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Review Article

Cost-Benefit Analysis Methods For Assessing Air Pollution Control Programs in Urban Environments — A Review

A. Scott VOORHEES[1], Ryoji SAKAI[1], Shunichi ARAKI[1,2], Hajime SATO[1], and Akiko OTSU[1]

[1]*Department of Public Health and Occupational Medicine, Graduate School of Medicine, The University of Tokyo, Tokyo*

Abstract

The most common method of evaluating beneficial impacts of environmental policies is cost-benefit analysis (CBA). In the present review, CBA methods for air pollution impacts are reviewed. Three types of air pollution effects are identified, including health, productivity, and amenity. Market valuation, stated preference methods, and revealed preference methods are identified for valuing benefits. Three types of costs are described, including private sector costs, societal costs, and governmental regulatory costs. A benefits valuation approach based on Freeman's principals is described. A costs valuation approach based on U.S. Environmental Protection Agency and Dixon et al. principals is described. Limitations associated with estimates of benefits and costs are summarized. Input assumptions and results are compared for several existing air pollution control analyses. The importance of CBA in environmental policy studies is discussed. Our conceptual approaches should be useful in analyses of urban air pollution impacts and air pollution prevention policies.

Key words: cost-benefit analysis, urban air pollution, environmental policy, benefit valuation, cost estimation

Introduction

Modern cost-benefit analysis (CBA) has been used for the last quarter century to estimate social costs and benefits, either possible or real, of certain societal choices. It is often applied as a decision making tool for development projects and is also useful in evaluating policies[1)]. However, the use of CBA as a tool for estimating environmental costs and benefits is relatively new and still evolving[2–4)]. Present interest in this field can be attributed to the 1987 report "Our Common Future"[5)] that outlined the concept of Sustainable Development, and a reorganization at the World Bank in the early 1990's that established the Global Environment Facility. In the United States, Presidential Executive Orders now require all major federal regulations to pass a cost-benefit test before they can be implemented. Section 812 of the U.S. Clean Air Act Amendments of 1990 stipulates that the U.S. Environmental Protection Agency (U.S. EPA) "should consider the costs, benefits and other effects associated with compliance with each standard issued for [each section of the Act]"[6)]. Several studies have been done to assess costs or benefits of federal pollution control policies and programs[7–10)].

Research into economic valuation techniques for use in environmental policy decision-making is occurring in North America and in Europe. African, Latin American and Southeast Asian countries are also interested in this field due to the desires of development banks and national governments to incorporate environmental concerns into policy and project appraisals[11)]. When governments require pollution control, massive expenditures by the public and private sectors may result. Estimates of the economic value of environmental clean up can help answer the question of whether diverting resources from the production of other goods and services makes a society better off[4)]. The most common method of evaluating environmental impacts where benefits can be valued is CBA[2)]. The use of CBA provides an account of the real costs and benefits of environmental policies by quantifying their environmental effects[1)]. There have been historical developments and disputes over the quantification of benefits and costs, namely, what items to include and how to calculate their importance in the overall valuation. Though CBA based results are not regarded as arbitrary, decisions made by analysts regarding input parameters and assumptions can have a significant bearing on results.

In the present review, we (1) reviewed the history of CBA for air pollution control programs, (2) identified benefits valuation techniques, (3) categorized costs of air pollution control, (4) described conceptual approaches for estimating benefits and costs, (5) compared CBA assumptions and results for several existing analyses of air pollution control in urban areas, and (6) summarized the importance of CBA in environmental policy studies.

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Reprint requests: A. Scott VOORHEES
Visiting Fellow, The University of Tokyo, 5109 Lansdowne Drive, Durham, NC 27712 U.S.A.
TEL: +1(919)541-5348 (work), +1(919)382-8477 (home), FAX: +1(919) 541-5509
E-mail: voorhees.scott@epa.gov
[2]Present address: National Institute of Industrial Health, Kawasaki

History

The methodological properties of costs and benefits analyses include time (i.e., prospective or retrospective), pollutant (i.e., single or multiple), type of pollution controls (i.e., single or multiple), and scale (i.e., urban, regional, or national). A chronology of urban- and regional-scale air pollution benefits and costs analyses is presented in Table 1. Such analyses include Loehman et al.'s CBA for electric power plants controls in Tampa, Florida[12], Chestnut et al.'s air pollution benefits analysis for lead, particulate matter, nitrogen dioxide (NO_2), sulfur dioxide (SO_2), ozone and carbon monoxide in Santa Clara, California[13], Chestnut and Rowe's particulate matter and ozone benefits analysis for Denver, Colorado[14], and Krupnick and Portney's prospective CBA for ozone pollution controls in Los Angeles, California[15]. Additional studies include Ostro's estimates of health impacts due to lead, particulate matter, NO_2, SO_2, and ozone in Jakarta, Indonesia[16], Dixon et al.'s follow up analysis of health impacts and medical expenses from particulate matter exposure in Jakarta[3], the U.S. EPA's analyses of the prospective costs of alternative pollution control strategies for particulate matter in Philadelphia, Pennsylvania and El Paso, Texas[17,18], Shin et al.'s estimates of the productivity impact of particulate matter, carbon monoxide and lead pollution levels in Bangkok, Thailand and other Asian cities[19], Austin et al.'s and Krupnick et al.'s prospective analyses of nitrogen oxides (NO_X) control costs in the Chesapeake Bay airshed[20,21] and our retrospective CBA of NO_2 in Tokyo[22].

