Investing in Water Quality

Measuring Benefits, Costs and Risks

Clifford S. Russell, William J. Vaughan, Christopher D. Clark,
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Inter-American Development Bank

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EXECUTIVE SUMMARY

The management of water quality and quantity is a complex undertaking because a multiplicity of sources, uses, and stakeholders are involved. In particular, improvement of ambient water quality, the focus of this study, has become a critical issue in many countries in Latin America and the Caribbean. Economic and demographic growth have stressed water resources and rendered water and the ecosystems it supports increasingly scarce and valuable. The costs of reversing the trend of ever-deteriorating water quality in rivers and estuaries near large metropolitan areas are simply enormous.

The Bank's *Integrated Water Resources Management Strategy* issued in 1998 suggests that an integrated, basinwide approach should be adopted to formulate wise water resource policies and develop cost-effective investment programs in the water sector. The knowledge and tools to do so are available, so the failure to take an encompassing view that can hold down the systemwide costs of maintaining acceptable levels of ambient water quality involves costly and serious deficiencies in focus and planning. This study shows how the general principle of integrated water resources management can be put in practice to develop cost-effective and beneficial programs that reduce water pollution in urban areas.

Water quality improvement initiatives can be characterized by their level of geographic scope and their ability to allow benefits to be monetized. A review of a decade of IDB experience in the design and analysis of projects in urban water pollution control reveals that the best among them have attempted to identify the minimum-cost package of investments that would achieve ambient water quality standards in a basin or subbasin, and then have subjected the least-cost program to a cost-benefit test as well as a subsequent analysis step. Decision support systems are available that can be used to make a cost-effective selection of projects on a basinwide scale.

Turning to the benefits side, various indicators suggest that improvement of ambient water quality has a high priority in developing countries. There are several routes to estimating the benefits of better ambient water quality. They include the direct valuation of human health, recreation, or property value effects based on observed behavior, or more omnibus measures based on the preferences of respondents that are stated in contingent valuation surveys. In the developing country context, where collecting and matching epidemiological, leisure time, or property value information to pollutant exposure data is extremely difficult, the contingent valuation of hypothetical scenarios is probably the most practical way to construct estimates of household willingness to pay for water pollution control programs. It has become the method of choice at the IDB.

The point estimates of willingness to pay for better environmental quality are inherently uncertain, whether they are obtained from contingent valuation or via another route. No single economic valuation method provides a unique benefit number that is independent of the analyst's judgment, and variation across methods is common as well. Although the extent of statistical uncertainty (the variance of the mean benefit estimate) can be objectively quantified, methodological uncertainty is more elusive, so of necessity analysts must make subjective judgments to arrive at a reasonable or preferred estimate of benefits. Moreover, investment costs, operating costs, and the timing of project impacts are also uncertain.

Often uncertainties are not made explicit in studies of project feasibility, and this omission can dangerously mislead prospective investors about a proposal's social profitability. If the possibility of negative net benefits (downside risk or opportunity losses) is suspected to be nontrivial, the components of a program of investments in sewerage and wastewater treatment that form part of a least-cost package at the basin level also should be subjected to a cost-benefit test that incorporates risk considerations. The early identification of large, costly, and risky investments gives decision makers the flexibility to search for better alternatives before making an irrevocable commitment.

An example based on an actual IDB project demonstrates how uncertainty about the timing and magnitude of investment costs and benefits can be successfully handled through Monte Carlo risk analysis, which produces a probability distribution for expected net discounted benefits. This technique provides prospective investors with more information than the conventional approach, which calculates a single measure of economic feasibility and engages in a sensitivity analysis that by definition cannot realistically portray the combined effects of simultaneous uncertainty about all the key variables determining the expected outcome.

The study is primarily intended to provide guidance for engineers and economists involved in the economic analysis of large wastewater treatment projects.

ABBREVIATIONS

AEQ	ambient environmental quality
AWQ	ambient water quality
BAT	best available technology
BCT	best control technology
BMP	best management practices
BOD	biochemical oxygen demand
BPT	best practicable technology
CA	conjoint analysis
CBOD	carbonaceous biochemical oxygen demand
CDF	cumulative distribution function
CEA	cost-effectiveness analysis
CO	coliform bacteria
COD	chemical oxygen-demanding organics
CV	contingent valuation
CVM	contingent valuation method
CVPI	conditional value of perfect information
DALY	disability-adjusted life years
DO	dissolved oxygen
DOC	dissolved oxygen carbon
DOD	dissolved organic deficit
DSS	decision support system
EIR	economic rate of return (IDB term)
ELTA	expected loss of terminal action
EPA	U.S. Environmental Protection Agency
ES	environmental summary
EVPI	expected value of perfect information
EVSI	expected value of sample information
GDP	gross domestic product
GIS	geographic information system
IPPS	Industrial Pollution Protection System
IRR	internal rate of return
LAC	Latin America-Caribbean
LCA	least-cost analysis
MWTP	marginal willingness to pay
NAAQS	National Ambient Air Quality Standards
NBOD	nitrogenous biochemical oxygen demand
NGO	nongovernmental organization
NOD	nitrogenous oxygen demand
NPV	net present value

OECD	Organization of Economic Cooperation and Development
OLS	ordinary least squares
O&M	operation and maintenance
OMR	operation, maintenance, and repair
Pdf	probability density function
PS	producer's surplus
PTS	pretreatment station
PV	present value
RUM	random utility model
SOD	sediment oxygen demand
SPMA	São Paulo metropolitan area
SS	suspended solids
TC	travel cost
TCM	travel cost methodology
TEV	total economic value
TSP	total suspended particulates
TWTP	total willingness to pay
VA	value added
WQM	water quality model
WTP	willingness to pay
YLL	years of life lost

Chapter 1

Background and Problem Settings for Economic Analysis

Poor ambient water quality (AWQ) is a serious problem in a substantial part of the developing world, including the Latin American and Caribbean (LAC) countries. The causes of the problem are intertwined and indeed the nature of the problem is more complex than the simple phrase "ambient water quality" implies. The causes and nature of the problem will be examined in a moment, but it is important to understand the outcomes in a public health sense, for these outcomes drive the responses, of which the Bank's lending program is an important one.

BACKGROUND

Developing countries are ravaged by two of the intestinal diseases transmitted by feces—diarrhea and intestinal worm infections. Together, these diseases cause an estimated 10 percent of the total disease burden in these nations, with diarrheal diseases alone killing more than 3 million children in 1993 and causing some 1.8 billion episodes of illness annually (WRI, 1996). In Latin America and the Caribbean, where waterborne diseases remain the number one cause of infant mortality in many countries, the result of these conditions is an average rate of 54 infant deaths per 1,000 births. In addition, in 1991, cholera, which is also spread through contact with human waste, swept down the west coast of South America, eventually infecting more than 1 million people and killing some 11,000 (WRI, 1998).

Why does this happen? First, urbanization has concentrated population.[1] While the rural population of Latin America and the Caribbean has hardly changed since 1970, holding steady at about 120 million inhabitants, the urban population has mushroomed, growing at over 3 percent a year over the past 30 years (IDB, 1995). Thus, the proportion of urban inhabitants in the region's total population has risen from 50 percent in 1970 to 75 percent today (ECLAC/IICA, 1997), and is expected to exceed 80 percent by the year 2020 (WRI, 1998). This rush to the cities has also involved a rush to the seacoasts. Of the 215 cities in LAC countries with more than 100,000 inhabitants, 76—representing some 58 million people—are located along the seacoast or on river estuaries (see also WRI, 1996).[2]

Second, the ways of dealing with the resulting concentration of human waste have not kept pace with the increased population in the cities. The magnitude of the problem can be seen

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[1] Much of this section is drawn from Montalvo (1994).

[2] "Future projections for Latin America are equally alarming. By the year 2010, the Brazilian coastal zone from Rio de Janeiro to São Paulo is expected to become one long urban area. A similar process is taking place along Chile's coast between Valparaiso and Concepcion. One of the consequences of increases in coastal population density is the worldwide increase in pollution. . . .Experts pointed to an overload of nutrients—mainly nitrogen and phosphorus from untreated or partly treated sewage, agricultural runoff and erosion—as the most serious coastal pollution problem." (Hinrichsen, 1996, p. 40)

in Montalvo's (1994) estimates that 30 million cubic meters of untreated wastewater are discharged every day in LAC countries, with a load of over 2 million tons of biochemical oxygen demand (BOD) per year. Approximately 70 percent of the region's 337 million urban inhabitants have adequate sanitation, but over 100 million people remain unserved. The bulk of these are concentrated in large countries like Brazil (57 million unserved), Mexico (10 million), Argentina (8 million), and Venezuela (7 million), or in smaller countries with very low coverage, like Peru (7 million).[3]

Third, those lacking adequate sanitation are likely to be the very poor, particularly the newly arrived migrants living in squatter communities without either piped potable water or waste removal infrastructure.[4] These people are more likely to be exposed to the dangers created by the latter (no sewers) because they have to use the available ambient water—for drinking, cooking, laundry, and recreation. And they may even find it difficult to be able to boil the water before ingesting it. This might be called the neighborhood water quality problem and is most likely the source of the greatest public health threat.

Despite these problems, a consensus on the need for increased urban wastewater treatment has been slow to develop. Local populations have demonstrated a willingness to pay (WTP) for wastewater collection and removal, but not for treatment and safe disposal (Briscoe, 1992). Western environmentalists have largely ignored the problem altogether.[5] The priorities of the populace and of the international environmental community, not surprisingly, have been reflected in the ambivalence of foreign aid agencies and local governments toward wastewater treatment.[6] These priorities, taken together, have led to a shortage of new treatment facilities and acute operational deficiencies at existing facilities.[7]

The low priority for wastewater treatment can in turn be attributed to four different factors. First, and most obvious, the expense and complexity of treatment facilities puts them beyond the reach of many communities in the developing world. Second, foreign aid agencies and local governments have traditionally preferred water supply projects over sanitation projects

[3] Calculated from coverage data in WRI 1998 (Table 7.4, p. 251), where adequacy is defined rather broadly as the proportion of the population with disposal facilities that can effectively prevent human, animal, and insect contact with excreta. In urban areas, this includes populations served by connections to public sewers or household systems such as pit privies, pour-flush latrines, septic tanks, communal toilets, and other such facilities. The WRI report notes (p. 128) that such coverage data are problematic, which may explain discrepancies among sources.

[4] In the rural setting, there is the possibility of adequate dispersion or dissemination of wastes. On-site disposal systems such as pit latrines or septic tanks can keep wastes from contaminating local water sources and therefore provide satisfactory solutions to rural waste problems. However, in the heavily populated urban areas, the volume of wastes and the scarcity of undeveloped land combine to make such systems much less effective.

[5] Easterbrook (1994) asserts that Western environmentalists have been preoccupied with the preservation of tropical forest and endangered species and consequently have been content to blame the wretched living conditions that prevail in many developing countries on these countries' high birth rates. Easterbrook's essential point is that this distinction is fallacious because high birth rates are largely a function of poor living conditions. This argument is similar to Massignon's about the distinction between urban and rural development: "It is now accepted that urban and rural development, far from conflicting, are mutually reinforcing and ought to be pursued simultaneously. The idea that the various problems of urbanization can be solved through strategies designed to improve life in rural areas has to be abandoned. If the rural labor force does not turn to urban activities, it will, to be able to grow crops, have no choice but to destroy forests or move onto land that is environmentally sensitive or ill-suited to farming. That process is already well under way in a large and steadily increasing number of countries." (Massignon, 1993, p. 20)

[6] It appears that attitudes may be changing. For example, cleaning up Mexico's rivers—some of the most contaminated on the continent—has recently become a national priority in Mexico, and all cities above a certain size are now required to install wastewater treatment plants (Coone, 1995).

[7] Again, in the case of Mexico, a relatively recent survey of treatment plants revealed that almost half were not in service (WRI, 1990). For example, the 27 plants treating Mexico City's wastewater generally operate at less than 50 percent efficiency and combine to treat only about 7 percent of the city's wastewater (Ezcurra and Mazari-Hiriart, 1996).

(Porter, 1996). This preference is due, to a large extent, to the fact that it is easier to charge for water than for sanitation. Also, whereas sewer projects may prove to be too expensive for many communities and lenders, treatment plants may be too small and complicated to lend themselves to loan approval processes (Porter, 1996). The third factor is the position of wastewater treatment in the sanitation triage that dominates city planning in many communities:

> *Households and communities typically go through three stages in their efforts to obtain (1) safe, hygienic conditions in their houses, (2) clean, sanitary neighborhoods, and (3) improved quality of surface waters. Stage 1 involves the removal of excreta and wastewater from the household's living space. In the course of solving their own individual sanitation problems, households often impose costs on their neighbors by discharging untreated human wastes and wastewater from their property onto streets and other public property. This creates the setting for Stage 2: neighborhood collection of household wastewater. Collecting and removing wastewater from neighborhoods improves neighborhood public health conditions, but the quality of the surface water receiving the wastes will likely deteriorate. Stage 3 is the improvement of the quality of surface waters. In most industrialized countries, cities built their sewer lines first and then later, when they could afford it, they built wastewater treatment plants. This staged approach improved public health conditions in cities because it removed the human waste from town. However, the rivers and lakes were often badly polluted by the discharge of untreated wastewater.* (Choe et al., pp. 520–521, 1996)

The pattern outlined in the quotation introduces the notion that cleaning up the immediate neighborhood sanitation problems of a rapidly growing city can lead to the creation of a different environmental problem—the pollution of rivers (or of coastal waters) because rivers are in effect transformed into extensions of the sewer system by the discharge of collected but untreated sewage. The results in terms of ambient water quality can be severe depletion of dissolved oxygen (DO), as evidenced by extensive reaches of rivers with zero dissolved oxygen levels (e.g., Argentina's Reconquista River, Brazil's Tietê River, and Colombia's Bogota River), or bacteriological contamination (e.g., Rio Grande de Tarcoles in Costa Rica), or contamination with toxic substances (e.g., Brazil's Paraiba River). These conditions lead to clear damages in the technical, economic sense of monetized losses to society, via downstream (or along shore) public health problems, increased downstream treatment cost for potable water intake, restrictions on the use of water for irrigating crops, loss of commercial fisheries, and in certain settings the discouragement of tourism. Less obvious in the developing country setting because of immediate concerns with poverty and health are the aesthetic problems (smell) created by low dissolved oxygen and the accumulating ecosystem damage, especially where coastal wetlands or coral reefs may be affected.

Now, to say that this complex of problems exists is not to say that no efforts have been made to solve them; international lending agencies have been central to these efforts. Thus, in its 30 years of operation, the Inter-American Development Bank (IDB) has placed over $11 billion (constant 1995 U.S. dollars) in loans supporting almost $26 billion worth of investment in sanitation, which includes basic sanitation, potable water supply, sewerage, and water pollution control. Lending in the sanitation sector is a solid and growing component of the Bank's portfolio; almost 45 percent of the total historical lending for sanitation between 1961 and 1995 has occurred since 1985, and over 25 percent since 1990. The 1997–1998 pipeline of potential projects contained $1.7 billion for sanitation, an amount roughly equal to or greater than constant 1995 dollar lending for sanitation in any of the five quinquenia between 1961 and 1985.

However, the future looks even more demanding than the past. Just holding the line at the current 70 percent sanitation coverage level over the next 20 years will require that new sanitary facilities be provided for over 6 million persons per year, the statistical equivalent of a new megacity annually. The major urban areas also have become centers of industrial concentration, and severe industrial pollution problems have appeared in most large cities. The net result of this urban and industrial growth is that in South America, for example, wastewater discharges from agriculture, industry, and municipal uses are forecast (Davis, 1996) to more than double between 1990 and 2025, from 60.3 to 134.3 km^3 per year. The increase in effluent from agriculture is trivial, but dramatic increases are expected from industry (an additional 35.6 km^3 of discharge per year, an increase of nearly 250 percent over 1990) and municipal uses (increased discharge of 33.6 km^3 per year, or 145 percent).[8]

IMPLICATIONS FOR THE INTER-AMERICAN DEVELOPMENT BANK

The considerable expense implied by the effort to respond to these numbers through investments in water quality protection and improvement makes it especially important that the Bank help its client nations design and implement projects that make economic as well as physical sense. The foundation for such help is the Bank's long-standing general operating policy on the preparation and analysis of the projects it finances in all sectors (IDB, Operational Policy 302, September 1980). Broadly, that policy requires, whenever possible, a cost-benefit analysis of project feasibility, where all inputs and outputs must be valued in opportunity cost or economic efficiency terms. Acceptable projects must have an economic internal rate of return[9] above 12 percent, or a positive net present value (NPV), using a 12 percent discount rate as a measure of the social opportunity cost of capital. The operational policy requires a full analysis, not just an economic justification of a single proposal, by saying "The analysis compares different alternatives of design, scale, location and timing so as to maximize the net present value, and thus contribute as efficiently as possible to economic growth." (p. OP-302-3) It also acknowledges that in situations where the value of economic benefits cannot be reliably estimated, the costs of alternative projects can be compared to select the one(s) that attain a given set of objectives at the lowest cost (referred to interchangeably in the policy as "least-cost" and "cost-effectiveness" analysis).

Standard IDB economic analysis methods and guidelines exist for the analysis of investments in potable water supply and household sanitary sewer connections. However, there are none for the treatment-plant end of the sewer projects. (This is true for both the net benefit optimization and the cost minimization versions of the analysis.) Two consultation workshops held with IDB staff in 1997 and 1998 confirmed that some explicit guidance on how to analyze investments that improve ambient water quality is desired by technical staff (also see Vaughan and Ardila, 1993). This is true even though it should be recognized that there is such wide variation in individual project circumstances and so many gaps in both data and available techniques that nothing even approaching a "cookbook" is possible. This book does not attempt to develop a step-by-step guideline.[10]

In the case of investments designed to improve water quality, the Bank's *Strategy for Integrated Water Resources Management* (IDB, 1998) emphasizes that the analysis of prospective

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[8] Discharge calculated approximately as the difference between withdrawals and consumption reported in Davis (1996).
[9] Since most investment project analysis uses shadow prices, the technically correct term, and the one used by the IDB, is the economic rate of return (EIR). However, since most general readers are familiar with the internal rate of return (IRR), this term is used here and includes the economic rate of return when shadow pricing of cost and benefit flows is required.
[10] Benefit analysis manuals (e.g., Foundation for Water Research, 1996) and computerized aids for investment planning exist. Several of the latter are reviewed in Chapter 5.

investments should, to the extent possible, be done in a basin context and should be integrated with other water resource management investment decisions, such as those involving water withdrawals for urban, industrial, or agricultural use; the creation of impoundments; and the management of watershed land (hence of soil, nutrient, and pesticide or herbicide discharges). The strategy calls for a recognition of water's economic value and calls on the Bank to account for the social, economic, and environmental value of water in its lending and nonlending operations. It does not, however, make any specific recommendations about how the economic analysis of investments for improving ambient water quality should be done, and does not explicitly mention net benefit optimization.

Translating the above general rules into terms that are specific to the public wastewater treatment investment problem by combining the general objective of net benefit maximization with an integrated basin approach to management of water resources produces an ambitious problem statement. Ideally, the location, scale, removal efficiency, and timing of wastewater treatment plant investments should be chosen so that in conjunction with mandated (or voluntary) private industrial pollution control efforts, they will maximize the present value (PV) of the net benefits from improvements in ambient water quality.

These are strong requirements for any project or program analysis to satisfy, mainly because economic benefits cannot be measured with the same spatial, temporal, and absolute precision as can natural world conditions and the physical effects of investments improving water quality. However, adopting an integrated and basinwide view of water resources management is possible and within the state of the art, if standard water quality models rather than ecological models are used. It does require the collection of reliable data on many features of the existing and proposed water use regime, and these data must cover the entire basin area. There must also be available predictive models that allow the analysts to say what will happen relevant to water use or water quality parameters at specific locations when some change in another use or policy is made somewhere "upstream." But these are hardly insurmountable obstacles. In fact, no environmental investments can or should be undertaken in ignorance of baseline ambient conditions or of the impacts that a series of investments and policy changes can have on a linked water resource system. So, the crux of the problem of achieving some sort of optimal water quality situation does not arise from the need to take an integrated, basinwide approach to water quality management by developing the data and models to support it. These are available or can be developed, as Chapter 5 demonstrates.

The critical limitation to achieving a basinwide optimum in water quality arises mainly from the economics. Our reviews of IDB benefit estimation practice (Chapter 2) and the general literature on environmental benefit estimation (Chapters 6 and 7) confirm that in most situations, variants of the stated-preference method of contingent valuation are the most popular and best way to estimate water quality benefits. In developing countries, revealed-preference methods such as hedonic property value models and travel cost analyses (which are based on observed, rather than hypothetical behavior) are less relevant, comprehensive, and applicable. Stated-preference methods, however, require interviews that ask households to place a monetary value on changes in water quality.

The first problem in benefit measurement lies in translating the improvement outcomes predicted by water quality models into realistic descriptive end-point characterizations that people can readily understand, such as "swimmable" or "fishable" water (Cropper, 2000). Households cannot be expected to know what a change in dissolved oxygen concentrations or fecal coliform (CO) counts from a given baseline might mean, let alone be able to value it. The second barrier to full optimization is that currently it is not possible to develop spatially tailored benefit *functions* that can place monetary values on ambient quality changes of varying magnitudes across a number of critical locations in a river basin, a marine coastline, or an estuary. The changes that can be valued in money terms are spatially broad *point estimates* that are usually restricted to a single scenario comparing environmental quality with and without the investment project (or specified package of projects) in order to value an improved state of the natural world that differs perceptibly from the existing situation. This falls far short of what is needed for full basinwide optimization—the ability to compare a larger number of finely differentiated water quality improvement scenarios associated with alternative bundles of projects with the status quo baseline. While the stated-preference method of conjoint analysis offers some promise as a way to obtain a multiattribute benefit function, to date it has had limited use in environmental economics (Farber and Griner, 2000).

Given these limitations in our ability to estimate finely tuned water quality benefit functions, it is not surprising that the ideal of basinwide optimization has yet to be attained. This is not true just at the IDB; the ideal is not a practical reality anywhere. Nonetheless, the ideal is a useful paradigm that characterizes the larger context of the problem. Most important, aggregate economic efficiency is adopted as the criterion for decision making, consistent with the rationale for cost-benefit analysis. In principle, the degree of net welfare improvement associated with alternative investments or packages of investments should be measured rather than assumed. From an anthropocentric social welfare point of view, all investments in better environmental quality are not automatically worth doing.[11] In addition, the idealized paradigm makes a judgment in principle that the costs of using the basin as the spatial unit of analysis are worth it—that the gains in net benefits achievable by doing a basinwide analysis are likely to outweigh (at least on some sort of average basis) the extra costs of obtaining and using additional information. The alternative method of drawing a more restricted spatial boundary around a single project or set of projects taken one at a time in isolation from each other unrealistically assumes away natural world synergies. In short, potentially large efficiency losses are a real danger with a circumscribed empirical approach that ignores the basinwide context, because with blinders on, the wrong projects may be selected at the wrong times and put in the wrong places, perhaps not achieving the favorable impacts expected of them.

Even with economists' imperfect ability to estimate the benefits of water quality improvement investments, it is still possible to take an integrated basinwide approach in a least-cost sense, and do cost-benefit analysis in a second step to compare the costs of attaining water quality goals with the benefits of those goals. While this is a second-best procedure that unlike the ideal does not guarantee that an optimum balance will be identified, it at least satisfies the spirit of Operational Policy 302 and the *Strategy for Integrated Water Resources Management.* Allowing for difficulties in estimating the benefit functions, but assuming that the political process has chosen ambient water quality standards (or permitted the exploration of a range of standards), leads to an effort to find the investment locations, designs, and capacities in a basin that minimize the costs of meeting ambient in-stream quality standards at least cost (Spofford, 1991, 1993a, 1993b; Spulber and Sabbaghi, 1994). This may be supplemented by requiring that any specific project or package of projects in a basin put forward for funding from the integrated least-cost solution be subject itself to a cost-benefit test.[12] The rest of this report recommends this approach as the most practical alternative and explains in broad outline how

[11] Hahn (2000, p. 393) comments that political forces can lead to less efficient environmental policy, noting that in the United States, "Environmentalists have been successful in framing the debate as being either 'for' or 'against' the environment, making it difficult to introduce the notion of explicit trade-offs."

[12] These choices have been modified and expanded upon by analysts offering broad guidelines for water resource management. See, for example, Lord and Israel (1996), who were advising the IDB directly; and Lee and Kirkpatrick (1996) writing more generally; as well as Mariño and Boland (1998), who suggested an entire institutional structure, involving extensive public participation, for making decisions about water quality investments at the World Bank. The goal of this good-practice document is more modest. It accepts whatever institutional structure is in place, both at the Bank and within client countries.

to do it. It accepts the notion that economic efficiency matters, but recognizes that efficiency is not the only criterion used to judge the merits of prospective investments, particularly when benefits are uncertain and hard to measure. It therefore concentrates on the problems, pitfalls, and possibilities for doing economic efficiency analysis in the less than ideal conditions that are often found in developing nations.[13]

This study builds on some practical lessons drawn from the reality of current IDB practice, as revealed in a survey of analyses of IDB water quality improvement projects discussed in Chapter 2. To anticipate slightly, this survey shows that over the years a variety of approaches at varying levels of detail and sophistication have been attempted in the context of actual loan documentation. In several of the projects, clients and their consultants faithfully tried to apply the general policy guidelines, usually by first selecting the combination of sewer connections, publicly owned wastewater treatment plant investments, and complementary industrial pollution control policies that would achieve specified ambient water quality targets at least cost, and then doing a cost-benefit analysis of the least-cost infrastructure package. Sometimes alternative levels of water quality targets were explored in both least-cost and cost-benefit terms to arrive at an approximately best choice. Admittedly, other cases have fallen far short of these exemplary attempts, but overall, the review of actual IDB practice reveals that a great deal is known about what economic questions should be asked when undertaking projects involving wastewater collection, possible treatment, and ultimate disposal, and how the answers should be obtained. However, none of this collective Bank experience has ever been codified or put in a broader context of what is in fact desirable or possible.

Rather than being a simple "how-to" manual, the book looks first at major issues and broad lines of attack, in an attempt to establish the rationale for a basinwide perspective. Then, later chapters propose some specific solutions to commonly encountered but frequently ignored economic analysis problems in benefit estimation and cost-benefit analysis that apply in almost any situation, whether it involves an integrated basinwide approach or the analysis of a single wastewater treatment plant. These include, for example, a critical review of the tricks and traps in economic analysis that may mislead decision makers, an evaluation of the relative merits of nonparametric and parametric measures of average benefit that can be extracted from contingent valuation (CV) surveys and how those alternative benefit measures can influence the economic attractiveness of an investment, how to determine the optimal sample size for contingent valuation surveys, and how to incorporate uncertainty, especially about benefits, in an economic cost-benefit analysis by using Monte Carlo simulation. Answers to these questions are rarely addressed in the available texts and manuals, which are usually unsophisticated and reluctant to openly admit that public good benefits are hard to measure with any great precision. Yet all of these issues can have a critical bearing on the quality of the information provided by any economic analysis of a prospective investment that changes the supply of a public good. They can be thought of as the most frequently overlooked and yet influential questions that should be on the project economist's watch list.

MORE ON THE ANALYSIS OF WATER QUALITY IMPROVEMENT INVESTMENTS

Between the idealized objective of basinwide net benefit optimization and the intensely practical but logically flawed (though far from trivial) imposition of a cost-benefit test on isolated projects lies a spectrum of application possibilities for efficiency analysis. The big problems with

[13] For a different perspective on the notion that projects be justified by cost-benefit analyses, see Chichilnisky (1997) and Davies (1997).

the ideal arise from the optimization requirement involving net benefits and, to a much lesser degree, the requirement for a basinwide scope. Taking the latter first, the obvious strategy is to reduce the scope by making "cuts" in the basin, taking everything upstream of the area of greatest interest as given and imposing quality constraints at some downstream boundary to reflect the interests of downstream communities (or even countries). The first of these steps will do greater or lesser damage to the economic efficiency of the results achievable by analysis, depending on the strength of the link that is being broken. Thus, if agricultural sources of oxygen-demanding organics, fertilizers, or pesticides play a substantial role in defining the water quality found at the discharge point of the prospective treatment plant, and if there are options for reducing those upstream discharges at reasonable cost, looking narrowly at the treatment plant alone can lead to the choice of a needlessly expensive plant.

As has already been noted, the net benefit optimization goal implies that benefit functions are available as well as cost (of discharge reduction) functions. As will be discussed later (in Chapters 4, 6, and 7 especially), this is not currently a realistic expectation. The alternative is to do a more or less elaborate simulation. That is, several alternative arrangements of pollution reduction investments can be posited and their implications for ambient water quality predicted. These results can be valued using point estimates of benefits.[14] The alternatives analyzed could be based on technical, engineering work (and even engineering intuition about what might be best). Proceeding in this way is almost certainly not going to produce a regional optimum, but it can never be known how far the actual cases are from the optimum because that optimum will not be known.[15]

It is not necessarily the case that client countries will couch their policies related to water quality in terms that exactly parallel the Bank's requirements for project justification. This introduces what might best be seen as political complications into the analytical enterprise. One alternative already mentioned is that a nation may adopt ambient water quality standards, perhaps from the World Health Organization, from a specific Organization of Economic Cooperation and Development (OECD) country, or from a U.S. state. These can form the basis of the constraint set for the cost-minimizing basin (or subbasin) problem. But the imposition of the cost-benefit test on individual projects identified and in effect designed by such a cost-minimizing program implies a second guessing of the original choice. It says that the Bank at least suspects that the chosen standards actually do not make sense in terms of economic efficiency. This may strike some nations as rather a hard line, given that in European nations and the United States such standards were chosen without reference to cost-benefit analysis. However, it should be recognized that many developing country governments face tight budget constraints and difficult resource allocation choices in addressing a host of environmental and social concerns. One or two expensive, poorly chosen projects can preclude other, more socially desirable interventions, so a hard line is almost a necessity if one takes the rhetoric of sustainability seriously.

Another possibility for project evaluation, also common in the OECD world, and discussed further in Annex 1-A in this chapter, is the definition of technology-based effluent standards. These have implications for the ambient water quality that will be attained after the standards become effective, but they themselves are not chosen with any particular ambient goal in mind. Rather, they seem to reflect a desire for forcing action on polluters without the necessity of making decisions on such intricacies as modeling techniques for predicting ambient water quality and the fine points of marginal cost of pollution reduction functions. Efficiency-

[14] The more simulations that are run, of course, the closer the analysis comes to requiring benefit functions. Two or three might be practical, ten or more would not be feasible.

[15] See Russell (1974) for more extensive comments along these lines.

based analysis protocols that only reluctantly accept politically chosen ambient quality standards cannot possibly be expected to accept the idea that technology-based discharge standards are economically sensible. However, Mexico may be choosing that approach, as is pointed out in Annex 1-A.

To summarize the background for this good-practice document: First, the ideal of a cost-benefit analysis that designs individual projects using basinwide maximization of net benefits is not a part of current reality, but it is possible to do an approximate search for economic efficiency at the basin level. Second, some national policies are consonant with this general notion, in that some simulation effort is made to pick the best of a small subset of possible investment projects, so that the Bank analysts' job is fairly straightforwardly to check the analysis behind the particular project or set of projects put forward. Third, political tensions between the Bank and client nations can be created if the clients insist on making environmental policies—and thus designing projects—on bases that, while common among OECD countries, have no necessary connection to a balancing of benefits and costs. If U.S. experience is any guide (Hahn, 2000), a meaningful application of the cost-benefit test to environmental policies and investment programs will continue to be a struggle in LAC countries. Fortunately, IDB protocols recognize the importance of promoting economic efficiency.

OUTLINE OF THE BOOK

Chapter 2 is a review of actual IDB project documentation in which attention is concentrated on 18 loan proposals considered during 1989 to 1997.[16] This review will make it clear just how wide the gap is between the most and the least careful project analyses.

Chapter 3 is devoted to a more complete discussion of what a full cost-benefit model would look like where multiple sources exist in a regional setting, with multiple routes by which benefits can accrue to society. The practical problems implied by this ideal approach are then discussed and successive possible simplifications explored. In Chapter 4, additional conceptual and technical complications are introduced (or in some cases reemphasized, since they were found in the project reviews of Chapter 2).

Chapter 5 expands on the knowledge required at the heart of the cost-benefit model; the water quality model that translates discharges into ambient water quality. These models are also discussed as parts of so-called decision-support systems (DSSs), which may be thought of as approximations to full cost-benefit analysis efforts.

In Chapters 6 and 7, the searchlight is turned more narrowly on the benefit side of cost-benefit analysis. Chapter 6 discusses the types of water quality benefits most likely to be important in LAC countries and their importance relative to other benefit categories (such as air pollution). It also includes further discussion of the notion of "routes" to benefits and to the link between route and estimation methodology. In Chapter 7 the methods of benefit estimation are examined more closely, with special emphasis on the contrast between methods based on observed behavior and those based on the posing of hypothetical questions to random samples of persons potentially affected by a project or policy. The current dominance of the latter method for practical project work is noted and the apparent reasons set out. Variations on the

[16] A single project often includes more than one facility. Commonly, loan requests include elements of municipal water supply, neighborhood sewers, connector and trunk sewers, and one or more wastewater treatment plants in large urban areas. These are appropriately analyzed together because of their unavoidable physical interconnection. This study contains no discussion of methods designed for large package loans for projects in a multiplicity of urban areas or basins (global multiple works), although the methods discussed will in principle be applicable to the individual components of such packages.

standard contingent valuation method (CVM) are described as well, and their potential contributions and pitfalls are cataloged.

Chapter 8 reveals that with referendum CV survey data, there are nearly a score of alternative ways to measure the mean or median of willingness to pay for environmental improvements, and that past IDB practice had been using an approach that systematically understated project benefits. Chapter 9 sheds light on what uncertainty means and how to approximate it empirically. More important, it emphasizes that uncertainty is inherent in project analyses that are based on CV benefit estimates, the part of project analysis that was found to be of very uneven quality in the Chapter 2 review. It explains how to take uncertainty fully into account in economic cost-benefit analyses by using Monte Carlo simulation rather than conventional sensitivity analysis, which provides a great deal less information about project risk. The chapter argues that when benefits and costs are uncertain, decision makers in the Bank's borrowing member countries would be well served by a full economic evaluation of the risks associated with prospective investments.

The illustration in Chapter 10 attempts to provide a model for future applications. It discusses the optimal sample size for obtaining an accurate estimate of benefits from CV surveys and relies on the notion that larger CV survey samples compress the variance in the probability distribution of an investment's net present value. The concluding chapter contains a summary, a reiteration of major recommendations, and a discussion of the problem of when any cost-benefit analysis may be justified.

One important general matter that this study does not address is the question of how any particular project will be paid for. This is a complex, multifaceted problem in its own right, although at one extreme it can be treated as just a matter of finding some approximation of average cost per unit of input and allocating units of input across sources connected to the water system and sewers. At the other extreme, if an agency wants to try to approximate marginal cost pricing in the context of an expanding water supply and wastewater treatment system, it will need to determine optimal sequencing over time in the presence of endogenously determined rates of demand growth.[17]

[17] For a discussion of the relevant literature and an examination, on the water supply side, of several competing marginal cost approximations, see Russell and Shin (1996a, 1996b). The second of these papers demonstrates that getting the project sequencing right may be more important than approximating marginal cost pricing. Other facets of the pricing problem include what mix of imputation and measurement is optimal for sewered systems and, in that context, how much freedom can be allowed dischargers in the opt-in/opt-out decision. Another pricing problem is raised by jointness in removal of specific pollutants.

Annex 1-A

Technology-Based Discharge Standards as a Basis for Water Quality Management

ALTERNATIVE COUNTRY GOALS

One need look no further than the United States to see that countries may not opt for something approximating national economic efficiency as the criterion for choice of specific water quality improvements.[18] This is also more generally true. Indeed, it would be no exaggeration to say that making such decisions on an aggregate efficiency basis is almost unheard of as a practical matter. The reasons for this can only be speculated, but two fairly obvious possible reasons are the previously discussed problem of producing persuasive benefit estimates and the fact that politics (for these *are* political decisions involving public goods) is not an aggregative game but a distributional one. That this is more generally true can be seen by reviewing such sources as Kinnersley (1994) (for the U.K.), Bower et al. (1981) (for France and Germany), and Hersh (1996) (for the more recent U.K. experience, Sweden, and the European Union).

One perspective on this question, in which regional optimization in a cost-benefit analysis setting is explicitly rejected, is offered by Mexico:

> *Setting Specific Discharge Stipulations (Condiciones Particulares de Descarga) as a function of water uses and assimilative capacity based on studies and Classification Statements (Declaratorias de Clasificación) for water in receiving bodies requires knowledge of the morphological and hydrological characteristics of the rivers, knowledge of all beneficial uses by specific stretch, and undertaking simulation modeling with previously calibrated models. Thinking about this option for setting quality requirements for each and every water body involves a complex undertaking which could lend itself to arbitrary considerations and reasoning, as much on the part of the users as those establishing the standards, which could produce mistakes because relevant information is either unavailable or overlooked.* (Mexico, Comisión Nacional del Agua, p. 29; See also *International Environmental Reporter*, 1998)

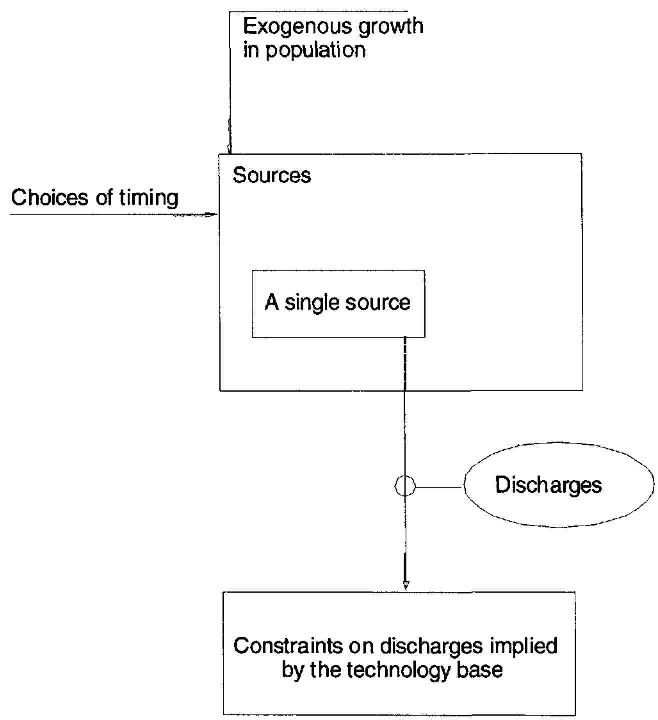
[18] An estimate that indicates how far the U.S. Clean Water Act program misses the efficiency mark is offered by Freeman (1990). Comparing his Tables 4-8 and 4-9, we find a most likely point estimate of benefits to be $14 billion per year, and an average annual cost estimate of $23.2 billion. Table 4-10 shows that another estimate of costs, this one from the U.S. Environmental Protection Agency (EPA), is $30.8 billion, or over twice the benefits. (All figures are in 1984 dollars and all refer to the decade of the 1980s.) However, as noted in this annex, the legislation was not designed for efficiency. Rather, it was designed to produce action.

OTHER ALTERNATIVES

In the United States, frustration with efforts to use ambient quality standards as the basis for enforceable discharge standards for pollution sources led to the adoption of technology-based standards. In the resulting law (Federal Water Pollution Control Act Amendments of 1972, subsequently amended and retitled the Clean Water Act), phrases are used that define a pollution control technology in terms of its availability and sometimes, implicitly at least, its cost relative to the status quo. Examples include "best available [technology] economically achievable," "best practicable [control technology] currently available," and "reasonably achievable [control technology]." U.K. legislation includes very similar definitional phrases, such as "best available techniques not entailing excessive costs" and "best practical environmental option" (Hersh, 1996). While such a legislative strategy opens the door to almost endless quarreling about the "real" meaning of the words in the label, the intended application is roughly as follows: Once the terms of art are defined operationally, the technologies that emerge can be applied to actual or representative pollution sources in paper studies. These studies show what discharges would result if the technology in question were applied to a particular source (e.g., to a 100,000-barrel/day petroleum refinery that produces gasoline for vehicles). If the source is an actual one, the result is a set of discharge standards. If the source is representative of a class, the result can be normalized for size (e.g., expressed as pollutant discharges per barrel of crude oil input) and applied to any member of the class of such sources to produce that source's allowable discharge levels.

This is clearly a very different basis for choosing projects than either aggregate efficiency or vector-constrained cost-effectiveness. As economists, we can hardly support or recommend it, except perhaps in countries where institutional weaknesses make it all but impossible to do any better, and only then in the very earliest stages of trying to develop a water pollution control

Figure 1A-1. Schematic of a Technology-Based Standards Approach to Project Analysis



program (Russell et al., 1998). Most important, the link to the ambient environment has been dissolved.[19] (See Figure 1A-1 for a schematic, and see Chapter 3 for corresponding diagrams that describe the cost-benefit and AWQ standard settings.) This broken link implies that the meaningfulness of a watershed or regional modeling approach also disappears, for it is the mutual effects of all sources upon the environment linking them that makes a simultaneous programming approach necessary. Technology-based standards lend themselves only to source-by-source cost minimization of the vector-constrained sort, where the vector is the discharge standard implied by paper application of the technology defined in the statute, and optimization is across alternative technologies that will at least satisfy that vector constraint.

From the point of view of the Bank and its project review process, the justification for a project loan provided by compliance with a nationally mandated technology-based discharge standard must be even less persuasive than that arising in an AWQ standard system. However, dealing with this seems an even more delicate matter. Checking on the aggregate efficiency implications of a particular project application would require embedding it in a model of at least the portion of the watershed affected by it. This in turn builds in potentially inefficient levels of treatment at all other interacting sources as a result of whatever assumptions are made about the discharges from those sources. This difficulty is likely to arise at some point, because Mexico has apparently chosen such a route for its national policy (see Mexico, Comisión Nacional del Agua, 1996).

[19] In the U.S. water pollution control system, this link has been maintained, but only at one remove. If application of technology-based discharge standards does not result in satisfying the AWQ standards that predate this legislation, the EPA administrator is able to require additional effort beyond those standards.

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Chapter 2

A Decade of IDB Experience in Review

No guideline that tries to make a constructive and relevant contribution can stand in isolation from what is already known and done in practice. Promising opportunities for improvements cannot be readily identified in a vacuum of historical fact. With that premise in mind, a desk review was undertaken of IDB projects approved since 1989 that had ambient water quality improvement as an explicit objective. This was supplemented by two consultative workshops with technical Bank staff familiar with the issues raised during the course of designing, approving, and executing these sorts of projects. The workshops set the stage by providing an insider's view of which questions in particular the guideline should explore out of a host of possibilities, while the project reviews summarized in this chapter form the backdrop for that exploration.

Candidate projects were identified by a computerized search of the Bank's database, supplemented by a survey of staff economists, and later screened by inspection of the documents themselves. Given the large number (27) of eligible projects having ambient water quality objectives, the desk review primarily consulted official IDB project documents (loan proposal, project report) and readily available published articles or World Bank reports. The environmental summaries (ES or ESIR) were used only when the environmental information in the main project documents was clearly unsatisfactory (there were several cases, many of them of recent vintage). Background technical and economic feasibility studies were not consulted, primarily because our intent was to determine what Bank management and project teams ostensibly regard as important for decision making; that information by definition ought to be contained in the principal project documents, not in ancillary reports, memos, or background design studies. Moreover as a practical matter, the sheer volume of background information and its uneven availability across various file locations in the Bank made a more in-depth and thorough review impossible to justify. Each project was reviewed using the sample form reproduced in Annex 2-A; that form was submitted to Bank staff for review before being used to characterize the attributes of each project in the sample.

Projects were grouped into two categories: Type I: investment loans concentrating on one predefined area (most frequently a metropolitan area) within which all the investments and the relationships among them were fully identified before project approval, and Type II: global multiple works loans, usually involving investments in several different cities or basins, only a sample of which were analyzed prior to loan approval.[1]

[1] According to the Bank's 1993 *Glossary of Terms*, a multiple works program is designed to finance a group of similar works, physically independent of each other, the feasibility of which does not depend on the execution of any given number of such works. In the past, the Bank has required that a representative sample of projects sufficient to cover 30 percent of total program cost must be exposed to ex ante feasibility analysis. This requirement often imposes a serious analysis burden (see Vaughan, 1994). Note that using our separation between projects in one metropolitan

Table 2-1. Percent of Projects Reviewed that Contain Adequate Information in Loan Proposals and Project Reports, Overall and by Subperiods

	1989–1997	1989–1993	1994–1997
Specific Investment Projects (18)	(18)	(10)	(8)
Technical information	44%	70%	13%
Economic analysis information	50%	70%	25%
Global Multiple Works Projects (9)	(9)	(4)	(5)
Technical information	22%	50%	0
Economic analysis information	22%	50%	0

SUMMARY OF THE RESULTS

The intent of this review was not to evaluate the merit of any particular analysis, but we did ask whether the information on the natural world and the economic analysis provided in the project reports were sufficient to convey a reasonable understanding of the problem setting, the project's beneficial physical effects, and the cost-effectiveness or net benefits accruing from the proposed investments. A subjective appreciation of adequacy was applied, defining adequacy as the presentation in the official project documents of at least 70 percent of the information that ideally could be expected by someone conversant with the technical subject matter.[2] The results are summarized in Table 2-1.

