Defra approach. Furthermore, the issue of benefit transfer is similarly present, as the underlying annoyance dose-response curve indicates significant variability in the subjective response of individuals to noise, and the mean best-fit of this curve may not be representative of the behavior of the me noise-affected population in the vicinity of the London airports.

Myocardial Infarction Damages Model

Three influential modelling choices are identified in the myocardial infarction (MI) damages model: the dose-response relationship, the MI fatality rate, and the valuation approach. While the link between noise exposure and MI is well established, there remain significant uncertainties in the physical pathway, the existence (or lack) of a threshold value, and the magnitude of the relative risk of MI with increasing exposure level. In particular, the relationship between aircraft noise specifically and MI is subject of only a few studies (Basner et al. 2014). As such, the functional form of the dose-response relationship is still uncertain. Two common approaches are the linear relative risk approach (Basner et al. 2014) and the polynomial odds ratio approach (Babisch 2006). Changing from the linear model to the polynomial model results in a decrease in total 2010 noise damages around LHR and LGW of £13.9 million, a change of 17%.

The fatality rate from MI depends upon the definition of causal fatality and the choice of underlying population and incidence dataset. The metric for causal mortality is dependent upon the length of time after incidence or hospitalization through which the death is attributable to the MI. Popular reporting metrics include 24-hour, 30-day, and 90-day fatalities per 1,000 incidents or per 1,000 hospitalizations. Furthermore, the MI survival rate may differ substantially from location to location and from year to year depending on changes in the health environment and because of natural variability. For example, the UK Interdepartmental Group on Cost and Benefits Noise Subject Group (ICGB[N]) uses a mortality rate of 72%, which is derived from total deaths due to acute myocardial infarction in the UK in 2001 (IGCBN 2010). However, using the same statistic from 2006 would reduce the fatality rate to 46% (BEL 2008). In the analysis in Appendix A, age and gender specific 30-day fatality rates for England from 2010 are used (Smolina 2012). Changing from this fatality statistic parameter to that of the IGCB[N] would increase total noise damages by £23.7 million, a difference of 29%.

Finally the value by which fatal and non-fatal MI's are evaluated is subject to uncertainty and social and policy-maker preference. The Value of Life Years lost approach, which monetizes each year of life lost, is used in the study presented in Appendix A and has been adopted in the United Kingdom for regulatory analysis. However, the value assigned to a year in full health has varied across analyses and agencies. For example, the Interdepartmental Group on Costs and Benefits Air Quality Subject Group has used a value of £29,000 for a quality-adjusted life year, less than half the value adopted by the IGCB(N) (IGCB(N), 2010). Furthermore, in the United States, the Value of a Statistical Life (VSL) approach, where all theoretical mortalities are evaluated identically regardless of onset age, is more commonly applied in regulatory analysis. The VSL approach was used in the analysis presented in Section 4.2. Adopting the VSL approach and the 2010 value adopted by the US Department of Transportation (DOT, 2014) would result in total noise damages in London that are 93% higher than total noise damages under the VOLY approach, a difference of £75.6 million.

Stroke Damages Model

Of the impact endpoints quantified by the model in Section 3.1, the effect of noise exposure on stroke incidence is the most uncertain, with some researchers believing that any causal relationship between the two is still unknown (Kolstad et al. 2013). In the model developed in this thesis, a best-guess estimate of stroke damages is calculated by applying a direct dose-response relationship between aircraft noise and stroke consistent with recent literature (Hansell et al. 2013, Floud et al. 2013). Alternatively, stroke incidents may be estimated by computing the increased likelihood of stroke given the expected increase incidence of hypertension (Defra 2014). Adopting this approach may reduce the uncertainty in the causal relationship, but it may also undercount strokes caused by physical mechanisms not related to hypertension. Adopting the hypertension-to-stroke methodology is described by Defra would decrease the total damages from noise estimated around London airports in 2010 by £3.4 million.

The magnitude of damages from an additional stroke incident is uncertain and dependent upon the valuation approach. Adopting the disability-adjusted life year approach (Cadhilac et al. 2010) and the £60,000 VOLY value adopted for MI fatalities would increase the estimate of total noise damages from LHR and LGW by £6.9 million, an increase of 8.5%.

Hypertension Damages Model

There is stronger evidence for a defined dose-response relationship between aircraft noise and hypertension relative to other health endpoints modeled in Section 3.1.2 and applied in Section 4.2 and Appendix A (Basner et al. 2014). Because hypertension is a chronic disease whose primary economic burden may be through its role as a risk factor for other health impacts such as heart disease and stroke, increased risk of hypertension is difficult to monetize. For instance, hypertension alone is not expected to decrease the lifetime earning potential of an individual (Cropper and Krupnick, 1990). The baseline model calculates the increase in direct medical costs from increased hypertension as the cost of this health endpoint. Alternatively, a model could calculate the mortality and morbidity impacts from secondary diseases for which hypertension is a precursor. Harding et al. (2013) identify stroke, myocardial infarction, and dementia as three relevant health endpoints. As the baseline model directly estimates stroke and myocardial infarction, the hypertension damages of the baseline model are compared to those of the dementia endpoint of the alternative model. The dementia modeling approach in Harding et al. (2013) produces total 2010 noise damages around London £3.4 million greater than the baseline hypertension model.

Hypertension is defined by an age-specific cutoff blood pressure. However, aircraft noise may exacerbate the condition in existing hypertensive individuals, which would be unaccounted for in either the baseline or the alternative model. Furthermore, additional health endpoints not modeled and monetized may be influential including renal failure and aneurysm. Finally, the alternative model has uncertainty in the dose-response relationship and additional uncertainty from the hypertension-to-secondary-impact relationship.

Discount Rate

While noise itself is transient and ephemeral, its impacts on the environment can last into the future, particularly where it causes mortality or morbidity. Because the impacts of noise are temporally diffuse, future impacts are monetized by applying a social discount rate accounting for social preference and the time-value of money. The socially optimal discount rate is a subject of debate and may differ whether formulated on a descriptive approach based on opportunity costs or a prescriptive approach formulated on ethical views of equity (Moore et al. 2004, Arrow et al. 2013). Results are presented for two linear discount rates, 3% as applied in the original model and 2%. Because a lower discount rate means that future impacts are valued more highly, in most environmental assessments a lower discount rate would lead to a higher total monetary impact. This is true in the model for health impacts. However, because the baseline model first calculates the amenity damages of noise through capitalized housing values, decreasing the discount rate effectively moves some of the current damages into the future while holding the total capitalized damage constant. Thus, a lower discount rate of 2% lowers the total noise impact by £8.0 million, a difference of 9.9%.

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