A chronology of national-scale air pollution control analyses is presented in Table 2. National analyses include Freeman's CBA of U.S. air and water pollution control policies for the year 1978[8], Hazilla and Kopp's comparison of their estimates of social costs from air and water pollution control policies to U.S. EPA's estimates based on engineering data for the years 1975 and 1981–1990[7], and Portney's summary of U.S. air pollution policy benefits and costs for the year 1981 compiled from other researchers[9]. Additional studies include the European Commission s calculations of health effect externalities associated with air pollution from energy production in western Europe during the early 1990's[23], U.S. EPA's retrospective CBA of air pollution policies instituted under the U.S. Clean Air Act for the period 1970–1990[10], Burtraw et al.'s study of the impacts of pollution controls on acid rain sources in the U.S. for the time period 1995–2030[24], and the Committee on Japan's Experience in the Battle against Air Pollution's analysis of the costs of controlling air pollution in Japan for the time period 1966–1995[25]. These studies have been made either in response to existing environmental crises in anticipation of the need for pollution control policies, or to evaluate past major environmental policies in these areas.

Benefits valuation techniques for air pollution effects

Three principal environmental benefits valuation techniques identified in the literature are summarized in Table 3: (1) Market Valuation of Physical Effects, also known as "Techniques in Which Market Prices are Used to Value Impacts", (2) Stated Preference Methods, also known as "Contingent Valuation Methods", and (3) Revealed Preference Methods, also known as "Techniques in Which Surrogate Market Prices are Used". Objective benefits valuation techniques, which use market prices to value environmental effects, are based on physical relationships that formally describe cause and effect relationships and provide objective measures of damage resulting from environmental change. Objective approaches assume that rational individuals were willing to pay an amount less than or equal to the costs

Table 1 Chronology of urban- and regional-scale air pollution control analyses

Principal Investigator	Year(s) of Analysis	Study Area(s)	Pollutant(s)[a]	Type of Analysis
Loehman et al. (1979)	late 1970's	Tampa	SO_2, PM	prospective costs and benefits of air pollution controls on electric utilities
Chestnut et al. (1987)	uncertain	Santa Clara	lead, PM, NO_2, SO_2, O_3, CO	prospective benefits of air pollution control
Chestnut and Rowe (1987)	uncertain	Denver	PM, O_3	prospective benefits of air pollution control
Krupnick and Portney (1991)	1990's	Los Angeles	O_3	prospective costs and benefits of air pollution control
Ostro (1994)	1990's	Jakarta	lead, PM, NO_2, SO_2, O_3	prospective health impacts of air pollution control
Dixon et al. (1994)	1990's	Jakarta	PM	prospective benefits of air pollution control
U.S. EPA (1995)	2007	Philadelphia	PM	prospective costs of air pollution control
U.S. EPA (1995)	2007	El Paso	PM	prospective costs of air pollution control
Shin et al. (1997)	1985	Bangkok, Beijing, Bombay, Delhi, Hong Kong, Jakarta, Kuala Lumpur, Shanghai, Tokyo	PM, CO, lead	productivity impact of existing air pollution levels
Austin et al. (1997)	late 1990's	Chesapeake Bay airshed	NO_X	prospective costs of air pollution control
Krupnick et al. (1998)	2005	Chesapeake Bay airshed	NO_X	prospective costs and benefits of air pollution control
Voorhees et al. (2000)	1994	Tokyo	NO_2	retrospective costs and benefits of air pollution control

[a] SO_2=sulfur dioxide, NO_X=nitrogen oxides, PM=particulate matter, NO_2=nitrogen dioxide, O_3=ozone, CO=carbon monoxide.

Table 2 Chronology of national-scale air pollution control analyses

Principal Investigator	Year(s) of Analysis	Study Area	Pollutants[a]	Type of Analysis
Freeman (1982)	1978	United States	SO_X, NO_X, PM, lead, O_3, CO	retrospective costs and benefits of air pollution control policies
Hazilla and Kopp (1990)	1975, 1981–1990	United States	SO_X, NO_X, PM, lead, O_3, CO	retrospective costs of air pollution control policies compared with engineering costs
Portney (1990)	1981	United States	SO_X, NO_X, PM, lead, O_3, CO	retrospective benefits estimates and costs estimates compiled from other researchers for air pollution control policies
European Commission (1996)	early 1990's	Western Europe	SO_2, NO_X, PM	health effect externalities from existing levels of air pollution due to electricity generation from fossil fuel combustion, nuclear fission and renewable energy sources
U.S. EPA (1997)	1970–1990	United States	SO_2, NO_2, PM, lead, O_3, CO	retrospective costs and benefits of air pollution control
Burtraw et al. (1997)	1995–2030	United States	SO_2, NO_X	prospective costs and benefits of air pollution controls on electric utilities
Japanese Committee (1997)	1966–1995	Japan	SO_X	retrospective costs of pollution control; impact of pollution control timing on economic efficiency

[a] SO_X=sulfur oxides, NO_X=nitrogen oxides, PM=particulate matter, O_3=ozone, CO=carbon monoxide, SO_2=sulfur dioxide, NO_2=nitrogen dioxide.

Table 3 Summary of benefit valuation techniques for valuing environmental effects[a]

Environmental Valuation Techniques	Principal Technique	Methods Used in Each Technique Category	Description of Method
Market Valuation of Physical Effects	uses market prices to assign value to the physical effects of environmental change	dose-response function	measures physical impacts of control on environment (e.g., air pollution on materials damage, crop damage or human health) *used to value productivity, health*
		damage function	uses dose-reponse data to estimate economic cost of environmental control (values physical effects using market prices of the units of output) *used to value productivity, health*
		production function	environmental input (e.g., air quality) is related to output (e.g., farm crop yields increase or decrease) *used to value productivity*
		human capital/cost of illness	estimates cost of ill health on worker output due to environmental change (human capital) estimates medical cost of bad health due to environmental change (cost of illness) *used to value productivity, health*
		replacement cost	estimates cost of damage using costs which injured parties incur in alleviating a harm *used to value productivity, health*
Stated Preference Methods	surveys people and directly asks them their preferences	contingent valuation method	asks people directly how much value they assign to a change in environmental quality (willingness to pay for improvement or to prevent deterioration; willingness to accept compensation) *used to value health, amenity, existence values*
Revealed Preference Methods	observes people's actual economic market choices and indirectly draws inferences about their preferences	travel cost method	examines time and cost incurred in visiting and enjoying a natural site as a surrogate for its value *used to value amenity*
		avertive behavior/defensive expenditure	uses data on what people spend to protect themselves against actual or potential environmental decline *used to value productivity, health*
		hedonic pricing method	assumes the price of land (i.e., real estate) reflects the quality of the environment in which it is located *used to value amenity*

[a] Table based on descriptions in Organisation for Economic Co-operation and Development[1] and Dixon et al.[3].

incurred as a result of the effect. Subjective benefit valuation methods, which use surrogate market prices, are based on assessments of possible damage expressed or revealed in real or hypothetical market behavior[3].