Overall, project documents on global multiple works do not contain much useful technical and economic information. This is perfectly understandable. The burden of performing an in-depth analysis of a score or more complex individual projects in a representative or indicative sample that is only a small share of the number of projects that the program will eventually finance (30 percent or less) is extreme, as is the challenge of describing the bare bones of so many analyses in a few document pages. Thus, because not much can be learned from the reports on these sorts of projects, they are not discussed here. However, the principles set out in this guideline are applicable to them, time and analysis cost permitting (Vaughan, 1994).

The results for individual investment projects are more unsettling at first glance, since it appears that projects are cogently described in technical and economic terms only half the time. However, that overall impression is largely a product of the Bank's structural reorganization in 1994 and its associated attempt to streamline project documents. Before that break, adequate information was directly available in the documents over two-thirds of the time. After it, less than one-fourth of the documents could be relied on to contain anything much beyond blanket generalizations about health and environmental quality improvements of unspecified magnitude and reassurances that the economic rate of return passed the Bank's 12 percent minimum rate. This result conceals the mountain of project feasibility analysis material developed to design the proposed investments and the oftentimes valuable information contained therein that is omitted from the summary documents prepared for upper Bank management.[3] Random

area versus projects in many, project 12 is borderline because while it is in one metro area, all of its components were not identified ex ante, and only a sample of separate works was analyzed. It was arbitrarily grouped with the investment loans, not multiple works.

[2] All individual ratings were done by a single senior reviewer, so while they may be biased in an unknown direction (either too generous or too harsh), they are at least internally consistent.

[3] It has been noted in discussions with staff experts responsible for internal project document review at the Bank that their roles have also changed since 1994. The effect of these changes is said to have been a serious reduction in the amount of time available for in-house project review. It seems that this phenomenon, in combination with the decrease in readily available information, reduces the chances that inadequate or misleading analyses will be caught before loan approval.

checks of this detailed background information could not confirm or deny a decline in information quality over the decade. With these caveats in mind, the next sections summarize what can be learned from the official documents.

General Project Characteristics

Annex 2-B contains a key for the projects summarized in Annex 2-C, which gives each project's components, objectives, and scope. The most important features of the specific investment projects (Type I) are summarized in Table 2-2. The total accumulated cost of these 18 projects was nearly $5 billion between 1989 and 1997, with 4 of them exceeding a half billion dollars each. One-third include potable water supply. More potable water at a lower price means more domestic wastewater that has to be disposed of. So these projects also contain sewer and wastewater treatment components that mitigate environmental damage by leaving receptor watercourses in no worse condition than they would have been without the additional household potable water hookups.[4]

The bulk of the projects (16) affect the quality of freshwater rivers and lakes. Only two are exclusively devoted to improving the quality of near-shore marine waters. However, marine waters are also affected by five projects that deal with fresh water, so coastal waters are involved in almost 40 percent of the cases. This has an effect on the sources of benefits, particularly those obtained from international tourism geared toward saltwater recreation activities, as explained in Chapter 6.

The direct disposal of sewage with no treatment, or just pretreatment, was a viable option in just four situations (counting projects 17 and 18 as one, since they represent two stages of a program using a common deepwater outfall). In two cases, disposal without treatment was chosen because the receiving water bodies had sufficient assimilative capacity. In a third case, project 2, no rationale for collection without treatment was supplied. In the fourth case (the bay in project 10), industrial and other sources of pollution that would not be directly controlled by the investments had already destroyed aquatic life. Here, collection and concentration of commercial and domestic discharges at a single underwater outfall instead of numerous existing outfalls along the shore could not cause further damage to an already very low quality bay, but would improve near-shore quality, thereby reducing human exposure to disease pathogens while swimming and bathing.

Treatment equipment investments appear in 13 of 18 projects. In three instances, only primary treatment is involved. In the remaining 10 cases, a mix of primary and secondary plants, or secondary plants alone, are planned. No tertiary treatment investments appear in the sample.

BASELINE ENVIRONMENTAL INFORMATION AND AMBIENT ENVIRONMENTAL QUALITY IMPACT ANALYSIS IN INVESTMENT PROJECTS

Table 2-3 displays what the Bank's official loan and summary environmental impact documents have to say about the available baseline information on the natural world (either ex ante or collection of data included in the project itself), the polluters and pollutants targeted, the extent to which the financed projects will achieve ambient environmental quality (AEQ) standards, and whether water quality simulation models were employed to gauge project impact. Credit is given if something is merely mentioned, however brief that mention may be. Reviewer remarks about the quality and depth of those references appear in Annex 2-D. Suffice it to say

[4] The economic justification for add-on treatment of sewered wastewater is discussed in Annex 4-A, Chapter 4.

Table 2-2. Characteristics of Type I Projects

Project No.[a]	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	Type I Total
Country[b]	AR	AR	BA	BR	BR	BR	BR	CO	CO	CO	EC	EC	GU	ME	ME	NI	UR	UR	
Approval year	93	97	92	92	92	93	93	93	97	98	90	94	96	90	94	96	89	96	
Types of Water Bodies Affected[a]																			
Freshwater streams/rivers	1	1	0	1	1	1	1	1	1	0	1	1	1	1	1	1	1	1	16
Lakes/reservoirs	0	0	0	1	1	0	1	0	0	0	0	0	1	0	0	1	0	0	5
Coastal marine waters/bays/estuaries	0	0	1	1	0	1	1	0	0	1	0	0	0	0	0	0	1	1	7
Groundwater	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Components[c, d]																			
Potable water	0	0	0	0	0	1	0	1	0	0	1	1	0	1	1	0	0	0	6
Flood control	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	4
Sewer/drainage	0	1	1	1	1	1	1	1	0	1	1	1	1	1	1	1	1	1	16
Deepwater outfall[e]	0	0	1	1	0	1	0	0	0	1	0	0	0	0	0	0	1	1	6
Pretreatment	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	2
Primary treatment	0	0	1	0	0	1	0	0	1	0	0	1	1	0	0	0	0	0	5
Secondary treatment	1	0	0	0	1	1	1	1	0	0	1	0	1	1	1	1	0	0	10
Total Nominal Original Cost[f] (million nominal U.S.$)	280	500	73	266	900	793	220	232	125	40	57	170	35	650	282	47	33	219	4,922

[a] See Annex 2-B for a key that includes project titles.
[b] AR, Argentina; BA, Barbados; BR, Brazil; CO, Colombia; EC, Ecuador; GU, Guatemala; ME, Mexico; NI, Nicaragua; UR, Uruguay.
[c] 1=yes; 0=none or unclear.
[d] Highest level of treatment indicated as 1 if all plants have the same technology (e.g., primary plus secondary is scored 0 for primary, 1 for 0 secondary). If several treatment plants are built with different technologies, both primary and secondary are indicated as 1.
[e] Projects 4, 10, 17, and 18 use an existing underwater outfall.
[f] Total project costs include components other than treatment costs.

Table 2-3. Baseline Information and AEQ Impact for Type I Projects

Project No.[a]	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	Type I Total
Country[b]	AR	AR	BA	BR	BR	BR	BR	CO	CO	CO	EC	EC	GU	ME	ME	NI	UR	UR	
Approval year	93	97	92	92	92	93	93	93	97	98	90	94	96	90	94	96	89	96	
Baseline Information[c]																			
AEQ monitoring information (available or included in project)	1	1	1	0	1	1	1	1	1	1	1	0	0	0	0	1	1	1	13
Industrial discharge inventory (available or included in project)	1	1	0	0	1	1	1	1	0	1	1	0	0	1	1	1	1	0	12
Discharge Reduction Targets[c]																			
Industrial/commercial control (including any complementary programs, financed or not)	1	1	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	17
Domestic collection/treatment	1	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	17
Pollutants targeted by treatment																			
BOD	1	NA	0	0	1	1	1	1	1	0	1	0	1	0	1	1	NA	NA	10
Chemical oxygen demand	0	NA	0	0	0	0	0	0	0	0	0	0	0	0	0	1	NA	NA	1
Fecal or total coliform	0	NA	1	1	1	1	1	0	1	1	1	0	1	0	0	1	NA	NA	10
Nutrients (eg., nitrogen, phosphorus)		NA	1	1	1	0	0	1	0	0	0	0	0	0	1	0	NA	NA	5
Others (e.g., suspended solids, oil, grease, heavy metals)	0	NA	0	0	1	0	0	1	1	0	0	0	0	0	0	1	NA	NA	4
Percent removal of pollutants specified?																			
Percent of BOD	95				38	40-90			40						70	71			
Percent of coliform					38	90-99										100			
AEQ Targets																			
AEQ standards specified?	1	0	1	0	1	1	1	0	1	1	1	0	0	1	1	1	1	0	12
Contribution of project to achieve standards (1=full; 0=partial; U=unclear)	0	U	1	U	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Use of water quality models to simulate AEQ impacts	1	1	1	1	1	1	1	1	1	1	1	1	0	0	0	1	1	0	14

[a] See Annex 2-B for a key that includes project titles.
[b] AR, Argentina; BA, Barbados; BR, Brazil; CO, Colombia; EC, Ecuador; GU, Guatemala; ME, Mexico; NI, Nicaragua; UR, Uruguay.
[c] 1=yes; 0=none or unclear; NA indicates projects without treatment plant components.

that a great deal is left unsaid about the performance of the treatment equipment being financed (beyond its general class, primary or secondary), the assumptions driving the AEQ models, and the link between the project's outcome and existing AEQ standards (when the two are discussed, clear connections between them are often not made).

Five projects do not mention baseline ambient water quality conditions or make provisions for collecting these data during execution, but of these, two refer to using AEQ models to simulate project impacts, so by implication only 3 out of the 18 investment projects appear to have been developed in ignorance of preproject water quality and the extent of improvement occasioned by the project. Similarly, 12 of the 18 projects recognize the importance of accounting for industrial as well as domestic wastewater discharges in pollution control by either referencing or providing for collection of information on industrial discharges.[5] In fact, 17 out of 18 projects impose some parallel conditionality on implementation of an industrial pollution control program or a discharge inventory in parallel with the public investments in sewerage and wastewater treatment the loans will finance.

Water quality simulation models appear to have been applied most of the time. The official documents rarely provide much detail, either relegating dissolved oxygen profiles by river reach or similar technical data to annexes, or including a few lines of text about ambient impact in the economic analysis discussion. Nevertheless, the evidence does suggest that technical Bank staff working on these projects recognize the close dependence between the environmental engineering and economic analyses for these types of investments.

Finally, the importance of public health considerations in these projects is revealed by the specific targeting of coliform bacteria in 10 of the 18 cases, the same frequency as biochemical oxygen demand.

Economic Analysis of Investment Projects

As shown in Table 2-4, regional least-cost optimization models have been used infrequently (3 of 18 cases). More common is a narrower cost-effectiveness analysis that seeks the least expensive wastewater treatment equipment design needed to achieve a prespecified set of pollutant removal targets (percentages). This exercise is usually a precursor to a cost-benefit analysis, but hardly ever is explained in much detail.

Cost-benefit analysis was commonly applied in a majority of the projects reviewed. In fact, in only two (projects 1 and 2) was it not attempted. Usually, projects with both sewer and wastewater treatment components contained a separate analysis for each, which is consistent with good practice, rather than lumping both sets of investments together in a combined analysis that might disguise economically unattractive components. The preferred benefit estimation approach for both sewer and treatment project components has overwhelmingly been contingent valuation. Although Table 2-4 does not show it, in most cases where multiple methods were applied, the bulk of the benefits were obtained by CV, using other methods to fill in the smaller gaps.[6]

An economic sensitivity analysis was missing in 4 of the 15 projects where cost-benefit analysis was applied, in apparent disregard of standard Bank requirements. Where it was done, it was generally a pro forma exercise, with three notable exceptions (projects 3, 4, and 7). (The next section summarizes these interesting applications, as well as other innovations that are particularly worth noting.)

[5] Here, credit was given for either gross regionwide estimates of total industrial pollution or more detailed plant-by-plant inventories.

[6] On occasion, some double counting may have arisen through the use of multiple methods. See the review remarks in Annex 2-D.

Table 2-4. Economic Analysis of Type I Projects

Project no.[a]	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	Type I total
Country[b]	AR	AR	BA	BR	BR	BR	BR	CO	CO	CO	EC	EC	GU	ME	ME	NI	UR	UR	
Approval year	93	97	92	92	92	93	93	93	97	98	90	94	96	90	94	96	89	96	
Type of analysis																			
Regional AEQ cost minimization	0	0	0	0	1	1	0	1	0	0	0	0	0	0	0	0	0	0	3
Cost-effectiveness/technology standard	1	0	1	1	1	1	1	1	0	1	1	0	1	1	1	1	1	1	15
Cost/benefit of																			
Sewer and/or flood control	0	0	1	1	1	1	1	0	0	1	1	1	0	0	1	1	1	1	12
Treatment	0	0	1	0	1	1	1	1	1	0	1	1	1	0	1	1	0	0	11
Major types of benefit and methodology																			
Total economic value (generic) of AEQ improvement	0	0	0	0	0	CV-D	CV-D	CV-D	0	CV-D	H-D	0	0	0	CV-D	CV-D	CV-D	0	
Aesthetics and odor elimination/method	0	0	0	0	0	0	0	0	CV-D	0	0	0	0	0	0	0	0	0	
Sewer connection/method	0	0	CV-D	CV-D	CV-D	CV-A	CV-D	CV-D	0	CV-D	RP-D	CV-D	0	0	0	0	CV-D	CV-D	
Health/method	0	0	CA-A	0	0	0	0	0	CV-D	0	0	0	0	0	0	CA-D	0	0	
Recreation/method	0	0	CV-D	0	0	CV-D	0	0	0	0	0	0	0	0	0	0	0	0	
Agriculture/irrigation method	0	0	0	0	0	0	0	0	PS-A	0	0	0	0	0	0	0	0	0	
Tourism/method	0	0	PS-A	0	0	PS-A	0	0	0	0	0	0	0	0	0	0	0	0	
Systems cost savings	0	0	CA-D	0	0	0	0	CA-D	CA-D	0	0	0	CA-D	0	CA-D	0		0	
Fishery/method	0	0	PS-A	0	0	PS-A	0	0	0	0	0	0	0	0	0	0	0	0	
Erosion or property damage/method			CA-A	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
Sensitivity	0	0	R	R	S	0	R	S	S	S	S	0	S	0	0	0	S	S	
Single or multistage (0=single; 1=multistage; U = unclear)	1	U	0	0	1	1	1	1	1	1	1	1	0	1[c]	1	1	1	1	13

Notes:
1=yes; 0=none or unclear. Benefit method type codes are CV, contingent valuation; TC, travel cost or other participation models; H, hedonics; PS, producer's surplus; CA, cost of damage avoided/averting expenditure; RP, revealed preference, demand function estimation. Information codes are A, mainly by assumption or benefit transfer; D, based on project-specific data from surveys or other sources. Sensitivity method codes are R, risk-based and S, standard which involves either arbitrary percentage changes in cost and benefits flows or solution for percentage changes that make the project economically unviable.

[a] See Annex 2-B for a key that includes project titles.

[b] AR, Argentina; BA, Barbados; BR, Brazil; CO, Colombia; EC, Ecuador; GU, Guatemala; ME, Mexico; NI, Nicaragua; UR, Uruguay.

[c] Last in four stages of IDB projects, thus analytically similar to a single stage.

Table 2-4 also indicates that two-thirds of the projects reviewed represent just one stage in a multistage program, which raises difficult analysis issues with respect to the time phasing of investments and in any given time period, over space within a basin. While dynamic programming has not been used, some analyses have considered alternative investment configurations in a more approximate way. For instance, the analysis for project 5 used regional least-cost mixed-integer programming to minimize the sum of treatment plant investment and operation and maintenance (O&M) costs, allowing construction to begin in either of two time periods (1992 or 1996 so operation comes on line in 1995 and 2000) subject to plant flow capacity constraints. Up to four plant locations were possible, and two alternative flow capacities at three of the locations (1.5 or 3 m^3/sec) and three capacities at the fourth location (2.5, 4.75, or 9 m^3/sec). Each plant could be built in either the first or second stage subject to BOD concentration constraints imposed at three critical river reaches.

Time phasing raises additional problems when cost-benefit analysis is undertaken instead of (or in addition to) regional least-cost analysis. The heart of the issue is that the program overall may produce a positive net present value, but the initial phase, as designed, may not (see Chapter 4 for a more formal discussion of this problem). In addition, even if an initial phase is justified when population is held constant, additional population growth may, by increasing wasteloads, erode the AEQ gains the initial stage achieves. Project 9 is an example of the first issue, where the benefits of the project, taken in isolation from subsequent stages, were on the narrow margin between acceptance and rejection, depending on the assumptions underlying the analysis (e.g., the size of the benefiting population, the legitimacy of stemming or multiplier-induced benefits, etc.). In project 16, the question of future growth was handled by adding additional costs midway through the project's lifetime to preserve the level of benefits initially achieved.

ISSUES ARISING FROM PROJECT ANALYSIS

Benefit Estimation

Contingent Valuation of Total Economic Value

The contingent valuation method is a stated-preference technique used to measure willingness to pay for certain nonmarket commodities. It may turn up so-called "nonuse" values, and is in general the only method that allows the estimation of both use and nonuse values, especially for goods not traded in markets. Furthermore, contingent valuation is the only technique that measures Hicksian surplus directly, without requiring additional manipulation. Under the CVM, a hypothetical scenario is created and individuals are surveyed through phone, mail, or personal interview methods. To obtain a monetary measure of a scenario involving welfare change, individuals are asked the amount they would be willing to pay for the good (the welfare improvement) in question. The CVM assumes that individuals respond the same way to a hypothetical situation as they do to a real scenario. If so, their stated willingness to pay will be a realistic monetary measure of the worth or utility gained or lost from changes in the availability or quality of nonmarketed environmental goods. The CVM has been widely used in environmental and natural resource economics.

This method originated in the early 1960s, when Robert Davis used questionnaires or interviews of 121 hunters and recreationists in the Maine area to estimate the benefits of outdoor recreation (Mitchell and Carson, 1989). In the late 1960s, Ronald Ridker (1967) used the CVM in Philadelphia and Syracuse to estimate air pollution benefits. In the 1970s, CVM was used to value various recreational amenities. Cicchetti and Smith (e.g., 1976) used it to estimate willingness to pay to reduce congestion in a hiking area; Arthur Darling (1973) used the CVM to value amenities of three urban parks in California; Alan Randall and colleagues (Randall et al., 1974) used it to estimate air visibility benefits in the Four Corners area in the Southwest (the area where New Mexico, Utah, Arizona, and Colorado intersect); Acton (1973) applied the method to valuing programs that reduced the risk of dying from a heart attack. By the late 1970s, CVM became a recommended method for determining project benefits for resources such as environmental amenities. In 1979, the U.S. Water Resources Council recommended CVM in its *Principles and Standards for Water and Related Land Resources Planning* (U.S. Water Resources Council, 1983). In the early 1980s, the U.S. Army Corps of Engineers began to use CVM to measure project benefits; CVM was also recognized under the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (Mitchell and Carson, 1989).

There are three basic survey designs in CVM: (a) open-ended surveys where individuals are asked to state a willingness to pay, (b) closed-end referendums where individuals are presented with a bid and respond with a yes/no binary decision (single-response referendum), and (c) closed-end double referendums where individuals are presented with a sequence of two payments to obtain binary decisions (double-response referendum). Most IDB applications have been of the single or double-bounded referendum sort,[7] following the protocols recommended by the National Oceanographic and Atmospheric Administration Panel of Experts (NOAA, 1993). The referendum type of design is the recommended model for contingent valuation surveys. By posing the elicitation question in the form of a referendum, CVM studies imply a certain imposition of the payment scheme (e.g., an increase in taxes). Furthermore, respondents are familiar with these methods of operation and their use in the political system (Mitchell and Carson, 1989).

CVM Innovators: Ducci and McConnell

As late as the mid-1980s, institutions like the IDB continued to be reluctant to employ CVM, regarding it as a frontier, nonoperational technique of questionable accuracy, particularly in developing country contexts where respondents may be unaware of or uninterested in the state of the ambient environment or its improvement. However, at about the same time, the ability of the traditionally favored hedonic pricing method to generate reliable estimates of willingness to pay for nonmarketed environmental attributes came under scrutiny (Vaughan, 1987, 1988). The Bank began to use the CVM in place of hedonics or travel cost models when circumstances warranted. Bank economist Jorge Ducci, in collaboration with Kenneth McConnell of the University of Maryland,[8] was the first to successfully implement the method for a Bank project in the 1988 economic analysis of the second stage of Uruguay's Urban Sanitation Project for the City of Montevideo (Ducci, 1988; McConnell and Ducci, 1989; McConnell, 1995). Their analysis paved the way for future applications and effectively sounded the death knell for estimates of water quality improvement benefits based solely on hedonic analysis.

Negative Expected Willingness to Pay: Ducci, McConnell, and Rodriguez

Disturbingly, the possibility of a negative gross project benefit estimate arose in the very first IDB application of the referendum CVM by Ducci and McConnell noted above, nearly dooming a novel experiment before it started. At the time, this outcome was wholly unexpected, given

[7] This method is also known as dichotomous-choice or discrete-choice contingent valuation.
[8] Pablo Gottret, Terry Powers, and William J. Vaughan of the IDB also collaborated in this effort.

the limited experience with the approach. Fortunately, a solution was found. McConnell mentioned the experience in passing several years later (McConnell, 1995), although at the time the project report did not elaborate on how the negative expected value of willingness to pay was overcome.

Since this experience, a considerable amount has been written on the origins of and cures for negative estimates for average WTP for a sample surveyed using a referendum format. While it is beyond the scope of this chapter to go deeply into the question, it is dealt with in some detail in Chapter 8 because the general phenomenon of WTP uncertainty is a central problem for sensitivity analysis. For now, a few general observations will suffice.

The instability of benefits estimated from referendum data, stemming from their sensitivity to econometric specification issues (see also Halvorsen and Sœlensminde, 1998) is worrisome, and can easily lead to relentless exploration of the data and alternative formulas for expected value of WTP until a benefit level that manages to justify the project is found. While academic or research applications usually provide a thorough account of what was done in commercial applications, such as those done by or for the Bank, the several steps (estimation, function evaluation) required in the referendum approach can pose difficulties and open the door for ambiguities, especially if no reporting is expected from the analyst on exactly how the benefits estimates were derived.

This suggests that the desirability of the open-ended method, the use of which was discouraged by the NOAA panel, should perhaps be reassessed (McFadden, 1994), especially in light of the institutional context of project evaluation as done by the Bank and its borrowers (Rodriguez, 1998). Alternatively, more thought must be given in the future to nonparametric methods for extracting expected values from referendum data (McConnell, 1995; Haab and McConnell, 1997a), or to estimation approaches that eschew the standard random utility difference model (Hanemann, 1984) in favor of an approach that in application is more consistent with theory, particularly that willingness to pay should have a non-negative lower bound and an upper bound no greater than income (Haab and McConnell, 1998a).[9] For a discussion of nonparametric approximations, see Chapter 8.

Double-Bounded Referendum: Ardila

In 1993, in the analysis of the Guaiba Watershed Management Program (project 7), Ardila (1993) introduced the application of a double-bounded variant of the referendum CVM developed by Hanemann et al. (1991). Rather than just asking respondents whether they would or would not be willing to pay a stipulated amount for a proposed water quality improvement, the double-bounded approach adds a follow-up question, which raises the payment amount for those who accepted the initial proposal and lowers it for those who did not, and asks a follow-up accept/reject question. The advantage of the approach is that at a given sample size it yields a more precise estimate (reduces the variance) of the expected value of WTP. Put otherwise, the sample size (and hence survey cost) needed to achieve a given degree of precision can be reduced by using two plays of the referendum game rather than one.

This approach has occasionally been used since, but it is not possible to determine how often because the descriptions of methods used are very terse in most official project documents. Apparently, however, it has not always met with success because respondents, who put

[9] Creel (1998) sounds a more optimistic note by demonstrating that the marginal expected value of willingness to pay, truncated from below at 0 and from above at a maximum that drives the probability of acceptance to 0, can be consistently estimated from the simplest possible logit model (intercept and price parameters only) providing the bids are spread uniformly between the upper and lower bounds, the upper bound is known a priori, and the acceptance probability is integrated only up to the upper bound in calculating the mean.

full faith in the realism of the initial play of the game, become confused and suspicious when the initial offer is altered in the second play (Quiroga, personal communication). For this reason, in project 16 (Lake Managua) only a single-bounded model was fit although double-bounded data were collected.

Valuing Alternative States of the World: Ardila

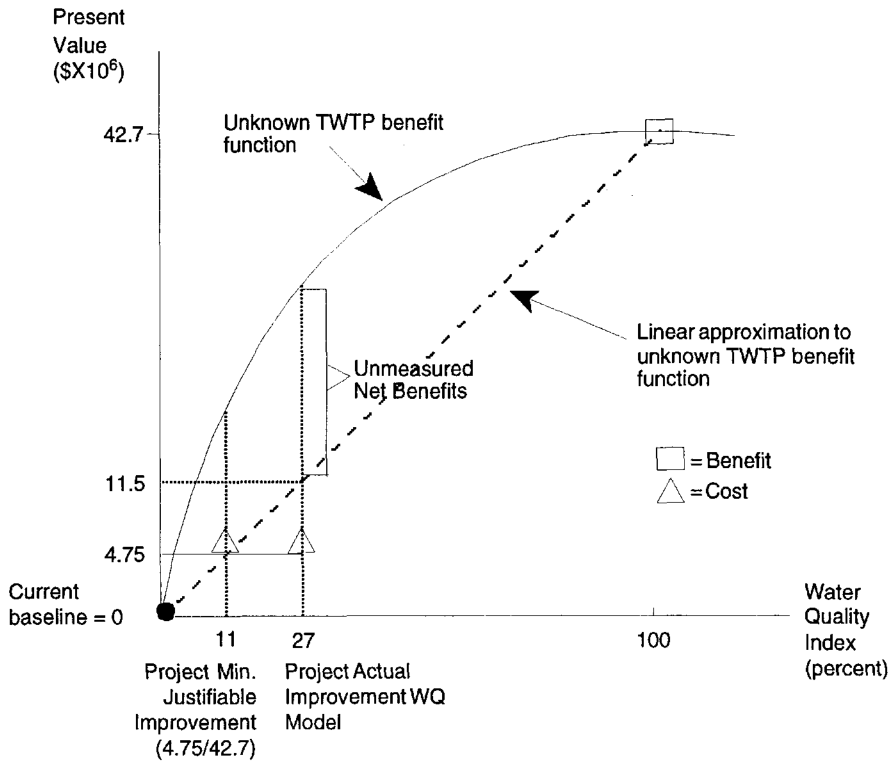
One of the major problems revealed in discussions with Bank staff is maintaining consistency between the actual effect of a project on water quality and the outcome that is described and valued in a CVM experiment used to generate project benefits. Obviously, if CVM survey respondents think they are paying for a greater degree of water quality improvement than the project will actually be able to supply, a form of benefits inflation sets in that makes economic justification pro forma. This can be a "trick" if it is premeditated, or a "trap" if it is the inadvertent result of poor communication between the project economist and the environmental engineers in charge of modeling the project's effect on water quality. A mismatch between benefits and achievements can also arise from a last-minute scaling back of project objectives because of, say, financial constraints, after the CVM exercise has already been conducted for the original more ambitious design. Finally, it can be a product of CVM benefit estimation for the ultimate outcome of a long-term, multistage program, of which the IDB-financed project is just a part or a first stage.

One way to deal with this potential problem is to anticipate it by conducting several independent CVM surveys to value alternative "states of the natural world" to rough out a stepwise approximation to benefits as a function of varying levels of AEQ. The analyses for the Bogota River wastewater treatment plant and the basic sanitation program for the Cartagena Bay basin projects in Colombia (Ardila, personal communication) attempted this. Another route is to conduct contingent choice experiments to produce continuous value functions for the various water quality attributes the project effects (an approach described in the literature by Mazotta et al., 1997, and Adamowicz et al., 1998, but never employed by the IDB in project analysis).[10] Either approach can be expensive and would normally be undertaken only for applying some variant of the regional cost-benefit optimization model approach described in Chapter 3.

When the analyst must improvise to correct for a mismatch, Ardila's (1993) analysis of the Guaiba Watershed Management Program suggests a shortcut. Figure 2-1 displays the calculation for two treatment plants that will be built to improve the quality of the Gravatai River in Brazil. The CV benefits are associated with a water quality level (Level II quality designation, which basically amounts to water fit for contact recreation, e.g., swimmable quality) that the project will not attain overall in the first stage, but only for a few hundred meters along selected stretches. To reconcile the project's limited achievements in the first stage with the ultimate Level II goal of a multistage program, the benefits of the entire program were scaled back using a linear approximation to the (assumed) concave, unknown total willingness-to-pay function for water quality, indexing Level II as a 100 percent improvement. (See also Chapter 4 on benefit function shape.) This linearization by definition understates benefits, assuming a diminishing marginal willingness to pay. As shown in the figure, the present value of known costs ($4.75 million) would just offset benefits if water quality improved from the baseline polluted condition by 11 percent. However, water quality model runs forecast that a weighted average quality improvement of 27 percent could be achieved by the project, which suggests gross

[10] Malarin and Vaughan (1997) illustrate how it *might* be applied to assess the environmental damage of solid waste landfills using a didactic integer programming model for optimal site selection.

Figure 2-1. Linear Benefit Interpolation: Guaiba Watershed Management Program Loan Document (TWTP, total willingness to pay)



discounted benefits of at least \$11.5 million, or net benefits greater than \$6.75 million (\$11.5 million minus \$4.75 million).

Combined Methods: Niklitschek and Leon

The work of IDB economists Niklitschek and Leon (1996) shows how to obtain an estimate of the total value of rehabilitating a recreational resource by combining hypothetical market and observed direct recreational use information. They demonstrate their approach by estimating an econometric model from survey data on value and use of ocean and bay beaches in a major metropolitan area in Latin America. Their technique allows use and nonuse values to be distinguished. The integrated econometric model can be evaluated to compute the effects of a beach carrying-capacity constraint in terms of the equilibrium entry fee it implies, the reduction in household benefits it imposes, and the potential revenue that can be collected under a fee regime rather than unrestricted access. It should be acknowledged that much simpler routes to benefits were actually taken in analyzing the project for which the data were collected; Niklitschek and Leon's published article went far beyond what normally can be expected or is found in applied project analysis.

Application of Indirect Methods

In general, revealed-preference (e.g., travel cost, random utility, hedonic, producer's surplus) or damage-avoided methods have only occasionally been employed and in most instances their empirical foundations have been shaky or missing entirely (as revealed by the comments in

Annex 2-D). The one exception appears to be the estimation of reduced potable water treatment costs from intakes downstream of a wastewater treatment plant. If the effects of the plant on water quality can be accurately modeled, the difference in potable water treatment costs with and without the project can be readily quantified by a straightforward engineering economics calculation of the savings.

Market-based measures of health benefits and tourism-related producer's surplus changes appear to be especially problematic. One possible reason is that in most cases the data needed to apply these methods reliably are simply not available off the shelf, because they are either proprietary, inaccurate, or extremely expensive to collect and analyze, leaving analysis by assumption as the only recourse. For example, one project in Colombia (project 23) calculated health benefits by applying medical treatment costs drawn from clinic records to gastrointestinal cases avoided due to improved river water quality. The cases avoided were calculated by applying reduction factors of 25, 50, and 75 percent, depending on the change in coliform counts in the river. However, no evidence from statistical epidemiology was provided in support of these decreases. In another case (project 6), tourism-related benefits of cleaning up a well-known bay were calculated by assuming that with the cleanup, half of the current visitors to the site would stay over one extra day, earning hoteliers an extra $35 per person per night. No analysis was cited in support of these assumptions. For project 25, tourism forecasts with and without the project and estimates of producer's surplus per visitor-day had to be built up from extensive interviews with hotel operators and potential investors (Kuhns, personal communication, 1993) to produce a judgment forecast of benefits whose statistical reliability was unknowable. In sum, the assertion that CV surveys are too costly for project appraisal would have to be evaluated by calculating, for example, what it costs to develop reliable data on health status and risk exposure, or tourism destination and length of stay in order to base health or tourism benefits for a single project on informed data analysis rather than reasonable assumptions.[11]

In the case of direct measurement of health damage, we need to know several things (Talcott, 1992), most of which are hard to determine in the developing country context. First we need to know the baseline, without-project incidence of the diseases in question (e.g., gastrointestinal illnesses) and their severity in terms of acute and chronic morbidity and mortality and treatment costs. Since many low-income individuals do not seek treatment, records from certified public or private health-care providers are not a reliable source of information. Next, we need a dose-response relationship and a model of exposure to separate out and predict the beneficial effects of the improvement in ambient water quality (physical benefits measured in terms of reduced health-care visits, work loss days, mortality), holding constant all the other possible assaults on good health, such as contaminated drinking water, spoiled foods, and the like. Controlled clinical tests to determine dose-response relations are not generally available because of ethical concerns, although even if a government agency were willing to do such tests, expense and technical difficulty might well still prevent them. Nor are there large databases on health status that could be used for statistical epidemiology. Finally, going beyond out-of-pocket treatment costs to get a fuller monetary measure of health damage avoided raises vexing valuation questions, especially when the affected parties are children, home-care givers, or the

[11] In commenting on an earlier draft of this chapter, Ducci (personal communication 1998) succinctly put the Bank's experience with CV and hedonics in perspective: "Survey data for hedonics is much cheaper than CV as the latter requires focus groups, pilots, better trained interviewers, etc. We usually quote CV surveys at $25 per questionnaire, while a comparable hedonic might be $15 per questionnaire. It used to be, however, that the time spent in the econometrics of hedonic models was large (to be able to torture the data to provide a decent coefficient) while for the CV a simple logit was sufficient. Nowadays the econometrics of CV has also grown, with double bounded models, Monte Carlo simulations or bootstrapping for variances, and other novelties. In the end hedonics is much cheaper but more unreliable with respect to whether you will be able to get a usable result."

unemployed. What is the value of reduced physical discomfort? a work loss day? a permanent decline in productivity or a premature death? These are serious research questions, but it is wildly unreasonable to expect that they can be convincingly answered through any single project analysis undertaken with a limited budget.

Tourism producer's surplus is equally hard to pin down. Ideally, to predict the tourist visitation response to project-induced changes in amenities, one would like to have an econometrically based model explaining tourists' choice of vacation destination and trip duration to broad regions (e.g., the Caribbean, Central America, the Southern Cone), countries within regions, and sites within countries as functions of trip cost, purpose, and environmental amenity attributes (including but not limited to ambient water quality). Again ideally, this information would have to be supplemented by cost information obtained from holiday service providers to identify surpluses in the transport, tour, food, and accommodation chain, recognizing that surpluses captured by non-national corporate entities remitted abroad must be netted out of the benefit estimate. It seems more than a bit unreasonable to expect that such an analysis could be built from the ground up just for a single project, particularly since in coastal areas the effect of ambient water quality on tourism levels may be hard to identify statistically if tourists are unaware of it. Many countries are reluctant to establish a policy of beach closings or health advisories for water contact recreation, so the effects of water quality on tourism based on water recreation are unlikely to be revealed in any econometric model of tourist behavior.

Given the several intractable data and analysis issues raised by the application of all but the simplest of the methods that rely on observed rather than hypothetical behavior, it is not hard to understand why CVM has been the favored route to estimating the benefits of IDB-financed sanitation projects and their sewer and wastewater treatment components. For more on this choice from non-Bank economists practicing in developing countries, see the comments in Chapter 7.

Sensitivity Analysis

Most of the sensitivity exercises conducted have been of the standard decision reversal type, taking key variables such as investment cost, operation and maintenance cost, total benefits or benefits by category, rate of population growth, and postponement and asking what percentage changes in baseline costs and benefits would drive the economic rate of return below 12 percent (or make net present value negative at a 12 percent discount rate). Along a similar line, sometimes arbitrary changes in the same variables have been posited and their effect on internal rate of return (IRR) reported. These approaches give a rough, subjective impression of whether the project's economic viability is robust to "small" changes in cost and benefit streams or the variables underlying them, taken one at a time. It does not define what "small" is, nor does it consider the possibility of the joint occurrence of several unfavorable events.

In the project sample, there are three innovative analyses that go beyond this conventional sensitivity approach; two explicitly incorporate the variance in statistically estimated benefits to construct a probabilistic statement about the magnitude of the benefit streams, and one undertakes a full Monte Carlo risk analysis. (For further discussion and a worked-out example, see Chapter 9.)

The Distribution of Benefits: Ardila and Savedoff

In the economic analysis of the Guaiba Watershed Management Program, a sensitivity analysis is presented using the empirical distribution function of expected benefits. As explained in Ardila (1993), this distribution was generated from the econometric estimate of a logistic probability-of-acceptance function estimated from survey data, which posited a referendum offer to supply improved environmental quality over a range of payments. The probability-of-acceptance function can be transformed into an expression for the expected value of willingness to pay, which depends on the parameter estimates of the logistic that are random variables. Repeated evaluation of this expected willingness-to-pay function over a large number of draws from the distributions of the parameter estimates (holding the explanatory variables fixed at representative levels) produces a rough (bootstrap) estimate of the empirical distribution function of expected benefits per household. With this empirical household benefits distribution in hand, sensitivity-type statements can be made about the probability that aggregate benefits will be above the critical value required for project acceptance.

The report on the Basic Infrastructure and Sanitation Program in Fortaleza also contains information on the IRR under reductions in per household benefits of one and two standard deviations. Unlike the Ardila (1993) approach, the variance of willingness to pay in this case was approximated with a first-order Taylor series expansion (Savedoff, personal communication and Savedoff, 1992). Although this innovation antedated the Ardila application, which was undertaken independently, the background paper on the 1992 Fortaleza economic analysis was not widely distributed, and did not elaborate on how to derive and apply the technique. Analytical and programming details on both the delta (Taylor series) and bootstrap methods for generating the empirical distribution of willingness to pay are carefully elaborated in Hazilla (1999).

Risk Analysis: Darling, Gomez, and Niklitschek

Monte Carlo risk analysis was creatively employed by Darling et al. (1993) to assess the net benefits of a Barbados South Coast sewerage project. While willingness-to-pay estimates for a public sewer system were available from a contingent valuation exercise, they alone were insufficient to justify the project. Other important categories of benefit (involving losses in producer's surplus in tourism and fisheries that would be avoided through the project's beneficial effects on coastal water quality and coral reef condition) were more difficult to quantify precisely with the data available. Therefore risk analysis was used to generate IRR and net present value (NPV) distributions because there was no information on the empirical relations between water pollution control and coastal water quality and reef condition, fisheries productivity, beach erosion, and tourism visits.

Excluding benefits from avoiding tourism declines, but including cost savings from not having to invest in and operate private sewage disposal systems, domestic willingness to pay for cleaner coastal water, health costs avoided due to reduced exposure to contaminated coastal water, fishery benefits, and prevention of beach erosion produced a range for the IRR from –0.2 to 6.6 percent, and a modal value of about 4 percent. This range is below the Bank's 12 percent cutoff. The permanent decline in tourism avoided by maintaining coastal water quality required to raise the IRR to 12 percent and justify the project was then calculated in a second "what if" step. Without the project, tourism on the whole island would have to decline by 4 to 7 percent, once and for all, owing to coastal pollution (or, alternatively, 10 to 17 percent in the South Coast only) in order to raise the IRR to 12 percent. The analysis concluded that, given the possibility that without the project, the tourism decline could be even greater than 10 percent, there is a high probability that the project was economically feasible.

CONCLUDING OBSERVATIONS

Two sorts of concerns emerge from the accumulated IDB experience: broad questions related to the definition of the problem setting and specific issues related to working within that setting once it has been defined.

Scope of Analysis

Paramount among the broad issues is how to define the scope of the analysis in the sense of going beyond the immediate project to incorporate the objectives and actions of the longer range program it fits into. Here, we run into the challenges raised by multistage programs, particularly whether they can be structured to optimally define the time phasing for investments that improve AEQ. Temporal scope has an analogue in spatial scope, which involves representing the natural world and the discharge behavior of nonpoint and point (domestic, commercial, and industrial) sources that affect natural world conditions in a basin, and choosing among possible ways to improve AEQ in which any particular project is usually only one of many. Setting the time dimension aside, this implies the need for integrated empirical modeling strategies that consistently meld and mimic economic and natural world conditions to produce static recipes for the optimal (or efficient) allocation across space of discharge reductions and the public investment part of the total package. The broadest and most difficult problem of all is combining the temporal and spatial dimensions to solve for the regional least-cost or net benefit-maximizing investment packages over a long planning period of 20 years or more, given assumptions about growth and change (both economic and demographic).

Most of the projects reviewed do not engage in such heroics, of course, although some have tried to do static regional least-cost analysis, and a few have tried to explore scenarios of alternative investment phasing. Usually the focus of the rest has been much narrower, so the issues that arise pertain mainly to ways to work within that more restricted setting.

Analytical Technique

Even within the ad hoc single-project approach, environmental simulation modeling is critical for establishing the degree of ambient quality improvement attributable to the project and identifying where the benefits will be registered. If the project treats only domestic wastewater, and industrial dischargers are not obliged to connect to the public sewer system, the project's effects must be separated from what else is going on (in particular, the effects of other dischargers). As such, AEQ models are an essential ingredient in project appraisal and a necessary condition for realistic economic analysis (either least-cost or cost-benefit). Although the Bank's official project documents, not being technical in nature, scarcely refer to quantitative modeling of natural systems, economic analysis cannot be divorced from it, as Chapters 4 and 5 demonstrate.

The decision about which benefit estimation method to use in general seems to have come down pretty firmly on the side of stated- rather than revealed-preference approaches. Having said that, however, it is still true that several specific issues remain. Most important, the predicted effects of a project on the natural world must be matched with the benefit scenario(s) portrayed in a CV survey used to generate gross benefits. Many of the projects reviewed produce rather modest AEQ improvements because they work on a very degraded baseline under a hard budget constraint. While CV may be able to accurately detect WTP for very modest improvements, very few "multiple states of the world" CV exercises have been done in order to verify that belief or help decide on the extent of treatment and AEQ improvement. Rather, benefit estimates for a total program sometimes have had to be scaled back to produce the benefits of less ambitious specific project components. Contingent choice or multiple scenario methods may be worth considering in this context to approximate a benefit function rather than a benefit point or single state of the world. Chapter 7 elaborates on this and several other technical aspects of benefit estimation touched on in this chapter.

Benefit transfer has been uncommon in Bank applications. By beginning to collect CV benefit estimates in a database, after enough observations accumulate, the Bank could undertake a statistical meta-analysis. It would relate benefits per household to the type of benefit (sewer connection, general ambient water quality improvement, etc.), the magnitude of change being valued, the average income level and other socioeconomic characteristics of the respondents, and the characteristics of the method used (e.g., open-ended, single-bounded, or double-bounded CV; hedonic analysis). The estimated benefit function could be used as a benefit transfer shortcut for prefeasibility screening, assessment of environmental damage, and valuation of global multiple works projects.

Sensitivity analysis has been handled in a routine fashion in most IDB projects. However, the more innovative applications suggest that water quality improvement projects are inherently uncertain because the natural world is not deterministic and because benefit estimates are random variables rather than fixed values having zero variance. Chapter 9 blends elements of these innovative analyses to demonstrate how a risk analysis can be constructed that incorporates information on the distribution of benefits as well as historically derived information on cost and performance to show their separate and combined effects.