Using market prices as a means of valuing environmental effects assumes that market prices accurately reflect the value that individuals in society place on those effects, and that all individuals behave uniformly in their consumption decisions. If the government provides financial assistance to regulated industries, it would be inaccurate to rely on market prices charged by companies for their products when calculating costs, because those market prices may not reflect the true cost of producing those products. Where the government provides subsidies or tax exemptions, or if there are price and quantity controls, the market price understates the true cost by an amount called the shadow price[11].

Using surrogate market prices as a valuation tool assumes that market prices do not or could not accurately reflect the value that individuals place on environmental effects, and assumes that all individuals do not behave uniformly in their consumption decisions. Surrogate market prices are identified by assessing stated or revealed market behavior. Stated market behavior relies on constructing hypothetical markets for various options to reduce environmental damage, and then uses surveys to ask people directly to express how much they would be willing to pay for an improvement in environmental quality or what they would be willing to accept for a loss of environmental quality. Revealed market behavior relies on observation of people's actual market behavior and then makes indirect inferences about their preferences[1,3].

Categorizing costs of air pollution control

As outlined in Table 4, costs associated with implementing environmental policies are divided into three economic sectors, including private sector costs, societal costs, and governmental regulatory costs[3,26]. The costs incurred by the private sector include direct and indirect costs. There are two types of direct costs, capital costs and operating costs. Capital costs include expenditures for facilities and equipment, as well as changes in production processes that reduce or eliminate pollution generation. Capital costs are amortized over time, because such costs are incurred in a short period of time, but provide pollution control benefits over a longer time frame. Amortized capital costs represent the real resource costs of tying-up funds in the purchase and installation of capital equipment or other fixed assets required by environmental regulation. Operating costs consist of all costs and expenses for the operation and maintenance of pollution control processes, including spending for materials, equipment leasing, parts and supplies, direct labor, fuel and power, services provided by private contractors, and research and development. In multi year analyses, annual costs include the operating costs for the year in question plus one year s amortized capital costs[26].

In addition to direct costs, the private sector incurs indirect costs (also known as "second order effects" or "opportunity costs") from pollution control requirements. Indirect costs exist for regulated industries and also for the private sector as a whole. In the case of regulated industries, a factory may decrease its overall production if it diverts capital to purchase and operate pollution control equipment, it may charge a higher price for its outputs of goods and services when it has to pay for pollution control, or it may lay off employees when it uses some of its resources to pay for pollution control. For the private sector as a whole, environmental regulations will result over time in macroeconomic impacts due to shifts in economic activity between industries, including changes in distributions of labor, capital, and other production factors within the economy, and changes in the distribution of goods and services[10]. Society also incurs opportunity costs, defined as the forgone income from other uses of a resource which cannot be realized because the resource was used to comply with environmental regulations[3]. The costs for national and local regulatory agencies consist of the portion of governmental agency budgets devoted to implementing and overseeing environmental programs, including personnel, contracts, and financial assistance to regulated businesses.

Conceptual framework of cost-benefit analysis in urban air

Estimating benefits and costs of air pollution control for urban areas can be divided into three components. First, the impacts of air pollution are estimated, then the benefits are estimated, and the costs are calculated. There are two types of analyses used to evaluate the temporal relationship between pollution and its impacts, prospective ("*ex ante*") and retrospective ("*ex post*"). There are two scenarios used in *ex ante* analyses, "no-control" and "control". In no-control *ex ante* analyses, the impacts of air pollution are based on *real* air pollution levels, the benefits consist of the expenditures for pollution control equipment that industry is not paying, and the costs consist of the value of *actual* medical expenses that society is paying while exposed to pollution. On the other hand, in control *ex ante* analyses, the impacts of air pollution are based on *hypothetical* future clean air, the benefits consist of the value of *potential* medical expenses that society would not pay in the future if it were not exposed to pollution, and the costs consist of the *potential* expenditures for pollution control equipment that industry would pay in the future. In *ex post* analyses, the impacts are based on *hypothetical* air pollution that was prevented by air pollution control policies in the past, the benefits consist of

Table 4 Costs of air pollution control[a]

Economic Sector	Type	Categories of Costs	Specific Cost Items
Private Industry	Direct-Capital	Expeditures for facilities and equipment	
		Changes in production processes	
	Direct-Operating	Operation and maintenance of control equipment/processes	Materials
			Equipment leasing
			Parts and supplies
			Direct labor
			Fuel and power
			Contractor services
			Research and development
	Indirect-Regulated Industries	Decreased production	
		Increased product prices	
		Employee layoffs	
	Indirect-Other Industries	Changes in distribution of labor, capital, other production factors	
Society	Indirect	Forgone income	
Government	Direct	Personnel	
		Contracts	
		Financial assistance	

[a] Derived from U.S. EPA[10,26] and Dixon et al.[3].

the value of those *potential* medical expenses that society did not pay because it was not exposed to pollution, and the costs consist of the *actual* expenditures for pollution control equipment that industry paid.