Annex 2-A

Form Used to Gather Information on the Analysis Reported in IDB Treatment Plant Loan Applications

PROJECT DESCRIPTION	
PROJECT NAME	
COUNTRYIYEARIPROJECT NO.ILOAN AMOUNTITOTAL COST	
MAIN COMPONENTS	
OBJECTIVE(S)	
SOURCES CONSULTED	
TYPE (I, II)[1]	
PHYSICAL SETTING	
Scope and Spatial Limits of Problemshed (Basin, River Reach, Point, Lake, Estuary, Political Jurisdiction(s), Metro Area(s), Not Clear, etc.) Clearly Specified and Rationale Supplied?	
Sources Targeted (Point, Non-Point, or Both)	
Polluters Targeted (Municipal, Industrial, Commercial, Agricultural)	
Types of Pollutants Targeted? (Specify)	
Discharge Reduction Targets? (Specify)	
Ambient Environmental Quality Targets?(Specify)	
Effect of Actions Outside Problemshed on AEQ in Problemshed Identified/Accounted For?	
BASELINE INFORMATION, ENGINEERING DESIGN and NATURAL SYSTEMS MODELS	
Baseline Ambient Quality Conditions Established via Monitoring?	
Discharge Inventory by Source Location and Pollutant?	

Alternative Project Options Explored?(Sector coverage; end-of pipe vs. non treatment options; use of incentives; comparison of treatment type options)			
Use of Natural Systems Models to Simulate AEQ Impacts of Project and Alternatives (Types of Models Used /Impact Types/Impact Magnitudes)			
ECONOMIC ANALYSIS			
Single-Stage or Multi-Stage Program. Discuss.			
Project Priority Established via Regional Scoping and Ranking of Alternatives over Space and Time (Yes/no/criterion Used)			
Simulation of a Few Pre-identified Alternatives (e.g. Municipal Treatment) or Optimization over a Much Larger Universe of Possibilities?			
Economic Analysis Method (None, Cost-Effectiveness, Regional Least Cost, Cost-Benefit, etc.)			
Types of Benefits Identified (e.g. health, recreation, aesthetic, downstream productivity/cost reduction) *Degree of Quantification* of Each (e.g. qualitative, physical effect, monetary) *Method Used to Value* (Hedonic, Travel Cost, Damage Avoided, Contingent Valuation etc.)	Type	Quantified?	Valuation Method
Benefits Targeted for Specific Geographic Locations?	Type	Principal Location(s) of Benefit Accrual	
Number of Beneficiaries by Type of Benefit	Type	Number of Beneficiaries	
Magnitude of Benefits by Type (Total and per Standard Unit: $/Family etc.)	Type	Tot. $ (1997)	1997 $ per capita
Distributional Analysis of Incidence of Costs and Benefits by Income Group, Location, etc.? Describe.			
Consistency of Cost-Benefit Analysis Outcome with Existing Discharge or AEQ Standards Discussed/Reconciled? Explain.			
Cost-Benefit Analysis Done for Multiple Stages in Sequence. Both Separately and En Toto (the Contingent Project problem)? Conclusions of Analysis.			
Nature of Sensitivity Analysis Performed, If Any. Non-quantifiables Identified? Implicitly Valued for Decision Reversal? Other. Describe. Conclusions of Analysis.			
Financing Arrangements for Subsequent Stages in Multi-stage Program Discussed? Explain.			

Annex 2-A continued

REVIEWER'S ASSESSMENT OF PROJECT DOCUMENTATION AND ECONOMIC ANALYSIS[2]	EXCELLENT	GOOD	FAIR	POOR
Baseline and Project Design Information				
Economic Analysis Information				
COMMENTS				

[1]Project types are: I, large, project/program for wastewater collection, treatment and/or disposal in a single metropolitan area; II, Global works projects in two or more cities, multiplicity of potential locations, treatment/septic systems.

[2]Clarity and completeness of information does not refer to the quality of the information in surveys, feasibility studies and consultant's background reports upon which the analysis was based. Rather, it means whether the final project document fully explains and discusses the methods used and the quality of the information available to apply them, clearly pointing out areas of uncertainty and information weaknesses and gaps. In short, can the project and the analysis approach be understood by reading the project document alone, or would other background sources be needed?

EXCELLENT: More than 85 percent of the information expected is available in official project document.
GOOD: Between 70 percent and 85 percent of information expected is available in official project document.
FAIR: Between 60 percent and 70 percent of information expected is available in official project document.
POOR: Less than 60 percent of information expected is available in official project document.

Annex 2-B

Key to Project Identities and Numbers Used in Chapter 2

		TYPE I PROJECTS
1	AR0038	Environmental Sanitation and Flood Control in the Reconquista River Basin
2	AR0136	Environmental Management of the Matanza-Riachuelo River Basin
3	BA0036	South Coast Sewerage Project
4	BR0186	Basic Infrastructure and Sanitation Program of Fortaleza — SANEFOR
5	BR0190	Tietê River Decontamination Stage I — Basic Sanitation
6	BR0072	Basic Sanitation Program for the Guanabara Bay Basin — Phase I
7	BR0073	Guaiba Watershed Management Program — First Stage
8	CO0082	Medellin River Sanitation Project — First Phase
9	CO0208	Rio Bogota Wastewater Treatment Plant
10	CO0227	Cartagena Sewer System
11	EC0161	Water Supply and Sanitation Project for the City of Cuenca
12	EC0025	Water Supply and Sewerage Project for the City of Quito
13	GU0073	Guatemala City Metropolitan Area Environmental Program
14	ME0138	Fourth Stage of the Monterrey Water Supply and Sewerage Project
15	ME0056	Guadalajara Water Supply and Sewerage
16	NI0027	Lake Managua and the City of Managua Environmental Improvement Program
17	UR0023	Second Stage of City of Montevideo Urban Sanitation Project
18	UR0089	Metropolitan Montevideo Sanitation Program, Stage III
		TYPE II PROJECTS
19	AR0116	National Water Supply and Sewerage Program — First Stage
20	AR0130	Water Supply and Sanitation Program — Stage IV
21	BO0125	Urban Basic Sanitation Program
22	BR0067	Program of Social Action in Sanitation
23	CO0198	Upper Bogota River Environmental Rehabilitation Program
24	GU0093	Municipal Development Program — Second Stage
25	HO0028	Bay Islands Environmental Management Project
26	ME0128	Water Supply and Sewerage Program
27	PR0064	Urban Water Supply and Sewerage Project

Annex 2-C

Abbreviated Project Descriptions

Project No.	1
Country	Argentina
Type	I
Main Components	1. Flood control. 2. Household and industrial pollution control, including (a) the construction of three plants and reconditioning of another for the treatment of wastewater and sludge generated by nonconventional household systems, and of works for final discharge of treated wastewater into receiving bodies; and (b) execution of a supplementary environmental protection subproject including, among other things, a plan of action for industrial pollution. 3. Relocation and indemnification.
Objectives	The main objective is to improve the quality of life, environment, and public health of a broad segment of the urban population in Greater Buenos Aires, which is affected by flooding from the polluted waters of the Reconquista River and its tributaries, through the construction of works for flood control and measures to begin cleanup of the river.
Scope and Spatial Limits	Reconquista River basin (167,000 hectares) in the NW part of Buenos Aires metro area. Sixty-five percent of basin population lacks potable water supply, drawing water from groundwater that is exposed to contamination due to improper sewage and solid waste disposal; 79 percent lack sewerage connections, and use septic tanks or bore-hole latrines. Sewage collected by tank trucks and discharged at unauthorized sites along course of river. Industry (4,242 plants, 280 of which account for 89 percent of BOD) contributes 148 tons/day of BOD; 1.7 million households with no sewer connection generate 10 tons BOD/day, and connected households 0.6 tons/day. Area is subjected to extreme flooding from heavily polluted Reconquista River (BOD of 409 mg/l, DO of 0 for 41.8 of 48.2 km of river in project area). Province of Buenos Aires has identified short-, medium-, and longer-term measures to reduce flooding and pollution.

Project No.	2
Country	Argentina
Type	I
Main Components	1. Industrial pollution control ($52 million total cost) including such activities as: (a) Industrial pollution control plan ($1 million) to implement environmentally motivated low-cost waste minimization and cleaner production measures for industries. The World Environment Center will provide direct technical assistance to most of the basin's 65 major polluters to develop and implement pollution control measures. In addition, 1,000 polluters will be inventoried and licensed (as complying with or having a timetable to comply with environmental legislation) by month 48 of the program; (b) building institutional capacity of public pollution control authorities ($10 million), with direct technical assistance to the province's Secretaría de Medio Ambiente and the municipality's Sub-Secretaría de Medio Ambiente; (c) hazardous industrial waste disposal facility studies ($0.4 million); (d) contingency planning for flood control ($18 million); and (e) removal of abandoned boats, vehicles, and cargo ships from the river and river bank cleanup ($22.4 million). 2. Flood control and drainage works. 3. Solid waste management. 4. Urban rehabilitation.
Objectives	The program's overall objective is to improve management of the Matanza-Riachuelo basin's natural resources through coordination of environmentally related actions. The objective of subprogram (1) relevant here is to reduce contamination levels. Degree of reduction is not specified.
Scope and Spatial Limits	Matanza-Riachuelo basin, whose catchment, along with the Reconquista River basin (see separate summary of loan AR-0038), occupies the metropolitan area of Buenos Aires. These two basins include most of the country's industrial base and generate 25 percent of the country's gross domestic product (GDP). The Matanza-Riachuelo basin occupies over 2,000 km^2 and has over 60 tributaries in the catchment. Eastern region is highly industrialized and densely populated (12,000 persons per km^2), and includes the poorest areas of Buenos Aires. Western region is largely agrarian, with population density as low as 6 persons per km^2 in some areas. Rationale rests on problems of flooding, land use planning, industrial and domestic pollution, which are discussed (no mention of agricultural nonpoint pollution). Mention of "integrated basin approach," the existence of a basin environmental management plan and use of a basin authority as executor.

Project No.	3
Country	Barbados
Type	I
Main Components	Project works would consist of (a) a collection system (sewers), (b) five lift stations, (c) a transmission line, (d) a primary wastewater treatment plant and main pumping station, (e) a 1.1-km outfall line into the sea, and (f) in-house connections of properties in the project area. Some complementary components (e.g., water meters).
Objectives	Primary objective of the project is to bring the near-shore waters on the south coast to a bacteriological standard that corresponds to the accepted international standards and to reduce the chemical contamination affecting the reefs, marine life, and beaches.
Scope and Spatial Limits	Twelve km of coastal strip running 0.5 km inland from Ostins in the south to Bridgetown in the north. Area of 500 ha includes high levels of commercial and tourism activities and high residential density (4,422 households, 182 retailers, 47 offices, 109 service businesses, 92 hotels, 51 public institutions, 14 industrial establishments, and 21 recreational facilities). Rationale clear. Restoration of near-shore (beach) water quality to accepted standards for human water contact and reduction of ground (sheet) water contamination requires replacement of absorption wells and septic tanks with in-house sewerage connections, primary treatment, and ocean disposal.

Project No.	4
Country	Brazil
Type	I
Main Components	1. Sanitary sewerage component will target the following: (a) Expand service coverage from the current 59,800 household connections to 208,000 by 1996, thereby benefiting an additional 796,000 users; by the end of the program, some 1,100,000 people will be served. (b) Reduce organic pollution in the Côco, Maranguapinho, and Ceara rivers and their respective tributaries, in a number of lakes, and at most of the beaches along Fortaleza's ocean front. (c) Improve the efficiency of the system for dumping waste in the ocean through construction of a pretreatment station (PTS). 2. Urban storm drainage. 3. Urban sanitation.
Objectives	The main purpose of the program is to improve the standard of living of Fortaleza's urban population by improving public health, environmental protection, and the public welfare through sanitary sewerage works, storm drainage works, and collection and proper disposal of solid waste.
Scope and Spatial Limits	City of Fortaleza, including (a) the maritime watershed with a population of 450,000 whose Atlantic coast beaches are heavily polluted (unsuitable for direct contact recreation) due to domestic and industrial effluents; (b) the Côco River basin with a population of 560,000 with pollution and flooding problems along the length of the river; (c) the Maranguapinho River basin with a population of 760,000 that has flooding problems in areas of haphazard low-income housing and serious pollution in Parangaba Lake, a receptor of industrial, hospital, and household sewage.

Project No.	5
Country	Brazil
Type	I
Main Components	1. Sanitary sewerage system. This consists of the following works: (a) laterals: approximately 1,500 km of sewers and 248,000 house connections, (b) mains: 315 km of mains, and (c) pumping stations (11). 2. Outfalls and treatment. This consists of the following: (a) outfalls: 37 km of outfall sewers with reinforced concrete pipe of diameters from 0.60 m to 2.50 m.; (b) treatment plants: construction of two wastewater treatment plants and expansion of existing plant for total direct cost of $58 million; increasing the proportion of wastewater treated from 19 to 45 percent by 1995; and (c) final disposal of sludge. 3. Supplementary activities include relocation of families, updating of the official system survey, comprehensive study for appropriate use of water resources, institutional strengthening of the Companhia de Tecnologia de Saneamento Ambiental (CETESB) for industrial pollution control, training of São Paulo State Basic Sanitation Authority personnel, and special studies.
Objectives	The objectives of the project are to (a) enhance the quality of life for the population of the metropolitan area of São Paulo, (b) improve health and environmental conditions in the area, (c) reduce the pollution of the Tietê River and its main tributaries, (d) study the use of the water resources and the subsequent stages of the project, (e) strengthen the legal and institutional structure of the state of São Paulo for control of industrial waste, and (f) train technical and administrative staff to operate and maintain the wastewater treatment plants.
Scope and Spatial Limits	Upper part of Tietê River valley extending along 97 km of the metropolitan area of São Paulo. Sewage makes up 50 percent of the upper Tietê's average flow. Daily pollutant loads are mainly 800 tons of household organic wastes and 350 tons of industrial organics. Only 14 percent of the wastewater discharged into the river is treated (secondary). Roughly 95 km of the river are anaerobic, and will reach 105 km by year 2000 without corrective action. Odor is a serious problem, as are health hazards. The waters of the Tietê and its main tributaries are similar in chemical and organic composition to the city's sewage.

Project No.	6
Country	Brazil
Type	I
Main Components	1. Sewage collection and treatment. Sanitation works consisting of (a) construction of four primary wastewater treatment plants to handle a total flow of 6.6 m^3/sec (Alegria, Sarapui, Pavuna, and São Gonçalo plants); (b) two secondary-level treatment plants for Governador and Paqueta islands, to treat a flow of 0.247 m^3/sec, including a 2.5-km underwater outflow for Paqueta; (c) upgrading of the Icarai and Penha plants to perform secondary-level treatment of 2.23 m^3/sec and a 4.7-km land and underwater outfall for Icarai; (d) 126 km of collectors, interceptors, and outfalls; and (e) 1,000 km of collector systems and 34 pumping stations. Total cost of sewage collection and treatment subproject is $405.9 million. 2. Potable water supply. 3. Solid waste collection and disposal. 4. Canal and river drainage. 5. Complementary environmental programs in industrial pollution control, environmental monitoring, and education. As part of the industrial pollution control activities, the State Environmental Engineering Foundation (FEEMA) will be supported institutionally to give continuity to the control actions in 50 industries considered critical and to initiate another 402, so as to bring under control by the end of four years a total of 455 industries that account for 90 percent of the organic industrial material generated in the basin. The program would also support FEEMA by enhancing its efficiency in handling technological mishaps in the area, including control of pollution by gasoline stations. 6. Digital mapping and municipal institutional development.
Objectives	(a) Clean up the Guanabara Bay and adjacent basin area. (b) Improve the quality of life of the 7.3 million inhabitants residing in the Guanabara Bay basin. (c) Strengthen those local government institutions whose activities can positively affect the bay.
Scope and Spatial Limits	Greater Guanabara Bay basin, including the bay, most of the municipality (and 70 percent of the population) of Rio de Janeiro, 5 municipalities to the west of the Bay, and 7 municipalities to the east. The area has a total population of 7.3 million and includes the country's second largest industrial park (6,000 industries), a major port, and a large oil refinery. Rio's rapid urbanization and weak enforcement of pollution control regulations have contributed to high levels of bay contamination. In 1991 it received a daily load of 400 tons of untreated sewage, 82 tons of industrial organic material, 4 tons of leachate from solid water dumps, 3.2 tons of petroleum products, and 0.4 tons of heavy metals. Only 14 percent of the sewage generated in the bay area is treated; untreated sewage enters the bay via canals, the drainage system, and 35 tributary rivers in the basin. All of the bay's beaches are closed most of the time because of excessive coliform levels.

Project No.	7
Country	Brazil
Type	I
Main Components	1. Prevention and control of industrial and residential pollution. The objective of this component is to reduce the harmful effects of (a) some of the discharges of raw sewage and untreated liquid industrial effluent that flow into receiving bodies of water upstream from water supply intakes and from beaches that in the past were used for recreation, (b) pollution caused in the water resources of the Gravatai River subbasin by the leachate from the landfill in the northern area of Pôrto Alegre, and (c) the lack of a system for solid waste collection. To control residential pollution, sanitary sewerage systems will be provided for the population of the cities of Cachoeirinha and Gravatai, approximately 171,560 inhabitants at the beginning of the period and estimated to increase to about 243,760 by the end of the first stage of the program (more than 95 percent coverage). Furthermore, sanitary sewerage systems will also be provided for 119,520 inhabitants of the city of Pôrto Alegre at the beginning and 165,390 by the end of the first stage of the program. To control industrial pollution, via a reduction by 50 percent of the pollution load, a licensing plan will be designed and established. 2. Soil management and control of toxic farm chemicals in priority microwatersheds. The objectives of this component are to (a) increase agricultural productivity based on principles of sustainable development and (b) increase the net income of productive units. 3. Consolidation of conservation units. The objectives of this component are to (a) ensure the protection, supervision, and improvement of the infrastructure in five existing conservation units; (b) conduct studies for the expansion or creation of new units within the watershed; (c) develop a program of environmental education for conservation; and (d) strengthen the institutional capacity of the Fundação Zoobotânica and the National Department of Renewable Natural Resources. 4. Education and raising of awareness of environmental issues.
Objectives	Overall objective is to restore environmental quality in rural and urban areas so as to improve the quality of life of the population and to promote the sustainable use of the natural resources of the Guaiba River watershed. Among the specific objectives is one to monitor and reduce urban pollution from residential and industrial sources.
Scope and Spatial Limits	Overall project setting is entire Guaiba basin, covering 85,050 km^2, but sewerage and treatment activities are mainly concentrated in the Gravatai and Lake Guaiba subbasins, of which the Pôrto Alegre metro area is part. Together they cover only 6.3 percent of the total Guaiba basin area, but have about 45 percent of the total population. Rationale is that the cities of Cachoeirinha, Gravatai, and Pôrto Alegre generate significant amounts of industrial and domestic wastewater that is discharged untreated into receiving bodies, and most rivers in the lower watershed are highly polluted. For example in the Gravatai River, a tributary of the Guaiba, coliform counts are in the hundreds of thousands to millions per 100 ml, BOD is up to 20 mg/l, and DO is near zero in several reaches above its confluence with the Guaiba. The Guaiba River/Lake is the chief source of water supply for Pôrto Alegre as well as a recipient of its untreated waste; while BOD levels are low due to the high flow and reoxygenation capacity, fecal coliform counts are in the billions in the upstream segments of the Guaiba, falling over the 50 km between the city and its outlet to a lagoon eventually draining into the Atlantic Ocean. These coliform levels present a serious public health threat to users of beaches and water-based recreation.

Project No.	8
Country	Colombia
Type	I
Main Components	1. Sanitation. This component is divided into two parts: (a) treatment and interception, which involve construction of a secondary wastewater treatment plant in the municipality of Itaguí in the upper Medellín River valley and 29 km of intercepting sewer mains, and (b) a sewer system, involving construction of sewer systems, collectors, storm drains, and overflow channels. Cost of secondary treatment plant $47.2 million. 2. Water supply. 3. Future plan and institutional strengthening. The Empresas Públicas de Medellín is planning to clean up the Medellín River in three phases, with the present project representing phase one of its sanitation plan.
Objectives	Overall objective is to improve the quality of life and health and environmental conditions for Aburrá Valley residents. Specific objectives are (a) cleaning up parts of the Medellín River and its tributaries, (b) treating a portion of the area's wastewater through construction of the San Fernando wastewater treatment plant, (c) upgrading and expanding potable water and sanitary sewerage services to urban areas still lacking such services, (d) optimizing operation of potable water distribution systems by using water more efficiently and reducing losses, (e) preparing for the next phase of the comprehensive sanitation plan for the Aburrá Valley, and (f) strengthening the institutional capacity of the Empresas Públicas de Medellín.
Scope and Spatial Limits	The stretch of the Medellin River running through the center of the Aburra Valley, home to a population of 2.5 million living in Medellin, Colombia's second largest city, and 9 other municipalities. Two hundred ravines in the valley act as open sewers by collecting all the rainwater and much of the industrial and residential wastewater, discharging it into the Medellin River. This creates health risks and prevents the use of the Medellin's polluted waters for irrigation further downstream.

Project No.	9
Country	Colombia
Type	I
Main Components	A BOOT (build, own, operate, transfer) concession to the project company for the design, construction, maintenance, and ownership of treatment plant in two phases; primary followed by secondary.
Objectives	Under the terms of the concession, the project company will design, build, own, operate, and maintain a wastewater treatment plant at the juncture of the Salitre River and the Bogotá River.
Scope and Spatial Limits	Metro area/river reach. Plant is located at juncture of the Salitre and Bogota rivers in the middle basin of the river, which contains the city of Bogota. Downstream from this junction, the Bogota is a "dead" river, as it receives sewage from the city's 6 million residents plus 10,000 industries. The plant will operate on wastewater collected from residential communities in the northern part of Bogota. Rationale clear; project complementary with another IDB project (CO-0198) to improve water quality in upper basin before it reaches Bogota proper.

Project No.	10
Country	Colombia
Type	I
Main Components	1. Laying of sewer lines in southwest Cartagena ($10,500,000). Investments under this component will expand sewer-system coverage in the southwest part of the city, in six subbasins draining into Cartagena Bay. 2. Reconditioning of Bocagrande sewer lines ($6,300,000). Investments in this component will recondition secondary sewer lines in the Bocagrande district. 3. Sewage disposal ($14,100,000). This component will fund the construction of collector sewers and pumping stations to convey sewage away from all the project areas, and from other parts of the city whose wastewater is running into the bay. The project does not contain a treatment plant component.
Objectives	The objective is to improve sanitation conditions in areas of Cartagena that drain into Cartagena Bay. The project is the first stage of a program to expand the city's water system, increase water production, extend the sewer system, and devise a sewage treatment and final disposal system. The project claims as a benefit, "Water quality will significantly improve when sewage is conveyed to the existing underwater outfall, ending wastewater runoff into the bay."
Scope and Spatial Limits	Metropolitan area of Cartagena, confined to the Cartagena Bay catchment, with sewer collection, chlorine application to reduce fecal coliform concentrations, and disposal via an underwater outfall into the bay for ultimate discharge into the Caribbean. Postponed for a later stage are sewer connections for the Cienaga de la Virgen catchment and a treatment plant.

Project No.	11
Country	Ecuador
Type	I
Main Components	1. Completion of Tomebamba potable water supply system, improving the catchment, potable water treatment plant, and existing distribution systems, and new distribution with 6,400 additional connections. 2. Build an additional water supply system using withdrawals from the Machangara River. 3. Build stormwater and sanitary sewer works in unserved areas, along banks of all rivers flowing through the city, and a final outfall. 4. Build a wastewater treatment plant. 5. Supplementary activities include designs for dams to supplement storage capacity, water loss reduction program, system for industrial waste control, automation/data processing for water company.
Objectives	Primary objective is to expand the capacity and coverage of the water supply and sanitary and stormwater sewerage systems of the city of Cuenca, to improve the living conditions and health of its inhabitants, including treatment of the wastewater to control pollution of the rivers. Specific goals are to (a) increase the percentage of coverage of the population with water supply connections from 85 at present to 95 percent at the end of the project and maintain that coverage to the year 2000; (b) increase the percentage of coverage of sanitary sewerage services from 75 at present to 85 percent at the end of the execution period and 90 percent in the year 2000; (c) provide for effective sanitation of the urban area, intercepting the numerous discharges of wastewater entering directly into the rivers going through the city and treating them before conveying them to the Cuenca River; (d) set up a system of ongoing control of industrial wastes to regulate them and prevent the entry of toxic materials into the sewerage systems and the rivers; and (e) improve the systems of Empresa Municipal de Telefonos, Agua Potable y Alcantarillado de Cuenca (ETAPA) for control of operations and marketing, for the purpose of reducing the water not accounted for by 25 percent and lowering the demand to 330 liters per person per day.
Scope and Spatial Limits	Polluted rivers and streams crossing through the urban area of the city of Cuenca in Ecuador (pop.= 200,000), and the Cuenca River, the ultimate receptor.

Project No.	12
Country	Ecuador
Type	I
Main Components	1. Water supply system ($79,232,000). 2. Sewerage system ($14,210,000). Approximately 300 km of sewers will be installed in about 22 districts of the city. In areas of recent growth, separate pipes will be used to collect sewage and stormwater. In the more established areas, a single or combined disposal system will be installed as being more compatible with the surrounding neighborhoods. In those cases in which immediate action needs to be taken to protect the water quality of the receiving body, small wastewater treatment plants will be built. 3. Master plan ($3,100,000). The integrated master plan for water supply and sewerage of the city will determine, at the feasibility level, the medium- and long-term investment program for optimum provision of the service. Special considerations will be given to (a) the predominance of a combined sewer system for wastewater and stormwater and the advantages of separating them; (b) the polluted condition of the Machángara and Monjas rivers, using a mathematical water quality simulation model; (c) definition of the level of municipal treatment and industrial pretreatment to be required, including treatment of the sludge product from the city's purification plants; and (d) the need to update, amend, or expand the rules, regulations, practices, and design criteria currently used. 4. Institutional strengthening ($13,717,000).
Objectives	Principal objective is to help improve the hygienic and health conditions of the population of the city of Quito by providing a more efficient and larger-capacity water supply and sewerage service.
Scope and Spatial Limits	Not fully known ex ante since program is multiple works. Only a general statement: "The sewerage system of the city of Quito is mostly the combined type, mixing sewage with storm water and discharging them untreated into the Machángara and Monjas rivers that cross the city. Coverage of the existing system is estimated at 74%. ...Areas that are not covered have individual disposal systems, such as pit latrines and septic tanks. ...there is an urgent need to expand the existing systems to the areas of greatest recent demographic growth (southern area of the city). .... The parishes adjoining the city have sewer systems that cover approximately 950 hectares, and serve a population of 110,000 inhabitants through systems totalling 158 km. None of the waste is treated, which means that sewage is directly discharged into streams and creeks."

Project No.	13
Country	Guatemala
Type	I
Main Components	1. Reform and institutional development of metropolitan environmental management. 2. Solid waste management. 3. Sewage management consisting of (a) reconditioning of 7 existing, priority treatment plants currently out of commission, (b) introduction of sewage collection and septic tank treatment systems in 5 slums, and (c) construction of 2 new secondary treatment plants (San Miguel Petapa and San Jose Pinula).
Objectives	A sustainable improvement in the quality of life of the Guatemala City metropolitan area. The specific objectives are to (a) improve metropolitan management of environmental services by encouraging private sector and community participation, and (b) upgrade the quality and coverage of environmental services, thereby reducing the environmental degradation caused by poor collection and disposal of solid waste and sewage.
Scope and Spatial Limits	Metro area

Project No.	14
Country	Mexico
Type	I
Main Components	1. Water supply component. This would include 7 basic subprojects: (a) the El Cuchillo Dam on the San Juan River, to store up to 1.5 billion m^3 to regulate the stream flow, providing up to 10 m^3/sec, (b) the Cuchillo-Monterrey water supply system, (c) works to expand the capacity of the San Roque water treatment plant to 12 m^3/sec, (d) expansion and improvement of water supply systems, (e) installation of secondary systems in low-income neighborhoods and private urban developments, (f) pilot plan for modernization and automation of the water service system, and (g) reduction of losses and more efficient use of water. 2. Sanitation component. This would include three basic subprojects: (a) expansion of secondary intercepting sewers, main intercepting sewers, and wastewater outfalls; (b) 3 secondary wastewater treatment plants; and (c) construction of secondary networks and household connections in low-income neighborhoods and private urban developments. The direct investment cost of the 3 treatment plants is $68.3 million, and adding in the cost of secondary sewer networks in the city and the main and secondary intercepting sewers brings the total to $109.5 million in July 1990 prices. 3. Training component.
Objectives	The objectives are to (a) increase by 5 m^3/sec the supply of water to the metropolitan area of Monterrey, (b) integrate the sanitation cycle with sanitation works to achieve secondary treatment of wastewater at a rate of up to 8 m^3/sec, (c) use water more efficiently, and (d) establish a pilot program to modernize system operations.
Scope and Spatial Limits	The metropolitan area of Monterrey, Mexico's third largest city. Current potable water demand is 12 m^3/sec, but current supply is only 8.8 m^3/sec, leading to rationing. Wastewater is discharged through 11 main sewers without any treatment. Eighty-one percent of the discharge flow comes from the water supply service and 19 percent from industries having their own wells. Roughly 60 percent of the wastewater is subsequently used for irrigation.

Project No.	15
Country	Mexico
Type	I
Main Components	1. Water supply. 2. Sewerage and sewage treatment ($127 million of total cost); this component includes the expansion of sanitary sewerage service and the necessary sewage treatment for recently urbanized areas located in the so-called Blanco River watershed east of the city and the El Ahogado River watershed in the south. 3. Institutional strengthening. 4. Environmental sanitation and protection.
Objectives	To improve the sanitary conditions and standard of living of the population of the metropolitan Guadalajara area, through works and other activities for (a) rehabilitation and upgrading of the water supply system; (b) expansion of water supply, sewerage, and sewage treatment services to newly urbanized districts in the city; (c) institutional strengthening of the utility (SIAPA); and (d) environmental sanitation and protection.
Scope and Spatial Limits	Two basins (El Ahogado Creek and Blanco River) in the Guadalajara metro area. Targeted at low-income districts where considerable population growth is observed. First-time sewage from metro area will be treated.

Project No.	16
Country	Nicaragua
Type	I
Main Components	1. Rehabilitation and modernization of Managua's sanitary sewer system. 2. Environmental sanitation of the shores of the lake fronting Managua. Investments of this component are aimed at reducing the incidence of vector-borne diseases, especially malaria. 3. Program monitoring and environmental evaluation plan. The purpose of this component is to define and evaluate the parameters that affect the environmental health of Lake Managua, especially the processes causing eutrophication.
Objectives	Primary objective of the program is to help improve the environmental conditions and quality of life of residents of Managua. This is a long-term program to be carried out in stages. The specific objectives of the first stage are (a) cleanup and drainage of the strip of the lakeshore facing the city, (b) rehabilitation and modernization of the city's sanitary sewerage services, (c) education and community participation in campaigns to control diseases and vectors, and (d) development and execution of a plan to monitor the human and environmental health indicators of the lake and lakeshore area.
Scope and Spatial Limits	The southern subbasin of the Lake Managua catchment, including Lake Managua and surrounding metropolitan area (888,000 inhabitants), especially direct positive health impact on 120,000 people living within 2 km of the lake's shore (malaria, cholera). Selected and prioritized within wider basin context.

Project No.	17
Country	Uruguay
Type	I
Main Components	1. Western interceptor sewer to impound sewage from the Paraguay basin and convey it to an underwater outfall at Punta Carretas. 2. Sewerage service of 156 km for 32,200 residents in Chacarita and Canteras basins, avoiding pollution of Carrasco arroyo. 3. Separate 32 km sewerage for 13,800 persons in Chacarita basin.
Objectives	Carry on with the decontamination of the Montevideo shoreline begun in the prior stage, extending the benefit to the western area of Punta Carretas and supplementing the benefits of the first stage by removing nonpoint pollution affecting the estuary east of Punta Carretas.
Scope and Spatial Limits	Impoundment of all wastewater flows from the sanitary sewer system of the Paraguay basin that were discharged along the estuary coastline from Punta Carretas to Sarandi. This flow will be trapped with the project and conveyed by pumps to the existing underwater outfall at Punta Carretas. Also, through sewage collection, the project will decontaminate the Carrasco arroyo and the Chacarita and Nascientes de Chacarita basins and reduce coastal pollution east of Punta Carretas.

Project No.	18
Country	Uruguay
Type	I
Main Components	1. Environmental sanitation: (a) expansion of sanitary sewerage and storm drainage systems; (b) construction or expansion of sewer mains, interceptors, pumping stations, outfalls and pretreatment plants; (c) relocation of settlements located alongside the watercourses affected by the works or within hazardous areas; (d) development of programs to monitor industrial pollution and water quality in receiving bodies; and (e) rehabilitation of existing sewer systems and lines. 2. Institutional strengthening of the Montevideo municipal government (IMM). 3. Solid waste master plan.
Objectives	The objective of the program is to improve the living conditions for people in metropolitan Montevideo by increasing coverage of sewerage service and by reducing industrial and household pollution in the city's streams, in particular in the Pantanoso, Miguelete, and Carrasco basins.
Scope and Spatial Limits	Metropolitan Montevideo, particularly the city's streams in the Pantanoso, Miguelete, and Carrasco basins. The current organic load from industrial and household sources discharged into streams is 85 tons BOD/day and the quantity of heavy metals dumped into the streams and Montevideo Bay is 998 kg/day. The greater part of the sewerage network in Montevideo is combined, carrying both wastewater and stormwater. Forty percent of the sewage is disposed through an underwater marine outfall following pretreatment to eliminate sand, oil, and grease. In areas lacking sewerage infrastructure, household septic tanks are used, which must be pumped out periodically by vacuum trucks. The waste is then discharged directly into watercourses.

Project No.	19
Country	Argentina
Type	II
Main Components	1. Type "A" projects on operational improvements which fall into two categories: (a) technical assistance with subprojects such as institutional development, marketing, personnel training, and initial operational improvement, and (b) supplementary operational improvements, which would be for operation and maintenance equipment, laboratory equipment, etc. 2. Type "B" projects on rehabilitation. Under this component, resources from the program would be used to finance the purchase of equipment and materials and the execution of works to (a) repair or replace the various components of water-related systems that hamper their efficient operation, and (b) optimize existing facilities, especially water and sewage treatment plants. 3. Type "C" projects for expansion and execution of new works. Funds from the program would be used to finance the purchase of equipment and materials and execution of works to (a) expand the capacity of all or some of the components of the water supply system such as production, treatment, transmission and distribution; (b) expand the capacity of all or some of the components of the sewer systems such as catchment, conveyance, treatment, and final disposal, including recycling the water; and (c) execute new water supply and sewerage works for localities that do not have those services. 4. Type "D" projects for preparation of studies and designs. 5. Project "E" on institutional strengthening.
Objectives	The objectives of this global line of credit are to (a) strengthen the decentralized system in the provision of services by reinforcing the utilities that supply those services in an environment of operating efficiency and financial improvement, and (b) increase the coverage and quality of the services by executing rehabilitation works, and optimization and expansion works, paying special attention to remedying any environmental problems that such works might create.
Scope and Spatial Limits	Not fully known ex ante since program is multiple works. Only a general statement: "At present only 35 percent of the urban and concentrated rural population have sewer service." The report also states that "A total of 17.4 million inhabitants and 3,594 communities (577 urban and 3,017 rural) are without sewer service. Nearly all cities with populations of over 60,000 have service, though the coverage is low and service coverage needs to be improved and/or expanded."

Project No.	20
Country	Argentina
Type	II
Main Components	1. Consulting services to (a) conduct studies and designs in addition to those of the representative sample, (b) update national drinking water standards, and (c) conduct environmental control of projects and compile data for future program evaluations ($10,442,000). 2. Supervision of construction work, program execution, and contract administration by the executing agency ($4,790,000). 3. Multiple works that would include the construction or expansion of water supply systems, sanitary sewerage, or individual sanitation systems ($197,831,000). This would include the reconditioning, improvement, and construction of approximately 50 sewerage systems in various localities. In addition, sanitation works such as septic tanks, appropriate technology plants for treatment of the sewage and sludge from the tanks and latrines will be executed in about 30 localities. 4. Acquisition of land and easements ($244,000). 5. Technical support and training, including institutional strengthening of the utilities providing the services ($3,435,000). 6. Health and environmental education, including community outreach ($2,427,000).
Objectives	The basic objectives are to (a) enhance living conditions in towns with 500 to 15,000 inhabitants throughout the country, by executing works to increase water supply and sanitation services, and (b) support the utilities providing water supply and sanitation services so they can maintain and improve their administrative, financial, and operating mechanisms.
Scope and Spatial Limits	Not fully known ex ante since program is multiple works. General statements such as: "36 percent of the population had sewerage systems....This situation means that of a total of 32.2 million inhabitants, more than 20 million did not have sewer systems." Furthermore, the document also states that "in rural areas, in other words in localities with less than 2,000 inhabitants, 100,000 (2.4 percent) had sanitary sewerage systems." In reference to towns in the range of 500 to 15,000 inhabitants, through May 1991, of the 4.3 million inhabitants of the 2,893 towns in this range, 3,866,000 inhabitants had no access to a sanitary sewerage system, 1.9 million (49.2 percent) used septic tanks, 800,000 (20.7 percent) had pit latrines, and 1.2 million (30.1 percent) had no privy.

Project No.	21
Country	Bolivia
Type	II
Main Components	Multiple works for cities with over 5,000 inhabitants. 1. Works involving water collection and supply and sewage collection, treatment, and disposal. The sanitary sewerage works will include the construction of secondary networks, trunk sewers, intercepting sewers, pumping stations, outfalls, and treatment plants. 2. Technical cooperation to support the development of a regulatory framework for the sector and the establishment of a water board, which will be the sector's regulatory agency. 3. Institutional strengthening of companies delivering the services, through actions aimed at improving their administrative and operational efficiency.
Objectives	The general objective is to improve the hygiene and health status of the Bolivian urban population, through the expansion and improvement of water supply and sanitary sewerage services in cities with over 5,000 inhabitants.
Scope and Spatial Limits	Several cities, so need for and degree of treatment depends on specific circumstances which are not fully known at time of ex ante analysis. Analysis done for 4 cities (Santa Cruz, Cochabamba, Montero, and Riberalta), with emphasis on water supply and/or sewerage, and less on wastewater treatment. Program, when completed, could cover as many as 10 cities. Bolivia has one of the lowest rates of water supply and sewerage coverage in Latin America (in urban areas, 75 and 36 percent, respectively).

Project No.	22
Country	Brazil
Type	II
Main Components	This is a global multiple works program covering most of the states in Brazil; projects in approximately 30 urban districts can be accommodated by the loan and there is a "repressed demand" for many more. The proposed program includes construction of the following: (a) approximately 2,300 km of sewer collector systems with ceramic, polyvinyl chloride, and asbestos-cement pipelines 150 to 450 mm in diameter; (b) approximately 218,000 external household connections with sewer lines 100 mm—and in some cases 150 mm—in diameter linking the meters outside the houses to the collector system; (c) approximately 105 km of trunk collectors, in which most of the pipelines are made of reinforced concrete 300–1,200 mm in diameter; (d) approximately 100 pumping stations with the respective ductile cast iron pressure pipelines 50 to 700 mm in diameter; and (e) approximately 10 sewage treatment plants.
Objectives	The objective is to enhance the quality of life of the beneficiary population by (a) improving and restoring the sanitation and environmental conditions of broad urban sectors, mostly low-income people, which have deteriorated because wastewater is emptied on the ground, or in stormwater ditches for lack of a public sanitary sewer system, and (b) job creation, on an emergency basis, in order to employ current idle labor.
Scope and Spatial Limits	Multiple municipalities across the country. The program is justified by a review of the state of the country's water and sanitation sector. Historically, provision of potable water supply has taken precedence; while 86 percent of the country's urban population is connected to a water supply system, only 37 percent has sewerage service, and less than 10 percent of the wastewater collected receives any treatment prior to disposal. This has led to unsanitary conditions in urban areas (sewage in the streets); health risks; and severe pollution of rivers, streams, and coastal areas. Government's multiannual investment plan intends to raise sanitary sewer coverage to 60 percent and sewage treatment to 20 percent. The IDB-financed program will boost sewer coverage from 37 to 39 percent.

Project No.	23
Country	Colombia
Type	II
Main Components	1. Sanitation, including (a) construction of 23 wastewater treatment plants in 21 municipalities on the upper course of the Bogotá River, (b) construction of 9 systems to treat waste from slaughterhouses in 9 municipalities in the area under CAR's (Regional Corporation for the Bogotá, Ubaté and Suárez rivers) jurisdiction, and (c) construction of 25 infills for final disposal of solid waste in municipalities in the same jurisdiction. Direct cost of this component: $29,034,000. 2. Irrigation and management of swamps and lagoons. 3. Land reclamation and forestation. 4. Environmental management.
Objectives	(a) Raise the standard of living of the population, through environmental sanitation in the area. (b) Rehabilitate and maintain water quality in the Bogotá River up to standards set by CAR. (c) Increase farm production by expanding the irrigation infrastructure. (d) Rehabilitate and conserve natural resources, particularly in the Funza-Bojacá area. (e) Boost CAR's capacity to manage the area's natural resources.
Scope and Spatial Limits	High basin of the Bogota River, covering 175 km in length, including 18 municipalities with 350,000 inhabitants upstream of Bogota. Rationale is clear—municipal and industrial discharges (e.g., tanneries near the headwaters in Villapinzon, chemical industries in Zipaquira, 200 other industrial dischargers) causing serious declines in water quality, increased treatment costs for the Tibito potable water plant supplying 25 percent of Bogota's drinking water, contaminated irrigation water in the Bojaca/Herrera irrigation district, risks of diarrhea and other water-related gastrointestinal diseases. Also, irrigation component will improve the flow regime and ecological productivity of the Herrera Lagoon, an important wetland.

Project No.	24
Country	Guatemala
Type	II
Main Components	1. Physical infrastructure works, consisting of (a) water supply systems; (b) sanitary and storm sewers; (c) markets, open-air markets, and terminal markets; (d) community centers; (e) paving stone and cobblestone roads; and (f) bridges and footbridges. 2. Institutional strengthening of National Municipal Development Board (INFOM) and the participating municipalities. 3. Preparation of the studies and designs for the remaining projects in the program and for a representative sample of a subsequent stage of the municipal development program.
Objectives	The overall objective of the program is to improve the conditions of people living in cities and to provide institutional strengthening for INFOM and the municipalities participating in the program and to improve their financial condition. To achieve the objective, financing will be made available for investment projects that would expand the existing physical infrastructure in the municipalities and provide technical assistance for INFOM and the participating municipalities with a view to boosting their capacity to operate and maintain urban services.
Scope and Spatial Limits	Not fully known ex ante since program is multiple works. Only a general statement: "Only 54 percent of 330 municipal administrations have sewerage service. In most of these communities the systems provide coverage to less than 50 percent of the homes and effluent is not treated, creating serious hazards for the health of the general population."

Project No.	25
Country	Honduras
Type	II
Main Components	1. Integrated management of natural resources, including marine environment assessment and monitoring program, consolidation of existing marine protected areas and establishment of a new one, quality control of coastal waters, terrestrial ecological evaluation, watershed management, environmental education. 2. Environmental sanitation, including potable water supply systems, wastewater treatment (sewers and 4 treatment plants and small-scale septic tanks and dry latrines for isolated areas), solid waste. 3. Real estate census and property registry. 4. Institutional strengthening and development.
Objectives	The main objective is to maintain and improve the quality of the environment of the Bay Islands as a basis for sustainable economic development. The specific objectives are to (a) protect and restore natural resources and coastal and marine ecosystems by establishing a system of integrated management; (b) develop and institutionally strengthen local capacity for planning, managing, and administering economic utilization of natural resources; (c) improve the conditions and quality of life of the inhabitants of the islands through improvements in water supply, introduction of basic sanitation, and maintenance of marine ecosystems; and (d) establish mechanisms of cost recovery for public investment in the environmental sector and generate the resources and financial income needed to support sustainable development on the islands.
Scope and Spatial Limits	Bay Islands archipelago 46 km off Honduras' northern coast. Project designed to maintain the quality of the island environment, permitting the sustainable use of natural resources important for ecotourism (coral reefs, marine ecosystems), rationalize the use of freshwater, improve wastewater treatment and solid waste disposal.

Project No.	26
Country	Mexico
Type	II
Main Components	The program will provide partial financing for investments under the National Water Supply and Sewerage Program through a credit line for infrastructure facilities and consolidation of the utilities. Among the subcomponents are (a) consolidation of municipal, intermunicipal, and state utilities by strengthening these companies and carrying out infrastructure works; (b) rehabilitation, upgrading, and expansion of water supply and sewage systems; and (c) a sanitation subcomponent, in which sewerage treatment plants would be built in priority areas.
Objectives	The main objectives of this operation are to support the implementation of a program of reforms aimed at developing the sector institutionally, improving the quality and coverage of water supply and sewerage services, promoting a proper rate and cost-recovery policy, reducing the federal government's participation in the financing of investments in the sector, and improving environmental conditions in Mexico.
Scope and Spatial Limits	Not fully known ex ante since program is global multiple works. Only general statement that "Of the few treatment plants in Mexico (392), only 50 percent of the municipal plants are in operation and these are only treating 10 percent of the volume of wastewater produced." Program to give "special consideration to comprehensive sanitation in urban areas."

Project No.	27
Country	Paraguay
Type	II
Main Components	1. Consulting services for the development of a legal and institutional framework for Paraguay's water and sanitation sector, preparation of feasibility studies and final designs for additional projects under the components for works in cities in the interior, program management, and institutional strengthening of CORPOSANA (Sanitation Works Authority). 2. Specific works in the metropolitan area of Asunción to include (a) increased raw-water pumping capacity, (b) conveyance lines for the main system, (c) regulating tanks and equipment for the distribution centers, (d) secondary distribution system and household connections, and (e) establishment of pitometric districts. 3. Multiple works in cities in the interior, including expansion or installation of water supply systems and expansion or installation of sewage collection, treatment, and final disposal systems.
Objectives	The objectives of the proposed program are to (a) support the establishment of a new legal and institutional framework for the sanitation sector; (b) improve the technical, financial, administrative, and business efficiency of CORPOSANA; and (c) improve the living conditions of residents of the metropolitan area of Asunción and cities in Paraguay's interior by increasing water supply and sewerage system coverage.
Scope and Spatial Limits	Not fully known ex ante since program is multiple works. Only a general statement: "Coverage of the existing sanitary sewerage system in Asunción is estimated at 40%. Sewage is discharged into the Paraguay River downstream from the uptake point, after pretreatment through screen chambers to remove thick solids. The low-water volume of the river is on the order of 1,000 cubic meters per second, which allows the dumped sewage to dissolve sufficiently, without compromising future uses of the resource." The report also states that only 4 of 34 towns in the interior of Paraguay have sewerage systems covering only 3.3 percent of the population in these towns. Overall, in Paraguay, 3.7 million people have no access to sanitary sewerage systems.

Annex 2-D

Comments on Project Information

Project 1: This is one of the few projects reviewed that presents sufficient information in the loan documents on the baseline water quality situation and the natural world impact of the project. It provides AEQ model simulation results (BOD, DO) with and without the project. In comparison, the economic analysis is very sketchy for the treatment plant component. No details are given on percentage removal by pollutant at treatment plants. Most of the economic analysis effort was spent on cost-benefit analysis of the largest component, flood control ($116 million investment), as opposed to sewage treatment (about $20 million).

Project 2: Overall, this document supplies a minimum of technical and analytical detail. The economic analysis effort was devoted to storm drainage, flood control, and urban rehabilitation, which do not figure in the assessment rating. It could be argued (as the document obliquely does) that an economic analysis of the achievement of alternative levels of water quality across space and time, and the alternatives for control by source, lies beyond the scope of this program, which only involves technical assistance to industry and hauling junked cars and boats out of the river. However, no existing source is cited in the loan documents that provides justification for the degree and timing of pollution reductions taken as given by the program, and in principle some analysis probably should have been done, or at least financed through the program.

Project 3: No analysis of natural systems solidly linking the project with AEQ outcomes in near-shore water is reported in the loan document, yet passing mention is made in the environmental summary, which provides little additional information. In both documents mention is made of such an analysis at the ocean outfall. The need for natural systems models of near-shore AEQ is not strong under the implicit assumption that connecting all wastewater dischargers in the coastal zone to a sewer system and discharging the wastewater far offshore after primary treatment would resolve the coastal pollution problem. However, the effects of nonpoint pollution are ignored, and no attempt appears to have been made to target primary contributors to the problem via a source discharge inventory. The economic analysis was hampered by the lack of hard evidence and resorted to subjective probability distributions for key variables in a risk analysis instead. There may be some double counting of benefits, since the preamble to the contingent valuation survey question explicitly stated a public sewerage system would "keep beach water clean for swimming and protect coral reefs" and noted that "contaminated or polluted beach water might also discourage tourism to our island." The WTP responses may therefore contain private valuations of the costs of private septic tank O&M avoided, health damage avoided, and loss of income from tourism, categories which were also monetized separately using health costs avoided and "what if/how much" sensitivity techniques. While the project document does not mention this issue, a paper by Darling et al. explains the difference in WTP between households in the zone of project activity ($178) and those outside of it ($11) on that basis: "The larger mean benefit derived for catchment households captured both the private and public benefit while the smaller mean benefits derived for the outside catchment households measured only environmental or public benefits." (Darling et al., 1993, p. 107) In sum, the analysis is creative, but lacks a strong empirical foundation (which the project document makes clear) and probably involves double counting of benefits (which is not made clear).

Project 4: This is primarily a sewerage project whose components can be justified exclusively on the basis of local or neighborhood benefits. The additional environmental quality benefits that presumably accrue via collection and disposal elsewhere (i.e., pretreatment plus deep ocean outfall) are not calculated. Given the character of the program, the economic analysis is narrow but adequate for the purpose, but it sheds no light on the magnitude of total benefits, or the real extent of AEQ improvement the project will provide. The environmental summary does not provide much information beyond what the loan documents have.

Project 5: This project presents a very difficult economic analysis problem: Is it worth only partially cleaning up a river that for all intents and purposes is an aboveground extension of the city's sewerage system? Rather than addressing that question, the analysis solves the more restricted one of identifying the regional least-cost configuration of treatment plants that can be undertaken. Most of the information required to understand the goals and the natural world impact of the project is contained in the loan documents; reference to the environmental summary to get additional key information is not necessary.

Project 6: The natural world impacts of the project can be understood from loan documents without reference to the environmental summary. There is also a fairly complete and well-explained economic analysis. However, no presentation is made of individual and family swimming benefits by location, which instead are subsumed in the aggregate benefit calculation for each combination of plants and locations presented in the annexes. It is impossible to reconstitute this because benefits are scaled down by congestion factors that are not reported by subproject. Tourism and fishery benefits are not firmly grounded in actual data, but confected by assumption. Of the projects reviewed, this is one of the few to integrate natural systems models (albeit simplified, static, and restricted to BOD, DO, and coliforms) with alternative treatment activities to find the degree of pollution control and ambient improvement that could be reached with alternative levels of financial expenditure. A subsequent World Bank analysis using a more recent water quality model of the bay produced in 1994 by the Japan International Cooperation Agency (JICA) highlights the problem of eutrophication in the bay and the need to control phosphorus by including chemical precipitation in primary treatment, which was not considered in the IDB project.

Project 7: Relative to other documents reviewed, this one stands out as being among the very best. It is outstanding in the completeness of the technical and economic information provided and the lucidity of its interpretation and explanation of both. It has good water quality simulation information and a good economic analysis of separate components and the program as a whole. The document also demonstrates the effect of variance in estimated benefits on the project acceptance decision.

Project 8: The description of the least-cost analysis of an overall wastewater treatment program provides no details, especially if or how ambient quality targets were specified over time and space, and whether an optimization or simulation approach was used. Nor is there any discussion of how the CV question was worded to value a partial improvement. Also, no explanation is provided for why benefits to industry (over and above the CV response) were estimated as equal to the marginal water intake treatment cost of 38 pesos/m^3. The environmental summary does not add much information to that in the loan document.

Project 9: The confusion this loan document induces led us to make an exception and inspect the background economic feasibility study. That document did little to dispel the confusion. The AEQ effect of the project is not identified in the loan proposal or the economic feasibility study, making it difficult to match what is being valued with what will be actually achieved. The economic analysis contains several questionable benefit categories, in particular, multiplier benefits from project costs, which customarily are not allowed in IDB project analysis (see Chapter 4). The cost streams are not phased as real resources that will actually be invested, but instead are based on the time pattern of financial payments by the Bogota district environmental agency (Departmento Administrativo del Medio Ambiente) to the private sponsors for treating a given annual flow. This has the effect of smoothing the timing of costs in the sense that although resources will be invested up front for construction, zero costs are shown for years 1–3 in the economic cost-benefit analysis. This too is at odds with Bank practice, which takes a social stance, not the financial stance of any particular public entity, when doing cost-benefit analysis. The size of the beneficiary population is extremely large in the base case cost-benefit results, which impute benefits to almost the entire city's current population.

Project 10: These loan and project documents are, from a technical and analytical standpoint, among the most unclear and superficial (though apparently overoptimistic) of all those reviewed. The technical information presented about the design and natural world impacts of this project is just short of vestigial, giving the impression that such information is either nonexistent, unreliable, or counter to the sanguine but vague claims of favorable impacts made in the project loan proposal. Only by reference to the separate ESIR can the environmental improvements occasioned by the project be understood. The economic analysis section of the document is telegraphic and compressed, does not report NPV, seems to assume that the project benefits would persevere in the face of population growth and increasing waste loads without any additional investments beyond those initially made, and only takes a casual swipe at sensitivity.

The striking paucity of technical information about the project and its impact spurred additional inquiries, despite the express intention to limit this review to official project documents rather than a full search of the analysis files, feasibility studies, etc. This inquiry revealed that the negative impression the document creates about the technical and economic integrity of the project appears to be false. In fact, the bay mathematical modeling exercise, barely mentioned in the official loan documents, but explained in the ESIR, generated a wealth of information about the impact of the project (coliforms, BOD, and DO at the surface, middepth, and bottom; by dry and wet season; with and without the project; and with and without chlorine pretreatment in the short and midterms). This analysis appears to solidly establish that the proposed collection and deepwater disposal scheme would indeed improve near-shore quality to levels suitable for direct water contact. Moreover, interviews with project team members revealed that the economic analysis took a careful look at the incremental benefits of achieving near-shore recreational use alone or the benefits of a more ambitious restoration of the entire bay, and found the latter to be lower on average and insignificantly different from the former. None of this is reported in the loan documents. In short, the background information and the ESIR solidly establish the technical and economic feasibility of the project, but the official loan documents do not.

Project 11: Given the several components of the project, the economic analysis burden was not trivial. More effort appears devoted to justifying the potable water and sewer connection components than evaluating the water quality impacts and the advisability of a wastewater treatment plant. The principal deficiency of the latter was the notion that the hedonic analysis captured all of the water quality benefits of the interceptor and wastewater treatment plant. No attempt was made to assess downstream damage to aquatic life, health, recreation, and irrigated agriculture, or the higher downstream costs of treating intake water if the treatment plant were not built. Nor was there any attempt to clearly present "with/without treatment plant" comparisons of dissolved oxygen and coliform profiles for the Cuenca River by reach, year, and season, or to compare those results with any water quality standards, Ecuadorian or otherwise. This situation generated some disagreement among Bank economists on the advisability of postponing construction of the plant. For example, during the comment period, the Operations Evaluation Office observed (CON/OEO-137/90, May 2, 1990), "There probably are several compelling non-economic arguments that could be brought to bear to justify postponement of the wastewater treatment plant, especially if such a plant could not be properly operated anyway due to technological complexity. However, the document emphasized economic (mainly cost/benefit) arguments. Such arguments are best used as supportive evidence because of the well-known difficulties in precisely quantifying the stream of benefits associated with pollution reduction. In particular, the hedonic method is notoriously unreliable. Thus we feel that a strong non-economic argument case needs to be made for postponement."

Project 12: The economic aspects are barely mentioned in the main document. The explanation of the analysis of the sewerage projects identified in the sample is sketchy. Furthermore, the initial WTP results were 13 percent of average income. This amount was discarded; instead, 8 percent of income was used. The only criterion used to choose the 8 percent cutoff rate was that the maximum WTP required for projects of the sample to have an IRR of 12 percent was 21,000 sucres/month (approximately 8 percent of income). A master plan will be financed by the project and therefore the mathematical water quality simulation model and the definition of the level of municipal treatment and industrial pretreatment required will be determined during execution; hence, it will not apply to the projects in the representative sample.

Project 13: Hardly any technical or economic analysis details are supplied for this project for NPV and they are presented in a table that appears to have some typographical errors (i.e., a negative NPV is associated with a reported IRR above 12 percent, which is impossible). There is no ambient quality, discharger, or treatment plant performance information at all. Plants are just said to be "primary plus secondary."

Project 14: The project economic analysis employs the logic that wastewater treatment costs are a lower bound proxy for the unpriced environmental damage imposed by expansion of potable water supply services (i.e., treatment costs plus residual damage are less than or equal to total damage of uncontrolled, untreated discharge), and hence must be offset by willingness to pay for potable water in the cost-benefit analysis. The economic analysis does not try to determine the optimal level of treatment by exploring the comparative levels of discounted benefit (damage avoided) and cost across alternative AEQ goals. Rather, it takes mandated secondary treatment as a given and performs least-cost design analysis of technical alternatives within that general category without exploring the AEQ impacts.

Project 15: The economic analysis explains how the project design was selected by a comparison of alternatives, which is uncommon in the documents reviewed. No reference is made to optimization in selecting the least-cost number, location, or timing of plant construction, but at least an ad hoc attempt was made. There is no reference to AEQ models, so no tight link appears to exist between treatment equipment removal rates, AEQ, and benefits. Neither are there details on baseline AEQ or treatment equipment characteristics. Most project design information appears in the economic analysis section—little is said in the document's technical feasibility section or annexes. The environmental summary provides little in the way of additional information on the physical setting, baseline, and project impact.

Project 16: The document is candid about information deficiencies (water quality models, lack of epidemiological data to conduct a rigorous health risk/health benefits assessment) and recognizes the need to provide improved ambient environmental quality information and analysis to evaluate future investments in subsequent stages. The report discusses the performance characteristics of treatment equipment options, least-cost analysis and ambient outcomes, but not in great detail. There is no sensitivity analysis.

Project 17: This project involved the Bank's first application of referendum contingent valuation, and the documentation is excellent overall. The interceptor benefits were relatively low because only 51 percent of interviewees were willing to pay at the lowest bid, $0.50 per month.

Project 18: The document contains some baseline information, but it is not connected clearly to the economic analysis. The final WTP elucidated in the document is sketchy and so is the economic analysis. Several studies to design a mathematical water quality model and to do alternative analyses for a sewerage connection will be financed as part of this program, but not ex ante.

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Chapter 3

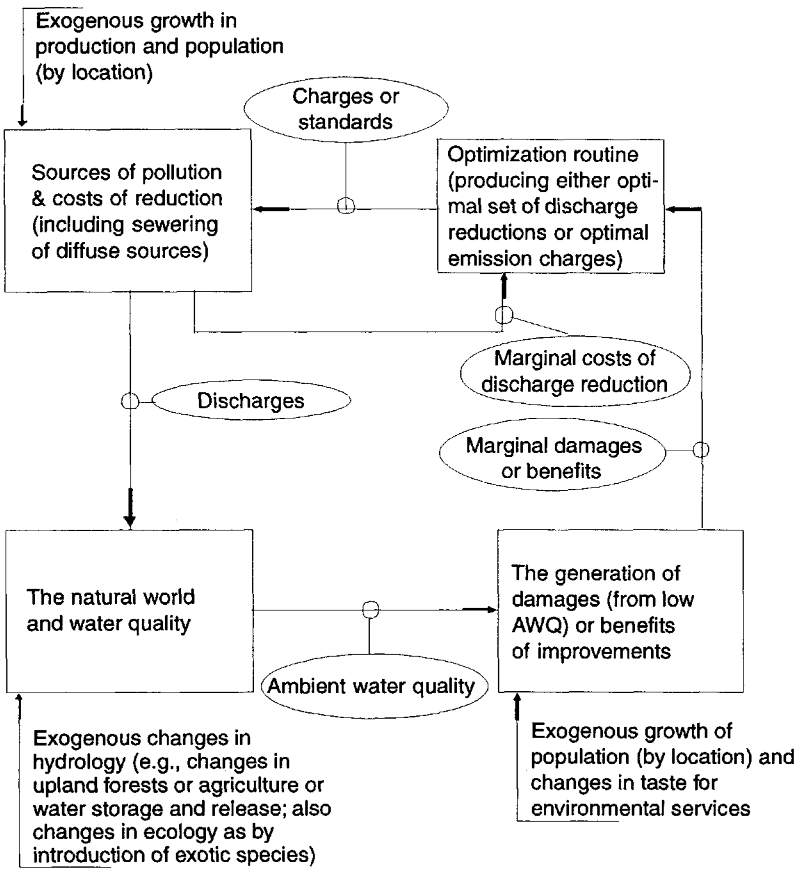
Cost-Benefit Analysis of Water Quality Improvement Projects in More Detail

In Chapter 1, several levels of possible application for cost-benefit analysis relevant to decisions about ambient water quality improvement projects were very briefly noted. This chapter extends the discussion of these alternatives through schematics designed to make clear their differences and include more formal versions of their structures. These latter can easily become quite bewildering, in particular because benefits arising from changes in water quality are in part enjoyed as "use" benefits at particular places (beaches, points at which water is withdrawn for treatment and delivery to households and industry) and in quite specific ways (swimming, fishing, boating, drinking). Each use in turn is usually sensitive to different elements of the group of measures (such as dissolved oxygen or bacterial contamination) that make up the overall description of ambient water quality. This means that keeping track of place and of use-related quality elements puts heavy demand on the "bookkeeping" (subscripting) side of model construction, and thus on the reader's patience. Beyond that, it can be difficult to pick up what might informally be called the "feel" of these models when one is drowning in *i*s, *j*s, and *k*s. With this in mind, an effort is made in this chapter to steer a middle road between the extreme simplicity of standard cost-benefit textbooks and the formalism often found in the most advanced treatments of the subject (e.g., Sasikumar and Mujumdar, 1998, are quite extreme in this regard, Loucks et al., 1981, much less so). Any particular reader may wish for either more or less detail in the mathematical structure. Some may even hope to find a true manual, complete with generalized benefit functions that can be adapted to specific problems by the estimation of one or two parameters. To these optimists it is necessary, unfortunately, to say that the state of the art is not there yet. While there do exist off-the-shelf ambient water quality models (of which more in Chapter 5) and catalogs of wastewater treatment cost functions (again, some numbers are to be found in Chapter 5), there is no simple route to the benefit side of such a model. In the end, at least at this point in the history of the development of these techniques, each basin and even each project model will have to be constructed for the specific application and will require considerable local data and estimation work. That warning or caution having been delivered, it will still be useful to look more closely at what a full cost-benefit analysis structure might look like.

REGIONAL OR BASINWIDE OPTIMIZATION

A schematic model of the basinwide optimization problem is shown in Figure 3-1. In it, the linked boxes indicate submodels, initial conditions, and choice options for a model whose objective is the maximization of the net present value of benefits from investments to improve

Figure 3-1. Schematic of a Regional or Watershed Optimizing Model



water quality over a basin. Models of pollutant generation, discharge, and discharge reduction indexed by source and location have an effect on AEQ, which is transmitted through or mimicked by models of the natural world and water quality, indexed by, say, river reach.[1] These physical effects are monetized by economic models of benefits (or damage reduction). At the optimum, equalization of the marginal cost of discharge reduction with resulting marginal benefits for every source and location will produce the solution for the optimal level of AEQ and the optimal way to achieve it via investment in sewering, treatment plant types, and their locations.[2] The ovals in the diagram indicate the flows of information among submodels.

Now, it is hard to argue with the proposition that especially where capital and institutional capacity are limited, working toward regional (river basin or coastal "problemshed") optimization in water and wastewater planning makes sense. The principal reason is easy to see and understand: Constraining the set of possible choices—for example, by concentrating on possible municipal waste treatment plants and ignoring industry and agriculture—more or less guarantees that more cost will be incurred than is necessary to produce the ambient quality improvements projected. An added advantage for both borrowers and the Bank would be that

[1] "Reach" refers in general to a length of river or stream—a complex bathtub that includes the full width and depth of the river at whatever volume of flow is appropriate to the modeling task.

[2] In general, reducing any single discharge (pollutant type and source location) will produce benefits at many other locations and via several routes. Thus, it is the sum of all specific marginal benefits that is to be equated to the marginal cost. This is what the programming algorithm will have been designed to achieve.

a single review of the analysis that produced the optimal basin plan would in effect provide precertification of all the individual projects subsumed in it, subject perhaps to regular updating of the assumptions (such as growth rates and changes in industry structure).

While it may be the most conceptually correct way to represent the problem, the case could be made that basinwide optimization is not a practical alternative at the moment because it is too demanding of information and modeling skill, and especially of benefit estimation technique. Certainly the project loan analyses reviewed in Chapter 2 do not generally reflect such efforts.[3] In addition, noneconomists may object to an emphasis on efficiency, even though that is an IDB requirement (see Chapter 1).[4]

A More Formal Model

The schematic of Figure 3-1 does not really convey the complexity of a regional or basinwide optimization model because it buries so much in the boxes. Without going so far as to work through a full model for a real situation, it is still possible to expand on the schematic. This exercise will make explicit the information that is required for even a fairly simple, static, and deterministic model. That information will be seen to fall into four categories: discharge quantities of individual pollutants, costs of reducing discharges, fates of discharges in the environment and their effects on measures of ambient quality of interest to the relevant population, and valuation of changes in ambient water quality by that population.[5] The whole enterprise is made more complex by the importance of space; the effects of discharges and hence of changes in discharges vary across the basin, depending on source locations and characteristics of the receiving water. At the same time, the human uses of the basin occur at particular places, sometimes for natural reasons, as where sand exists for a beach, sometimes because other humans have created a desirable setting, such as a park with picnic tables and boat-launching facilities. So the model cannot deal with some sort of basinwide average water quality, but must reflect spatially specific vectors of quality.[6] This requirement reveals yet another complication—that the valued activities sensitive to water quality are differentially sensitive to different elements of a reasonably complete quality description. Thus, swimming will only be possible

[3] One or two projects are embedded in the regional context in less formal ways. Three, however, stand out as including substantial, if by no means complete, information on the basin context. These are AR0138 (Environmental Sanitation and Flood Control in the Reconquista River Basin), BR0072 (Basic Sanitation Program for the Guanabara Bay Basin-Phase I), and C00227 (Cartagena Sewer System).

[4] Because environmental decisions involve public goods (reduce public bads), they are inescapably political, which is to say that distribution is at least as important to the decision makers as efficiency. So there is often more interest in seeing that politically desirable (or at least acceptable) designs are justified than in identifying projects that win the efficiency contest. Indeed, in the United States, as discussed in Annex 1-A to Chapter 1, no effort was ever made to choose economically efficient policies for either air or water pollution control, and even the more limited goal of controlling water pollution in an enforceable way was compromised by a reluctance to deal with the nonpoint source pollution from agriculture. It is only since the 1990s, and only in limited settings such as the Eastern Shore of Chesapeake Bay, that politicians seem to be summoning the courage to force real control efforts onto farmers.

[5] It is a good deal harder to identify the "relevant population" than it is to say it must be done, and this will be discussed again in Chapters 6 and 7. For now, just note that there can be two bases for an answer. One is: Who cares about the water quality? The people who care may live either in or out of the basin and they may or may not care because they actually use the river. A second basis is: What population is the government unit contemplating the quality improvement willing to include? The answer to this will generally be based on citizenship, not on caring. If basin and government units don't coincide, the citizenship requirement can be very awkward.

[6] In ambient water quality modeling, "spatially specific" usually means reaches of a river or sections of a lake or bay that are modeled as perfectly mixed "bathtubs," so that at each point in the reach, water quality is assumed to be the same while quality varies from reach to reach. Clearly, how much inaccuracy the perfect-mixing assumption introduces depends to some extent on how large the reaches are taken to be. The longer the reaches, the poorer the approximation, all things being equal, but the smaller the computational problem.

(safely) where fecal coliform bacteria counts (as a proxy for disease-causing potential) are very low, water clarity is above some minimal level, and there are no toxic chemicals that might be swallowed or harm eyes or skin.

As already noted, all these parts of an optimizing model can be written formally in quite general notation, with (N) discharges of (M) pollutants, measures of ambient water quality covering (A) elements and predicted in (R) reaches, with (K) benefit routes covered at those locations. This can be "economical" in the sense of saying a lot while using only a little paper and ink, but for many people it lacks something as a way of communicating. So the model discussed here will be less general and more informal. It describes a hypothetical, small, and quite uncomplicated basin with only a few polluters, reaches, measures of ambient water quality, and routes by which the benefits of improved quality accrue to the relevant population. It will be assumed that the WTP of that population is for changes in quality in the reaches.

Rather than trying to include uncertainty and dynamics in all their complex glory, the first will be handled as it often is by using a low flow that is quite unlikely, thus ensuring that the predicted or a better quality will apply much of the time. (Notice though that this element of uncertainty will in general be relevant to the WTP for the predicted quality. That is, WTP can be presumed to be greater for the same in-stream conditions if those conditions will be equaled or bettered 98 percent of the time than if they are equaled or bettered only 90 percent of the time.) The second element, dynamics, will just be assumed away, and time will implicitly enter the model in only two ways: First it will pass, so discounting will be necessary for a correct multiperiod analysis and second, population and incomes will change (but outside the control of the interested government unit "exogenously"). (See Chapter 4 for a further discussion of the dynamic elements of such problems.)

Further, in the interests of simplicity the basin is confined to six reaches of a river, the five downstream reaches each receiving pollution from one discharger. Five pollutants will be allowed for: biochemical oxygen-demanding organics (measured by the oxygen-using potential, BOD), nitrogen (N), phosphorus (P), suspended solids (SS), and fecal coliform bacteria as a proxy for other pathogens (CO). The ecological model of each reach will accept these pollutants as inputs along with background information, such as flow, temperature, and the quality of the water entering the reach from upstream. These models will predict six measures of ambient water quality (AWQ): dissolved oxygen (DO), turbidity (T), algal densities ($A\ell$), the "concentrations" of two sorts of fish, rough (U) and game (G), and the concentrations of fecal coliform bacteria.[7]

User benefits will be associated with three "routes": aesthetics (A), health (H), and recreation (REC), and there will be a separate user benefit per year for each reach and each route. These will be assumed to have been estimated on a per user basis, and the number of users by type and reach will also be assumed as known.[8] Nonuser benefits per "interested" person will depend on all the quality measures in all the reaches. This population of "interested people" will in general include some (but probably not all) of the users, plus others. Again, any discussion of the underlying problem of determining the size of this population (the "extent" of the nonuser "market") is postponed to the fuller discussions of benefits and estimation methods in Chapters 4, 6, 7, and 8.[9]

---

[7] "Rough" refers to the robustness of the species involved in the face of low water quality and the stresses it imposes. In general, the less robust, usually carnivorous, game fish, which are more susceptible to poor water quality, are more highly valued by recreational anglers.

[8] The major difficulties associated with allowing for the effects of other "competing" rivers and the shifting of use patterns in response to changes in quality will be ignored here.

[9] What leads to nonuse valuations is another subject with its own substantial literature. An example might be a general concern with ecological integrity, or the naturalness and resilience of the aquatic ecosystem. There is reason to think that nonuse benefits will be relatively less important in developing than in industrialized nations, certainly when the willingness to pay of only the nation's citizens is to be counted.

Even with all this simplification, there will be a formidable amount of notation, although the choices made above may serve to keep the expressions intuitively meaningful. The result is shown as expression (3-1).

$$\max[\,[P_{1R}\,B_{1REC}\,(DO_1, T_1, A\ell_1, G_{11}, G_{21}, CO_1, Y_{P1R}) + P_{1H}\,B_{1H}\,(A\ell_1, CO_1, Y_{PIH}) + P_{1A}\,B_{1A}\,(DO_1, T_1, A\ell_1, Y_{P1A}) + P_{2R}\,B_{2REC}(DO_2, T_2, A\ell_2\,, G_{12}, G_{22}, CO_2, Y_{P2R}) + \ldots + P_{5A}\,B_{5A}\,(D_5, T_5, A\ell_5, Y_{P5A}) + P_{Nu}\,B_{Nu}\,(DO_1 \ldots CO_5, Y_{PNU})] - [C_1(R_{1BOD}, R_{1N}, R_{1P}, R_{1SS}, R_{1CO}) + C_2(R_{2BOD}, R_{2N}, R_{2P}, R_{2SS}, R_{2CO}) + \ldots + C_5(R_{5BOD}, R_{5N}, R_{5P}, R_{5SS}, R_{SCO})]\,] \qquad (3\text{-}1)$$

The P_{ij} are populations relevant to benefit route j in reach i. The per capita benefit functions are intended to be fairly easily understood, with subscripts for each reach and route, which applies to the entire river (NU = nonuse). The arguments of the use benefits are the relevant quality characteristics of the relevant reach, O, T, $A\ell$, and so forth. The income arguments (Y_{Pik}) are, ideally, the incomes for the relevant populations (at place i, for use k). In these functions, again for simplicity, the "before" levels of quality are not included, but recall that benefits are a function of improvements in quality from some base.

The cost functions for pollution removal (R_{ij}) by the sources are shown as entirely non-separable. That is, mathematically, the marginal cost of, say, BOD removal at source 1, R_{1BOD}, depends in general on the level of removal of all the other pollutants by that source. The relationships inherent in the actual cost functions may very well create nonconvexities in the space of feasible solutions and thus lead to practical difficulties with optimization. The actual situation as defined by engineering reality is likely to be more even complex and harder to deal with because treatment technology has evolved to be used in stages, with joint removal of a subset of pollutants at each stage. Thus, in a rough schematic, the choices are as in Figure 3-2. For a finer breakdown within each stage, see the example in Chapter 5.[10]

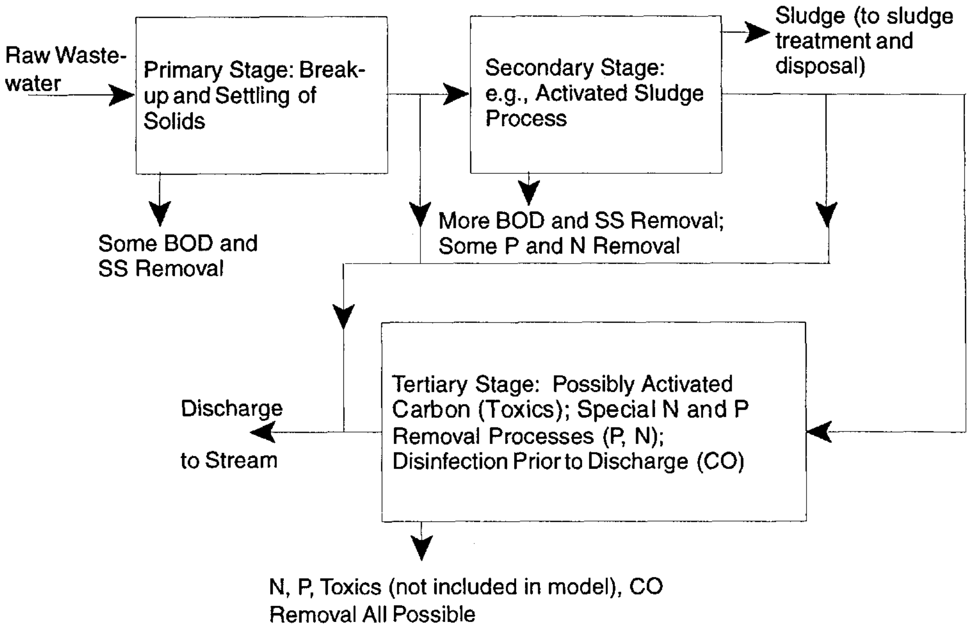
The model's choices involve whether to treat at each stage and how much treatment should be done at each stage. However, to carry out any part of the secondary stage, some amount of primary stage capacity must be in place, and so forth. Further, within each process at each stage there is generally joint removal of more than one pollutant. This implies that the marginal cost of BOD removal, for example, is not well defined, even though the conceptual model uses cost functions that imply a well-defined partial derivative.

The link between the pollutant reductions and the benefits is the model of ambient water quality, along with some auxiliary relationships. The latter include the identities $D_{ij} = P_{oij} - R_{ij}$, for each source i and pollutant j; with discharge, D; reduction, R; base or raw load, P_0; and the river flow identities:

Given flows from the tributaries not being modeled (F_0)
Given inflows to each reach I_k
Given outflows from each reach L_k
The net change in flow in the reach is $\Delta F_k = I_k - L_k$
For each reach going downstream, the flow is $F_k = F_{k-1} + \Delta F_k$ (3-2)

[10] This is the standard treatment sequence, as presented, for example, in Tchobanoglous (1996). In situations with cheap land, ponds may be the secondary (biological) treatment setting, but primary treatment to remove (or break up) solids seems to be considered a necessary precursor. "Natural treatment systems" may involve spreading sludge on vegetated land, the use of a natural slope to carry water over such land, application of wastewater to infiltration basins, or filtering through natural or artificial wetlands. All attempt to make use of the natural processes in an ecosystem to take pollutants out of the water body that is being protected. All of these methods must also be preceded by some version of mechanical pretreatment for gross solids that could clog up the system and lead to odor nuisance.

Figure 3-2. Wastewater Treatment Stages



So the flow is measured at the downstream end of the reach. The next reach down receives that flow and in the reach, net inflows are added to get the flow at the bottom of that reach, and so on.

The river water quality model itself may be very complicated, as discussed in Chapter 5, but the essence can be conveyed by the following: Given DO_0, N_0, P_0, SS_0, CO_0 in the upstream reach (as well as F_0 and all the net inflows, as above):

$$
\begin{aligned}
&DO_1 = Z_{DO}(DO_0, N_0, P_0, SS_0; F_0, \Delta F_1; D_{1BOD}, D_{1N}, D_{1P}, D_{1SS})\\
&T_1 = Z_T\,(DO_0, N_0, P_0, SS_0; F_0, \Delta F_1; D_{1BOD}, D_{1N}, D_{1P}, D_{1SS})\\
&A\ell = Z_{A\ell}\,(DO_0, N_0, P_0, SS_0; F_0, \Delta F_1; D_{1BOD}, D_{1N}, D_{1P}, D_{1SS})\\
&U_1 = Z_{G1}\,(DO_1, A\ell_1, T_1; \text{Temp}_1)\\
&G_1 = Z_{G2}\,(DO_1, A\ell_1, T_1; \text{Temp}_1)\\
&CO_1 = Z_{CO}\,(CO_0, D_{1CO})\\
&\vdots\\
&DO_2 = Z_{DO}(DO_1, T_1, A\ell_1, BOD_1, N_1, P_1, G_{11}, G_{21}, F_1, \Delta F_2; D_{2BOD}, D_{2N}, D_{2P}, D_{2SS})\\
&\vdots\\
&CO_5 = Z_{CO}\,(CO_1, D_{2CO}), \text{ etc.}
\end{aligned}
\tag{3-3}
$$

through six equations for each of the five reaches to be modeled (which are influenced by the discharges whose control levels are the decision variables). That is, for each stream characteristic (output of the water quality model) in each reach, there is a prediction equation, which, pathogens excepted, depends on all the inputs of pollution to the reach and the inputs of at least some of the characteristics and pollutants from the reach above. Pathogens are dealt with separately, as though they do not participate in the ecological relationships that transform the other discharges into stream characteristics. (This seems to be consistent with the way available, off-the-shelf, stream-quality models deal with them.) The bacterial concentrations decay as the bacteria die off in the open water, exposed as they are to sunlight, unfamiliar chemicals, and other organisms that eat them.

While this is not intended to be a manual on how to build and use aquatic ecological models, a few additional remarks are in order. First, in current reality, models with even this slight degree of complexity are not usually available for specific basins in Latin America. (For more on what is generally and publicly available, see Chapter 5.) What is most often modeled is just the BOD/DO connection, with dissolved oxygen standing for water quality in a broader sense. This in turn implies that the benefits of water quality improvement cannot be dealt with in an entirely satisfactory way, since dissolved oxygen is only one of the characteristics that influences people's enjoyment of natural water bodies—although at low levels it is key to the problem of bad odors—and does not even figure in the health benefits unless it is taken to be a surrogate for pathogens. For an effort to make the most out of a limited amount of information, including dissolved oxygen, by translating it into an index number that attempts to capture use suitability, see the description of the RFF Water Quality Ladder in Annex 7-B to Chapter 7.

A second note is that the equations written above do not have intrareach simultaneity reflected in them. In reality, the DO in a reach will, for example, be influenced by the level of algal growth (and death). This will have its greatest effect in widening the daily swings in DO. Growing algae will add to the oxygen content; dead, decaying algae will use up DO. Sufficiently dense algae can produce shade that shuts down other sun-driven process in the water column. (See Chapter 5 for more on such interactions.)

Also lacking in the above discussion is any explicit notice of time. Since long-lived investments producing multiperiod returns are involved, time cannot be ignored. Bringing it in, even in the simplest sense of passing time and changing populations and incomes requires first, differentiating investment and operating costs; second, repeating the net benefit expressions for each year over the "horizon" (the period agreed on as relevant to the decision)[11] ; and third, doing the necessary discounting and summing over the horizon. Without going back and redoing the complicated model in expressions (3-1) – (3-3), it will be possible to indicate what is involved in a simpler way. Thus:

- The cost part of the problem would include an initial investment term at each source related to each stage of pollutant reduction. This would be the cost of a certain amount of capacity and would imply a particular variable cost function.[12] Then the variable costs of reduction for each period would be included, discounted as appropriate for the period.
- The per capita benefit functions would be different for each year because of the dependence on income. An additional and much more difficult challenge would be to reflect increasing leisure time and changing tastes. As a general matter, all these changes would

[11] This "agreement" is usually based on some notion of the "useful life" of the facilities involved in the discharge reductions (municipal wastewater treatment plants and industrial facilities designed to deal with particular problems such as oily water or acids, or complex organic chemicals). A big simplification is usually included here as well. It is that the facilities continue to operate as designed and built until after the horizon, when they may (or may not) just fall apart. It would be possible to build in exogenously determined declining efficiency (the cost for the same level of removal could rise over time), but this raises the possibility of intervention to raise efficiency through rehabilitation (a sort of partial investment in a new plant). And this gets perilously close to a true dynamic problem because of the choice of rehabilitation timing and the logical link between how bad things have gotten and the cost of the rehabilitation (see Chapter 4).

[12] This is also quite a bit easier to say than to accomplish, both in the sense of determining the actual cost functions and relations between fixed and variable costs, and in the sense of solving the resulting programming problem, where the choice of capacity and type of treatment is not continuous but involves picking one from a menu of choices. In actual examples it is often convenient to approximate the correct formulation by making the cost function into total cost, with capital charges being applied on top of variable costs in each year. If raw pollution loads are also changing, however, this shortcut will no longer work. It will be necessary to solve a dynamic programming problem.

probably increase the recreation and aesthetic benefits attached to a specific water quality. Their effect on health benefits is not so clear (see Chapter 6).

- Populations could be growing at different rates, so that total benefits for any route and reach in any year t would differ from those in the first year, both because of changes in the per capita valuation and in the population that cares. (Again, of course, the annual benefit numbers would be discounted to the present using the agreed-on interest rate. Or possibly a range of rates would be used to explore the sensitivity of the result to this choice.)

Notice that a fuller elaboration of the problem could have the cost side changing as well to reflect increasing production and therefore increasing raw pollution loads, so that achieving the same discharge in year t would cost more than in year 0. (As the footnote points out, combined with the possibility of choosing different capacity levels in the first period, this creates a more difficult problem.)

Knowledge Requirements for Basinwide Optimization

The knowledge requirements for basinwide optimization include discharges, the costs of reducing discharges, ambient quality, and benefits.

Discharges

While it is not a trivial task to obtain and maintain an inventory of a basin's point dischargers, it does appear to be something that LAC nations are undertaking, often with Bank assistance. It is not a conceptually difficult job, but is more a matter of establishing an initial survey technique and then making sure that regular sampling is done to keep on top of changes. Such sampling will in any case be necessary as part of an enforcement policy, regardless of how the policies that affect dischargers are arrived at.

Costs of Discharge Reduction

It is not necessary for LAC countries to reinvent the wheel by creating their own databases of cost-of-discharge reduction functions for the sources identified. The OECD countries have made a very large investment in such information and a substantial fraction of the available information can be obtained through off-the-shelf programs. [For example, a library of cost function information is available as part of the "Streamplan" model used for illustration in Chapter 5 (De Marchi et al., 1996).]

Ambient Quality Modeling

Again, there are many models available publicly and through consulting engineers. These provide a mathematical structure and make use of generally accepted biochemical and hydrological relations, but they do need to be adjusted for local conditions. This requires some investment in obtaining whatever background geographic and climatic data are required (such as hours of sunlight, air and water temperatures, channel depth, and surface roughness). It is also necessary to have enough ambient quality data corresponding to the particular chosen model's predictions to allow verification of the adjustments.[13] The important choice here is the

[13] The ambient water quality model used in the illustrations in Chapter 5 is simply one representative of a large class of publicly available models. Some other publicly available models are cataloged in Chapter 5.

model's complexity. At one extreme, the choice could be confined to oxygen-demanding organics (represented by biochemical oxygen demand) and their effect on in-stream oxygen concentrations. This is the modal path for the project analyses reviewed in Chapter 2. Such a model is used for illustrative purposes in Chapter 5. However, in many settings this will not really capture the problem. Thus, in Brazil's Guanabara Bay, the straightforward BOD/DO connection missed the substantial role of available nutrients and their effect on algal growth. The algae, in their cycles of growth and death, played a complex part in wide short-term swings in DO. [QUAL2E, an EPA-developed model discussed and used in Chapter 5, contains a model of this system (Brown and Barnwell, 1987).]

Fecal coliform bacteria counts (as a proxy for pathogens) are another ambient water quality characteristic of great interest when any kind of contact with the water is expected. It is not complicated to include these in a regional water quality model. In QUAL2E, for example, they are modeled as a decaying "substance" that does not interact with the other parts of the model (Brown and Barnwell, 1987).

Benefits

The necessity of producing benefit *functions* rather than just point estimates, and functions that reflect all routes to benefits (all effects on humans) at all points in the environment of interest, is the biggest obstacle to successful application of the basinwide optimization approach. There are several reasons that this obstacle still looms large after a bit over two decades of research into benefit estimation methods (and one very intense decade since the Exxon *Valdez* grounding in Alaska). First, it will never be practical to develop comprehensive benefit functions for a regional pollution control problem using combinations of the revealed-preference or indirect methods, such as hedonics and travel cost modelling. Russell and Clark (1997) explain that, looking at the several routes to benefits (see Chapter 7), there is no perfect mapping of routes onto available methods, ignoring the problem of data availability. There are overlaps, as when two methods each capture parts of the same benefit route. For example, hedonics and some dose-response case-valuation approaches could both capture some health benefits involving some of the same people. But hedonics will capture more parts than the health route (aesthetics and recreation, for example). Doing both will result in double counting. Doing only one or the other will result in an "underlap," since the dose-response approach will miss everything not health related, while hedonics will miss everything not capitalized in property values; that is, everything not capturable by property ownership.

Further, there is the great difficulty of dealing with where benefits accrue, although this is less problematic for a river than for a comprehensive regional effect such as air pollution control. Thus, while it is easy to identify beaches and to try to value the benefits of changing water quality at any beach, how does one deal with the variation in quality along a river as it relates to the fishing and boating routes to benefits? Where do these accrue and what method and data can capture this relation?

Accepting the first assertion about method applicability leaves the field to the hypothetical, or direct, or contingent valuation methods. These have several huge advantages in the context of this discussion. First, they work at the level of the individual, not the piece of property or the recreation site, so in large part they avoid the problem of location. That is, a careful description of the projected effects of a particular policy or decision on the quality of the watercourses in a basin leaves the respondents free to decide what matters to them. Second, the same is true of what routes to benefits matter to them. Assuming for the moment that their answers are credible, those answers can be used to reflect all the ways in which the described changes affect them.

However, it is still not straightforward to obtain benefit functions using some version of CVM. A major reason is what might be called "information overload." Reviewing all the potentially relevant changes in basin water quality in ways that lay respondents can actually relate to, for even one level of control, could tax the patience of a high percentage of those surveyed. And the benefit function requires information on reactions, not just to varying levels of control seen as a scalar, but to varying vectors, with the elements made up of ambient water quality indicators such as those discussed earlier. While techniques that ask directly for vector comparisons might ultimately prove advantageous in such a setting, the sheer number of vectors that would have to be asked about and the size of each vector (characteristics and locations in the watercourses) imply the need for very large numbers of respondents to produce enough comparison data to allow inferences about the shapes of the relevant WTP functions.[14]

The discussion immediately above suggests the possibility of not insisting on net benefit optimization while retaining the regional (ideally, basin) focus.[15] This leads to a cost-effectiveness analysis in the form of cost minimization subject to vector constraints on ambient water quality. Another possibility is to relax the optimization requirement and reduce the number of possibilities to be analyzed from the infinite to a small number (small enough that one could imagine costing and obtaining benefits for each). Some comments on these two versions of a second-best approach follow.