Both *ex ante* and *ex post* analyses can be thought of as comparative in nature. Prior to the implementation of a regulation, it may be informative to policy makers to compare the benefits of no-control (i.e., avoided equipment expenditures by industry), with the costs of no-control (i.e., medical care expenditures by citizens). Both before and after the implementation of a regulation, it can be illustrative to calculate the benefits of control (i.e., avoided medical care expenditures by citizens) and compare them with the costs of control (i.e., equipment expenditures by industry). The distinction between *ex ante* and *ex post* analyses is temporal in nature. Depending on the timing of the analysis, costs of no-control can be considered to be synonymous with benefits of control, and benefits of no-control can be considered to be synonymous with costs of control.

Ostro proposed a four step process for estimating pollution impacts in an *ex ante* analysis of Jakarta, Indonesia: (1) estimate the dose-response function, (2) multiply the slope of the dose-response function by the exposed population, (3) estimate the change in air quality, and (4) calculate the economic value of the predicted health effects[16]. Ostro's process was chosen by Dixon et al. as a case study for how benefits analysis should be conducted for air pollution impacts, and they carried the analysis further by estimating the dollar value of reducing air pollution[3]. Winpenny included similar steps for calculating environmental effects: (1) determine emissions, (2) estimate air concentration, (3) establish dose-response functions, (4) define the population at risk[2].

The U.S. EPA applied a seven step approach in its *ex post* CBA study of the Clean Air Act. Step 1 was the estimation of direct costs and Step 2 was the modeling of macroeconomic impacts. Steps 3 through 6 were the benefits estimation steps. Step 3 was the modeling of air pollutant emissions. Step 4 was the modeling of air quality, the identification of ambient monitoring data, and the estimation of no-control concentrations of air pollution. Step 5 was the use of control and no-control air quality data in combination with dose-response functions to estimate health and environmental effects. Step 6 was the estimation of the economic value of a change in the incidence of adverse effects. Step 7 was the aggregation of results and uncertainty characterization[10]. In either case, the core components are the same, which emphasizes the relatively straight forward nature of the CBA process. By application of these steps, the following four sections describe procedures for identifying critical benefit and cost components and outline some of their limitations.

Framework for benefits valuation

According to Paul Portney of Resources for the Future, A. Myrick Freeman's 1979 text "The Benefits of Environmental Improvement: Theory and Practice" was considered the definitive reference on environmental benefit estimation for many years, and became the *de facto* standard for use in applied cost-benefit analyses of environmental regulatory actions[4]. Freeman's model of environmental and resource valuation in his updated text "The Measurement of Environmental and Resource Values: Theory and Methods" serves as the basis for a conceptual benefits valuation approach. His model is based on the premise that the environment is a resource which provides services that can be valued. For example, air provides oxygen so humans and other animals can survive, crops can grow, and industry can operate. It also provides a sink for by-products of the economy, including a place for industry to discharge waste, a place for humans to exhale, and vegetation to release oxygen. Freeman described three functional relationships that allow for a thorough analysis of benefits. He referred to these relationships as "some measure of environmental or resource quality [related] to the human interventions that affect it", "the human uses of the environment or resource and their dependence on [the quality of the resource]", and "the economic value of the uses of the environment"[4]. As applied specifically to air pollution, these relationships can be expressed in the form of three questions: (1) How are levels of air pollution affected by humans? (2) What are the effects of air pollution? (3) What is the economic value of the effects of air pollution?[22].

The first question relates air quality to those human actions that affect it. When the government imposes environmental restrictions on sources of pollution, industries respond by altering their operations to comply with the new requirements. The effect on air quality of a change from not regulating air pollution to regulating air pollution depends on the private sector response. Not all regulated industries may accept legal requirements equally, but clearly, as the degree of compliance with an environmental policy rises, air quality improves. In other words, the quality of air is affected by environmental regulations, pollutant emissions, and control of those emissions.

The second question concerns the effects of air pollution. Based on Freeman's classification of environmental effects, three separate "systems" use air: human systems, ecosystems, and nonliving systems[4]. The level of activity involving air is dependent on the quality of that air after the inputs of labor (e.g., the efforts people make to protect themselves from dirty air), capital (e.g., money spent to clean up air that is not pure enough, money spent to relocate away from a polluted area) and other resources, such as time (e.g., time spent traveling to a location to exercise or work where the air is cleaner).

The people, plants, animals or nonliving things that are potentially affected are considered to be the susceptible populations, or subjects at risk, and the effects on those subjects can be divided into three categories: health, productivity and amenity[1]. For example, in human systems, the effect of air pollution on a citizen's health is considered to be a health effect. The effect on a worker's work output is considered to be a productivity effect. The effect on noise or odor levels on a person's mental or physical well-being is considered to be an amenity effect. The third question concerns the economic value of the effects of pollution in a society that result from using the air resource. Value is expressed in monetary terms based on what the society considers those effects to be worth.

Limitations of estimating benefits

The effects of air pollution that are most amenable to market valuation include human health effects and productivity effects (e.g., work output, crop yield, fishing yield, damage to industrial equipment and soiling). Amenity effects, including visibility, odor and noise, are the most amenable to surrogate market valuation using either contingent valuation or hedonic pricing. Ecosystem health, however, typically does not have market value, nor can it be readily valued with surrogate market prices since people's behavior is difficult to link to any preference for a healthy natural ecosystem over an unhealthy one. Materials damage can be assigned a market-based value using a combination of dose-response function followed by a damage function, but often is unquantified due to the absence of dose-response functions. Likewise, natural ecological impacts which may occur cannot be valued without relevant dose-response functions. Avertive behavior (e.g., relocation by asthmatics to avoid breathing polluted air) and defensive expenditures (e.g., purchase of air purification machines for clean indoor air) are also difficult to isolate and quantify given the data that are typically available for an urban area. Both require surveys that demand the use of resources and time beyond the scope of most analyses.