BASINWIDE, VECTOR-CONSTRAINED COST MINIMIZATION

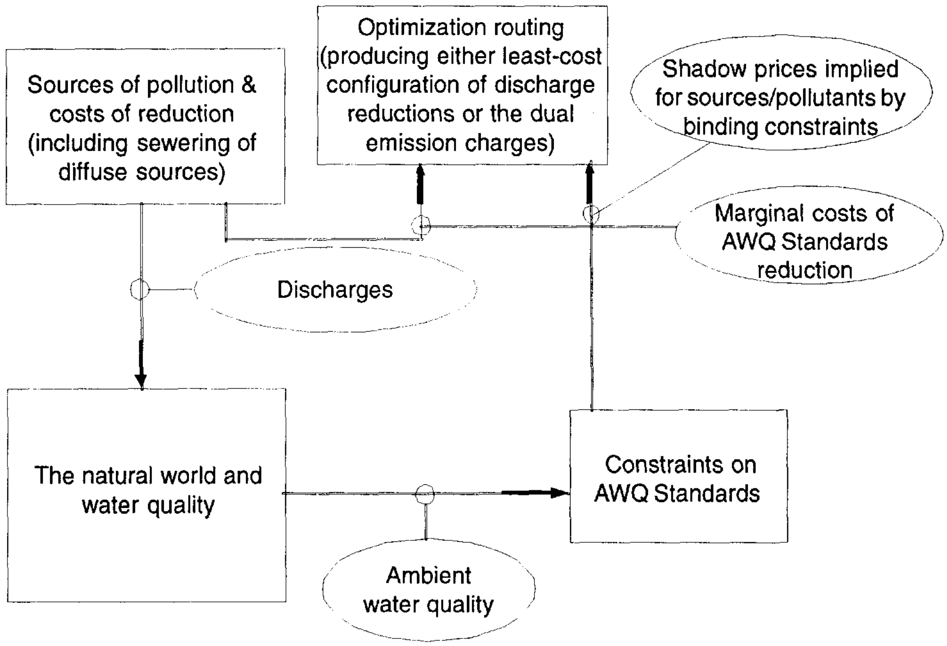
Vaughan and Ardila (1993) have made the case for the legitimacy of cost-effectiveness analysis (CEA) for Bank purposes in the face of great imprecision in benefit estimates. The background requirement for such an approach is the choice of ambient water quality standards that will be imposed on the watercourses to be analyzed. These may be chosen by a national legislature, or the environmental agency may be directed to develop them. The basis for such standards might include safety, health, aesthetic, or ecological considerations. For example, turbidity might be limited to guarantee visibility for the safety of swimmers. The same is true for fecal coliform bacteria and swimmers' health. Dissolved oxygen could be limited to protect fish of interest to commercial fishermen or recreational anglers. Or a more general "health of the ecosystem" justification might result in a similar DO constraint. Constraints on the presence of oil and grease would protect against surface "sheens," while constraints on floating sewage solids and DO at very low levels could also be justified on the grounds of minimal aesthetic protection. Constraints on algal bloom densities could be related directly to other requirements such as those on DO or turbidity, or could be aimed at protecting against toxic threats such as those from certain blue-green algae or coastal red tides. (For internationally recognized standards from the World Health Organization, see Jianming et al., 1983.)

Wherever the AWQ standards come from, they can be treated as constraints in a more or less complicated and sophisticated watershed cost-effectiveness model, as shown schematically in Figure 3-3, which may be contrasted with Figure 3-1. "More or less complicated" means the model can be, in principle, dynamic, stochastic, and locationally flexible. It can reflect all the chosen AWQ standards and contain natural world models that relate all the known (or understood) pollutant discharges to these standards via aquatic ecological models. (For a discussion, see Russell, 1995.)

[14] See Chapters 7 and 8 for more on this, including references dealing with the problems of using the responses to such surveys econometrically.

[15] For a brief review of 20 examples of basinwide modeling efforts, see Lee and Dinar (n.d.).

Figure 3-3. Schematic of a Regional or Watershed Cost-Effectiveness Model



What has been abandoned from the first approach is the benefit function part of the model. There is no balancing here. Rather, each element of the standard is of the same absolute importance. The interest of the designers of water quality improvement investments is only in the cost side. As long as all the standards are met (constraints satisfied), all that matters is minimizing the costs of this achievement.[16] In present value terms, of course, this will be relevant to the ranking of alternative solutions only to the extent that the timing of projects is allowed to vary because of the dynamics of the situation or because of constraints on fiscal or engineering "capacity" that force postponements. Again, see Chapter 4 for further discussion.

Practically speaking, with multiple constraints (both in the sense of multiple places and multiple dimensions of ambient quality), and the actual ambient quality levels, which are not fully spatially independent of each other because of the structure of the natural world, it is inevitable that only some subset of the constraints will be binding at the optimum, with different subsets binding at different feasible solutions. A comparison across the feasible set purely on the basis of cost involves, then, a rather strong assumption—that it is irrelevant which constraints (AWQ constituents, locations) are more than satisfied and by how much. If this creates discomfort, it is possible to examine the sensitivity of the choice to other knowledge by introducing credits for doing better than the standard. These can be specific to quality elements and locations. But this is an inherently arbitrary exercise, and in a large, complex watershed model could at best only tell you that there was or was not "much" sensitivity. (For an illustration of the penalties that ignorance of the real benefits can incur when doing regional cost minimization on ambient quality constraints, see Smith and Russell, 1990.)

As in full efficiency modeling, there are layers of complications that can be stripped away in the interest of practicality, and the meaning of "optimal" is correspondingly variable. A few

[16] As already noted, it is quite common for loan applications to involve some form of least-cost analysis, supplemented by a computationally independent check on whether the least-cost alternative passes the net benefit test.

other observations about cost-effectiveness analysis and its "legitimacy" for the Bank seem to be in order, however.

First, the cost-effective meeting of politically chosen standards has no built-in economic legitimacy, although it may be highly practical and internally logically consistent. It is always possible for economists or others to question the efficiency of the standards. For example, the executive orders of successive U.S. presidents have had the effect of requiring the EPA to examine the efficiency justification for the National Ambient Air Quality Standards (NAAQS), even though the legislation establishing these standards requires that they be based on protecting health (the primary standards) or welfare (the secondary standards) without regard to cost.[17] Chapter 9 offers an example of the dangers of vector-constrained optimization, drawn from IDB experience. The approach implicitly assumes that it is worth meeting the standards, but if benefits are insufficient, a least-cost program may not contribute to social welfare.

Analogously, the IDB could question the standards (constraint set) imposed by a loan applicant. This might well be a natural reaction, for example, if an LAC nation adopted wholesale the AWQ standards chosen by one or several U.S. states. [These were chosen as part of the standard-setting exercise for interstate waters carried out in the 1960s, prior to the change of emphasis to treatment capability contained in the 1972 Federal Water Pollution Control Act Amendments (Davies and Davies, 1975). For an example of a set of state standards, see Tennessee Department of Environment and Conservation, 1997.] At this point, however, the situation becomes at least as political as economic. The very act of questioning is significant and, in a sense, threatening. The practical difficulty of doing such questioning correctly is considerable because of the optimality requirement already noted.

Second, in the absence of such a fundamental question, the projects that enter the optimal (least-cost-while-meeting-standards) solution would be "precertified" in the same sense as those found in the full aggregate efficiency solution. The same roles for continuing oversight apply as well—specifics of the overarching models and continued checking of projections against reality are all fair game.

Third, if the proposing agency or nation is prepared to adopt a unidimensional (scalar) definition of effectiveness, and if the scale of the problem is held constant by the definition of the watershed or region, it is possible to look at the problem as one of maximizing the ratio of effectiveness to cost.[18] (This is not possible if the goal being sought is multidimensional, as described earlier. Then there is no single ratio to maximize.) What would be a scalar effectiveness measure? Possibilities include the minimum of DO found in any stream segment in the watershed; the value of some index defined by combining predictions of DO and suspended solids; or a summary ecological prediction such as habitat quality units. (On the latter see, for example, O'Neil et al., 1991.) Note that this may be seen as even more problematic than vector-constrained cost minimization because so much more of the full problem is left out. That is, all that matters to the mathematics is the single effectiveness measure. The values of all other outputs are ignored.

Finally, only in very simple problem settings would it be possible to impose the requirement that every solution create *exactly the same* level of effectiveness. Where that is possible,

[17] There is a serious "catch-22" here, however. Such an examination of the efficiency of standards should, in principle, be based on a comparison of costs and benefits for the optimal (least-cost) solution to an aggregate efficiency model. (These could be regional models summed across the nation.) Any arbitrary comparison away from the optimum would imply an arbitrary "penalty" because you would not be seeing the largest attainable (positive) difference (or smallest negative difference) between benefits and costs.

[18] Holding scale constant protects against a solution that chooses only small, highly effective actions (projects) while leaving much of the region alone because it is harder to deal with. This is a general problem with ordering choice on the basis of ratios. See, for example, Krier (1994).

least-cost analysis (LCA) is the method of choice,[19] and the ambiguity introduced by varying levels and locations of "slackness" disappears.

Examples of regional vector-constrained, cost-effectiveness models include the Lower Delaware Estuary Model (Spofford et al., 1976) described briefly in Chapter 5. Much more recently, a Chesapeake Bay Model (Linker, 1996) with a similar structure has been operated. Directly relevant to the LAC setting is the Guanabara Bay Model (NRERPD, 1996). The latter involved an attempt to deal with both BOD and DO and the algae and nutrient cycle without building a complex water quality model embodying the relevant processes and their links.

MULTI-OBJECTIVE ANALYSIS: A SLIGHTLY DIFFERENT USE OF OPTIMIZATION

One related thread of the project (or plan) evaluation literature that is worth mentioning at this point is multiobjective, or multicriteria decision analysis, which involves a priori rejection of the notion that all project effects can be meaningfully transformed into terms of a single numeraire—money, for most purposes. An example of such an effect might be the distributional consequences of a project, where the fractions of costs and benefits going to particular neighborhoods or income classes are estimated, as they are for Bank projects. However, these fractions do not translate into dollar terms that are equivalent in meaning to the efficiency benefits. The multicriteria methods, and there are many (see, for example, Janssen, 1992), involve supplying new, all-encompassing scales on which all the criteria can be measured in the same units. This might be achieved, for example, by asking the (somewhat mythical) "decision maker" for his/her relative weights for results along the several dimensions. Or, hypothetical weights can be used to explore the sensitivity of decision to this choice. These methods may be seen as attempts to mimic part or all of the political choice process when alternative weights are used for outcomes that include other criteria than aggregate efficiency. Another possibility is to create a sort of production frontier for the several project goals and to find the "shadow prices" of, say, distributional effects, in terms of net efficiency benefits forgone at the margin. This technique might be seen as an attempt to more usefully inform the political process. (See Van Pelt, 1993, for a discussion of techniques and examples of case studies.)

AD HOC, OR PROJECT-BY-PROJECT COST-BENEFIT ANALYSIS

The most common type of loan application received by the Bank involves a single water quality improvement facility, or possibly a small number of facilities in a metropolitan area, with a minimum of context provided. These are almost always wastewater treatment plants designed to deal primarily with domestic sewage, although the connection of some industrial sources may be contemplated. That is, so far from there being a watershed optimization model in which the project is part of the optimal solution, there may only be a description of existing water quality in the stream segment or other water body to be affected by the project and a projection of that effect. The effect may be monetized by more or less persuasive methods, but typically no information will be supplied about alternative possibilities for attacking the perceived water quality problem (for example, no information will be given about the costs and benefits of transferring more cleanup effort to industrial dischargers along the river or coastline in ques-

[19] There is a tendency for Bank documents to use the expression "least-cost analysis" as a synonym for cost-effectiveness analysis. The terminology difference is maintained here only to highlight the difference between guaranteeing that every option has the same physical result (LCA) and allowing for "slack" so that all the alternatives are different though none is worse than what the constraints require (CEA).

tion). In effect, the Bank is presented with a take-it-or-leave-it choice for a project that may in isolation seem justified, but which would actually be part of a relatively inefficient regional solution or vice versa.

The practical advantage of this analytical route has already been implied. Defining a single project possibility eliminates the necessity of pursuing information on all dischargers (although, as discussed in Chapter 4, their effects on water quality where it is also affected by the project in question generally cannot be ignored) and benefit functions covering the entire basin. (A point estimate corresponding to the projected effect of the project will be enough.)

SUMMARY

Table 3-1 compares three of the uses of cost-benefit analysis. Unfortunately, there is no basis for making judgments about the relative values of the advantages and disadvantages of an isolated project analysis, or a subset analysis as opposed to basinwide optimization; i.e., the cost of extra data gathering, estimation, and model construction vs. the gain in efficiency net benefits from the way water quality is eventually managed. Such a basis will depend on research that contrasts the implications of the broader and narrower approaches for costs and benefits *and* keeps track of the costs of the approaches themselves. In the absence of any agency's willingness to fund such studies, it is fair to say that there will be no way to settle the disagreement between those who want to push toward basinwide optimization and those who think too little will be gained to justify what they expect will be vast expense and years (or decades) of frustrating research. In the circumstances, it may not be amiss to suggest an intermediate-term compromise: Work toward requiring basinwide, vector-constrained, cost-effectiveness analysis, but subject specific projects that are identified as part of the solution to such a problem to an isolated cost-benefit analysis. This is not too far from what is already common, the difference being the basinwide character of the cost-effectiveness model.

The remainder of the book pursues the threads of inquiry described in this introductory material, with the ultimate goal of providing the guidance identified as needed by involved professionals at the Bank. Most immediately, in Chapter 4, some special challenges and pitfalls that apply to several of the techniques compared in this chapter are taken up.

Table 3-1. Summary Comparison of Cost-Benefit Analysis Applications

	Basinwide Optimization	Subset Simulation[a]	Isolated Project or Policy Justification
Analytical Goal	Finding the optimal policy or project (no possibilities eliminated up front)	Determining the best policy or subset of projects from a given set of possibilities	Determining whether a particular project or policy is justified in efficiency terms
Criterion	Maximize net benefits using all available control variables	Find highest $B_j - C_j$ over all predefined policies or projects j	Is $B - C > 0$
Information Required about Dischargers			
1. Locations	All affecting situation	All chosen for analysis	Single location as defined for project or multiple if single policy
2. Base discharge	All affecting situation	All chosen for analysis	Single location as defined for project or multiple if single policy
3. Costs of discharge reduction	Functions for all considered	Point estimates corresponding to chosen alternatives	Point estimate(s)
Information Required about Ambient Environment			
1. Accepting as "input"	All discharges from sources affecting situation	All discharges from sources chosen for analysis	Discharge from single project or discharges implied by single policy
2. Producing as "output"	Ambient quality everywhere relevant to situation	Ambient quality at all locations chosen for analysis	Ambient quality as affected by single project or policy (e.g., downstream)
Information Required about Benefits			
1. Relevant "routes" to benefits	All relevant to situation	All relevant to situation	All relevant to project or policy
2. Form of benefit information	Functions relating to ambient quality arguments	Multiple point estimates, one for each alternative	Point estimate(s)
Difficulty of Implementation	Highest difficulty	Medium difficulty, depending on number of alternatives	Lowest difficulty

[a] Vector-constrained optimization is a restricted version of subset simulation that changes the criterion to minimization of the present value of costs over alternative projects j to meet specified ambient standards, and eliminates the need to collect information about benefits.

Annex 3-A

A Case Study of Adaptation to Information and Budget Constraints: Guanabara Bay

Roughly speaking, the cost-effectiveness modeling work reported in NRERPD (1996) (see Chapter 2) represents a clever response to information and budget limitations in the context of Guanabara Bay, the water body that for many people symbolizes Rio de Janeiro. Not that there was *no* discharge or abatement cost information or any existing aquatic model on which to base an analysis. But the World Bank team judged that the water quality problem of greatest moment was eutrophication involving phosphorus fertilization of algae, with the phosphorus coming from current discharges and resuspended sediment deposits. Because previous work had been focused on DO and bacterial contamination, the available water quality model captured some features of the eutrophication processes but was clearly deficient in other respects. A better model would have been expensive to build and run, so the decision was made to use what was available and to simplify the problem to more closely fit the capabilities of the model.

The approach used the available water quality model, plus assumed weights for the relative importance of the indicators of water quality predicted by the model, to generate what amounts to an index number for pollution discharges. The water quality indicators were BOD, DO, and organic phosphorus (OP); the latter was used as a surrogate for algal problems and eutrophication. Three weighting schemes were reported, one emphasizing BOD to the exclusion of the other two, and two that weighted all three indicators positively, with one of these giving four times the weight of the other to OP. In addition, two alternative sets of geographic weights were used to reflect the importance of parts of the bay. Finally, a "valuation function" scored how closely any particular combination of discharges (and hence any resulting vector of weighted ambient results) approximated the ultimate quality target—levels equal to those in the open sea. All this work, when combined for alternative discharge levels, implied weights on discharge reduction of BOD and phosphorus by location in bay subregions (the embayment equivalent of perfect-mixing reaches in a river).

Then, using these weights, cost-effective solutions could be found for the question: How can total discharges be reduced by R percent at least cost? That is, the method creates a single (a scalar) discharge from each source based on its location and on the underlying mix of phosphorus and BOD in its wastewater. But the cost-of-abatement models reflect the raw loads of both BOD and phosphorus and include options for reducing these loads prior to discharge. So the optimization routine looks at a single number for each source, the weighted sum of BOD and phosphorus discharged, but is in effect solving a problem that reflects both the BOD/DO concern and the eutrophication concern, with relative spatial concern built in.[20]

To summarize, techniques emphasizing phosphorus removal, in particular primary treatment with enhanced (chemically induced) precipitation to remove extra phosphorus, are gen-

[20] The base (1.0) for the weights is 1 ton of BOD discharged to the northwest segment of the bay. The highest reported relative weight for phosphorus is 66.9 to 1—which is to say that removing 1 ton of phosphorus (discharged to the southwest segment) was equivalent to removing 66.9 tons of BOD discharged to the northwest segment.

erally preferred to secondary treatment aimed primarily at BOD. In addition, treatment efforts are concentrated in a few of the segments, at least until the total removal level required goes above 70 percent.

Now, it must be emphasized that this approach does not transfer directly to any other setting. In particular, the ultimately important discharge weights are situation specific. They are also model specific—they would change if a new water quality model were used in the underlying calculations. However, the technique seems interesting as an example of how a cost-effective regional solution can be sought even in the absence of totally satisfactory computational infrastructure and information. (For a conceptually similar effort with some additional complications involving citizen input in the weighting, see Ibrekk et al., 1991.) The beauty of it is that the optimization model, being inexpensive to solve, can easily be used for sensitivity analyses exploring which gaps one should attempt to close. (For examples of such sensitivity analyses that go well beyond what the World Bank team undertook, see Darling et al.,1993; Vaughan et al., 2000a; and Chapter 9 of this study.)

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Chapter 4

Temptations, Pitfalls, and Technical Complications

This chapter is divided into two parts. The first and shorter of these is a discussion of several temptations for manipulating benefit and cost numbers, even in simple static cases in which the technical problems raised by dynamic and contextual complications and the matter of functional form are ignored. The second part addresses items on this list of technical complications. For these items, it is useful to be aware of both the potential difficulties that are being avoided in the simplified forms of analysis generally undertaken and of the assumptions that must be made to support these simplifications.

TEMPTATIONS AND PITFALLS

There is widespread agreement that the key concepts in cost-benefit analysis are surpluses and opportunity costs. Thus, benefit estimation for consumers involves obtaining Hicksian or Marshallian measures of willingness to pay for actual or hypothetical quantities of the environmental goods or services to be provided (or for avoiding the existing level of damage to such goods or services). Analogously, when firms benefit, the appropriate measure is producer's surplus, which is the difference between revenue and variable production cost. On the cost side, what is important is what is being given up by using real resources in the proposed project, i.e., the opportunity costs of such resources.

Measurement problems aside, however, these apparently simple prescriptions leave some room for maneuver, and this freedom has traditionally not been lost on those with an interest in seeing projects approved. Indeed, some of the temptations discussed in this section are about as old as systematized cost-benefit analysis, although others have only become important since contingent valuation and nonuse values became part of the potential toolkit. These temptations are organized according to the following two questions: Whose surpluses count? and, What is "surplus" anyway?

Whose Surpluses Count?

Probably the most venerable of the benefit inflation temptations is to count the surpluses of businesses drawn by, or stemming from, a project. This effort may actually be launched from the cost side of the analysis by applying a regional multiplier to the project's costs, or some part thereof, thus producing an estimate of the increase in economic activity attributable to project spending. Using some assumption about the level of surplus per unit of activity, an apparent addition to benefits has been discovered and produced.[1] After all, this additional surplus is

[1] The question of whether the multiplier actually produces a "surplus" is taken up later. For now, this additional problem is ignored.

attributable to the project through the technique of regional multiplier analysis, which has its own long (and largely respectable) history of application to regional economic modeling.

An example of this practice from IDB experience is found in the Economic Evaluation of the Salitre Wastewater Treatment Plant Project (for Bogota, Colombia). In this evaluation, construction and O&M costs are converted into project benefits via multipliers (0.6177 for the former and 0.5621 for the latter) and a further multiplication by 0.8 in each case to correct for "leakage" from the region. The gross benefits generated by these multipliers during the 27 years of project operation analyzed are a bit more than 15 percent of the annual totals ($\$5.5 \times 10^6$ of $\$35.62 \times 10^6$). As a fraction of a typical year's net benefits, these secondary benefits are about 40 percent. Thus, it is worth some effort to sort out the conceptual correctness of this exercise.[2] Two questions are intertwined in this sorting out: What "accounting stance" is agreed to for the project assessment? and, To what extent are resources unemployed in the proposing nation or region?

Accounting stance determines what counts as a benefit and what is seen as merely a transfer into the project area from elsewhere. The Bank requires that a national stance be used in project evaluation. This corresponds to the National Economic Development Account as set out for U.S. water agency cost-benefit analysis in the principles and guidelines of 1983 (U.S. Water Resources Council, 1983). In this account, "stemming" or "secondary" benefits do not count because they are assumed to represent transfers of activity from other regions to the project region in a fully employed economy. The surpluses generated by the transferred activity (be it a barber shop, a supermarket, a locksmith, or a retail broker) would have been generated in any case—just in a different place within the nation.[3]

Unemployed resources seem relevant because there is no opportunity cost to employing them. Why not then allow counting of stemming benefits that involve the employment of such resources? There are two reasons: First, by Bank policy, credit is already taken for the direct use of unemployed resources through the use of accounting or shadow prices for inputs (suitably justified). Thus, if project labor is unemployed, its cost will be zero and net benefits will be larger.[4] Second, and even more to the point of this discussion, the appropriate basis for comparison in this context (stemming benefits) is not the no-project case. Rather, it is the "some-other-project" case. That is, any public project will involve the creation of additional employment via the multiplier; and it would be analytically challenging to have to demonstrate that project A was better than project B in this regard (Powers, 1980, makes this point). Further, even if one project can be argued to create more stemming employment than another because the projects are located differently with respect to unemployed resources, then the resulting comparison is a distributional, not an efficiency issue.

The Salitre project evaluation document provides another related example here, this one involving what amounts to a cross-city transfer rather than a cross-regional one. Thus, the evaluation includes as benefits of the project the WTP for odor elimination by future assumed residents (and for the later project phase, enhanced visual aesthetics of the river). Two assumptions are needed to include these amounts as project benefits. The first is that building the

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[2] The annual costs used in the net benefit analysis differ from those used to calculate the additional "economic activity" benefits. The former are measured using some unspecified "tariff" formula that produces substantial year-to-year variance, while the latter are constant over the life of the facility. Thus, only an approximation of their importance can be offered.

[3] The principles and guidelines do recognize that for some purposes, a regional stance may be useful. Accordingly, a regional economic development account is defined. This measures the sum of the national effects and the transfers into the region from outside it. This account is part of the subsidiary information that may accompany project cost-benefit analyses.

[4] If the direct costs of the project are to be reduced on this basis, then, at a minimum, a case should be made that the project will actually employ the local labor. If the project involves laying trunk sewers and building fairly complex treatment plants, and the local labor is unskilled, one might be skeptical of such a claim.

proposed water treatment plants (at different treatment levels) will allow the development of the land affected by the reduction of odors at higher densities than is currently true. The second assumption is that the incoming residents can properly be credited with a WTP to reside there that equals the WTP of the current residents for improving the odor and visual aesthetics along the river. What this procedure does, in effect, is to assume that the new residents would be willing to pay the amount credited to them for the chance to live along the river rather than live anywhere else in the metropolitan area. Another way of saying this is that it assumes away the fact that all those people would, in fact, be paying to live somewhere, whether or not the project were built. At most, what could be credited to the project would be a difference in WTP for one location as opposed to another, and there is no reason to think that that number is related in any neat way to the WTP of the current residents of a polluted neighborhood to be free of the pollution damage.

However, the correct number could be found via a metropolitan area hedonic housing value study, if there were enough information to allow a projection of the effects of cleaning up particular neighborhoods. Such a model would have to produce what amounts to a new general equilibrium arrangement of housing values, producing a new aggregate. It would *not* look only at one improved area.

Another way of proceeding would be via a CV study that asked respondents their WTP for living in the cleaned-up area as opposed to where they were currently living or where they might live, if they moved into the rapidly growing metropolitan area. This last phrase brings up yet another problem that seems to fit under the label of a temptation—defining the extent of the market for hypothetical questioning purposes. For this problem, the first question is how to identify and sample the population of potential future residents. A second question is how to give those sampled enough information about a place they do not (by hypothesis) yet live, in order that their responses might be taken to be informed. One option for the first solution—albeit an expensive one—would be to sample from the entire national population and ask about migration intentions before asking about WTP. Another option, requiring an up-front investment in research on migration patterns, would be to concentrate on those subpopulations that are especially heavily represented in existing in-migrants (see, for example, Freeman, 1993).

This "extent-of-the-market" temptation is probably more important and even more difficult to come to grips with when nonuse benefits are deemed to be part of the legitimate benefits package from a project.[5] How does the analyst identify the relevant population of people who do not now use the body of water in question, but who would still be willing to pay something to clean it up? This is not a question with an obvious and nonarbitrary answer. By definition, behavior directly related to the water body in question cannot be observed. The option of sampling from the entire national population will often prove to be too expensive. One ingenious but inherently arbitrary answer is to identify the relevant population by some other overt behavior. For example, Stavins, in a 1984 study of the preservation value of a whitewater river in California, attributed nonuse values taken from other studies (as a ratio to use value) to people who are members of the Sierra Club both in California and (with the per capita dollar figure reduced) throughout the United States (Stavins, 1984).[6]

[5] See the commentary in Chapter 6 on ambient water quality benefits as estimated for Bank treatment projects, where doubt is cast on the notion that nonuse benefits are important in the LAC setting. It may be that the nonuse benefit category is more relevant to OECD country decisions, but this may be a temporary phenomenon, linked to low incomes and aggressively bad current conditions.

[6] Kopp and Smith (1989) provide natural resource damage estimates from actual court cases that illustrate the importance of the extent of the market assumed in any particular case. They mention the possibility of trying to determine this extent empirically, but note that this could be a major research task in itself. The only nonempirical solution that is guaranteed to be consistent with the Bank's position is to use national samples and let the data define market extent.

What Is a "Surplus," Anyway?

The Salitre project also helps to illustrate another problem commonly found in efforts to expand benefits via regional multipliers: What these efforts produce is a "value-added" (VA) measure of economic activity, which is consistent with the national accounts, but not with cost-benefit analysis. Thus, value added is defined (e.g., Hyman, 1990) as total transactions minus intermediate transactions. For a firm we can write this as

$$\begin{aligned} \text{VA} &= \text{revenue} - \text{cost of purchases from other firms} \\ &= \text{payments to factors of production} \\ &= \text{wages} + \text{interest} + \text{rent} + \text{depreciation} + \text{profit} \\ &= \text{cost of labor} + \text{cost of capital} (+ \text{land}) + \text{profit} \\ &= \text{cost of labor} + \text{fixed costs} + \text{profit} \end{aligned}$$

Let us compare this with producer's surplus (PS) (e.g., Dinwiddy and Teal, 1996).

$$\begin{aligned} \text{PS} &= \text{revenue} - \text{variable cost} \\ &= \text{revenue} - \text{variable cost} - \text{fixed cost} + \text{fixed cost} \\ &= \text{profit} + \text{fixed cost} \end{aligned}$$

If we identify, as above, the cost of capital and land as the fixed costs, and simplify the notation by writing Π for profit, W for wages, and K for fixed costs, we find that

$$\begin{aligned} \text{VA} &= W + K + \Pi \\ \text{PS} &= K + \Pi, \text{ or} \\ \text{VA} - \text{PS} &= W \end{aligned}$$

So value added exceeds producer's surplus by the amount paid in wages. Using VA on the benefit side of cost-benefit analysis therefore exaggerates benefits.

One other temptation to exaggeration of surplus that is especially relevant to evaluation of water quality enhancement projects arises from commercial fishing. The notion is that cleaner water will increase fishery yield per unit of fishing effort and (here is where the problem enters) that this will lead to increased producer's surplus in the industry.[7] For example, two of the projects discussed in Chapter 2 were justified in part by projected surpluses from enhanced commercial fishing: Barbados South Coast (BA0036) and Guanabara Bay (BR0072). The problem here is not one of definitions, but rather follows from the economics of open-access resources. In brief it is this: If the fisheries in question are not managed so as to limit access, through, for example, traditional "commons" management or a state-imposed system of marketable licenses, there will be no surplus in the new equilibrium. Open access will guarantee that the surplus is competed away until average costs equal average revenues (see, for example, Munro and Scott, 1985, p. 635).[8] If commercial fishing surpluses are to be counted as project benefits, there should be an accompanying statement outlining how the fishery is or is going to be managed to control access.

[7] Whether this improvement is predicted using a persuasive aquatic ecosystem model or merely assumed is irrelevant to the point being made here.

[8] McConnell and Strand (1989) assert that water quality improvements could actually decrease the social returns from an overharvested, open-access fishery, if the improvements in water quality increase demand for the fish.

TECHNICAL CONCERNS

This section discusses a number of problems that arise in project analysis. They include context, functional form, phasing of the investment, and dealing with time in analyzing water quality projects.

The Regional Context

In Chapters 1 and 3, one view of the water quality improvement problem was taken to be the analysis of a single project or set of projects in the context of the situation in the rest of a basin. For example, the project could be the sewering of a neighborhood in a large city, with the collected sewage introduced into the river from a new outfall.[9] Alternatively, it could be the upgrading of wastewater treatment before the sewer effluent is discharged into the river, as in putting in primary treatment where no treatment existed, or going from primary to secondary treatment at an existing outfall. More complicated analyses may involve looking at several options, such as no treatment, primary treatment, and advanced (or enhanced) primary or secondary treatment. The engineers who design the project alternatives to be analyzed can provide the predicted discharge levels associated with each one. There may be only one pollutant tracked, which is almost always biochemical oxygen-demanding organics, measured by the quantity of oxygen required to stabilize them, or there may be more, such as heavy metals, oil, organic phosphorus or nitrate, or suspended solids. But whatever is being tracked, it would seem that only what happens downstream of the project matters (except in lakes or tidal reaches, where significant upstream mixing can occur). Why, then, bother with context?

There are two major reasons for discussing the relation between project and region, even if the focus will almost always be on the project. The first of these reasons is that we need to be clear that this focus will in general have costs because it will involve ignoring other opportunities to influence ambient water quality. The second is that under some circumstances, complete analytical isolation is simply not correct. That is, analytical isolation can lead to the wrong answer, even if we agree to ignore other options for affecting ambient water quality than the project in question.

There are two ways of approaching the question of regional context: thinking about the goal of the analysis and what it implies for information needed and functional assumptions, or thinking about the information-gathering strategy to be pursued and what it implies for a feasible analysis.

For the first, a range of possible analytical goals was discussed in Chapter 3. They include seeking the basinwide (aggregate) efficiency optimum (maximizing net benefits), a basinwide cost-effectiveness analysis involving minimizing the costs of meeting specified ambient water quality standards, options looking at the project's net benefits to see if they exceed 0, showing that a particular configuration of the project does (or does not) allow meeting some downstream ambient quality constraint, or simply demonstrating that the project meets some specified discharge standards.

For the basinwide goals, unsurprisingly, basinwide information is required. In both the net benefit and cost-effectiveness approaches, there must be information on all relevant sources of pollution—baseline discharges of pollutants of interest and the costs of reducing these. For the aggregate efficiency goal, there must be estimates of the benefits to affected populations

[9] The question of how to decide whether a sewering-plus-outfall project is justified and whether to add treatment before the outfall is discussed in Vaughan and Ardila (1993), and illustrated in the evaluation of Guaiba Watershed Environmental Management Program (BR0073). Annex 4-A elaborates on this. It represents an adaption of Annex A in Vaughan and Ardila (1993).

(which may not be confined to the basin but also may not include all the people in the basin; see the discussion under "Temptations and Pitfalls") of varying levels of ambient water quality.[10] This in turn implies that there is a water quality model capable of translating source discharges (and background information) into predicted ambient quality throughout the basin. Such a model is also essential to a regional cost-effectiveness goal. Given comprehensive information on context, the technical details concerning the formulation of the water quality models and the way the benefits relate to quality—whether only the changes or also the starting (or ending) quality levels—do not matter as long as all the pieces are consistent. If, however, the analytical goal is to determine whether a project considered in isolation is justified in efficiency terms, or which project configuration produces a desired ambient target at least cost, these details will in general be important, as discussed later.

The second (less desirable) way into the question of context is roughly the opposite of the first. That is, instead of starting with analytical goals and deriving the requirements for context information, it is possible to begin with the strategy chosen for gathering this information and the characteristics of the water quality model, and then determine what analyses are going to be possible. Again, it should be clear that if there has been no effort to gather comprehensive information or to build a basinwide model, there is no chance of doing basinwide cost-benefit analysis or cost-effectiveness analysis. Also, again, the interesting problems arise in the case of the isolated project.

Whichever way into the context question makes sense in a particular setting, the key pieces of the puzzle are first, the nature of the dependence of benefits on ambient water quality and second, the nature of the model that predicts levels of quality and changes in those levels.

For these purposes, what matters about the benefit relation is whether benefits depend only on how large a change in quality is predicted or on both the change and the level before (or after) the change. That is, the contrast is between:

$$B = f(\Delta AWQ)$$
$$B = g(AWQ_0, \Delta AWQ)$$

With the first relation, it seems intuitively likely that it should be possible to obtain the required prediction based only on what is projected to happen to the source being analyzed—the project.[11] With the second, there is a clear need to know something about what is going on independent of the project. However, the water quality model's characteristics become relevant at this point because with some models the effect of the project on downstream quality (the required ΔAWQ) is not itself independent of the level of ambient quality. (See Chapter 5 for an illustration.)

Combining consideration of the benefit function and the water quality model leads to the following propositions about the importance of regional context in the analysis of an isolated project: First, if the model's predictions of the change in water quality due to the project are independent of the level of water quality just above the project (which in general is unlikely)

[10] This discussion supresses, for clarity of focus, the complications raised by uncertainty, dynamic elements, and technical interconnections among pollutants on both the cost and benefit sides.

[11] A reviewer of an earlier version of this document objected that it makes no sense to assume the possibility of a benefit function depending only on ΔAWQ—that where on the scale of quality the change occurs inevitably determines what it is "worth." This is an empirical assertion about benefit functions and is more likely right than wrong. But environmental economists have long been ready enough to assume the can opener of constant marginal damages or benefits, so the assumption here hardly constitutes a radical departure. In any case, the purpose of introducing the possibility is to make clear the narrow limits of the situation in which context does not matter. It is not a prediction of what to expect in the real world of project analysis.

and if the benefit function depends only on ΔAWQ, then a cost-benefit analysis for the isolated project is possible *without additional context information.* [Of course, this can only tell the Bank and borrower whether the project is justified or, if several versions of the project are considered, which produces the largest (predicted) net benefits. It cannot say anything about other options for changing the quality of the river.] Second, if the water quality model is of the interdependent and more realistic sort, then ΔAWQ attributable to the project is a function of the background quality level created by upstream discharges, and regardless of the form of the benefit function, additional context information will be required. This information does not, however, have to include a full source inventory. In principle, AWQ information from just above the project source would be sufficient to start the AWQ model off and produce ambient quality predictions downstream from the project. Third, if the benefit function has both AWQ and ΔAWQ as arguments, then even if the water quality model is separable, there will be a need for context information in order to produce the required predictions of the absolute level of quality as well as its change. Again, though, this need not be in the form of a full basin discharge inventory. Ambient quality information from just above the project should be sufficient, whichever type of water quality model is being run. In a simple 2 × 2 table form, these propositions may be summarized as shown in Table 4-1 for the isolated project cost-benefit analysis problem.

If the analytical effort is to be directed at cost-effectiveness analysis, the form of the benefit function is irrelevant. But if the constraint involves the *level* of AWQ that must be attained, it is necessary to model AWQ as well as ΔAWQ, so information reflecting the regional context will be necessary. In effect, the second column of the table applies. If the question is how to put together subunits to make a wastewater treatment plant that will produce a particular difference in AWQ, the first (or left) column applies, and the need for regional context will depend on whether the water quality model to be used is separable or not.

Finally, it is worth returning to what was called the "second way" into the problem. The above discussion takes the goals as given and asks what is required to achieve them under different assumptions about functions and models. If the constraint is that the borrower (or the Bank) requires the use of an interdependent model, such as QUAL2E, then it will not be possible (in general) to do valid cost-benefit analysis in isolation from the regional context, no matter what form the benefit function is thought to take. However, the context can be captured by in-stream sampling upstream above the project.

Functional Form

Functional form is most often brought up in the context of optimization problems, where a "wrong" shape can lead to multiple local optima or to situations in which the first-order conditions identify worst rather than best results. However, even in the case of single project analyses for which the maximization of net benefits is not attempted, there are two related reasons to be concerned about functional form.

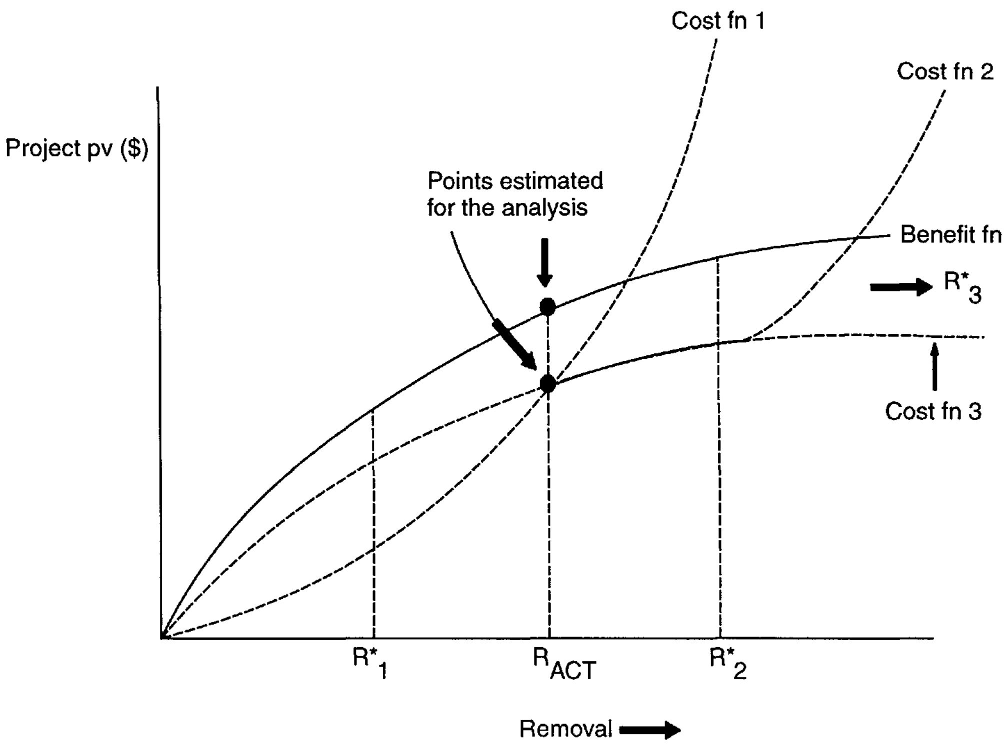
Table 4-1. Information Needs for Project Cost-Benefit Analysis under Four Different Assumptions

	Benefit Function	
Water Quality Model	**B = f (ΔAWQ)**	**B = g (AWQ, ΔAWQ)**
Additively separable for each source	Isolated benefit-cost analysis possible w/o context information	Need additional information to predict ΔAWQ
Interdependent (ΔAWQ due to source j depends on other discharges)	Need additional information to predict ΔAWQ	Need additional information to predict both ΔAWQ and other AWQ

First, without knowledge of functional form, there is not even a hint about what is going on on either side of the proposed project. Consider, for example, the possibilities illustrated in Figure 4-1. Here, the decision required is whether to build project ACT, which produces R_{ACT} percent reduction of the pollutant load (or pollutant index for Guanabara Bay). Assume that costs for achieving that reduction level have been minimized in a pre-cost-benefit analysis calculation. Let the discharge level corresponding to project ACT after the R_{ACT} percent reduction be the entry point to benefit estimation; that is, the benefits are a function of the difference between the "with" and "without project" discharges. Whatever method or set of estimation methods is used, the result will be a benefit corresponding to R_{ACT}. And, since R_{ACT} is already costed, the net benefits are determined for R_{ACT}. If, however, there is no knowledge of the shape of the benefit function through B_{ACT}, or of the corresponding cost function, or both, it is impossible to say whether the point, R_{ACT}, is locally better or worse than other nearby points on either side of it in terms of removal levels. This can be thought of as an opening for sensitivity analysis, where the analyst tries different functional forms rather than simply varying net benefits by arbitrary percentages.

Here R_{ACT} indicates the actual pollutant removal level (degree of cleanup). The optima for cost functions 1, 2, and 3 are designated as R^*_1, R^*_2, and R^*_3, respectively. The last of these is shown as farther out because the cost function is shown as continuing to exhibit increasing returns to the edge of the graph. More interesting (or at least more likely) are the first two—one above and one below the analyzed project level. R^*_1 is consistent with a cost function for which marginal removal costs increase at all levels. R^*_2 is consistent with one of the family of

Figure 4-1. Potential Problems Traceable to Lack of Knowledge of Cost Function Form (fn: function; pv: present value)



more complex functions often taken to characterize a primary-plus-secondary wastewater treatment plant—exhibiting initially decreasing and then increasing marginal costs.[12]

The second problem that arises when functional forms are unknown might be thought of as a sort of dual to the first. This second problem arises if and when the proposer or the Bank's analysts want to explore the implications of doing less than the hypothesized full project (pollutant reduction R_{ACT}). This might be the case for a phased project, for example, when the benefit estimation work is done on the basis of comparing the full project (all phases complete) with some base case. This was the case in the Guaiba River project proposal in Brazil (BR0073). The shortcut used to develop benefits for the earlier phases was to assume that the benefits could be scaled down linearly from the single known point (full project operation). It was further assumed that the benefit function was really concave (exhibiting declining marginal benefits with increasing R_{ACT}). In that case, linearization is conservative and, if the project is justified with the interpolated benefits, it would look even better with the correct benefit measure. (This was discussed in Chapter 2.)