Economic externalities consist of those benefits of a resource which are not sold as market products, but which have inherent value[3)]. Benefits estimates should attempt to value the importance of air to natural ecosystems and its importance to the productivity of industries, farms and forests. However, it is often not possible to quantify those benefits due to a lack of a sufficient database of ecosystem health and ecosystem/nonliving system productivity effects to develop dose-response functions.

In our previous Tokyo study, we assumed that changes in emissions of NO_X resulted in the same level of change in air concentrations of NO_2. This assumption was also used by the U.S. Environmental Protection Agency in its national ex post analysis. The U.S. agency noted that changes in air quality were treated as proportional to estimated changes in emissions because, for NO_X, "changes in ambient concentrations in a particular area are strongly related to changes in emissions in that area"[10)]. Sensitivity analysis performed in our previous study indicated that the use of different assumptions for NO_X emissions would have affected our benefits calculations and our costs calculations more than any other input variables. Regarding benefits, if we had applied an alternative assumption of NO_X emissions from motor vehicles, which would then impact our estimates of NO_2 concentrations, the benefits estimate would have been reduced by 35%. Likewise, our cost estimates could have increased dramatically by 420% or decreased by 16% if the upper and lower bounds of our ranges of emission volumes emitted by factories and other stationary sources were assumed respectively, rather than the midpoint[22)].

Framework for costs valuation

The U.S. EPA's control cost document "Environmental Investments: The Cost of A Clean Environment"[26)] was the basis for U.S. EPA's *ex post* CBA of national air pollution programs for the United States[10)]. Dixon et al.'s costs procedures in "Economic Analysis of Environmental Impacts" are used by the Asian Development Bank and the World Bank for analysis of environmental impacts[3)]. Based on the U.S. EPA and Dixon et al. references, eleven data items are identified as important to estimate costs. (1) "Environmental regulations" defines the stringency of control. (2) "Extent of coverage by regulations" defines the types and percentages of pollution sources that are ultimately required by the regulations to adopt pollution controls. (3) "Degree of compliance with regulations" defines the number of sources that are complying with the regulations. (4) "Types and numbers of sources/motor vehicles in the study area" defines how many sources and motor vehicles there are in the study area. (5) "Air pollution control technologies" defines the types of technologies used to control air pollution. (6) "Percentages/numbers of sources/motor vehicles with controls installed" identifies the type and number of sources and motor vehicles whose deadline for using control methods has already passed. (7) "Capital and operating costs of pollution control technologies" defines the costs of purchasing, installing, operating and maintaining each type of control equipment. (8) "Direct pollution control costs for sources/motor vehicles in the study area" defines the direct costs for sources in the study area. (9) "Indirect costs for sources/motor vehicles and the macroeconomy" defines indirect costs for sources, motor vehicles and the macroeconomy. (10) "Societal opportunity costs" defines the ripple effects of regulations on society. (11) "National and local government costs" defines the costs incurred by governmental regulatory agencies.

Limitations of estimating costs

Reliance on engineering costs has been criticized because capital and operating expenses alone do not account for dynamic and general equilibrium impacts of regulation. Rather, costs should be measured by the amount of money required to compensate individuals for unfavorable effects associated with regulatory policies and which leave those individuals no worse off after the policy than they were before the policy was implemented[7,9,15)].

Due to the absence of data, our previous study in Tokyo was not successful in estimating the indirect costs incurred by regulated industries. The EPA noted that indirect (i.e., second-order) benefits were excluded from its benefits calculation, and the resulting CBA included a comparison of only direct costs and direct benefits. This decision was supported by EPA's external review panel of scientists and economists[10)]. Our previous study was also unable to calculate the indirect macroeconomic costs resulting from regulations. A computer simulation model would have been needed to derive these impacts. In discussing macroeconomic costs in its national-scale CBA, the EPA has noted that macroeconomic modeling provides information on macroeconomic costs but not on macroeconomic benefits, and concluded that "estimated second-order macroeconomic effects were small relative to the size of the U.S. economy"[10)].

Comparison of CBA assumptions and results

In calculating the benefits and costs as described in the framework above, the choice of assumptions and inputs can substantially affect the resulting benefits and costs valuations. Data of importance include health effects and duration, medical costs and wages, and in particular pollutant emission volumes and air concentrations. Sufficient attention should be given to these assumptions when reading, comparing, or applying CBA across pollutant, location, and time. Different pollutants produce differing impacts with varying degrees of severity depending on exposure concentration and duration. Benefits and costs reflect local (domestic) prices, and any cross-cultural comparison should address possible biases. Time discounting may be necessary, as in the case of comparing options for future policy changes when the time horizons for both the imposition of the costs and the enjoyment of the benefits are measured in years. The use of discounting techniques including net present value, internal rate of return, or benefit-cost ratio is critical to the proper incorporation of the trade-off between present and future consumption, known as "time preference"[3)].

Duration of illness, medical costs and lost wage costs

Herein, we compared the assumptions and results of various air pollution-related CBA from the literature. As shown in Table 5, Dixon et al. assumed 2 weeks per episode, irrespective of pollutant, for duration of lower respiratory illness in children, based on cost of illness[3] and we assigned 2 to 3 days duration per incidence for phlegm in adults and lower respiratory illness in children based on cost of illness[22]. Dixon et al. assumed $210 per case for lower respiratory illness in children using cost of illness[3] and we assigned a mean value of $210 in medical costs per incidence based on cost of illness[22]. Medical expenses in Tokyo would tend to be higher than expenses in most other world megacities due to the relatively high cost of living. We assumed $320 to $480 per incidence in lost wages in adults using human capital valuation[22]. Using human capital valuation, Dixon et al. assumed either $116 or $232 in lost wages by parents for lower respiratory illness in children based on the assumption of two Restricted Activity Days (RAD) per parent for care per episode, valued at $58 for each RAD[3]. We assumed $200 to $310 in lost wages for parental care per episode of lower respiratory illness in children using human capital valuation[22]. Kenkel's values[27], which were based on pre-1994 citations in the literature, were lower for disease duration, medical cost, and lost wages.

In our previous study, our duration of illness and medical costs were based on mean values from actual outpatient cases for treatment of pollution-related health impacts in Tokyo over a one month period in 1995 (n=15,239)[28]. Likewise, our wage rates were specific to Tokyo. Using site-specific information on illness duration, medical expense, and wages is preferable when available. Dixon et al.[3] indicated that "[e]conomic costs for changes in morbidity are, of course, very country-specific". We recommend the use of region-specific data where available; otherwise, generic values can be used, such as those cited by Kenkel[27], acknowledging that medical costs and wages have risen in the intervening years since Kenkel's work was published. For duration of illness, which can vary widely depending on the severity of the health effect being evaluated, we suggest including a scenario in a sensitivity analysis which assumes a reasonable duration different from the default value.

Per-person work loss days

As shown in Table 5, we estimated the hypothetical number of work loss days (WLDs) per person in 1994 for workers as 4.7 and for working mothers as 0.61[22]. Work Loss Days are defined as the excess number of days that illness or injury prevent an individual from working. We were unable to identify any other analyses that estimated WLDs for NO_2 exposure. However, Shin et al. estimated total annual WLDs in several Asian cities due to existing particulate levels and we calculated their per-person equivalents to range from 1.1 WLDs in Kuala Lumpur to 4.5 in Bangkok. Shin et al. also estimated the number of Restricted Activity Days (RADs) in one year, which is defined as WLDs plus days where activity was restricted but the person works anyway[19]. We derived per-person equivalents, including 8.7 days (Bangkok), 7.3 days (Beijing), and 4.4 days (Shanghai). Kenkel, Dixon et al., Ostro, and Shin et al. all cited U.S. cohorts for health effects data.

As noted above, in our previous study we relied on the average duration of illness value cited for cases of pollution-related illness in Tokyo. We applied actual demographics data on the number of workers and data from a survey of working women in Tokyo to calculate work loss days. Even though per-person work loss days can be compared across analyses to validate the "reasonableness" of a given study's results, each analysis is obviously unique to some extent. Pollutant type, concentration and dose-response functions, and thus the calculated per-person work loss days, will vary depending on the region being analyzed. Nonetheless, given the similar results which we predicted compared to

Table 5 Comparison of cost-benefit analysis of nitrogen dioxide control and air pollution control benefits in various cities

Principal Investigator (year)	Pollutant[a]	Target Illness	Study City	Avoided Incidence of Illness Per Capita mean (LL, UL)[b]	Duration of Illness	Medical Costs Per Incidence mean (LL, UL)[b]	Lost Wages - Workers Per Incidence mean (LL, UL)[b]	Lost Wages - Mothers Per Incidence mean (LL, UL)[b]	Work Loss Days Per Capita mean (LL, UL)[b]
Ostro et al. (1994)[d]	NO_2	respiratory symptoms	Jakarta	0.20 (0.12, 0.28)					
	TSP	respiratory symptoms	Jakarta	5.2 (2.6, 7.9)					
	TSP	asthma	Jakarta	0.078 (0.039, 0.66)					
Dixon et al. (1994)[e]	TSP	respiratory symptoms	Jakarta	3.4					
	TSP	asthma	Jakarta	0.052					
	TSP	LRI[c]	Jakarta	0.012	2 weeks	$210		$116 or $232	
Kenkel (1994)[f]	not specified	all respiratory diseases			4.1 days	$87	$56		
Shin et al. (1997)[g]	TSP		Kuala Lumpur						1.1
	TSP		Beijing						3.7
	TSP		Bangkok						(0.81, 4.5)
	carbon monoxide	fatigue & headaches	Bangkok	0.40					(0.024, 0.060)
Voorhees et al. (2000)[h]	NO_2	phlegm	Tokyo	2.6 (2.4, 2.7)	2–3 days	$210 ($170, $240)	($320, $480)	($200, $310)	4.7 (4.4, 5.0)
		LRI[c]		0.33 (0.30, 0.35)					0.61 (0.56, 0.66)

[a] NO_2=nitrogen dioxide, TSP=total suspended particulates [b] LL=lower limit, UL=upper limit [c] LRI=lower respiratory illness [d] Ostro[16] [e] Dixon et al.[3] [f] Kenkel[27] [g] Shin et al.[19] [h] Voorhees et al.[22].

Shin et al., it would be reasonable to assume a 2 or 3 day loss of work per case of illness due to uncontrolled air pollution, if site-specific data on work loss days are not available.

Per-person illness incidence

Our estimated numbers of avoided health cases in Tokyo were derived by multiplying the slope of the dose-response function times exposed population times exposure concentration. The resulting numbers of avoided cases of phlegm in adults and lower respiratory illness in children (30 million and 3.8 million respectively), were then divided by the 1994 population of 11.6 million. As shown in Table 5, our per person estimates of avoided phlegm in adults in 1994 was 2.6 and for lower respiratory illness in children was 0.33[22]. There is a dearth of research into the incidence of NO_2-related illness on urban populations. Ostro was the only researcher we identified who has estimated the number of cases of phlegm due to NO_2 exposure within a specified urban population. Applying the same Schwartz and Zeger[29] dose-response function for phlegm in adults as we did, he estimated the impact of reducing NO_X levels in Jakarta from their estimated annual average values of 50–350 μg/m^3 (0.025–0.175 ppm) based on dispersion modeling of emissions, to a proposed Indonesian standard of 100 μg/m^3 (0.05 ppm). His concentration isopleth mapping showed most of Jakarta with annual average concentrations of 100 μg/m^3 (0.05 ppm) or less and two small areas with concentrations between 300 and 350 μg/m^3 (0.15–0.175 ppm), and his estimates of avoided incidence of phlegm in adults were 1.77 million[16]. Assuming a population of 9 million, we converted this to a per-person incidence of 0.20.