Some Evidence on Forms

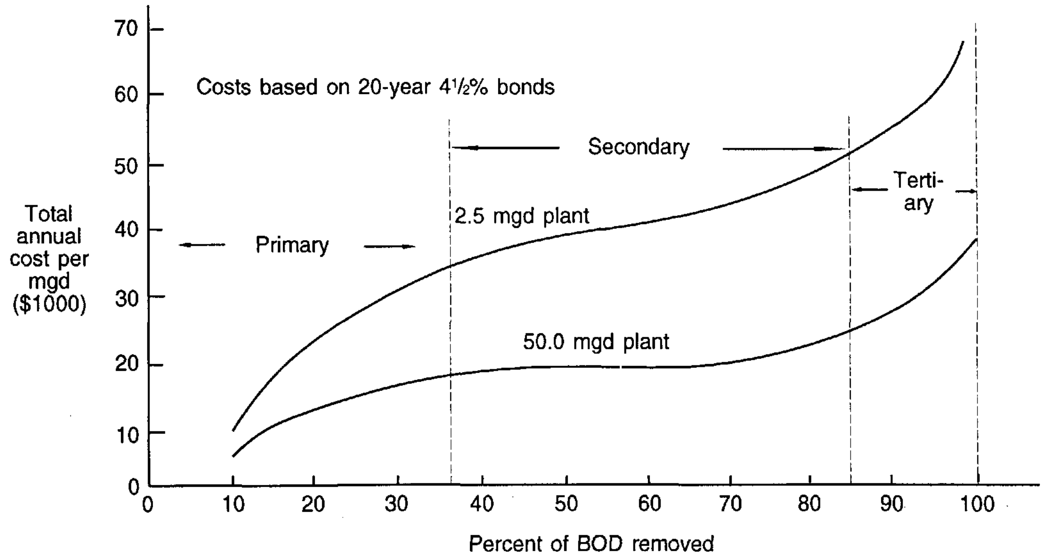
What about empirical evidence on the matter of functional form? On the benefit side, a first observation is that benefit studies reported in the literature are most often aimed either at illuminating some question of technique (e.g., dichotomous vs. open-ended WTP questions, or the functional form for travel-cost visitation functions) or at producing a benefit estimate for some condition or policy change.[13] Thus, it is hardly surprising to find a dearth of published work speaking to the question of the form of the benefit function. A recent exception is to be found in Adamowicz et al. (1998), in which some evidence is presented to support the intuitively appealing notion that marginal benefits decline for increasing levels of provision of environmental goods—in this case, caribou populations and acres of land managed as wilderness. However, it is possible to make a priori arguments for the opposite pattern for other kinds of goods; visibility measured in miles or congestion at remote recreation areas are two examples (Repetto, 1987).[14] A second concern over the shape of benefit functions is whether they are continuous. Environmental regulations that set ambient air or water quality standards effectively promulgate a threshold where damage levels become "unacceptable." These thresholds implicitly assume (or at least are more easily supported by) discontinuous damage functions, such as step or ramp functions. However, some comfort can be drawn from the arguments presented in Dewees (1995) supporting linear approximations of benefit-damage functions.

[12] Notice that this problem of lack of knowledge looks a bit like the one occurring when the feasible space for optimization is not convex, in the sense that the analyst's techniques are "myopic." The myopia here is of a different sort. In optimization, the problematic points of interest are identified by the marginal equivalences of the first-order conditions. The point here is more arbitrarily defined—by the original choice of R_{ACT}. Notice also that rather than better places lurking unseen in the vicinity of R_{ACT}, there may well be worse ones. Thus small problems with achieving R_{ACT} could significantly penalize the nation.

[13] Even if a study aims at valuing different levels of a multilevel policy, it will not always be true that it implies a meaningful functional form in a generally applicable sense. Thus, for example, Vaughan and Russell (1982) examined the freshwater sport fishing benefits of three levels of mandated waste water treatment technology in the United States. But because of the hundreds or thousands of different source types affected and the different definitions that affect different types, the technology definitions (see Chapter 1, Annex 1-A) do not map neatly into removal levels. So the several benefit levels cannot be said to characterize either a concave or a convex benefit function.

[14] However, Crocker and Shogren (1991), in two separate contingent valuation exercises, find that WTP for improvements in visibility increases at a decreasing rate as visual range increases.

Figure 4-2. Total Costs of BOD Removal over a Range of 10 to 98 or 99 Percent Removal



Source: Kneese and Bower (1968, p. 53) (modified to label treatment levels). (Note: Cost figures are not corrected to current dollars since our interest is in function shape, not absolute value.)

On the cost side there is considerably more experience with the sorts of wastewater treatment alternatives that are the heart of proposals to the Bank. For a quite complete account, it is possible to go to environmental engineering texts such as Eckenfelder (1980, especially Chapter 14). The costs of these treatment processes tend to be presented in terms of volume of wastewater to be treated, and some manipulation is required to convert the cost estimates into terms more obviously useful in the context of cost-benefit analysis, i.e., percentage removal for given influent volume and characteristics. An early source in the economics literature that drew on engineering sources is Kneese and Bower (1968). Their Figure 8, attributed originally to Frankel (1965), is reproduced here as Figure 4-2. Note that it includes costs all the way down to 10 percent BOD removal—a very low-efficiency primary treatment. The upper end of primary treatment capabilities (screening, grit chambers, sedimentation assisted by electrolytes) is said to be 35 to 40 percent removal (Kneese and Bower, 1968). This section of the function seems especially relevant to the Bank, for many of the loan proposals examined in Chapter 2 dealt with the introduction of primary treatment.

The upper end of the primary capability level falls at or near the top of the concave section of the cost curve, and the inflection point separating negative from positive second derivatives follows closely in the figures. The secondary treatment portion of the curves, from roughly 40 to 85 or 90 percent, displays increasing marginal costs.[15]

[15] It is interesting to note that much of the later economics literature implicitly takes primary treatment as given and focuses almost exclusively on higher percentage removals. This seems to have resulted from the fact that the available data driving these models came from primary plus secondary plants, which became the norm as the Federal Water Pollution Control Act Amendments began to take effect. Examples include Fraas and Munley (1984), whose reported marginal costs begin at 80 percent removal; and McConnell and Schwarz (1992), who report marginal costs from 70 percent removal. The results of both studies confirm that marginal costs seem to rise very steeply for higher removal levels. A recent engineering study showing similar results for the removal of nutrients (nitrogen and phosphorus) is Vanrolleghem et al. (1996).

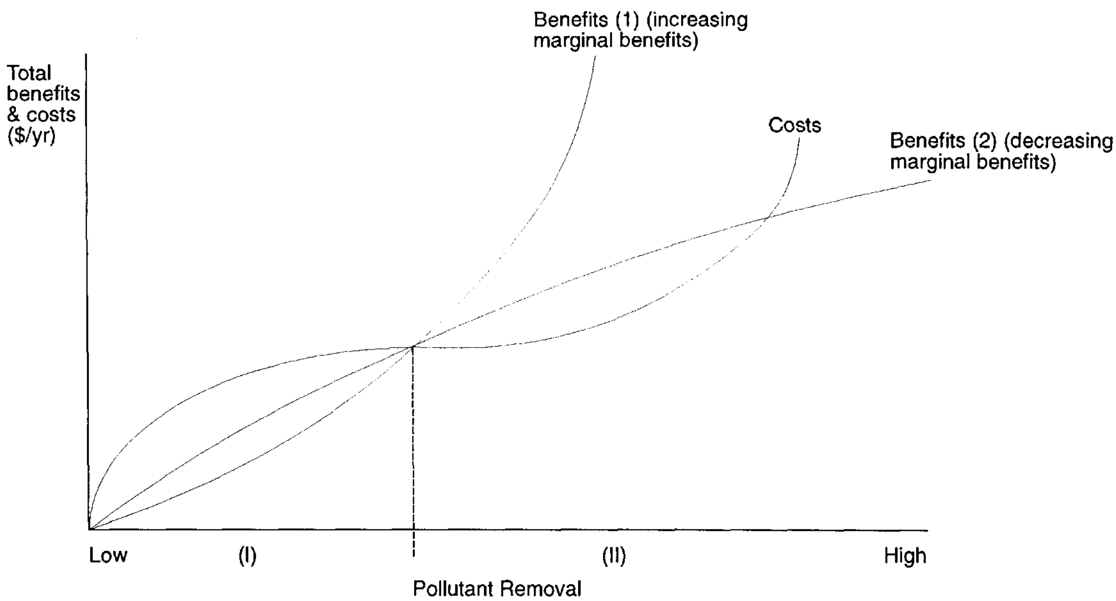
Phasing of Project Investment

One decision setting that troubles responsible Bank staff and that is rarely faced in the usual discussions of cost-benefit analysis is the phasing (or postponement) of project elements. A typical example of this phasing would be to apply for a loan to build a primary treatment plant now, with the promise to add a secondary treatment plant at some later time. A major reason for this strategy appears to be the shortage of capital for doing more in the near term. Specifically, what troubles Bank analysts is that the initial phase often cannot be justified by cost-benefit analysis; and even though the combined plant might be justifiable, there is no guarantee that the promise to go ahead later will be honored when the specified time comes.

The failure of a primary treatment plant to pass a cost-benefit analysis test can be understood intuitively by looking at the interplay of the classic cost relation for BOD removal with a benefit function, whether this function displays increasing or decreasing marginal benefits. Thus, the situation could look like that depicted in Figure 4-3, which represents a fairly small variation on the theme of Figure 4-1. In it, the cost curve in the primary treatment range (zone I) exhibits declining marginal costs of removal and lies above both illustrated benefit functions. (Function 1 involves increasing marginal benefits and function 2, decreasing marginal benefits.) In such a situation, building and operating a primary plant would not be justified under a cost-benefit analysis criterion. At higher removal levels (zone II), benefits do exceed costs.

For benefit function 1, the gap widens until presumably some very high removal is reached (off the figure). For function 2, the second intersection of cost and benefit functions occurs on the figure. Short of that intersection, between the two intersections there is an optimal removal level. In the absence of constraints on the country's (or agency's) capital budget, it would clearly be better to build the higher-removal plant corresponding to the optimum for the actual benefit function, assuming this is known. It would not be a good decision to build the primary plant as phase I if there were a significant probability that the second phase would never be built. A better solution would be to have the borrower reapply

Figure 4-3. Illustrating Cases in Which Primary Treatment Alone Would Not Be Justified



when it is ready to do the full sequence at one time. However, there may actually be stronger statements that can be made (statements depending on weaker, less restrictive assumptions). This section explores this possibility for a particularly simple version of a postponement model.

Some Assumptions and Notation

There are two possible strategies for the proposing country:

- Build a sequence of two projects, one (primary treatment) now and one (secondary treatment) "later;"[16]
- Postpone both parts of the project and build them at the same "later" time.

Separating the building of the two phases, which can be thought of as primary and secondary wastewater treatment, implies a cost penalty.

Annual benefits and O&M costs for each alternative (primary plant only, primary plus secondary) are known and are constant over the life of the project to which they apply.[17]

B_1 = annual benefits of the primary plant alone
B_{12} = annual benefits of the primary plus secondary plant
K_1 = undiscounted capital costs for the primary plant
K_2 = undiscounted capital costs for the secondary plant
K_{12} = undiscounted capital costs if both plants are built together
P = cost penalty for constructing primary and secondary plants at different times

So $K_1 + K_2 - P = K_{12}$

O_1 = annual undiscounted O&M costs for primary treatment alone
O_{12} = annual undiscounted O&M costs for primary plus secondary treatment

For purposes of discounting and summing, define:

u_1 as the "present worth factor" for a uniform stream of either costs or benefits over the first period of interest, i.e., the period of time between the building of the primary and secondary plants, when the latter is postponed;

$u_1 = \dfrac{(1+i)^t - 1}{i(1+i)^t}$, where i is the discount rate and t is the period of time involved;[18]

[16] Phasing could also be a matter of size alone: Build a plant with capacity F_0 in the current period and add capacity F_1 in a later period. This would not affect the discussion that follows. But the more usual case is the one described, in which plants are "ordered" by technical requirements, so that one must be first and the other second.

[17] The assumption of constant benefits does not consider the complications of growing populations, rising incomes, and changing tastes. These forces interact via their effects on the supply of wastewater and on the demand for better AWQ. The assumption of constant O&M costs also does not consider problems raised by unavoidable deterioration of project condition as time passes and parts wear out. (See the last section of this chapter.) Both the benefit and cost assumptions finesse any discussion of project "life" and of the problem raised by the fact that when the primary plant is built first, it can be expected to wear out first, so that some provision would have to be made for replacement to allow the secondary plus primary combination to reach its full lifetime.

[18] The present worth factor is the reciprocal of the capital recovery factor.

u_2 as the analogous factor applying to the period after the construction of either the secondary plant add-on or the primary plus secondary plant if both are postponed. This takes the stream of annual totals back to the time of the secondary plant's construction;

ρ as the discount factor connecting the present with the date of the secondary plant's construction (the time period of the phasing or postponement).[19]

$$\rho = \frac{1}{(1+i)^t}$$

Some Results

Given these assumptions and definitions, the present values of the net benefits for the phased and fully postponed options can be expressed as follows:

	Phased	*Postponed*
PV (costs)	$K_1 + \rho K_2 + u_1O_1 + \rho u_2O_{12}$	$\rho(K_1 + K_2 - P) + \rho u_2O_{12}$
PV (benefits)	$u_1B_1 + \rho u_2B_{12}$	ρu_2B_{12}
PV (net benefits)	$u_1B_1 + \rho u_2B_{12} - (K_1 + \rho K_2 + u_1O_1 + \rho u_2O_{12})$	$\rho u_2B_{12} - \rho(K_1 + K_2 - P) - \rho u_2O_{12}$

For phasing to be preferred to postponement, it must be true that

$$u_1B_1 + \rho u_2B_{12} - K_1 - \rho K_2 - u_1O_1 - \rho u_2O_{12} > \rho u_2B_{12} - \rho K_1 - \rho K_2 + \rho P - \rho u_2O_{12} \quad (4\text{-}1)$$

or, simplifying:

$$u_1B_1 - K_1 - u_1O_1 > - \rho K_1 + \rho P \quad (4\text{-}2)$$

which can be rewritten as

$$u_1B_1 > K_1 + u_1O_1 + \rho(P - K_1) \quad (4\text{-}2a)$$

which says that phasing will be preferred if the present value of gross benefits of the first phase is large enough to exceed the present value of total cost of the first phase, corrected by the discounted difference between the construction cost penalty for phasing and the primary plant construction cost.

Now, the situation of greatest interest is that in which the primary plant fails an isolated cost-benefit analysis, so that[20].

$$u_1B_1 < K_1 + u_1O_1 \quad (4\text{-}3)$$

When this is true, and when the construction-cost penalty for phasing is greater than the capital costs of constructing the primary treatment plant ($P > K_1$), phasing will never be preferred to postponement of both plants. [The term $\rho(P - K_1)$ in Eq. (4-2a) will just make the comparison worse.]

When the primary plant fails the cost-benefit analysis test but $K_1 > P$, so that $P - K_1$ is negative, the situation is more complicated. For a short postponement period, u_1 will not be

[19] For what follows, the period between phases and that until construction of the postponed pair are assumed to be equal.

[20] The Guaiba watershed proposal discussed in other contexts in this chapter also involves phasing. However, according to the proposal analysis, the first phase is justified on its own.

very large (because not many years will count); and ρ will not be much less than 1. If ρ is close to 1, then the negative contribution of the term $\rho(P - K_1)$ will be large relative to its size under a longer postponement. (See Table 4-2, which provides some illustrative values.) Thus it is possible that for shorter periods between phases, phasing will be preferred, while for longer periods, ρ becoming smaller but maintaining the assumption that $u_1B_1 < K_1 + u_1O_1$, postponement will be preferred.

It is possible, of course, that even if phasing *is* preferred to postponement, neither is actually desirable, i.e., neither passes a cost-benefit analysis test. For phasing to be justified in this sense, it must be true that

$$(u_1B_1 - K_1 - u_1O_1) + \rho(u_2B_{12} - K_2 - u_2O_{12}) > 0 \qquad (4\text{-}4)$$

If the first phase is not justified in isolation, then the net benefits of the second phase (the second parentheses in the equation) must not only be positive, they must be large enough to offset the first phase's negative present value. For a given set of net benefits for each phase, this requirement is more likely to be met when the time to the second phase is shorter, so that ρ is closer to 1.

None of the above seems surprising or counterintuitive. Rather, it just systematizes the suspicions within the Bank that phasing is generally not a good way to go unless the prospective borrower has developed a fairly complete design of the entire project and is reasonably sure that it can finance and execute a second phase that follows closely on the heels of the first. More specifically, it focuses attention on the size of the penalty for separating construction phases and on the length of time between phases. (See Chapter 9 for a sensitivity analysis of the impact of timing variations.)

Dealing with Time in the Analysis of Water Quality Investments

The way in which projects should be analyzed, at least where time is concerned, depends on the assumptions that are made about what happens to the project as time passes, and to what extent the future project managers will be able to intervene to change the situation. That is, generally and as already briefly noted, demand for a project's services can grow (or shrink, but growth seems most likely in the LAC setting) and its ability to deliver those services can decline as its physical condition deteriorates. Future project managers can often intervene to adjust project size and physical condition. However, these phenomena are usually assumed to not be

Table 4-2. Illustration of Values for Discount and Present Worth Factors

	$\rho \equiv$ **Discount factor** $\left[\frac{1}{(1+i)^t}\right]$		$u \equiv$ **Present worth factor** $\left[\frac{(1+i)^t-1}{i(1+i)^t}\right]$	
t	*i = 0.06*	*i = 0.12*	*i = 0.06*	*i = 0.12*
1	0.943	0.893	0.943	0.893
2	0.890	0.797	1.833	1.690
5	0.747	0.567	4.212	3.605
6	0.705	0.507	4.917	4.111
10	0.558	0.322	7.360	5.650
11	0.527	0.288	7.887	5.938
20	0.312	0.104	11.470	7.469
21	0.294	0.093	11.764	7.562
30	0.174	0.033	13.765	8.055
31	0.164	0.030	13.929	8.085

a concern when new investments are assessed, an assumption that, not surprisingly, makes for simpler analysis. But, as this section shows, the simplicity comes at a cost, which may either be large or small, depending upon the circumstances involved.

The Base Case

In classic cost-benefit analysis as most standard texts present it, the only role time plays is to stand between the present and the future and thus to imply the need to discount future cost and benefit (or damage) streams to reflect the productivity of the generalized investment opportunities available in the economy in which the project is embedded. (There is also the matter of consumer time preference, but it will not be necessary to consider it for present purposes.)

The classic approach does not account in any systematic way for changes in demand or in the condition of the project (i.e., the project's ability to provide benefits or avoid damage) over time. This approach need not imply that costs and benefits are treated as unchanging over the period defined as the project life, but usually it does. Sometimes, however, benefits are assumed to be exogenously changing, perhaps because demand for the project's output is predicted to grow. This is a very modest change and does not really add any difficulties to the analytical method; it just implies additional information costs and (depending on the pattern of growth predicted) increased difficulties in simplifying the present value expressions. In addition, in the base case, it is assumed that at the end of a project's life the project, in effect, simply falls apart. In this simplest case, all of the projects being compared have the same lifespan and share this same pattern of no decay followed by instant collapse.

Complications of the Base Case

A first complication, which is sometimes treated as a curiosity or a test of discounting "knowledge," is the recognition of different lifetimes for different projects in the set to be compared. An important result that allows a very simple way to deal with this is that when any shorter-lived project can be extended to equal the longest-lived project by making investments that have returns equal to the discount rate (i.e., unspecified investments in the economy at large), discounting over each project's *own* life gives the right comparison.

If reinvestments for lengthening a project must be done in new projects that have returns that are not equal to the discount rate, then they must be treated explicitly. This was, for example, the case faced by Krutilla and his colleagues who worked on the Hells Canyon Dam case over two decades ago (see Krutilla and Fisher, 1985). There, the shorter projects were nuclear power plants designed to supply the same amount of electric power as the dams in question. Since it was assumed that nuclear plant generating efficiency would be rising over the dams' lifetime, the productivity of new investment was growing. The model for analyzing the dams had to include an explicit series of alternative plants so that the total life of that series equaled the life of the dams.[21] It is important to note that this work treats condition as the simplest case does. None of the projects deliver worse service over time; instead, they perform flawlessly and then fall apart. (Or at least it is agreed that there is no interest in their condition.)

Another variant of the base case is the classic capacity planning model for water supply in which the challenge is to find the optimal pattern for adding capacity, given a predicted rate of growth of required output. A somewhat more complicated version involves less-than-infinite

[21] Harberger (1996) has written about how to deal with this situation in which the (N_L) life of the longer project is not an integer multiple of the life of the shorter (N_S).

penalties for not having a capacity at least equal to "demand" at every time period. Yet another version uses a demand function for output and worries about pricing, its effect on demand, and its relation to investment. What is important about these techniques in the context of this section is that they involve an even stronger assumption about condition: The facilities that are built are all implicitly assumed to have lives that exceed the horizon for the planning process. They are not replaced explicitly, although things get more complicated when price enters and capital costs must be annualized.

Also interesting is the fact that the papers in this literature usually assume a generalized cost function for adding supply capacity. If a real-world set of actual alternatives, involving different costs per unit of added capacity, is what faces the analyst, then the problem becomes one of finding an optimal sequence by some criterion. If, in addition, the investments made determine the prices subsequently charged, and if these prices affect demand levels, the new problem is intrinsically dynamic and cannot be solved by elegant analytical methods. It must be attacked by dynamic programming. The reason is that investment decisions for the current period affect the payoffs in future possible decisions.[22]

What if project condition is *not* taken to be constant; i.e., what if the project's ability to produce benefits is recognized as declining over time to the horizon? First, it is necessary to be clear about why a decline in condition should ever be assumed. Is not the maintenance in O&M (or the M and R in operation, maintenance, and repair, OMR) spent on maintaining condition? Is that not the whole idea of preventive maintenance? The answer seems to be that the kind of maintenance that can realistically be done on an operating project cannot routinely deal with everything that can be affected by operation, exogenous shocks, and sheer age. Thus, a ship slows down over time as its bottom is fouled. A lock gate in a river navigation system can be damaged in a cumulative way by ice and collisions. Key mechanical or electrical parts of a treatment plant that are difficult to change can wear out.

Because the act of correcting the cumulative problem is under the control of the decision makers—in both timing and extent (i.e., how much, and when, improvement is bought)—the situation created looks like the facility investment problem of ordering an array of options over time. But an added twist may be introduced by the assumption that the costs of the options are not independent of the timing of their application. That is, if the cost of a maintenance item that reads "increase ship speed at 'standard' power by 4 knots by cleaning the bottom" is a function of how badly the bottom is fouled when the work is done, then a dynamic link has been introduced. The ordering of such a set of projects is a dynamic programming problem. It is symmetric with the link through price on the benefit side.

However, there are ways to avoid the creation of a dynamic programming problem. For example, decay of condition may be recognized—just as is increase in demand—but it can be taken to be entirely exogenous. Specifically, it can be assumed that it is not possible to invest to fix it beyond such efforts as are included in maintenance. This "works" as long as all alternatives are similarly left alone, and as long as none of them would become a "ruin" (cease to function) short of some horizon. This horizon must be far enough in the future that society does not care about the differences in condition among the alternatives that are bequeathed to the future beyond it. Another alternative, with more arbitrary assumptions required, and with implications for the way the analysis must be carried out, is to allow for deteriorating condition, leading to facility breakdown with some increasing probability per period. The rule for dealing with condition is taken to be "fail-then-fix." But, in order to "see" failures, the future must be simulated, so that Monte Carlo methods become part of the required toolkit. Many alternative paths through time are simulated using the "fail-then-fix" rule, for each initial investment

[22] All of this is discussed in Russell and Shin (1996a and 1996b).

alternative. The choice among the alternatives depends on applying some decision rule to the information generated, the simplest being to pick the project with the lowest expected sum of costs (including the "fix" costs), and damage from the failures.

It seems that there are at least two situations in which the assumption of exogenous decay may not be satisfactory, with the result that analyzing an investment requires a truly dynamic approach. The first of these arises if a realistic assessment of condition decay rates says that at least some of the set of investment alternatives will end up as ruins before a horizon that is agreed to be far enough in the future that society can ignore the posthorizon future. [It is possible to adopt a Harbergerlike view and say that the only problem is defining a project or partial project that will take each of the ruined projects out to the longest lifetime (the shortest agreed-to horizon). This is not a dynamic problem as long as the cost of that extender is not dependent on what has been allowed to happen to the original facility.]

The second possibility is that the choice of original investment projects reflects some choice of ease of maintenance. This could be highly relevant in a developing country setting, where maintenance of all kinds is widely recognized to be problematic. If it is possible to in effect invest in ease of future maintenance when a new facility is built, it will be necessary to see how this plays out over time in order to make a valid comparison with different levels of this feature.

What should the practical implication of all of this be for IDB project analysis and review? One obvious and appealing answer is: "Nothing. Things are complicated enough as they are This dynamic business is just an academic frill that should be ignored by practical people." But before settling on this answer, it is worth giving some thought to the following three points. First, project maintenance really does seem to be a major problem for developing-country government agencies. (See, for example, Howe and Dixon, 1993, who provide a good deal of anecdotal evidence to this effect.) Second, it is almost certainly possible to design into waste-water treatment plants different levels of "robustness," in the sense of resistance to the ravages of time, weather, and poor operation. Encouraging the use of more robust designs might be as important to the attainment of positive net benefits in the long run as any of the more familiar complications, such as looking at alternative plant locations. And, third, the Bank could find or develop a dynamic programming package capable of handling exactly the question posed in this section and could then begin a process of encouraging its voluntary use, with the ultimate aim of making it a required part of the cost-benefit analysis of water treatment projects.[23]

CONCLUSION

Even leaving aside the very great obstacles to persuasive benefit estimates, the cost-benefit analysis of something as complicated as a major investment in AWQ improvement is potentially a great deal more difficult than the usual discussions imply. Simple models of the water body can be misleading and appear to make possible inferences that are really only allowed when complete knowledge of sources is combined with sophisticated aquatic models. While functional forms remain unknown, the threat exists that the level of AWQ improvement chosen for analysis is not the best attainable and may even approach quite inefficient levels. A phased approach may be assumed to be viable even though chances are that unless the primary

[23] Although some popular software packages such as Microsoft Excel, Mathematica, and MatLab either inherently or through plug-in components have the ability to perform dynamic programming, it is not clear that they can handle the type of analysis described. Even if a suitable dynamic program is not available as an off-the-shelf commodity, they are cheap to build and run for simple settings.

treatment phase itself passes a cost-benefit analysis, postponement of both phases would be better. And finally, everything relevant to project "robustness" may be assumed away to avoid having to deal with dynamics (in the strict, technical, sense).

At the same time, there are temptations for analysts who are really advocates. These temptations include ways of transforming costs into benefits by counting interregion transfers, conveniently confusing the notion of surplus, and pushing the envelope in identifying the sources of surpluses by assuming the existence of a huge beneficiary population (and by ignoring problems associated with management of open-access resources). The long and short of it seems to be that a review of project justifications requires considerable vigilance and subtlety if the ultimate project results are really to be beneficial in the aggregate efficiency sense.

Annex 4-A

Cost-Benefit Analysis of Projects Involving Sewering and (Possible) Associated Wastewater Treatment Plants[24]

A common situation lying behind more than a few investment proposals analyzed by the Bank is the perceived need to provide sewering to one or more neighborhoods currently unserved by such infrastructure. This may or may not be associated with a project to provide piped-in potable water to the same households. (That is, piped water may already be available. It would be unusual for an applicant to consider sewering a neighborhood without piped water.) The effect of the sewering will generally be threefold. First, it will allow an increase in the use of water by each household via increased convenience of disposal. Second, it will reduce some localized (neighborhood) damage resulting from inadequacies in the current systems for wastewater disposal. This damage may be quite large if latrines of some design that traps human waste are not available so that the neighborhood is effectively laced with open sewers. Or it might be quite modest if there are well-functioning septic systems with regular pumping of tanks or with suitable soil for leachfields. Third, it will increase pollution discharges to the water body at the end of the system into which the neighborhood sewer discharges.

There are two economic questions that require answers in such situations:

- Is the sewer system justified, even taking into account the new damage to the receiving water?
- Would it be an even better idea to add a wastewater treatment plant to the end of the sewer to reduce the damage to the receiving water?

To explore these questions in a very simple setting assume:

- That all neighborhood environmental damage existing in the without-sewer case is done away with by sewering;
- That the private benefits arising from the greater convenience of in-home water use and the associated greater use of water are straightforwardly measured using the predicted shift in the demand curve for potable water in the neighborhood;
- That the wider damage under each situation (no project, sewer only, sewer plus treatment plant) is reliably estimated;
- That there is only one practical treatment plant alternative, having a known cost and effect on water quality below its outfall.

[24] This annex draws heavily from Annex A in Vaughan and Ardila (1993) and includes an attempt to clarify the discussion to be found in the loan application documentation for BR0073, Guaiba Watershed Environmental Management Program—First Stage.

Then define:

Benefits and Costs	No sewer (before project)	Sewer only (no (treatment plant)	Sewer plus treatment plant
Private water use benefits (net of private costs)	0	NB_S	NB_S
Neighborhood damage	D_0	0	0
Damage in receiving water body associated with water use by neighborhood[25]	R_0	R_1	R_2
Treatment plant cost	0	0	C_2

The answer to the first economic question comes from the comparison of $NB_S + D_0$ with $R_1 - R_0$, where to make the problem interesting, it must be true that $R_1 > R_0$. If $NB_S + D_0 > R_1 - R_0$ (all in appropriate present-value form) then the sewering is justified.

But the second question asks if it is possible to do better by adding a treatment plant. This is addressed by first writing the comparison that allows a judgment on the treatment plant project itself: If $NB_S + D_0 > R_2 - R_0 + C_2$, then the treatment plant option is justified. (The right-hand side involves first the net regional damage, $R_2 - R_0$, which is negative if $R_2 < R_0$ and 0 if $R_2 = R_0$, which is to say the treatment plant is designed to just "make up for" the sewering. To this is added the cost of the treatment plant, so the right-hand side is the net cost of the plant.)

For the treatment plant option to be preferred to the sewer-only option, the net benefits of the former must exceed those of the latter (and must themselves be positive) or:

$$NB_S + D_0 - R_2 + R_0 - C_2 > NB_S + D_0 - R_1 + R_0$$

which reduces to:

$$-R_2 - C_2 > -R_1 \text{ or } R_2 + C_2 < R_1 \text{ or } C_2 < R_1 - R_2$$

That is, the cost of the treatment plant must be less than the damage reduction it achieves (its benefit). Notice that the same answer is obtained if $R_0 = R_2$ is assumed, for the important comparison becomes

$$NB_S + D_0 - R_2 + R_2 - C_2 > NB_S + D_0 - R_1 + R_2$$

or

$$-C_2 > R_2 - R_1 \text{ is required}$$

or

$$R_1 - R_2 > C_2$$

Now it could be true that the treatment plant option is preferable to the sewer-only option, but that neither is justified. This would involve:

$$R_1 - R_2 > C_2$$

and

$$NB_S + D_0 + R_0 - R_2 - C_2 < 0$$

or

$$R_1 - R_2 > C_2 > NB_S + D_0 + R_0 - R_2$$

or

$$R_1 > C_2 + R_2 > NB_S + D_0 + R_0$$

which says that if $NB_S + D_0$ (the private benefits plus neighborhood benefits) are small enough, sewering plus treatment will not be justified.

[25] It is assumed that some of the neighborhood's wastewater affects the receiving water body, even in the absence of sewers, possibly via stormwater flushing of open sewers or infiltration of septic system leachate.

In the Guaiba watershed application, however, a strong test of the treatment plant option is suggested. This test requires that the treatment plant alone be justified by the improvement it creates relative to the base (no sewer) case; that

$$R_0 - R_2 > C_2 \text{ (where } R_1 > R_0 \geq R_2 \text{ is assumed)}$$

If the plant passes the strong test, then if there are *any* private and local benefits, the treatment plant-plus-sewer project must pass the cost-benefit test. Thus:

$$R_0 - R_2 > C_2$$

and

$$NB_S + D_0 + R_0 - R_2 < C_2$$

implies:

$$R_0 - R_2 > C_2 > NB_S + D_0 + R_0 - R_2$$

or

$$0 > C_2 + R_2 - R_0 > NB_S + D_0$$

which is impossible if NB_S and D_0 are positive. Therefore, if the treatment plant alone can be justified solely on the basis of its effect on the pre-sewer quality of the receiving watercourse, and if it is reasonable to assert that there will be private and neighborhood benefits from the sewer project, then there is no need to estimate those components of the total project's effects.

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Chapter 5

Tools of the Trade: Decision Support Systems and Water Quality Models

This chapter discusses two different but closely related aids to water resource planning and investment decisions: decision support systems (DSSs) and water quality models (WQMs). The discussion begins with an overview of DSSs, including a brief review of their history and current status. This overview is followed by a list of some decision support systems currently available. The examination of DSSs is followed by a similarly structured examination of water quality models.

For introductory purposes it is sufficient to say that in the realm of water quality planning or investment analysis, a DSS amounts to a structure in which some form of WQM is embedded. The latter is, again roughly speaking, a mathematical representation of the processes that go on in a water body or linked series of water bodies. The representation transfers, dilutes, and transforms pollutants entering the water and traces the responses of at least part of the aquatic ecosystem contained in the water. The structure that constitutes the DSS may also include one or more of the following: databases allowing the WQM to be tailored to fit several or many different specific situations; economic components that reflect the costs of changing pollution loads or the value of damage due to those loads; sets of alternative possible effluent or ambient water quality constraints (standards); an optimization routine to allow efficient identification of the best way to approach a water quality situation; and a way of organizing and displaying results designed to be useful to decision makers, whoever they may be. Thus, jargon aside, the descriptor DSS may contain a suite of software packages that offers either more or less than the cost-benefit analysis framework set out in Chapter 3.

After the discussions of decision support systems, attention is focused on the WQM component itself. Since this label actually applies to a very wide variety of quite different structures focused on very different problems, an effort will be made to note the relevant distinctions and to locate the models of interest in this guide within the larger space of possibilities. Then the briefest of historical sketches is provided to illustrate how the field has evolved over the past 75 years. A description and comparison of two specific models, both used in later illustrations, rounds out the section on water quality models.

Those illustrations form the last part of this chapter. One illustration is of the effect of complicating the WQM structure so that the ambient quality effect of a particular source depends not only on that source's discharge but also on the existing level of pollution in the water that receives those discharges, a possibility noted in Chapter 4. The second uses the simpler of the two models and its associated decision support system to provide examples of vector-constrained, basinwide optimization (cost minimization subject to given wastewater quantities and ambient water quality constraints). These examples explore the effects of such changing background conditions as the initial arrangement of sources and the levels of the ambient quality constraints.

First, however, since these topics may appear to some readers as inappropriate to a good-practice guide written by and largely for economists, it might be useful to say a word about why they have been included. The water quality modeling terrain is not easily mapped, even by those who frequently traverse it, and it can be an especially confusing place for those unfamiliar with the dialect. Then why bother with it? Why not continue to leave it to the experts?

First, developments in the fields of computer technology and water quality modeling are rapidly making these programs more accessible to a greater number of people. This increased accessibility could be a tremendous advantage to an institution seeking to do analyses of water quality-related investment projects at low cost. In particular, it holds out the possibility of being able to place individual projects in a regional context in a meaningful way, and at a reasonable cost.

However, the Bank cannot afford to be an uninformed consumer of these models or their output. Indeed, the Bank might even want to proactively become involved in the promotion and design of water resource decision support systems. (The World Bank has already done this. See the later discussion for a brief description of the result.) This would provide an opportunity to highlight (and improve) the decision-making processes that occur prior to the Bank input, i.e., selection of a project proposal.[1] Proactive involvement would also mean that the Bank could enjoy some degree of control over, or at least input into, the design process for a future DSS, resulting in a design that is more useful to the Bank. Indeed, the experience of those involved in preparing this document with several off-the-shelf DSS and WQM versions suggests that it may be dangerous to naively trust these products for the analysis of multimillion-dollar investments. A major reason is the old problem of unreadable or inadequate documentation and user instruction. It is all too often impossible to determine where a particular number, for example a treatment plant capital cost, came from—either its original source or the applicable functional form, or both. Second, for exercises supposedly aimed at producing flexible, widely applicable frameworks, DSSs and WQMs are surprisingly loaded with inflexibilities. These comments are further illustrated in the discussions that follow.

Finally, this chapter in effect examines further some of the obstacles to true, basinwide or regional optimization. In this sense, the chapter can be viewed as an expansion of parts of the discussions in Chapters 3 and 4 that cover some of these problems.

DECISION SUPPORT SYSTEMS

This overview of decision support systems is divided into four sections. First, the structure or components of DSSs are examined, providing an organizational definition of a DSS and pointing out the close analogy to the regional cost-benefit analysis framework. Second, the potential uses or benefits of DSSs are briefly explored. Third, the history and current state of DSSs are discussed. Finally, a few specific examples of DSSs are described.

Structure

One way to describe a DSS is to list the three general components from which it is constructed. These, according to Reitsma et al. (1996), are state information, dynamic or process informa-

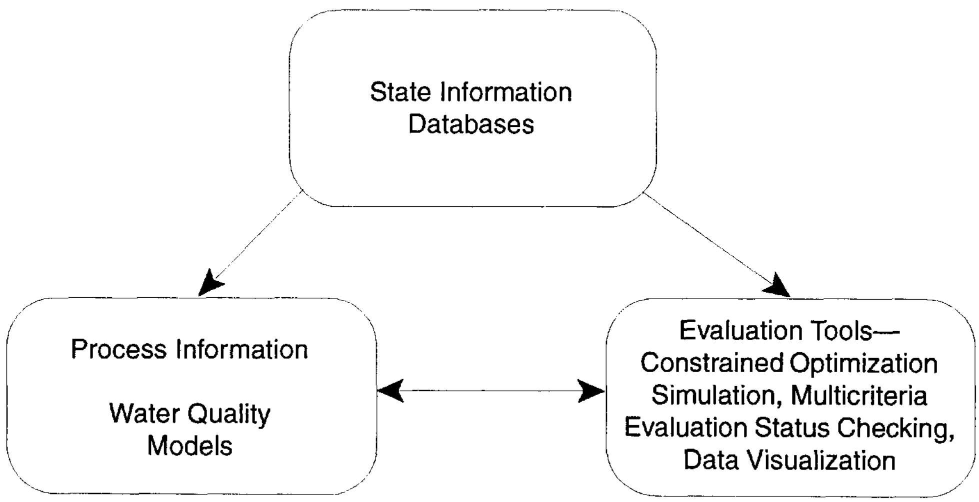
[1] Jamieson and Fedra make a similar argument on behalf of the public: "There are other advantages to be gained from adopting the DSS approach. For instance, with growing concern about the environment, public accountability for decisions affecting the water environment is assuming increased importance. Openness about how decisions are reached is greatly facilitated through the use of a DSS in which the effects of alternative development policies can be explained and their impacts assessed in a form which can be comprehended by the non-expert." (Jamieson and Fedra, 1996a, pp. 164–165)

tion, and plan evaluation tools. Figure 5-1 is a schematic view of these components. The reader can compare this figure with the schematics in Chapter 3 and see that the latter might be taken to be special cases of the more general, but not necessarily more useful, DSS framework. It seems useful for the Bank to know that this parallel terminology exists.

State information consists of the data that represent the water resource's condition or state at any point in time. The actual DSS component in which the state information is generally housed is some form of database program. The sophistication achieved by current database technology, including relational databases, allows a DSS, in principle, to offer powerful, relatively simple means for importing, organizing, manipulating, and exporting vast amounts of data. In fact, STREAMPLAN, one of the DSSs discussed in greater detail later, resides entirely within the Microsoft Excel database program. Perhaps the most important database advance for DSSs has been the advent of geographic information systems (GIS) technology. A GIS is essentially a database program that is designed expressly for use with spatial or geographic data and includes a wide variety of tools for collecting, storing, retrieving, transforming, and displaying data about spatial or geographical objects and the nonspatial attributes of those objects. Another DSS, BASINS, also discussed later, resides entirely within Esri's ArcView GIS program. A DSS integrating GIS functions is sometimes called a "spatial DSS" or SDSS (Valdes and Restrepo, 1996, 1997).

The set of functions that represent the principles governing the water resource's behavior over time comprise the *process* (or *dynamic*) *information* component of a DSS. Process information typically comes in the form of a model that simulates the physical, chemical, and biological processes or attributes of the resource, i.e., a water quality model. These models are often either potentially or originally stand-alone software programs. For example, BASINS employs two different stand-alone water quality models: QUAL2E (Version 3.2) and TOXIROUTE; and one nonpoint source model, HSPF (Version 11).

Figure 5-1. Schematic of Decision Support System Structure



The third component of a DSS, the *evaluation tools*, consists of utilities that transform raw system data into information relevant for decision making. The following passage from Reitsma et al. indicates something of the range of possibilities here:

> *Plan evaluation tools are those tools that filter, modify, and present parts of the system's data in a form suited for alternative evaluation and decision making, thereby transforming a system's data into information. At least two rather different forms of plan evaluation can be recognized: formal and informal tools. Formal plan evaluation involves the use of mathematical techniques for the systematic evaluation of various choice alternatives. . . . Unlike formal plan-evaluation tools, informal ones do not rely on mathematical formalism to synthesize the overall utility of choice alternatives. Instead, they are aimed at representing the various relevant characteristics of the options whereas the combination of these into utility or attractiveness judgments [is] left to the human interpreter.* (Reitsma et al., 1996, pp. 33.11–33.12)

Again, the reader will note the strong parallelism with cost-benefit analysis that is evident despite the different jargon.

Potential Benefits

DSS models have the potential to provide two broad types of benefits. First, they offer a technical framework for integrating all of the different components of a water resource planning decision under one "roof." As Jamieson and Fedra put it:

> *Although the principles of integrated river-basin management have been aspired to in many countries, more often than not the problems have been considered in a piecemeal fashion, with experts from different disciplines using separate models (water resources, surface-water pollution control, groundwater contamination, etc.), to tackle parts of the overall problem in a reactive way. River-basin agencies frequently invest large financial resources in uncoordinated modeling activities, conducted either internally by in-house staff or externally by consultants, with the inevitable incompatibilities between the various approaches. Moreover, advances in scientific understanding frequently do not find application in practice through lack of a framework in which scientists can appreciate the real problems that need to be addressed, and at the same time managers cannot always find a mechanism for absorbing this new understanding. In recent years, there has been growing interest in the use of decision-support systems (DSS) to address some of these issues.* (Jamieson and Fedra, 1996a, p. 164)

That this is not just an abstract concern is illustrated by a recommendation from a scientific peer review panel chaired by one of the authors of this report (Russell). This panel was commenting on a technical progress report prepared by the staff of the South Florida Water Management District and describing the staff's work on understanding and devising solutions for the environmental problems of the great wetlands of South Florida that lie upstream of the Everglades National Park. After reviewing a 10-chapter report, with each chapter aimed at a different facet of the problem, the committee wrote:

> *The Panel recommends more effort be put into integrating process research, model building and monitoring. The general message . . . is . . . that the integration of individual models should be pursued. We do not mean to imply . . . any lack of skill or effort on the part of modelers. But we do sense that . . . [the] construction of new,*

narrow-purpose models to address new questions and problems [is a] common response and not [one] that make[s] the best use of talent and resources. (Peer Review Panel 1998. Final Report, p. 55)

Second, DSSs can increase the practical quality of these individual component models by improving their accessibility for nonexperts.

In essence, the DSSs integrate water resources models within a system that makes it easier to use and understand model outputs for better decision making. . . . Models incorporated within the DSS framework are often better adapted for decision making than those models that are designed only for the technical specialist. (Walsh, 1993, p. 159)

With increasing emphasis on exploiting research results for the benefit of society, focusing on the quality of the decision also reveals the relevance and worth of scientific investigations designed to enhance understanding of the processes involved. Moreover, by maintaining upgrade paths for new model components, this improved knowledge can be incorporated and utilized where relevant. (Jamieson and Fedra, 1996a, p. 164)

History and Current State of Decision Support System Technology

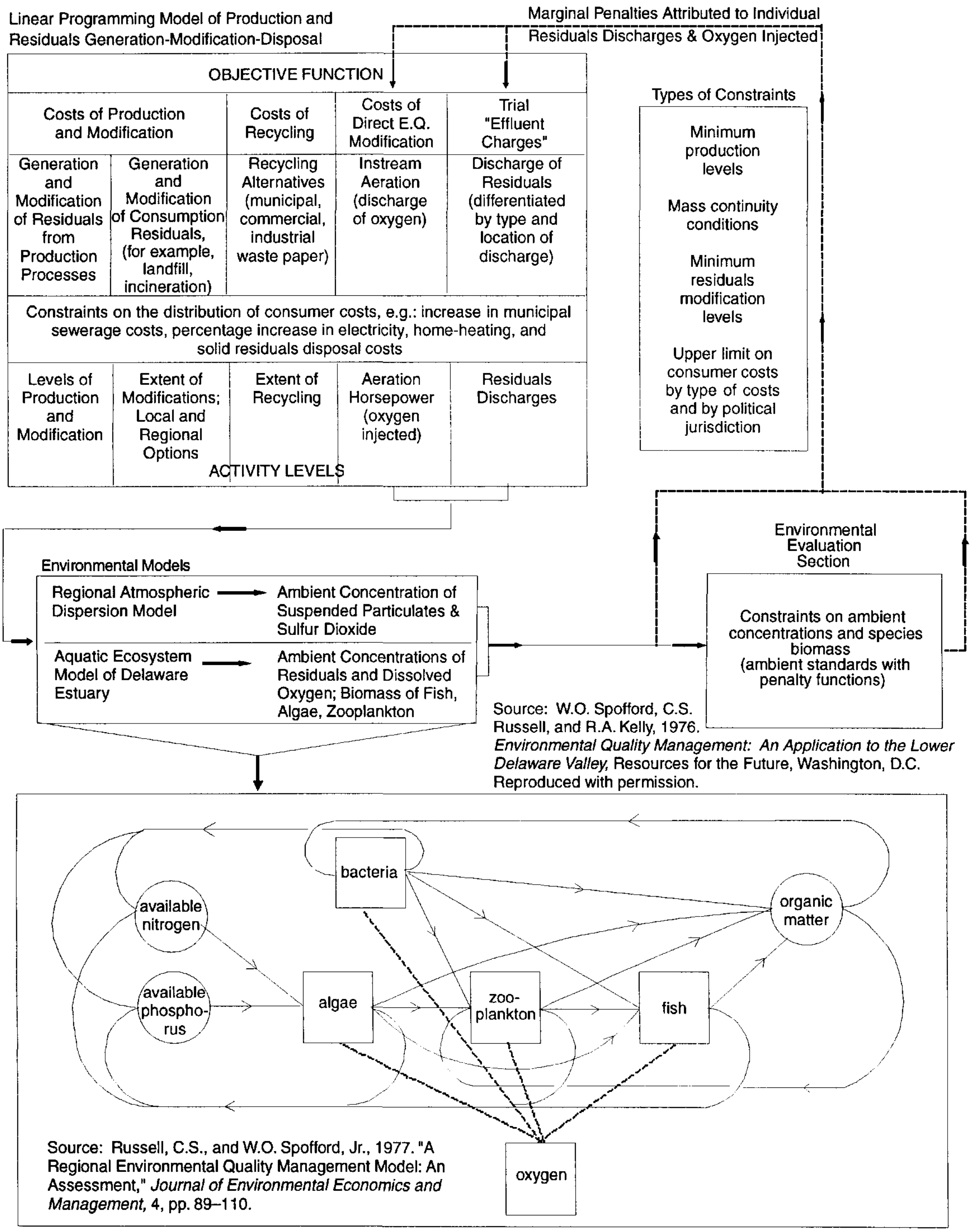
DSS programs have been around essentially since the advent of the computer, although much of the time their use was confined to the obscurity of military and, to a lesser extent, business applications. Although DSS technology was slowly being adopted into other highly specific, complex task environments, including water resource management, it remained largely unnoticed, in part because the choice of policy approaches in U.S. water pollution control law (technology-based effluent standards) made models linking economics (cost of discharge reductions) and ecology (ambient quality results) seem less than vital. But some of the earliest work of the DSS type predated the Federal Water Pollution Control Act Amendments of 1972 by more than half a decade. And by the time that choice of policy instruments was made, a substantial effort to build regional, multimedia pollution control models was under way at Resources for the Future (RFF) in Washington, D.C. These might well be seen as the ancestors of more recent modeling enterprises (see Russell and Spofford, 1977; Bower, 1977). Some idea of what the most ambitious of the RFF efforts, the Delaware Estuary Model, looked like, is provided by Figure 5-2, with more detail about size and complexity set out in Table 5-1.

More recently, the development of DSS tools has been aided by advances in computation, information availability,[2] computer graphics, and optimization algorithms. Nonetheless, actual, as opposed to demonstration, applications are very rare.

The Delaware model notwithstanding, the traditional target audience for water resource-related DSSs has been resource managers and as a result there has been an emphasis on water *quantity* and the resolution of conflicts over water use (e.g., conflicts over recreational vs. energy generation) and the capability of addressing "short-term operational decision making (daily and hourly operations of the dams and power plants) as well as more infrequent long-term planning decisions (new construction, long-term environmental planning, and so on)." (Reitsma et al. 1996, p. 33.2) This focus may also reflect the early part of a progression in the development

[2] When the Delaware model was constructed, there was essentially no readily and publicly available information on pollution discharges, even from publicly owned wastewater treatment plants.

Figure 5-2. Schematic Diagram of the Lower Delaware Valley Residuals—Environmental Quality Management Model



Source: W.O. Spofford, C.S. Russell, and R.A. Kelly, 1976. *Environmental Quality Management: An Application to the Lower Delaware Valley,* Resources for the Future, Washington, D.C. Reproduced with permission.

Source: Russell, C.S., and W.O. Spofford, Jr., 1977. "A Regional Environmental Quality Management Model: An Assessment," *Journal of Environmental Economics and Management,* 4, pp. 89–110.

Table 5-1. Lower Delaware Valley Model: Residuals Generation and Discharge Modules

Module Identification	Size of Linear Program Rows	Columns	Discharges	Description[a]
MPSX1	286	1,649	130	Petroleum refineries (7) Steel mills (5) Power plants (17)
MPSX2	741	1,482	114	Home heat (57) Commercial heat (57)
MPSX3	564	1,854	157	Over 25 μg/m³ discharges (75)[b]
MPSX4	468	570	180	Delaware estuary sewage treatment plants (36)
MPSX5	951	1,914	88	Paper plants (10) Municipal incinerators (23) Municipal solid residuals handling and disposal activities
MPSX6	229	395	117[c]	Other Delaware estuary industrial dischargers (23) In-stream aeration (22)
Total	**3,239**	**7,864**	**786**	

Source: Russell and Spofford (1977).

[a] The numbers in parentheses indicate the number of plants or activities that are included in the module with residuals management options.

[b] Industrial plants whose gaseous discharges result in maximum annual average ground-level concentrations equal to or greater than 25 μg/m³. To determine this group, all stacks (point sources) in the U.S. Environmental Protection Agency's Implementation Planning Program's (IPP) inventory of gaseous emissions were considered except those in MPSX1 (petroleum refineries, steel mills, and thermal power plants). The maximum annual average ground-level concentrations of sulfur dioxide and suspended particulates were computed for each stack. For all stacks at the same x-y location (i.e., same plant), the maximum ground-level concentrations were added together. Those plants associated with maximum ground-level concentrations, for either sulfur or suspended particulates, equal to or greater than 25 μg/m³ were placed in this category.

[c] Does not include 22 oxygen discharges from in-stream aerators.

of DSSs from the relatively straightforward and noncontroversial to the more esoteric and more controversial. In any event, the focus here is on water quality DSSs.

Examples of Basinwide Decision Support Systems

This section briefly describes six DSSs focused on basinwide water quality.

BASINS

BASINS, or Better Assessment Science Integrating Point and Nonpoint Sources, Version 2.0, was developed by the U.S. Environmental Protection Agency's Office of Water to "promote better assessment and integration of point and nonpoint sources in watershed and water quality management" (BASINS User's Manual, Lahlou et al., 1998, p. iv) by facilitating the examination of environmental information, supporting the analysis of environmental systems, and providing a framework for examining management alternatives. BASINS has six components, including (a) extensive water quality-related databases organized by watershed for the entire United States; (b) assessment tools that allow the user to generate a variety of characterizations of water quality on either a large scale (e.g., state, region, or nation) or small scale (i.e., river, river reach, or watershed); (c) utilities to facilitate the organization and evaluation of data, including user-provided data; (d) a reporting utility that allows the compilation and output of information on selected watersheds; (e) two water-quality models, QUAL2E and TOXIROUTE; and (f) a non-point source model and postprocessor, which allows integrated assessment of watershed loading and transport. Taken together, these components allow users to distinguish and prioritize bodies of water on the basis of water quality; gather information on the magnitude and significance of point and nonpoint sources; evaluate and compare the potential effects of different control strategies; and generate tables, graphs, and maps that aid in the visualization and communication of environmental conditions (Lahlou et al., 1998).

BASINS' orientation is essentially that of a nationwide screening model.[3] (Screening-level models are typically rather simple models with limited data requirements that are used to indicate if and where potential problems might exist.) As such, BASINS has the capacity to organize and manipulate a substantial amount of already existing, easily accessible water quality-related data. In addition, it has the advantage of running on a PC. However, for purposes of regional cost-benefit analysis, BASINS suffers from the absence of cost and socioeconomic data and related modeling capabilities. BASINS can be downloaded from the EPA's website at *http://www.epa.gov/ostwater/BASINS/index.html*, which also contains additional information on BASINS and the data available for use with it.

STREAMPLAN

STREAMPLAN, or Spreadsheet Tool for River Environment Assessment Management and Planning, Version 1.0, was developed by the International Institute for Applied Systems Analysis (IIASA) to "assist in the evaluation of alternative strategies for water quality management at the river basin level." (De Marchi et al., 1996, p. 1) Pollution control strategies that can be considered include uniform emission reduction and effluent-standard-based strategies, ambient water quality criteria and least-cost strategies, total emission reduction under minimized costs, mixed strategies, local and regional policies, and strategies with economic instruments. STREAMPLAN, which became available in 1996, was created using the Visual Basic capabilities of the Microsoft Excel 5.0 spreadsheet software and it can only be run from within Excel 5.0. The basic elements of STREAMPLAN are hydraulic, water quality, socioeconomic, and optimization models, which are organized into six "workbooks." An optional workbook can also be linked to STREAMPLAN to generate cost information and effluent wastewater quality for treatment alternatives. STREAMPLAN can be downloaded from IIASA's website at *http://www.iiasa.ac.at/Research/WAT/docs/stream.html*, which also contains more detailed information on STREAMPLAN. STREAMPLAN is used in this chapter to illustrate the conduct of regional, ambient quality, vector-constrained cost minimization.

DESERT

DESERT, or Decision Support System for Evaluation of River Basins Strategies, Version 1.1, was also developed by IIASA. The package, which became available in 1996, uses dynamic programming to find the least-cost solution among wastewater treatment alternatives for multiple point sources along a river system, subject to ambient water quality constraints for the system.[4] In addition to the optimization algorithm, DESERT contains a data-handling module with a dBase III style database engine and assorted water quality simulation and calibration capabilities.

While similar to STREAMPLAN in several ways (for example, it contains the same set of wastewater treatment plant alternatives as STREAMPLAN), it appears overall to be somewhat less ambitious. For example, it lacks the water pricing feedback possibility embedded in

[3] However, it seems to be intended that improvements in the program and data will increase its planning and management relevance.

[4] The technique is the same as that used in dynamic problems, in that possible branches of the "possibility tree" are pruned by induction, but the problem is a spatial rather than a dynamic one.

STREAMPLAN and does not allow for discharges of phosphorus or follow it in the stream. While the user's manual is no more helpful than the average of such documents, it appears that there are no wastewater transport costs in DESERT, which implies that the location "optimization" would be of limited use in real planning problem settings. DESERT can be downloaded from IIASA's website at *http://www.iiasa.ac.at/Research /WAT/docs/ desert.htm,* which also contains more detailed information on DESERT.

RIVERWARE

RIVERWARE is a general river basin modeling package developed by the Center for Advanced Decision Support for Water and Environmental Systems (CADSWES). It allows users to build and manage river basin models. RIVERWARE is designed with extensive water quantity scheduling, forecasting, and planning capabilities for use by water resource managers. While it also has the capability to model water quality in terms of temperature, total dissolved solids, and dissolved oxygen, it is not designed to do the sort of cost-effectiveness or cost-benefit analysis needed for water quality investment work. Rather, its strengths seem to lie in river basin flow and storage management. Thus, it is currently being used by the Tennessee Valley Authority for simulation and optimization for daily scheduling of more than 40 reservoirs and hydroplants, while the U.S. Bureau of Reclamation is using RIVERWARE for long-term policy and planning and midterm operation of the Colorado River. In addition, an interagency team including the Army Corps of Engineers, Bureau of Reclamation, and the U.S. Geological Survey (USGS) is currently developing a daily operations model of the Upper Rio Grande basin using RIVERWARE. More information on RIVERWARE is available at the CADSWES website at *http:// cadswes.colorado.edu/.*

WATERWARE

WATERWARE is the outcome of a collaborative research program involving three universities, a research institute, and two commercial companies. The aim was to develop a comprehensive, easy-to-use DSS for integrated river basin planning that would be capable of addressing a wide range of issues, including determining the limits of sustainable development; evaluating the impact of new environmental legislation; deciding what, where, and when new resources should be developed; assessing the environmental impact of water-related development; and formulating strategies for river and groundwater pollution-control schemes. This system has the capacity not only to predict what is likely to happen under different scenarios but also to offer expert advice on decisions that need to be made. While modeling techniques are used for predictive purposes, the artificial intelligence is provided by a mixture of optimization techniques and expert systems. These are claimed to allow the user to evaluate options, draw conclusions, and determine appropriate actions.

For modeling surface water pollution, the DSS contains a one-dimensional, stochastic river water quality model. The waste-load allocation module acts on the model output using a combination of heuristic search and linear programming to identify the least-cost solution for achieving the prescribed river quality standards by selecting the appropriate technologies for each effluent discharge. Alternatively, the module can identify the most effective allocation in terms of river quality improvements, for a fixed budget. WATERWARE also allows users to estimate irrigation requirements using a modified version of the U.N. Food and Agriculture Organization's CROPWAT model and a similar methodology for estimating domestic water demands for water resources planning purposes. WATERWARE has been used in developing the Rio Lerma/Lake Chapala master plan in Mexico. Information on WATERWARE is available at *http://www.ncl.ac.uk/wrgi/wrsrl/projects/waterware/waterware.html.*

IPC

The Decision Support System for Integrated Pollution Control (the IPC), is a software package for personal computers developed by the World Bank in cooperation with the World Health Organization and the Pan American Health Organization. The IPC uses databases and computational modules to generate rapid assessments of the pollution characteristics and control alternatives for specific geographic areas where little or no information on pollution is available. That is, IPC is a screening model that depends on transferring data from one context where it is available to another where it is not.

The assessments provided by the IPC can be partitioned into four different components or stages. First, estimates of air, water, and solid waste emissions are obtained by applying standard emission factors to measures of local economic activity aggregated by industrial sector.[5] These factors are calculated from databases containing information on pollution-intensive technological processes across all sectors of economic activity, grouped according to U.N. International Standard Industrial Classification at a four-digit level, and on the principal control options available for each process, including waste prevention programs. The emission factors associated with specific processes and control options are then applied (or transferred) to data supplied by the user on local industrial processes and pollution control measures. So, although the IPC is designed to operate in a climate of scarce or nonexistent pollution data, it does require the users to be able to supply relatively detailed information on local industrial employment, value added, or output. In any event, these emissions estimates can then be fed into simple dispersion models to generate estimates of ambient concentrations of air and water pollutants. Third, the total costs of various control options for these pollutants can be estimated using standardized engineering-type cost functions. Fourth, estimates of long-run marginal cost schedules for achieving a level of emission reduction or decline in ambient concentration for a chosen pollutant can be made. These treat the region as a point—a one-to-one "map" of pollution in total to average concentration of some pollutant. There may also be the capability of looking at effective emissions as related to stack height. Finally, for reference, the IPC also includes a database composed of health guidelines for various air and water pollutants. More specific information about these computational modules and databases is provided by the user's manual, along with brief summaries of example applications (World Bank, 1997).

The IPC should be viewed as a work in progress, given the World Bank's level of commitment and the promise for validating and/or fine-tuning the emission factors during the second phase of the IPPS project and the promise of incorporating more complex dispersion models as confidence in data accuracy grows.

Finally, it should be noted that there are some specific Latin American examples of the development and application of DSSs. For example, as one step in the process of formulating integrated management plans for the Paz River basin in El Salvador and Guatemala and the San

[5] The emissions factors were generated by the Industrial Pollution Projection System (IPPS), which was created by the World Bank to improve the design of effective pollution control strategies in developing countries where little or no industrial pollution data exist. The development of IPPS is occurring in two phases. The first stage estimated emissions factors from detailed U.S. data on industrial activity and emissions levels. Hettige et al. (1994) provide a detailed description of the derivation of the original emissions factors. Similarly, the cost schedules were also developed using U.S. data, as described in Hartman et al. (1994). The second phase of the IPPS project, currently ongoing, is to attempt to "calibrate" the estimates for country- and region-specific variations by gathering plant-level data from many different countries to estimate these differences. As a result of this second phase, a limited number of emissions factors generated from pollution data from Mexico and China are now also available. Although the applicability of U.S. data to developing countries is obviously limited, the developers of IPPS are counting on the presence of general patterns across countries or regions that will be sufficient (with appropriate calibration) to justify the creation of a screening tool that relies on such data.

Juan River basin in Costa Rica and Nicaragua, these local governments, with help from nongovernmental organizations (NGOs), are designing and constructing information systems to aid in decision making and planning.

WATER QUALITY MODELS

The label "water quality model" applies to a large number of quite different products addressing different problems and making different assumptions. While only a corner of this large territory is of direct interest in the context of this book, it is worth very briefly sketching the major distinctions—problem settings and techniques—that subdivide the larger field.[6]

- *Type of water body(ies) represented*
  - Groundwater
  - Surface water
  - Rivers
    - Lakes/reservoirs
    - Estuaries
    - Embayments
    - Open seas or oceans
  - Combination of ground and surface
- *Type of pollution sources accepted*
  - Point discharges (wastewater treatment plants, factories, urban runoff when it is sewered)
  - Nonpoint discharges (farms, forests, unsewered urban runoff)
  - Atmospheric deposition
  - Combinations of source types
- *Types of pollutants accepted*
  - Nondegrading, dissolving (e.g., sodium chloride)
  - Nondegrading, suspended (and settling out) (e.g., soil)
  - Organics that degrade and use up dissolved oxygen (e.g., biochemical oxygen demanding organics; chemical oxygen-demanding organics, COD; nitrogenous oxygen demand, NOD)
  - Nutrients (e.g., nitrates, phosphates)
  - Toxics (e.g., pesticides, herbicides, heavy metals)
  - Disease pathogens
- *Types of in-water processes and compartments modeled*
  - Settling and resuspension of solids
  - Decay of oxygen-demanding organics
  - Growth and death of algae
  - Growth and death of microscopic animals (zooplankton)
  - Growth and death of other plant eaters
  - Growth and death of meat eaters (usually fish such as bass and trout)
  - Processes that go on at the bottom of the water column (benthic processes) such as decay of settled organics; chemical reactions involving, for example, mercury; and growth and death of populations of benthic plants and animals

[6] Nothing is said here about pure water quantity models. Such a model may be embedded in a water quality model, but without such embedding, quantity is not of interest here.

- *Relation to time*
  Steady state (analogous to an economist's static model . . . timeless, or said another way, assuming unchanging background conditions over a long enough period to have produced equilibrium)
  Nonsteady state (in which the changing trace of outputs over time is sought)
  Short term (as in a daily model of algal growth and dieoff)
  Long term (as in a model that tries to capture chronic effects of low-level exposures to one or more toxics)
- *Relation to space*
  Near-field or mixing zone models concentrate on the dynamics of mixing
  Far-field models smooth out these dynamics, either by using spatially continuous equations for such processes as settling or decay, or by compartmentalizing utilization of the water body and assuming perfect mixing within each compartment (only far-field models are of interest herein)
- *Relation to certainty*
  Deterministic (all background conditions, discharges, and model parameters are assumed to be known with certainty)
  Stochastic (some elements of background, discharges, or model structure are not known with certainty, but are drawn from known discrete or continuous probability distributions)

Finally, this chapter concentrates entirely on models that some authors call "theoretical" and others "mechanistic." That is, the equations are derived from chemical or physical laws or from ecological theory. The actual parameter values are often chosen through an iterative process of trial and error, sometimes called "tuning," in which adjustments are chosen to make a model reproduce output from reality. And at some point the resulting model is more or less formally compared with another set of data from reality, a process often called "validation." The possibility exists of creating a model entirely empirically—on the basis of data, not theory—in which relations are fitted statistically using very general functional forms. However, this is not the way any of the models that have come to the attention of the authors have been built. Finally, in the days before computers capable of rapidly solving complex sets of differential equations were readily available, some physical models were built for predictive purposes. Sometimes these are referred to as "scale" models, but there is no longer any obvious reason for taking this approach.

For the purposes of this book, the model varieties of greatest interest are those with the following features:

- They cover surface water of all kinds. (However, see the following section for a brief discussion of ocean outfall models as a distinct variety.)
- They accept pollution at least from point sources. Nonpoint-source capability is desirable, but the resulting quality problems could be dealt with as part of background conditions. The investment projects being analyzed will (almost) always be point source treatment alternatives.[7]
- The best situation would be to have a model that accepted all the listed pollutants. In actual cases, it is likely that the best available models will accept degradable organics and nutrients. Suspended solids and the effects of specific toxics may be important in particular situations.

[7] Projects may involve sewering plus treatment as an option. The economics of this choice were discussed in Chapter 4. The water quality problems in the nonsewered situation will generally involve quite small, neighborhood water-courses. It is unlikely that water quality models of the nonsewered situation will be part of the decision tools required.

- The best available models are likely to be capable of dealing with settling of solids, the BOD/DO nexus, and the growth and death of algae. Models that have zooplankton and fish compartments are much rarer, even though the Delaware model of the mid-1970s already had this capability.[8]
- For practical cost-benefit work, steady-state models, perhaps even linearized versions of these, are generally sufficient.[9] Even though dynamic excursions, as in short periods of drastically low oxygen levels, may be important to higher forms of in-stream life, the model is unlikely to be able to reflect this anyway.
- Only far-field models are of interest.
- Again, while the world is stochastic, the important implications of this for water quality, quality excursions, are unlikely to be modeled. Most often, the uncertainty problem will be handled by choosing a low flow as part of the background conditions and the model itself will be deterministic.

The models actually used are also very likely to be one-dimensional, which is the field's label for models consisting of horizontally linked compartments (reaches) within each of which perfect mixing is assumed so that each compartment may be collapsed to a single vector—a single point.

Ocean Outfall Models

The models, methods, and concerns on which this chapter concentrates for its illustrative material reflect the classic situation of a freshwater river polluted by oxygen-demanding organics and possibly nutrients. Some very different policy and analytical problems are presented by the case of an urban area on the edge of the open ocean. In particular, the depletion of dissolved oxygen is unlikely to be a problem, except possibly in the immediate vicinity of the discharge, because the massive dilution and availability of DO in the ocean beyond this near zone creates what amounts to a marine treatment system (Markham, 1993). The problem that is likely to dominate discussion and planning of an ocean outfall system is the fate of disease-causing pathogens (bacteria and viruses) in the sewage plume. The concentrations of these will, of course, be diluted during advection and diffusion, and they will be disabled or killed by ambient conditions, especially exposure to sunlight. Nonetheless, it may be possible for potentially harmful concentrations to build up at beaches downwind or down current.

There are any number of ocean outfall models described in the literature and (at least) several are commercially available. Examples of the latter include EPA's CORMIX (Cornell Mixing Zone Expert System) and PLUMES and the model found, with code, in Markham (1993). An optimization model for planning coastal wastewater treatment and disposal systems is described conceptually (with a few equations) in Chang and Wang (1995).

Between the extremes of the river and the open ocean are found estuaries and embayments. Some of these are themselves quite open (e.g., the estuary of the Rio de La Plata shared by Uruguay and Argentina), and some are quite closed and are closer to a saltwater lake (e.g., Cartagena Bay on Colombia's Caribbean coast). The range of possible settings means that

[8] Thus, for example, Ambrose et al. (1996, Table 14.4) list and briefly describe eleven computer models for water quality analysis. Only two of these include zooplankton as a variable (a compartment or state). None include fish at all.

[9] Linearization is possible when discharges do not themselves interact and when decay rates are independent of the existing quality levels. The process produces the transfer coefficients so beloved by economic modelers in discussions of, for example, the static efficiency properties of effluent charges. These coefficients translate units of pollutant (x) at point (i) into some measure of ambient quality (y) at point (j).

substantial tailoring of any generally applicable model will be necessary for a particular application. However, the economics will not be different in any important way, only in the details of what considerations are important. Further discussion of technical distinctions and details is provided in Annex 5-A.

Brief History of Water Quality Modeling

The earliest water quality model generally acknowledged was that developed by Streeter and Phelps (1925) on the basis of work on the Ohio River. It may usefully be thought of as a simple, summary representation of what goes on in a moving watercourse when oxygen-demanding organic material is introduced by point sources. For any one source, a differential equation that mimics the use of dissolved oxygen in organic decay and the regaining of oxygen (reaeration) from the surface produces an oxygen "sag" downstream from the source. The equations can be solved analytically for equilibrium (steady-state) values for given constant discharges. Multiple sources are treated as independent or separable. That is, the key coefficients for decay and reaeration rates do not depend on what else is going on in the watercourse (on how much or how little BOD there may be in addition to that added by the source to which the equation applies).

The advent of civilian computers in the 1960s made it possible to create more complex versions of the Streeter-Phelps model. In particular, the stream or estuary no longer had to be treated as perfectly mixed in the direction perpendicular to the flow. Parallel "boxes" could be structured and differentiated and numerical solutions found using the computer. The field was freed from the constraints implied by the need for analytical solutions (Chapra, 1998). Two names prominently associated with the advances of this period are O'Connor (1960) and Thomann (1963).

Further advances that took advantage of increasing computational power and the declining cost of using it pushed the field toward looking inside the black box of the BOD/DO models. That is, efforts began to be made to represent the aquatic ecological system rather than just to summarize what happens to DO. Most prominently, this involved the addition of the connection among nutrients (nitrogen and phosphorus, phytoplankton or algae) and DO. This advance required modeling the connections among the different chemical forms of nitrogen and phosphorus to mimic how the forms that enter the water change to become available to aquatic plants or leave the system by volatilizing or settling. Some of these efforts layered on top of the nutrient/algae chain a representation of zooplankton (grazers on the algae) and fish. For several descriptions of early efforts along this line, see Russell (1975). These models demanded numerical solutions to complex sets of simultaneous, nonlinear differential equations. They could not even have been used as simulation devices without electronic computers.

More recently, according to Chapra (1998), the research frontier has moved on to the incorporation of chronic toxics and the representation of the relations between the sediments and their inhabitants at the bottom of the water column and the food web in the water column itself. Since the toxics often enter the water attached to soil or other particles that tend to settle (as with pesticides and herbicides from farming), these two frontier concerns are related. That is, toxics may first appear in available form on the bottom of the water source and enter the food web (the stream ecosystem) via feeding at the bottom by animals in the water column such as zooplankton (Thomann, 1971).

Another theme of work over the past two decades seems to have been to try to replace the previous reliance on special purpose-built, place-tied models with generic model frameworks often tied to databases of parameter and background-condition values. In principle, these allow the creation of "new" models by users rather than only by researchers. For fairly obvious reasons, there has been tension between the desire to elaborate and complicate and the desire to make more broadly applicable. This tension can be seen in the next section.

Two Examples of Surface Water Quality Models

This section briefly describes the water quality models that have been used in some of the illustrations found later in this chapter. One of them is the WQM embedded in STREAMPLAN, described earlier. The other, QUAL2E, represents one facet of the current status of multiyear investments by the EPA aimed at creating transferable model frameworks. (More models, several of them also from EPA, are described in Bouchard et al., 1995.) The QUAL2E description is taken largely from Lahlou et al. (1995) and from Brown and Barnwell (1987).

These models share a great deal of structure. In the terms just discussed, they are both designed for moving fresh water, but not for tidal estuaries. They both accept point source pollution discharges, but neither deals in an entirely satisfactory way with nonpoint sources.[10] Both accept as discharges carbonaceous (5-day) BOD (CBOD) as the major pollutant of interest. As explained later, however, both include nutrient (nitrogen and phosphorus) cycling, although the STREAMPLAN versions are vestigial compared with those in QUAL2E, which ties both cycles to algal growth and decay. For purposes of those cycles, they both accept discharges of organic and other forms of nitrogen and organic phosphorus. STREAMPLAN reflects the oxidation of organic nitrogen (NH_4 radical) to nitrate (NO_3 radical) in one step and the settling of organic phosphorus. QUAL2E includes the intermediate nitrogen states of NH_3 (ammonia) and NO_2 (nitrite radical); and in the phosphorus cycle it includes conversion of organic phosphorus to the biologically available dissolved form. The related discharges accepted are:

STREAMPLAN: organic nitrogen, nitrate, organic phosphorus;
QUAL2E: organic nitrogen, NH_3, NO_2, nitrate, organic phosphorus, dissolved phosphorus.

While QUAL2E models algal growth and death and decay, and connects this cycle to DO, it is not clear from the manuals whether it is possible to constrain chlorophyll-a concentrations, which stand for algal density, as part of a water quality management run.

Both models are built to be run for steady-state predictions, although QUAL2E can apparently be used to simulate daily variations in algae. Neither is designed to follow the modeled part of the ecological system through variations in flows or temperatures. Neither model has any stochastic elements, and both are treated as one-dimensional, with the definition of the reaches and subreaches (called "elements" in QUAL2E) a fairly complex matter.

Since they are so much alike, why bother with both? The answer lies in two features of QUAL2E that make it into what is being called here a nonseparable-source model:

- The inhibition of the rate of oxidation of nitrogen to NO_2 and NO_3 that is caused by already low DO in the watercourse, and
- The feedback effect on the algal growth rate, and thus on the supply of oxygen, for algae in the water column.

The effect of these two features, which are described more formally in Annex 5-B, is to make the rates at which discharges of biochemical oxygen-demanding organics (and nitrogen compounds) are oxidized functions of the quality of the stream just above the point of discharge. The effect that treating wastewater (reducing the organic loading) has on river quality is therefore

[10] Nonpoint sources have to be converted to quasi-point source loads, by accumulation along a tributary, for example.

different, everything else remaining the same, when the river is already quite heavily loaded than when it is lightly polluted. This effect is illustrated in the next section. Before turning to the example, however, it is worth stressing that this interdependence means that the benefits resulting from a particular choice of treatment plant will in general depend on the rest of the regional context. (Refer to Chapter 4, especially Table 4-1, for a schematic view of this complication.)

An Illustration of Nonseparability

To illustrate that existing water quality is important in determining the net effect of discharge reduction, this section contrasts the predictions of the QUAL2E model under two different assumptions about context. The illustration is composed of four scenarios distinguished on the basis of the assumed quality of the receiving waters (clean or dirty) and of the characteristics of a point source (treated or untreated).[11]

The stream in the illustration is divided into reaches, and the reaches into elements of equal length. In this example, each element is 1.43 km long. The basin has five reaches; Reach 1 has 13 elements, Reach 2 has 18 elements, Reach 3 has 18 elements, Reach 4 has 17 elements, and Reach 5 has 20 elements (see Figure 5-3). There are three "headwaters" and one point source, located at the upper end of Reach 1. The change from clean to dirty receiving water amounts to an assumed reduction in the dissolved oxygen at "Headwater 1" from 8 mg/l to 1.2 mg/l. The characteristics of the other two headwaters remain unchanged throughout the application. For simplicity, there is only one point source, and it is either treated or untreated. The water quality and discharge characteristics under the different scenarios are shown in Table 5-2.

The resulting interaction is detailed in Table 5-3, which shows the DO levels for every element in Reaches 1, 3, and 5 (2 and 4 are omitted because they are the same under all sets of assumptions) under the four scenarios. It also displays the improvement in DO from treating the point source under both the clean and dirty receiving water settings. The interdependence is depicted by the divergence of these two differences, i.e., when the improvement in ambient water quality from treating the point source in the "clean" ambient water quality setting is greater than the improvement from treating the point source in the "dirty" ambient water quality setting. Finally, Figures 5-4 and 5-5 display this interaction graphically.

AN ILLUSTRATION OF BASINWIDE AMBIENT QUALITY-CONSTRAINED COST MINIMIZATION

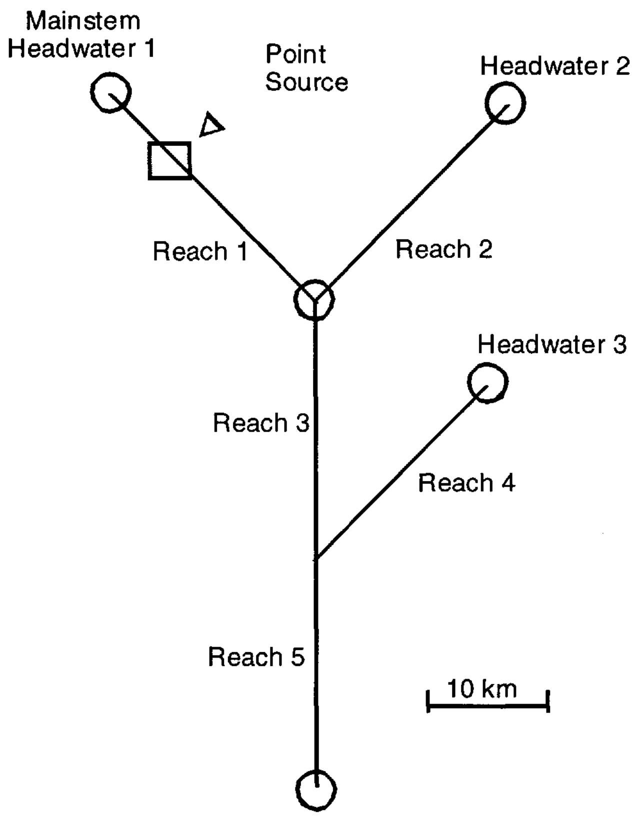
In the second- or third-best world of available off-the-shelf tools already described, no illustration of basinwide cost-benefit analysis for a hypothetical basin was feasible because the available models do not incorporate a benefits side. Rather, they are examples of the basinwide, vector-constrained cost-minimization approach discussed in Chapter 3.

What is possible is an illustration of the capabilities of one of the DSSs that are publicly available. STREAMPLAN, described earlier in this chapter, was chosen for the exercise because its package included three vital elements: a treatment plant cost module, a water quality model (contrasted with QUAL2E in the previous section), and an optimization routine.[12]

[11] The data used to set up the models were taken from EPA's BASINS database.

[12] One might call this optimization realistic rather than elegant. As the reader will see, the solutions involve each treatment plant operating fully with particular treatment technology (removal level). This produces a fairly small set of possible solutions to the regional problem so that what amounts to brute force can work to find the lowest-cost combination. A more elegant piecewise linearization of the individual plant objective function would in general produce lower overall costs with combinations of process choices at each plant.

Figure 5-3. The River Basin Used for the Interdependence Illustration



The hypothetical river basin serving as a basis for this illustration was modified from an example application that accompanies the STREAMPLAN software, and apparently represents the application of STREAMPLAN to an unnamed Eastern European river. The resulting hypothetical river basin is depicted in Figure 5-6. The network is divided into reaches or river segments. The main branch of the river is composed of six reaches (Mainstem 1 through Mainstem 6). The main branch has two tributaries, one of which is divided into three reaches (FirstTrib 1, FirstTrib 2, and FirstTrib 3), and the other of which is simply one reach long (FirstTrib 4). The first tributary has its own tributary that is one reach long (SecondTrib 1). In addition, there are four headwaters (Head Reach 1 through Head Reach 4), which are all treated

Table 5-2. Data for the Illustration

	Headwater 1		Headwaters		Point Source Discharge	
Parameters	Clean	Dirty	2	3	Treated	Untreated
Flow (mgd)	88	88	0.81	0.79	12	12
Temperature (°C)	24	24	24		25	25
Dissolved oxygen (mg/l)	8	1.2	8		6	0
BOD (mg/l)	5	5	5		10	273
Nitrate Nitrogen (NO_3-N) (mg/l)	0	0	0		4	20
Organic Nitrogen (NH_4-N) (mg/l)	0.7	0.7	0.7		1	30
Organic P (mg/l)	0	0	0		1	40

Table 5-3. Effect of Background Dissolved Oxygen (DO) on the Gain in DO from Wastewater Treatment

	Location		Clean Initial Conditions Dissolved Oxygen mg/l			Dirty Initial Conditions Dissolved Oxygen mg/l			Interaction
Distance from Headwater 1 (km)	Reach	Element	Source Treated	Source Untreated	Gain from Treatment	Source Treated	Source Untreated	Gain from Treatment	Clean Gain Minus Dirty Gain mg/l
1.4	1	1	7.95	7.89	0.06	1.33	1.28	0.05	0.01
2.9	1	2	7.70	6.79	0.91	1.95	1.07	0.88	0.03
4.3	1	3	7.67	6.58	1.09	2.03	1.00	1.03	0.06
5.7	1	4	7.64	6.37	1.27	2.11	0.94	1.17	0.10
7.2	1	5	7.62	6.17	1.45	2.19	0.88	1.31	0.14
8.6	1	6	7.59	5.97	1.62	2.27	0.82	1.45	0.17
10.0	1	7	7.56	5.78	1.78	2.34	0.77	1.57	0.21
11.5	1	8	7.54	5.60	1.94	2.42	0.72	1.70	0.24
12.9	1	9	7.51	5.42	2.09	2.49	0.68	1.81	0.28
14.3	1	10	7.49	5.24	2.25	2.57	0.64	1.93	0.32
15.8	1	11	7.46	5.07	2.39	2.64	0.60	2.04	0.35
17.2	1	12	7.44	4.90	2.54	2.71	0.56	2.15	0.39
18.6	1	13	7.42	4.75	2.67	2.78	0.53	2.25	0.42
20.0	3	1	7.41	4.68	2.73	2.90	0.62	2.28	0.45
21.5	3	2	7.39	4.54	2.85	2.98	0.61	2.37	0.48
22.9	3	3	7.37	4.40	2.97	3.06	0.59	2.47	0.50
24.3	3	4	7.35	4.26	3.09	3.14	0.58	2.56	0.53
25.8	3	5	7.33	4.13	3.20	3.22	0.57	2.65	0.55
27.2	3	6	7.31	4.00	3.31	3.29	0.57	2.72	0.59
28.6	3	7	7.30	3.88	3.42	3.36	0.56	2.80	0.62
30.1	3	8	7.28	3.76	3.52	3.43	0.55	2.88	0.64
31.5	3	9	7.26	3.64	3.62	3.50	0.55	2.95	0.67
32.9	3	10	7.25	3.53	3.72	3.57	0.55	3.02	0.70
34.4	3	11	7.23	3.42	3.81	3.64	0.54	3.10	0.71
35.8	3	12	7.22	3.32	3.90	3.70	0.54	3.16	0.74
37.2	3	13	7.21	3.22	3.99	3.77	0.54	3.23	0.76
38.7	3	14	7.19	3.13	4.06	3.83	0.54	3.29	0.77
40.1	3	15	7.18	3.03	4.15	3.89	0.55	3.34	0.81
41.5	3	16	7.17	2.95	4.22	3.95	0.55	3.40	0.82
43.0	3	17	7.16	2.86	4.30	4.01	0.55	3.46	0.84
44.4	3	18	7.15	2.79	4.36	4.07	0.56	3.51	0.85
45.8	5	1	7.15	2.80	4.35	4.16	0.67	3.49	0.86
47.3	5	2	7.14	2.73	4.41	4.22	0.67	3.55	0.86
48.7	5	3	7.12	2.65	4.47	4.27	0.67	3.60	0.87
50.1	5	4	7.11	2.58	4.53	4.32	0.67	3.65	0.88
51.6	5	5	7.10	2.51	4.59	4.37	0.67	3.70	0.89
53.0	5	6	7.09	2.45	4.64	4.42	0.67	3.75	0.89
54.4	5	7	7.09	2.39	4.70	4.47	0.68	3.79	0.91
55.8	5	8	7.08	2.33	4.75	4.52	0.68	3.84	0.91
57.3	5	9	7.07	2.27	4.80	4.57	0.68	3.89	0.91
58.7	5	10	7.06	2.21	4.85	4.61	0.69	3.92	0.93
60.1	5	11	7.05	2.16	4.89	4.66	0.70	3.96	0.93
61.6	5	12	7.04	2.11	4.93	4.70	0.70	4.00	0.93
63.0	5	13	7.04	2.07	4.97	4.75	0.71	4.04	0.93
64.4	5	14	7.03	2.02	5.01	4.79	0.72	4.07	0.94
65.9	5	15	7.02	1.98	5.04	4.83	0.72	4.11	0.93
67.3	5	16	7.02	1.94	5.08	4.87	0.73	4.14	0.94
68.7	5	17	7.01	1.90	5.11	4.91	0.74	4.17	0.94
70.2	5	18	7.00	1.87	5.13	4.95	0.75	4.20	0.93
71.6	5	19	7.00	1.84	5.16	4.99	0.76	4.23	0.93
73.0	5	20	6.99	1.81	5.18	5.03	0.77	4.26	0.92

Figure 5-4. DO by River Kilometers under Four Pairs of Assumptions

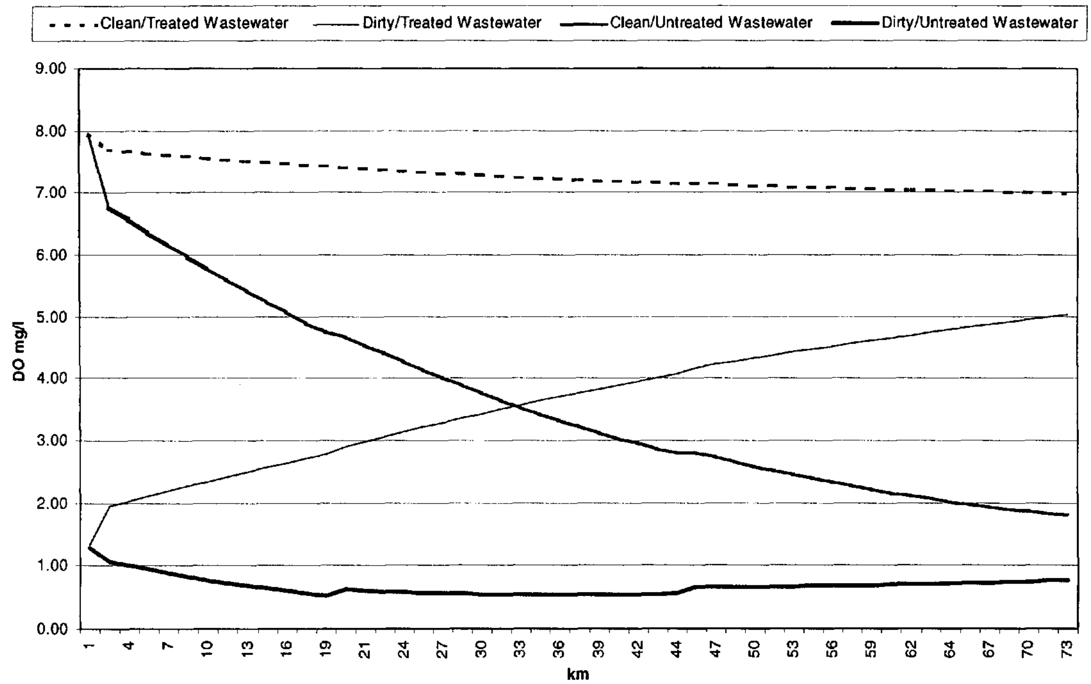


Figure 5-5. Change in DO by River Kilometers under Clean and Dirty Initial Conditions

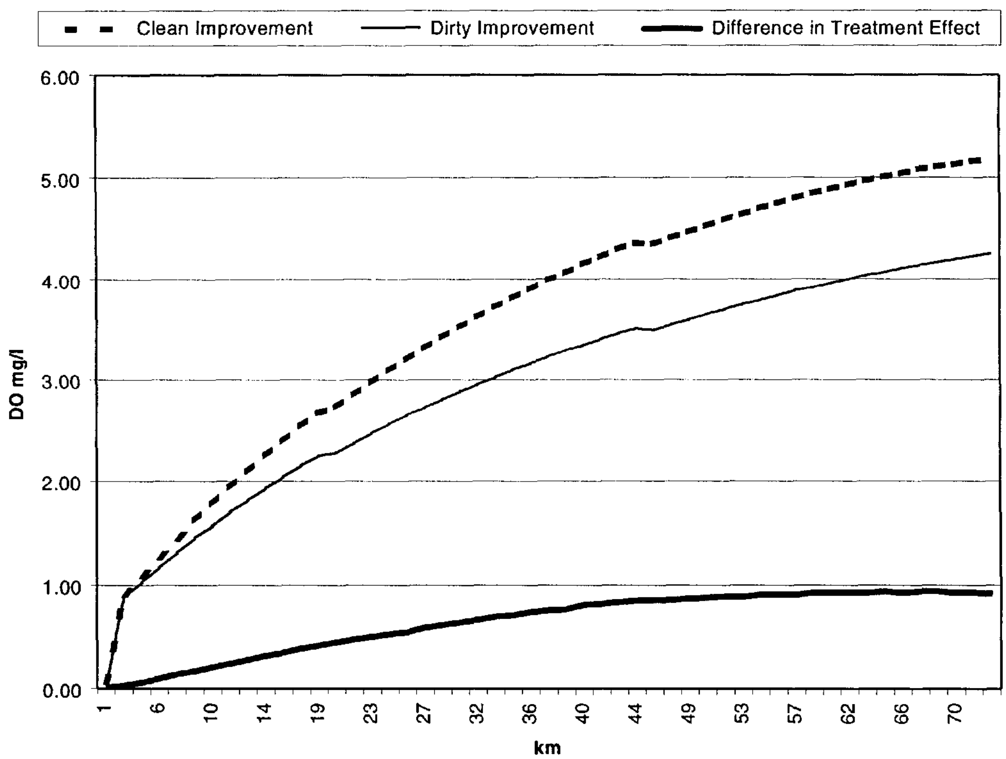
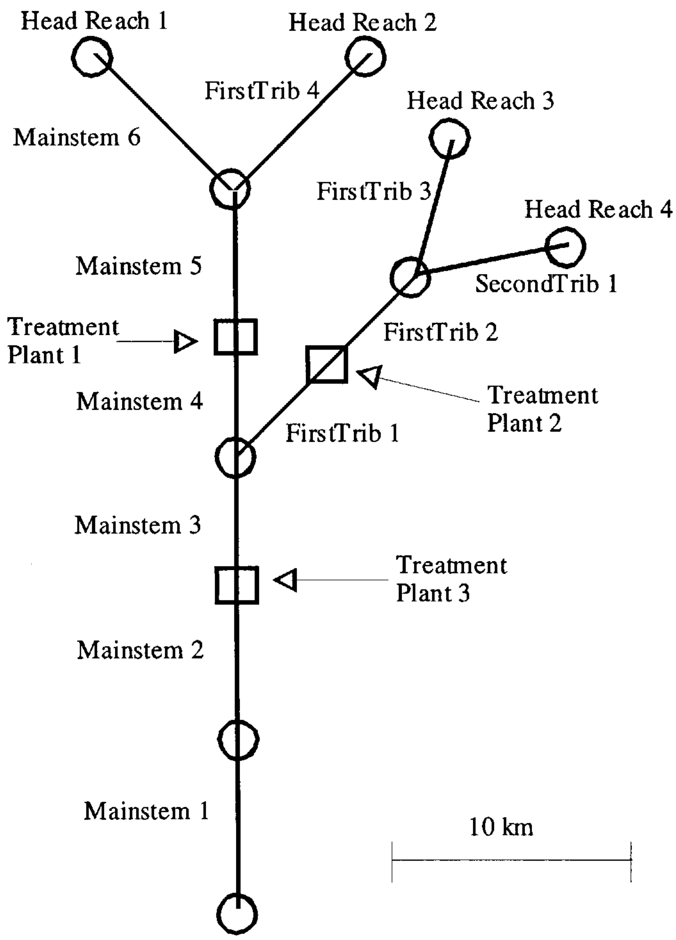


Figure 5-6. Hypothetical River Basin Network



as being of zero length for modeling purposes. Finally, there are three treatment plants (TP1, TP2, and TP3), the effluent of which is treated as flowing into the reach that ends at the plant (the plant actually serves as the point of transition from one reach to another).[13]

The water quality characteristics of the influent (sewered wastewater) at all three treatment plants depend on plant size, as displayed in Table 5-4. This variation in concentration with size results from an assumed change (by the DSS) in the ratio of industrial and domestic wastewater accompanying a change in the size of the plants.[14] In the example, industrial effluent has lower concentrations of BOD, organic nitrogen, and organic phosphorus than domestic wastewater, and the larger treatment plants have lower relative amounts of industrial wastewater arriving at them, so the concentrations of BOD, etc., are higher at larger plants. Three plant sizes are used in various combinations in the illustrations. To give some indication of what these sizes mean, assume an average per person wastewater flow of 50 gallons per day. Then the smallest plant is serving about 200,000 people (or their equivalents), the medium-size plant 600,000, and the large plant 1.4 million.