The principal reason why the results in our study were higher was because of a much broader area of hypothetical high NO_2 concentrations in Tokyo. Although our highest estimate of no-control NO_2 concentration (0.15 ppm) was predicted at only one site, all but three of our 87 ward and city monitoring sites throughout Tokyo were calculated to have no-pollution-control annual average concentrations above 0.05 ppm and 31 of those sites were estimated to have concentrations at or above 0.1 ppm. Our per-person incidence rates were based on ward and city specific population and concentration data, unlike Jakarta's rates which were based on city-wide data and thus diluted by population in low pollution parts of the city. If more geographically specific values were provided within various parts of Jakarta, we would have estimated higher per-person incidence in the inner city high concentration zones. A second possible explanation for our higher incidence results was our use of an average NO_2 monitored concentration in Tokyo cities (n=29) for eight of the cities, out of 27 cities in total, which had no monitoring data.

Ostro estimated the impact of reducing TSP levels in Jakarta from their estimated annual average values of 50–350 μg/m^3 based on dispersion modeling of emissions, to a California standard of 55 μg/m^3. His concentration isopleth mapping showed most of Jakarta with annual average concentrations of 100 μg/m^3 or less, with one area with concentrations between 300 and 350 μg/m^3, and three areas with concentrations between 200 and 300 μg/m^3. His estimates of avoided incidence of respiratory symptoms in adults were 47 million and 705,000 cases of asthma attacks[16]. Assuming a population of 9 million, we converted this to a per-person incidence of 5.2 for respiratory symptoms in adults and 0.078 for asthma attacks. Using Dixon et al.'s estimates of avoided illness due to exposure to particulates in Jakarta based on dose-response functions[3], we calculated per-person rates of 3.4 fewer cases of respiratory symptoms, 0.052 fewer asthma attacks, and 0.012 fewer cases of lower respiratory illness. Using Shin et al.'s estimates of illness due to exposure to current levels of carbon monoxide in Bangkok based on dose-response functions[19], we calculated a per-person rate of 0.40 cases of excess fatigue and headaches.

In our previous study we applied actual demographics data on the number of adults and children and site-specific air concentrations data. Even though per-person illness incidence can be compared across analyses, each analysis is obviously unique to some extent. Pollutant type, concentration and dose-response functions, and thus the per-person illness incidence, will vary depending on the region being analyzed. Given the relatively high illness incidence which we calculated, caution should be exercised in the estimation of uncontrolled pollution levels, which has a direct impact on the estimation of adverse health impacts. If uncontrolled pollution levels reflect current actual conditions, clearly the potential for over-estimation is less than if uncontrolled pollution must be estimated in a location where pollution has already been reduced.

Control cost-effectiveness of NO_X

As shown in Table 6, the cost effectiveness of NO_X control in our Tokyo study, expressed as dollars per ton of NO_X emissions controlled was approximately $1,400/ton for motor vehicles, $21,000/ton for all NO_X sources, and $91,000/ton for stationary sources[22]. This compares to $5,600/ton from motor vehicles in Virginia[20], $26,000/ton from all NO_X sources in the Chesapeake Bay Airshed[21] and $4,500/ton from all non-utility stationary sources in the Chesapeake Bay Airshed[21]. The high value that we calculated for stationary sources might have been lower if the NO_X control equipment also controlled other pollutants, thus allowing the control costs to be apportioned among the controlled pollutants. However, the types of controls used for NO_X do not typically reduce other pollutants, especially for combustion processes[30].

In our previous study, we applied actual site-specific NO_X control costs data in Tokyo. Even though control cost-effectiveness can be compared across analyses, each analysis is unique to some extent. Cost of control, fuel use and size of factories, and thus cost-effectiveness, will vary depending on the region being analyzed. However, our extensive use of site-specific cost data and the similarity with other researchers in our cost-effectiveness values for motor vehicles and all NO_X sources combined, suggests that our control cost values are not too dissimilar from other studies, and may have utility in future analyses of other urban locations if site-specific data are not available.

Ratio of benefits to costs of NO_X

As shown in Table 6, in our Tokyo study of NO_2 exposure, our best estimate of benefits exceeded the costs by a ratio of approximately 6 to 1[22]. In Krupnick et al.'s study of NO_X control from all NO_X sources in the Chesapeake Bay airshed to avoid ozone effects plus protection of the aquatic ecosystem[21], the benefits-to-costs ratio ranged from 0.07:1 to 0.8:1. The health effects which they valued using willingness to pay were due to ozone exposure. This is a reflection of differing political decisions between control of NO_X to reduce ozone levels in the United States and the control of NO_X to reduce ambient NO_2 in Japan.

Table 6 Comparison of cost and benefit-cost ratios for NO_2 control in various urban areas

Principal Investigator (year)	Pollutant[a]	Study Area	Sources Controlled	Cost Effectiveness of Pollution Control ($1,000/ton of NO_X controlled) mean (LL, UL)[b]	Benefit-Cost Ratio mean (LL, UL)[b]
U.S. EPA (1995)	NO_X	Philadelphia	all NO_X sources	5.4	
U.S. EPA (1995)	NO_X	El Paso	all NO_X sources	5.6	
Austin et al. (1997)	NO_X	Northern Virginia[c]	motor vehicles	2.7	
		Virginia[d]	motor vehicles	5.6	
Krupnick et al. (1998)	NO_X	West Virginia[e]	all NO_X sources	(0.24, 1.5)	(0.07:1, 0.8:1)
		Chesapeake Bay airshed[f]	stationary NO_X sources	(2.1, 4.5)	
		Chesapeake Bay airshed[f]	all NO_X sources	(17, 26)	
Voorhees et al. (2000)	NO_X	Tokyo	motor vehicles	1.4 (1.3, 1.5)	6:1 (0.3:1, 44:1)
			all NO_X sources	21 (19, 23)	
			stationary NO_X sources	91 (84, 98)	

[a] NO_X=nitrogen oxides, SO_2=sulfur dioxide.
[b] LL=lower limit, UL=upper limit.
[c] Includes the cities of Arlington, Alexandria, Reston, McLean and others.
[d] Includes the cities of Northern Virginia plus Richmond, Norfolk, Lynchburg, Charlottesville, Roanoke and others.
[e] Includes the cities of Charleston, Wheeling, Huntington, Parkersburg, and others.
[f] Includes the major cities of the United States east coast, including Boston, New York City, Philadelphia, Washington, D.C., Richmond and Atlanta, plus mid-western cities such as Pittsburgh, Detroit, and Cleveland.

Krupnick et al. did not value lost productivity as a benefit, which may explain why their benefits/costs ratio was an order of magnitude below ours. In Burtraw et al.'s study of NO_X and SO_2 control from electric utilities to avoid premature mortality, improve morbidity and visibility, a benefit-cost ratio of approximately 7:1 was reported. They included dose-response functions for particulates, SO_2, NO_2 and nitrates. They included eye irritation and phlegm for NO_2[24]. They estimated premature mortality due to particulate exposure, impacts on visibility and recreational fishing, plus estimates of morbidity from several pollutants, which would explain a relatively high benefits estimate. However, it appears that lost productivity was not valued, nor did they value health impacts in children, which would lead to an underestimate of benefits. Finally, their study was focused on power plants alone, rather than the complete universe of NO_X emission sources assessed in our Tokyo study. This would either lead to a higher or a lower ratio of benefits to costs than a complete NO_X source inventory assessment. Thus, this large disparity across past studies can be attributed, at least partly, to the assumptions and methodologies used by the respective researchers.

The choice of assumptions used in CBA can impact the outcome significantly. In our previous Tokyo study, for example, applying an alternate assumption of uncontrolled pollutant emissions decreased our benefits estimate by 35%. Assuming a shorter duration of absenteeism from work by employees decreased our benefits by 30%. Including different types of health effects increased our benefits by 200%[22]. Applying site-specific input data should be a primary goal of CBA where feasible. This is important especially for data that vary widely across cultures, such as medical costs and wages. If such data are lacking for the study area, a standard of living adjustment factor should be applied, or the potential bias should be quantified in a sensitivity analysis. Dose-response functions are rarely available for non-United States cohorts. The use of U.S. health effects data in a developing country context may introduce significant bias due to major differences in factors such as baseline health status, access to health care, and occupational exposure[3]. Given these caveats, CBA is nonetheless a valuable tool to evaluate environmental policy options. In the next section we summarize some important features of this analytical procedure.

Importance of CBA in environmental policy studies

In conducting CBA studies, we need a set of reliable data, as well as a number of assumptions which can be scientifically justifiable. If reliable data and justifiable assumptions are used, completing a CBA can perform several valuable functions in evaluating environmental policies. A necessary first step is identifying the severity of existing health impacts in polluted areas. Such information can be a powerful means of convincing policy makers that some effort should be taken to clean up the air or water. Ostro predicted significant reductions in mortality and morbidity if Jakarta's particulate matter, lead and NO_2 concentrations were lowered to World Health Organization standards[16]. Shin et al. estimated hundreds of deaths, millions of work loss days and restricted activities days in Asian cities that could be avoided if existing levels of particulate matter were reduced to United States air quality standards[19]. In our previous study we reached the conclusion that past NO_2 air pollution control policies in Tokyo were economically very effective, with a benefit-cost (B/C) ratio of 6 to 1. This is one important use of CBA, namely to ascertain if past environmental policies were economically worthwhile. Freeman reached the same conclusion in his nationwide analysis, while at the same time pointing out that stationary source pollution controls showed a higher B/C ratio than controls on mobile sources[8]. A second important use of CBA is to evaluate future controls to provide information to policy makers in order to inform their decisions. Krupnick and Portney found that the costs of future controls to reduce ozone concentrations would exceed the benefits[15]. Burtraw et al. estimated that the future benefits of reduced mortality and morbidity, and improved visibility, would exceed the costs[24].

More sophisticated applications of CBA allow for preferential decisions regarding control allocation and identification of externalities. Austin et al. completed a cross-media analysis of air and water pollution controls and made recommendations for cost-effective allocation of controls along geographic lines or according to preferences for air versus water quality improvement[20]. Krupnick et al. reviewed existing controls of NO_X in the Chesapeake Bay airshed and reached the conclusion that the costs of reducing emissions could be lowered by reallocating emission reductions based on type of source and also on geographic location[21]. Finally, the European Commission assessed the marginal environmental costs of energy production and identified key externality issues to be addressed in future policy[23].

Closing remarks

Numerous urban area and national scale studies have analyzed costs and benefits of air pollution and pollution control. Conducting a thorough CBA requires an estimate of impacts on health, productivity, and amenities. With this information, it is then possible to estimate the benefits and costs of either polluted air without pollution control, or clean air with pollution controls. Most analyses are prospective and provide estimates of what the benefits and costs would be if future regulatory actions were taken. Very few analyses are retrospective and provide estimates of the benefits and costs of regulatory actions in the past. Herein, we described the historical background of CBA, summarized existing benefits and costs methodologies, and proposed a conceptual approach for CBA in an urban setting, based on our experience in a CBA of the Tokyo metropolitan area. Much attention should be paid in reading and applying CBA studies, since the choice of assumptions and inputs can affect the results, especially for emissions volumes, air concentrations, health effects and duration, medical costs, and wages. If assumptions are fully described and the bounds of variables are quantified in sensitivity analyses, CBA can be a powerful tool for assessing both past and future policy choices and preferentially allocating pollution controls or identifying externalities.

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