[13] In effect, STREAMPLAN creates a reach where there is a point source, but chooses to put the source at the downstream rather than the upstream end. This lacks something as a matter of representing reality, since the perfect mixing assumption implicitly carries discharged pollutants upstream in the reach.

[14] The ratio changes because the change in size is accomplished by reassigning the total population among the individual plants without altering the allocation of industrial wastewater among the three plants.

Table 5-4. Flow and Influent Characteristics for the Illustrated Treatment Plants

Size	Flow (m^3/sec)	Flow (mgd)	CBOD (mg/l)	NH_4-N (mg/l)	Organic P (mg/l)
Small	0.48	10.96	291	334	6.40
Medium	1.39	31.73	297	442	8.76
Large	3.21	73.27	299	475	9.46

Table 5-5. STREAMPLAN Treatment Alternatives and Removal Efficiencies

Alternative	BOD Removal (%)	Ammonia Removal (%)	Total Phosphorus Removal (%)
Primary (P)	30.0	15.0	15.0
Chem. enhanced primary (CEP)	55.0	25.0	75.0
Prim. with chem. treatment (PC)	70.0	30.0	90.0
Primary + act. sludge (B)	90.0	98.8	30.0
CEP + act. sludge (BC1)	90.0	98.8	90.0
PC + act. sludge (BC2)	95.0	98.8	92.2
BC1+ partial denitrification (BC1DN)[a]	95.0	99.9	90.0
BC2 + partial denitrification (BC2DN)[a]	95.0	99.9	92.2
BCC2 + "full" denitrification (BCDN)[b]	97.0	99.9	92.2

[a] Partial denitrification removes ammonia, in part by oxidizing it to nitrate (NO_3).
[b] "Full" denitrification removes more NH_4 but also reduces the nitrate concentration compared with the levels under partial denitrification.

Each treatment plant, independent of size, can be built with any of nine technology combinations: three primary, three secondary, and three tertiary. As discussed in Chapter 3, the choice of a secondary treatment option assumes the presence of a primary plant before the secondary unit. The choices and their assumed performances in pollution removal are set out in Table 5-5.

The costs of the nine treatment alternatives at the three alternative sizes are summarized in Table 5-6, with total capital cost (C) and annual operation, maintenance, and repair costs shown. It is worth pausing a moment over these costs, if only to highlight an example of the sorts of problems one can run into with off-the-shelf packages. Most troublesome is something that does not show up in the table. Using the STREAMPLAN software, the total capital costs of the treatment alternatives begin to decline with plant size somewhere in the neighborhood of 2 m^3/sec in flow (F). There is clearly something wrong with the cost package, but it is impossible to tell what it is or to fix it without learning the programming language and going through the code line by line. The approach taken for this example was ad hoc. The scaling factor implied by the ratios of the first two sizes and their capital costs was just applied to the size ratio of the second pair, 1.39 m^3/sec and 3.21 m^3/sec to produce the capital costs found for the largest plant.[15] The other problem is visible in the table because no effort was made to fix it. This problem again involves capital costs, which are seen to decline between the simple primary plus activated sludge and more advanced versions of primary plus activated sludge. Since those more advanced primary processes have higher capital costs than the simple one, this does not seem sensible. Because there are differences in OMR costs in the "right" direction,

[15] That is, an estimate of β in the expression $C = \alpha F^{\beta}$ can be obtained from $C2/C1 = (F2/F1)^{\beta}$ or $\log(C2/C1) = \beta \log(F2/F1)$. For every alternative treatment process, this is roughly 0.72, so that factor was used to scale up the costs of the midsized plant to the largest size.

Table 5-6. Total Costs of Treatment Alternatives (millions of dollars)

	0.48 m³/sec (10.96 mgd)[a]		1.39 m³/sec (31.73 mgd)		3.21 m³/sec (73.27 mgd)	
Alternative	Capital	OMR (10^6/yr)	Capital	OMR (10^6/yr)	Capital	OMR (10^6/yr)
P	0.98	0.97	2.11	2.81	3.85	6.49
CEP	1.86	1.56	3.98	4.52	7.27	10.44
PC	5.83	1.85	12.46	5.35	22.76	12.37
B	12.95	1.60	28.03	4.65	51.21	10.74
BC1	10.40	2.16	22.56	6.27	41.22	14.49
BC2	11.79	2.50	25.68	7.24	46.92	16.72
BC1DN	18.55	2.54	40.22	7.37	73.48	17.03
BC2DN	21.83	3.03	47.30	8.78	86.42	20.27
BCDN	24.66	3.19	53.53	9.26	97.80	21.39

[a]Million gallons per day.

Table 5-7. Ambient Quality Constraints: Example 1

Reach	DO (mg/l)	CBOD (mg/l)	NH_4-N (mg/l)	NO_3-N (mg/l)	Total Phosphorus (mg/l)
Mainstem 1	3	50	25	25	12
Mainstem 2	3	50	25	25	12
Mainstem 3	3	50	25	25	12
First Trib 1	3	50	25	25	12

it is not clear a priori that this glitch will result in obvious solutions to problems.[16] This is another example of the sort of traps for the unwary that lurk in these packaged programs.

Be that as it may, the simple regional example can still be used to illustrate some of the effects that make use of the regional setting worthwhile. The first solution is to the problem of meeting the ambient constraints set out in Table 5-7. In this first example, all of the constraints other than DO are set at artificially high levels so that in-stream DO levels will provide the only binding constraint in the optimization.

The example is pursued using four different configurations of the three treatment plant sizes, as set out in Table 5-8. Thus, the inflows to the treatment plants total 4.17 m³/sec in every configuration, but the location of the largest and smallest loads changes.

Before exploring the solutions to the cost-minimizing problem, it is important to emphasize that this problem setup treats the bottom of Mainstem 1 as the end of regional interest—as though it were a national border or the mouth of the river. Some of the effects seen later are driven by this assumed configuration. In particular, if the model were set up to follow the river farther downstream, it is nearly certain that a different choice of treatment levels would be necessary to continue to meet the 3.00 mg/l DO constraint in the lower reaches.

That said, the solutions of the regional problems described earlier are set out in Table 5-9. One immediate observation is that there is very little variation in the treatment levels chosen across the configurations. Plant 2 is always built as primary plus activated sludge. Plant 3 is primary only when it is small and primary plus activated sludge when it is medium or large. Plant 1 is primary plus activated sludge except when it is upstream of the largest plant, when

[16] With a capital recovery factor of 0.142, the total annual costs (OMR and annualized capital) do increase between B and BC1, and between BC1 and BC2, as one would expect. But it is nonetheless true that the size of the increases must be "too small" relative to what would be true if, for example, one added the appropriate capital-cost differences for the other primary units to the B (simple primary plus activated sludge) capital cost.

Table 5-8. Plant Flow Sizes by Configuration (m^3/sec)

Configuration	Upstream/Mainstem Plant 1	First Tributary Plant 2	Downstream/Mainstem Plant 3
1	1.29	1.39	1.39
2	3.21	0.48	0.48
3	0.48	3.21	0.48
4	0.48	0.48	3.21

it goes to advanced primary plus activated sludge. A second thing that may strike the alert reader is that in two of the configurations, none of the DO constraints is binding. This has to be attributable to the discrete-choice nature of the choices available: Primary, even advanced primary, is not enough, but primary plus activated sludge is "too much."[17]

Another way of looking at the results is as preliminary information about the effects of moving sewered wastewater around. The results are preliminary because the package does not provide an interceptor sewer design and costing package. The lessons are also limited because of the assumed lack of interest in anything downstream of Mainstem 1 and because of the limitations of the discrete choices of sizes and treatment levels available. Nonetheless, it is interesting to observe that with the largest quantity of effluent at the highest plant (configuration 2 in Table 5-8), the costs are lowest and the DO constraint is just met. However, it seems likely that with more flexibility at the lowest treatment plant, so that the DO constraint is just met in the bottom reach, the option of moving wastewater downstream would look better.

A second example of exercising STREAMPLAN involves effectively adding another ambient water quality constraint—this one for phosphorus concentrations, which can be viewed as a surrogate for preventing or delaying eutrophication. The level chosen for this example is 1.6 mg/l, as shown in Table 5-10. This change makes a difference only in configuration 3 (where plant 2 is large and the other two small), because for the other configurations, meeting the

Table 5-9. Base Cases and Solutions to the Constrained Optimization Problems, Example 1

	Configurations							
	1		2		3		4	
	Base	Optimum	Base	Optimum	Base	Optimum	Base	Optimum
TP1 treatment level	NA[a]	B	NA	B	NA	BC2	NA	B
Ann. cost ($\$10^6$)	NA	8.64	NA	18.03	NA	4.18	NA	3.44
TP2 treatment level	NA	B	NA	B	NA	B	NA	B
Ann. cost ($\$10^6$)	NA	8.64	NA	3.44	NA	18.03	NA	3.44
TP3 treatment level	NA	B	NA	P	NA	P	NA	B
Ann. cost ($\$10^6$)	NA	8.64	NA	1.11	NA	1.11	NA	18.03
Total Regional Cost ($\$10^6$)	NA	25.92	NA	22.58	NA	23.32	NA	24.91
DO Levels (mg/l)								
First Trib 1	0	5.76	3.85	5.89	0	5.52	3.85	5.89
Mainstem 4	0	5.96	0	5.92	0	5.89	4.35	6.07
Mainstem 3	0	5.48	0	5.43	0	5.41	2.95	5.63
Mainstem 2	0	4.84	0	4.15	0	4.15	0	4.97
Mainstem 1	0	4.27	0	3.01	0	3.01	0	4.38

[a] NA, not applicable.

[17] Since the transformations of both CBOD and organic nitrogen use dissolved oxygen, both removal efficiencies are relevant (see Table 5-5).

Table 5-10. Ambient Quality Constraints: Example 2

Reach (mg/l)	DO	CBOD	NH_4-N	NO_3-N	Total P
Mainstem 1	3	50	25	25	1.6
Mainstem 2	3	50	25	25	1.6
Mainstem 3	3	50	25	25	1.6
First Trib 1	3	50	25	25	1.6

Table 5-11. Ambient Organic Phosphorus Results: Example 2

	Configurations							
	1		2		3		4	
	Base	Optimum	Base	Optimum	Base	Optimum	Base	Optimum
First Trib 1	1.67	1.57	1.46	1.43	2.06	1.35	1.46	1.43
Mainstem 5	1.33	1.21	1.81	1.53	1.07	1.04	1.07	1.04
Mainstem 4	1.44	1.35	1.54	1.42	1.54	1.19	1.24	1.22
Mainstem 3	1.56	1.43	1.57	1.44	1.57	1.19	1.57	1.43
Mainstem 2	1.56	1.43	1.57	1.43	1.57	1.18	1.57	1.43
Mainstem 1	1.56	1.43	1.56	1.43	1.56	1.18	1.56	1.43

Table 5-12. Contrasting Optimal Arrangements in Examples 1 and 2, Configuration 3

	DO Constraints Only Example 1	DO and Phosphorus Constraints Example 2
TP1 treatment level	BC2	B
Annual cost ($\$10^6$)	4.18	3.44
TP2 treatment level	B	BC1
Annual cost ($\$10^6$)	18.0	20.37
TP3 treatment level	P	CEPT
Annual cost ($\$10^6$)	1.11	1.82
Total Regional Cost ($\$10^6$)	23.32	25.63

Table 5-13. Contrasting Optimal Arrangements in Examples 1 and 3

	Configuration 1		Configuration 2	
	DO@3.0 mg/l Only	DO@3.0 mg/l P@1.5 mg/l	DO@3.0 mg/l Only	DO@3.0 mg/l P@1.5 mg/l
TP1 treatment level	B	B	B	BC1
Annual cost ($\$10^6$)	8.64	8.64	18.03	20.37
TP2 treatment level	B	BC1	B	B
Annual cost ($\$10^6$)	8.64	9.48	3.44	3.44
TP3 treatment level	B	B	P	P
Annual cost ($\$10^6$)	8.64	8.64	1.11	1.11
Total Regional Cost	25.92	26.76	22.58	24.92

DO $\geq$ 3.0 mg/l constraint also meets this phosphorus constraint (Table 5-11). The reaction in configuration 3, as seen in Table 5-12, is to upgrade the treatment levels at plants 2 and 3: from B to BC1 at plant 2 and from P to CEPT at plant 3. At the same time, however, the treatment level at plant 1 is relaxed from BC2 to B. The net additional annual cost is $\$2.3 \times 10^6$, and the constraint is more than satisfied with the highest phosphorus level only 1.35 mg/l.

Tightening the phosphorus constraint by another 0.1 mg/l, to 1.5 mg/l spurs changes in treatment choices in both configurations 1 and 2, but not in configuration 4 (Table 5-13). Under this new constraint set, in configuration 1, the treatment level at plant 2 is upgraded from B to BC1 at a cost to the region of about $\$0.8 \times 10^6$ per year. In configuration 2, the response to the tightening is to upgrade plant 1 from B to BC1, but the annual cost increment is $\$2.34 \times 10^6$ because plant 1 is the large plant in that configuration. The reductions in the highest phosphorus concentration that are purchased by the upgrades are 0.2 mg/l in configuration 1, reach 1 of the first tributary; and about 0.55 mg/l in mainstem 5 in configuration 2. [In configuration 2, the downstream reaches see reductions of about 0.2 mg/l, while in configuration 1 these are about 0.1 mg/l (Table 5-14).]

As a final example, a treatment technology standard is applied uniformly across the region. Under this requirement, each plant must have primary plus activated sludge treatment (designated B). In Table 5-15, the results of imposing this requirement are contrasted with those for the other examples. To reduce the information overload, the reporting is confined to treatment levels, minimum DO levels, and maximum phosphorus levels wherever they occur, and to total annual cost to the region. A first observation is that nothing changes in configuration 4 across the set of four different constraints. Technology "B" everywhere is the optimal way to achieve each of the AWQ constraints and is, of course, required anyway for the final example. For configuration 1, there is only slightly more flexibility. To meet the stricter phosphorus constraint (1.5 mg/l), BC1 is required at plant 2. Because of the discrete-choice limitation, meeting the standard, which is only 0.07 mg/l lower than the highest level under an "all-B" solution, requires pushing the highest level down a full 0.2 mg/l to 1.37 mg/l.

In configuration 2 (largest plant upstream), the regional treatment setup that meets 3.0 mg/l of DO also meets the added phosphorus requirement at 1.6 mg/l. However, to get down to no more than 1.5 mg/l everywhere requires upgrading the large plant to BC1. This adds almost $\$2.5 \times 10^6$ to annual regional costs and improves the highest P level by 0.10 mg/l, which is 0.07 mg/l more than necessary to just meet the constraint. It is interesting to see that in this configuration, the all-B treatment requirement produces a much higher value for the worst DO level (4.22 vs. 3.01 mg/l) than any of the other three solutions, but misses the P standard by only 0.03 mg/l. The regional cost is the same as that chosen to meet 3.01 mg/l DO and 1.5 mg/l P. In effect, the examples have produced points on an isocost frontier for attaining minimum DO and maximum P levels as sketched in Figure 5-7.

Finally, in configuration 3, the all-B requirement does not correspond to *any* of the regional setups that optimally meet the three alternative AWQ constraints. As in configuration 2, however, the minimum DO level resulting from the all-B constraint is well above the corresponding number for the other constraint sets. This is because B at plant 3 is more stringent

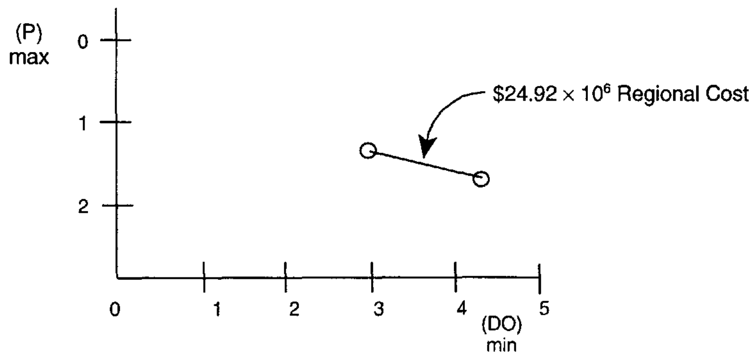
Table 5-14. Ambient Organic Phosphorus Results: Examples 1, 2, and 3 Contrasted for Configurations 1 and 2

	Configuration 1			Configuration 2		
	Base DO ≥ 3.0 Only	DO@3.0 P@1.6	DO@3.0 P@1.5	Base DO ≥ 3.0 Only	DO@3.0 P@1.6	DO@3.0 P@1.5
First Trib 1	1.67	1.57	1.37	1.46	1.43	1.43
Mainstem 5	1.33	1.21	1.22	1.81	1.53	0.98
Mainstem 4	1.44	1.35	1.26	1.54	1.42	1.19
Mainstem 3	1.57	1.43	1.34	1.57	1.44	1.21
Mainstem 2	1.56	1.43	1.34	1.57	1.44	1.21
Mainstem 1	1.56	1.43	1.34	1.56	1.43	1.20

Table 5-15. Contrasting Four Constraint Sets: Three AEQ Vectors and One Uniform Treatment Requirement

	1				2				3				4			
	DO@ 3.0 (mg/l)	DO@ 3.0 (mg/l) P@1.6 (mg/l)	DO@ 3.0 (mg/l) P@1.5 (mg/l)	B Req.	DO@ 3.0 (mg/l)	DO@ 3.0 (mg/l) P@1.6 (mg/l)	DO@ 3.0 (mg/l) P@1.5 (mg/l)	B Req.	DO@ 3.0 (mg/l)	DO@ 3.0 (mg/l) P@1.6 (mg/l)	DO@ 3.0 (mg/l) P@1.5 (mg/l)	B Req.	DO@ 3.0 (mg/l)	DO@ 3.0 (mg/l) P@1.6 (mg/l)	DO@ 3.0 (mg/l) P@1.5 (mg/l)	B Req.
Plant 1	B	B	B	B	B	B	BC1	B	BC2	B	B	B	B	B	B	B
Plant 2	B	B	BC1	B	B	B	B	B	B	BC1	BC1	B	B	B	B	B
Plant 3	B	B	B	B	P	P	P	B	P	CEPT	CEPT	B	B	B	B	B
Min DO (mg/l)	4.27	4.27	4.27	4.27	3.01	3.01	3.01	4.22	3.01	3.19	3.19	4.20	4.38	4.38	4.38	4.38
Max P (mg/l)	1.57	1.57	1.37	1.57	1.53	1.53	1.43	1.53	1.83	1.35	1.35	1.83	1.43	1.43	1.43	1.43
Cost ($$10^6$)	25.92	25.92	26.76	25.92	22.58	22.58	24.92	24.92	23.32	25.63	25.63	24.92	29.92	24.92	24.92	24.97

Figure 5-7. A Segment of Isocost Frontier for Minimum DO and Maximum P (Configuration 2)



in CBOD removal terms than the CEPT option used to meet the lower P requirements. At the same time, the highest P level under the all-B constraint is about a full half milligram per liter higher than under the low P constraint. All-B costs about $700,000 less per year than the low P optimal solutions in configuration 3.

CONCLUDING COMMENTS

This chapter has covered a lot of facts and formulas, and there is no point in trying to trace back through all of these. Rather, a few broader conclusions seem warranted and useful.

First, any number of general (or "tunable") surface water quality models are publicly available, both standing alone and embedded in one or another variety of decision support systems. On close inspection and based on the experience gained in preparing this study, none of these will prove really easy to tune, run, or interpret. In a sense, the water quality modeling part of regional cost-benefit analysis or ambient quality vector-constrained cost minimization is still best left to someone with special skills and experience—although that person does not have to be capable of building one of these models from scratch.

Surprisingly, at least to those aware of the long history of more fully ecological modeling, the readily available WQMs seem to be limited to representing and predicting the BOD/DO nexus, with occasionally, as in QUAL2E, a reasonably full algal compartment model added on and interwoven. A look at a recent review paper (Ambrose et al., 1996) seemed to confirm that it is most unusual for a model to include even microscopic algae grazers (zooplankton). And none of the models mentioned in that review include even one "fish" compartment. This may not be seen as a handicap by the Bank, because it operates in Latin America, where, as may be inferred from Chapter 2, the water quality problems being addressed are often at the very severe end of the DO scale. (DO at 0 or only slightly above so that aquatic life generally must be hard to sustain.) Further, with health effects a prime concern (see Chapter 6), even models that have quite rudimentary ecology may have a nonconservative and noninteractive compartment for coliform bacteria from human or animal waste. And finally, it may be some time before such routes to benefits as recreational angling become important to the Bank's pollution control decisions.

A potentially bothersome problem is that the mechanics of embedding one of the WQMs in a regional cost-benefit analysis model are far from trivial, and this does not seem to have been done despite the substantial number of decision support system efforts that are publicly available. It is, of course, easy to see that this would be hard to do. Most important, the generalized benefit functions that can be adjusted to local conditions just do not exist. Beyond that, the optimization routine necessary to take on even the static task of maximizing benefits minus costs would be a formidable nonlinear programming task. Finally, it seems likely to be difficult to provide a generic code for this.

The DSSs explored in this chapter were designed for the easier task of cost minimization in the presence of ambient quality constraints. The problem can be simplified even more by requiring that the treatment plant sources each represent only one technology level (one vector of removal capabilities) chosen from a set of nine possibilities. Economies of scale are built into the costing of the plants within each technology, but since sizes are fixed for any particular run, the optimization can be done by what amounts to brute force (looking at every possible choice at every plant). Beyond these shortcuts, which might well be acceptable in a project screening context, the DSS programs suffered from at least two cost function problems that had no explanation in the available documentation. One problem was corrected offline and the other was ignored because it was less serious. Errors such as these have to make one cautious about accepting the run results as gospel without a large investment of time in exploring the model's response to changing constraint sets, reach setups, and treatment plant sizes and locations.

Overall, it does not seem that the IDB can expect to take a generic model, or a system with an embedded model, and be able to apply it to each project or regional plan proposal that is submitted for review. (Equally, the Bank could not realistically require that all proposals use a particular DSS.) If the Bank thinks that a uniform proposal or review technology is a desirable goal, it will have to invest in its creation. This might be as simple a matter as embedding QUAL2E in a package that contains municipal and industrial treatment plant alternatives in the form of cost functions reflecting size and joint pollutant removal vectors. (However, some effort would have to be put into gathering the input data required by QUAL2E for many or most of the most-likely target river settings in the LAC countries. And the model itself would have to be adjusted to reproduce conditions in each regional setting.)

A more substantial investment might involve a more thorough and systematic search of the DSS universe than was possible in this project, followed by a thorough exercising and evaluation of potentially useful candidate systems. The best of the lot might then be subject to upgrading—including the addition of model background data and new capabilities, such as the ability to deal with nonpoint sources.

At the extreme, the Bank might set out to create its own DSS, tailored to its own needs and to the realities of the LAC countries. This would be a big job, and it is easier to write about it than to carry it out. It is not explored here.

Annex 5-A

More about Water Quality Model Distinctions and Characteristics

First, a brief primer on mass transport and balance equations may prove useful. For this primer, we will rely on two authoritative texts:

> *Mechanistic water-quality models are based on the* conservation of mass; *that is, within a finite volume of water, mass is neither created nor destroyed. In quantitative terms the principle is expressed as a mass-balanced equation that accounts for all transfers of matter across the system's boundaries and all transformations occurring within the system. For a finite period of time this can be expressed as Accumulation = loadings ± transport ± reactions. . . . The movement of matter through the volume, along with water flow, is termed* transport. *In addition to this flow, mass is gained or lost by transformations or* reactions *of the substances within the volume. Reactions either add mass by changing another constituent into the substance being modeled or remove mass by transforming the substance into another constituent. . . . Finally the substance can be increased by external* loadings. *. . . By combining all the above factors in equation form, the mass balance represents a bookkeeping exercise for the particular constituent being modeled. If, for the period of the calculation, the sources are greater than the sinks, the mass of the substance within the system increases. If the sinks are greater than the sources, the mass decreases. If sources are in balance with sinks, the mass remains at a constant level and the system is said to be at steady-state or dynamic equilibrium. The mathematical expression of mass conservation, therefore, provides a framework for calculating the response of a body of water to external influences.* (Chapra, 1998, pp. 13–14)
>
> *The fundamental principles of conservation of mass and conservation of energy comprise the bases for this approach. In their most elementary form these principles assert that mass and energy can neither be created nor destroyed, although they can be altered in form. Given that all residuals are either material (mass) flow or energy flow, these principles provide a set of accounting equations for keeping track of the flow of mass and energy in a natural system. These are the so-called mass balance and energy balance equations. These equations indicate that for any size volume in space — in air, water, or soil — the increase or decrease of mass or energy over any given time interval must be accounted for by either or both: (1) inputs to or outputs from the volume; or (2) transformations in the form of mass or energy within the volume over the given time period. Mass and energy cannot simply disappear. Analyzing a natural system using these principles entails dividing the system into volumes and tracing the movement over time of material and energy flows from volume to volume, using mass and energy balance equations.* (Basta and Moreau, 1982, p. 44)

Second, one-dimensional models are not the only possibility. McCutcheon describes the four dimensional categories of water quality models:

> a. ***Zero-dimensional model:*** *A segment of the stream is described by a single computational element, ignoring any lateral, vertical, and longitudinal variation that may occur. The single element is treated like a completely mixed reactor. This approach may be most useful in a screening-level analysis of a mixing zone. It is rarely expected to be used in typical stream studies.*
>
> b. ***One-dimensional model:*** *Where lateral and vertical variation is unimportant, the stream is described by a series of computational elements extending downstream and describing the longitudinal gradients that are prevalent in streams. This is the most common approach to describing stream water quality.*
>
> c. ***Two-dimensional model:*** *This is a model that describes the variation in two directions. The most useful riverine type describes lateral and longitudinal gradients and assumes that vertical variations are unimportant (i.e., water quality can be depth averaged). These models are occasionally used to define mixing zones in the vicinity of where point sources of pollution enter the stream. Rivers are not frequently stratified; but when this occurs, vertical gradients are usually important as well. Therefore, width-averaged, stratified river models do not seem to [be] very useful at present.*
>
> d. ***Three-dimensional model:*** *This is a model that describes vertical, lateral, and longitudinal gradients of water quality parameters. These models are occasionally used to describe complex mixing zones such as those formed by cooling water discharges from electrical power generation plants where the stratification results in significant vertical gradients.* (McCutcheon, 1989, p. 8)

Third, a summary categorization of water quality models in terms of complexity or sophistication is sometimes used. Ambrose et al. (1996) proposed a hierarchy—the Stream-Reservoir Model Classification. This has also been summarized in McCutcheon:

> ***Level I.*** *Simple manual or graphical methods based on statistical or deterministic equations that are easily used for crude screening over extensive areas to isolate existing or potential trouble spots for detailed follow-up analysis. Important issues and preliminary management options may be suggested by the results, or some preconceptions may be ruled out, but uncertainties are typically large and are not quantified. . . .*
>
> ***Level II.*** *Simple computerized model used for fine screening or crude planning and assessment for extensive areas over extended periods of time. Model equations are usually deterministic in nature but only approximate the basic processes. As a result, any management projections are somewhat uncertain and wise use requires considerable experience in interpreting the resulting calculations. Formal uncertainty analysis is usually not included. Data collection requirements are usually limited to one preliminary data collection study.*
>
> ***Level III.*** *Computerized model of intermediate complexity used as a fine planning model or a crude engineering design or resource management model. Extensive areas and extended periods of time can be simulated but at significant cost in data collection and preparation, and in computing time. Some approximation of the basic processes limits the applications for design and management. Data collection involves at least two independent data sets to bracket important conditions. Uncertainty analysis is*

> *typically included as a part of the computer model or as part of a supplemental analysis. . . .*
> **Level IV.** *Advanced mechanistic computerized model used for detailed design and management. Data requirements are usually intense and usually involve at least a preliminary data collection study and model screening to design at least two intensive synoptic data collection efforts. Simulations are typically limited to smaller areas and short time periods to avoid extensive data collection and computing costs. Procedures using these models are not defined well enough to specify typical uncertainty analyses required.* (McCutcheon, 1989, pp. 7–8)

Essentially, then, Level I models can be characterized as steady-state solutions with simple kinetics, while Level II models include steady hydrodynamics, either specified or handled empirically with steady or time-variable water quality, where the time resolution is on the order of weeks to a month. Level III models are characterized by nonsteady-state hydrodynamics, but with a simplified solution, dynamic water quality, and time resolution on the order of less than a day. Finally, Level IV models include nonsteady-state hydrodynamics with full equation routing, the ability to handle backwater and stratified reservoirs, and dynamic solutions to water quality variables with time resolution on the order of hours (McCutcheon, 1989).

Annex 5-B

More on the Comparison of the STREAMPLAN and QUAL2E Water Quality Models

The key distinctions between the models are shown in Figure 5B-1. There the shaded boxes and lines reflect elements that are shared by the two models. The elements without shading are found only in QUAL2E.

More formally, but still suppressing as much of the math and notation as possible, consider two dissolved oxygen formulations. In the first, the STREAMPLAN version, the effect of any particular source, j, on DO is independent of the levels of discharge of all other sources. In the other one, QUAL2E, the ΔDO at point k attributable to a change at the particular source, j, is a function in general of all the other discharges to the river. This is because certain "rate constants," real constants in the first model, are functions of ambient water quality levels in the second. Thus, the first model, in rate-of-change form has, for dissolved oxygen deficit (DOD):

$$\frac{dDOD}{dt} = K_1CBOD + K_4K_2(NH_4\text{-}N) - \frac{1}{H}[K_5(DOD)] + \frac{1}{H}K_6 \qquad (5B\text{-}1)$$

where:

K_1 = decay rate of BOD
K_2 = NH_4-N decay rate
K_4 = O_2/unit of (NH_4-N) decay
K_2 K_4 = oxygen demand from (NH_4-N)
K_5 = reaeration of coefficient
K_6 = sediment O_2 demand
H = mean reach depth

All the above are real constants with respect to CBOD, NH_4, and DOD, although not with respect to temperature. The notation (NH_4-N) means nitrogen concentration as ammonia.

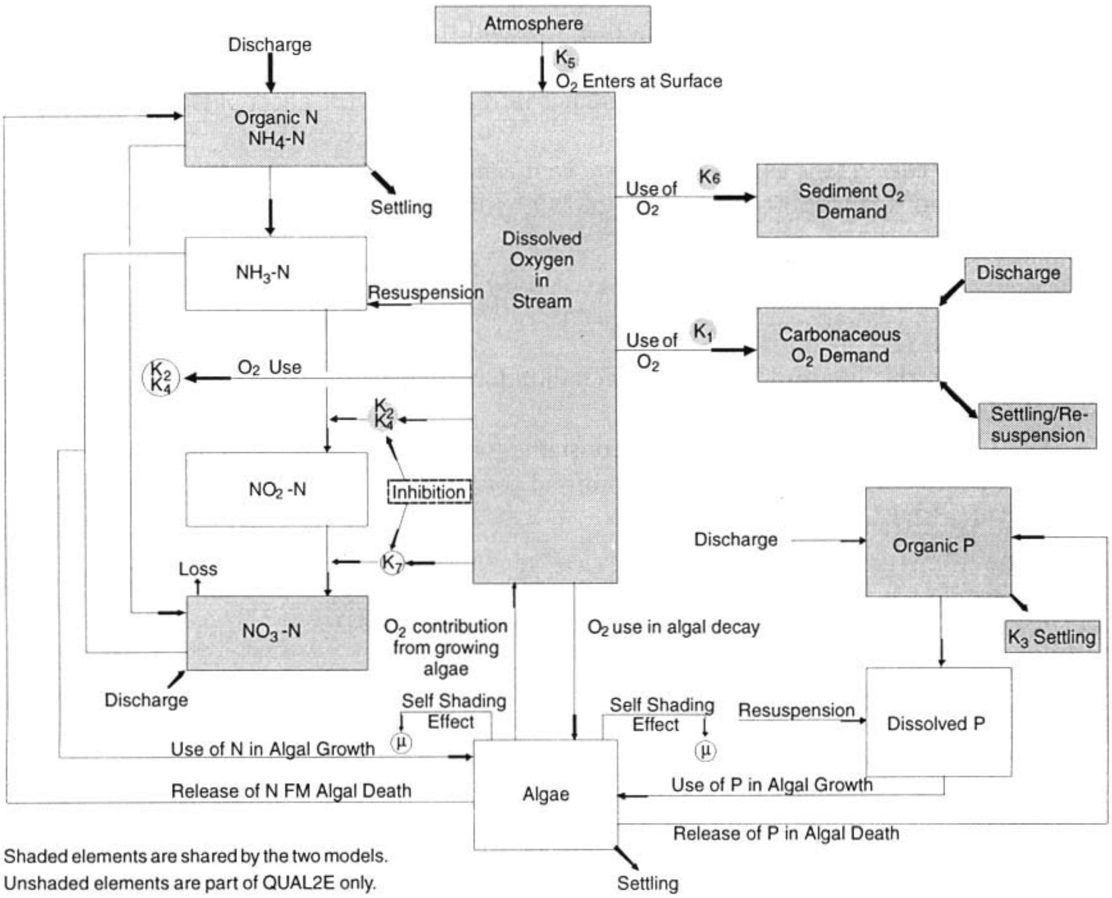
Solving (5C-1) for the equation for DO(t) as a function of H, and the other constants as well as DO_{sat} (also a function of temperature) produces the following form:

$$\begin{aligned} DO(t) = & -L_1(H, K_1, K_5, t)\ CBOD\ (0) - L_2(H, K_2, K_4, K_5, t)\ [NH_4\text{-}N(0)] \\ & + L_3(K_5, H, t)\ DO(0) + DO_{sat} - L_4(K_5, H, t)DO_{sat} + L_5(K_5, H, K_6, t) \end{aligned} \qquad (5B\text{-}2)$$

where x(0) indicates quantity of variable x at t = 0 and the $L_j(K_j, K_k \ldots t)$ are more or less complex functions of the indicated constants and time. However, they are constants with respect to CBOD, NH_4-N, and DO.

If there are two sources of BOD and/or NH_4-N to reach j (one to be thought of as everything upstream of the top of the reach containing source j; and one source, j, discharging at the top of the reach), then the total effect at any time (read "distance") downstream from the

Figure 5B-1. Process Distinctions between STREAMPLAN and QUAL2E



Shaded elements are shared by the two models.
Unshaded elements are part of QUAL2E only.

top of the reach is the sum of the effects of the two sources. In particular, the effect of the discharger is independent of what is going on above it. Another way of saying this is that the partial derivative of DO(t) with respect to the discharge of $CBOD_j$ is a function only of constants, not of upstream discharges.

The second model, in rate-of-change form, but for dissolved oxygen instead of the deficit (with translation of notation to make this as consistent as possible with the STREAMPLAN model), is

$$\frac{dDO}{dt} = K_5(DO_{sat}-DO) + (\alpha_3\mu-\alpha_4\rho)A - K_1CBOD - K_6/H - K_2K_4(NH_4\text{-}N) - K_7(N_2\text{-}N) \qquad (5B\text{-}3)$$

where: K_1, K_2, K_4, K_5, K_6 and H are as for (5B-1), and:

A = algal biomass concentration
α_3 = oxygen production per unit of algal photosynthesis
α_4 = oxygen uptake per unit of algae respired
μ = algal growth rate
K_7 = oxygen demand for oxidizing nitrite to nitrate
ρ = algal respiration rate nitrogen (N_2-N)

Now the first term is roughly the same as the reaeration term in the simpler model of Eq. (5B-1). The CBOD decay term is also exactly the same, as is the sediment term. On the surface the nitrogen terms look the same, except that a second stage in the oxidation of ammonia (N) has been included. The only obvious difference is the inclusion of algae, although it turns out that in QUAL2E there is also an interdependence introduced via the effect of DO on the quasi-constants K_2K_4, and K_7.

The latter effect is the easiest to explain, so it can be gotten out of the way first: QUAL2E makes K_2K_4, and K_7 functions of the level of DO by introducing a correction factor for each, of the form:

$$CF = 1 - e^{-KNITRF(DO)} \tag{5B-4}$$

and KNITRF is the first-order nitrification inhibition coefficient, with a value taken to be somewhere between 0.6 and 0.7.

KNITRF acts to reduce the normal constants for the oxygen uptake nitrification in the simplest way: $K'_j = CF(K_j)$ where K_j is the normal constant and K'_j is the corrected one. Thus, using the numbers from QUAL2E:

$$K_2K_4 = (CF)\ \alpha_5,\beta_1 \text{ or } K_2K_4 = g_1(DO,\alpha_5,\beta_2) \tag{5B-5}$$
$$K_7 = (CF)\ \alpha_6,\beta_2 \text{ or } K_7 = g_2(DO,\alpha_6,\beta_2)$$

And CF looks like this:

DO (@20°C)	−0.65 (DO)	$CF = 1 - e^{-0.65(DO)}$
9	−5.85	0.9972
8	−5.20	0.9945
7	−4.55	0.9895
6	−3.90	0.9798
5	−3.25	0.9613
4	−2.60	0.9358
3	−1.95	0.8578
2	−1.30	0.7275
1	−0.65	0.4780

Therefore, the contribution of a discharger of NH_4-N to the rate of change for DO depends on the DO at the point of discharge, which in turn is a function of upstream discharges and the extent to which they have affected DO at the point of discharge of interest.

The algal relationships are more complicated and themselves depend on nitrogen and phosphorus loadings as well as on light. The algal growth rate, μ, depends on growth limitation factors related to light, nitrogen, and phosphorus availability. One possible formulation is

$$\mu = \mu_{max}[(FL)(FN)(FP)]$$

where μ_{max} is the maximum possible rate of growth and the F_js are the limitation factors. Now FL depends itself on A, the algal concentration via the "self-shading" effect; or FL = f(A). The FN and FP depend on the concentrations of nitrogen and phosphorus relative to the half-saturation constants for each.

However, each of these nutrients has its own cycle in which algae figure as determinants of the rates of change of nutrient concentrations, and those concentrations help determine the rate of change of algal concentrations. So, through these cycles, the algal term in the dDO/dt equation can be written as

$$\{\alpha_3\mu[A(N,P)] - \alpha_4\rho\}A(N,P)$$

An analytical solution for the DO(t) equation is not possible. However, as long as the elements that are true constants in the simpler model appear in QUAL2E as functions of DO or nutrient loads (from upstream), the effective partial derivative of the implicit DO(t) equation with respect to any discharge, j, *should not* be independent of the upstream loads of CBOD and nutrients.

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Chapter 6

Benefit Categories, Evidence about Willingness to Pay and Relative Importance by Categories and Countries

This chapter and the next discuss the important matter of the benefits of water quality improvement in considerably greater detail than the earlier chapters on general analytical methods and problems. This is a reflection of a judgment that, for project justification purposes, the "action" is on the benefit side. The goal is to provide a common background concerning categories of benefits,[1] empirical evidence about how large benefits of water quality improvement seem to be by categories, how water quality benefits (or damage) in developing countries compare with analogous measures related to air pollution, how benefit and damage numbers from developing countries compare with analogous estimates for developed countries, and what benefit numbers are to be found in the documentation of loan applications to the IDB. Chapter 7 concentrates on examining the methodologies available to produce such estimates, and the applicability of these methodologies to developing countries.

PRINCIPAL CATEGORIES OF BENEFITS

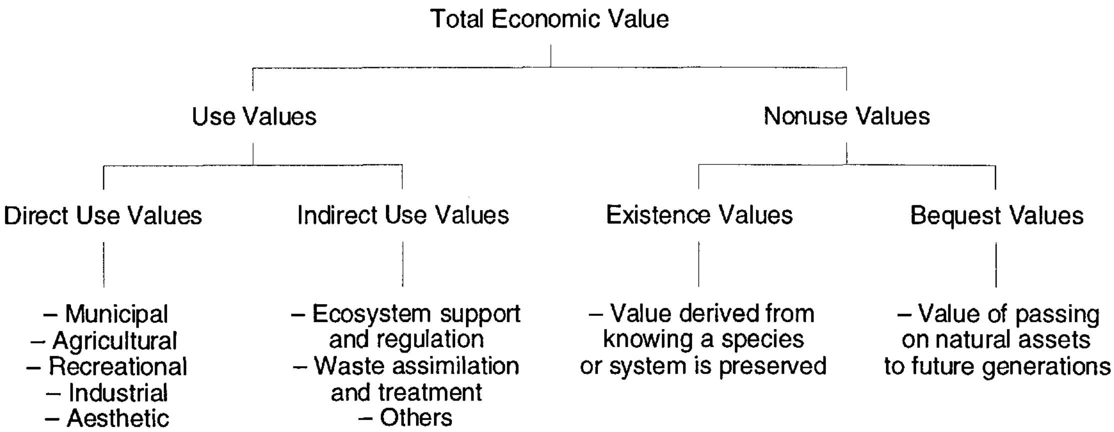
The benefit framework used throughout this chapter and the next consists of the traditional division of values into use and nonuse values. Although this taxonomy is undoubtedly familiar to many readers, a brief review may be helpful. Figure 6-1 illustrates this taxonomy.

Use values follow from the use of an environmental good or service in a consumptive or productive process. Direct use values of water can be apportioned into five different categories: municipal, agricultural, recreational, industrial, and aesthetic. They can include both goods (environmental commodities), such as drinking water or shellfish, and services (environmental functions), such as water-based transportation or recreation. The key distinction between direct and indirect uses is that in the case of direct uses, the environmental components or environmental functions themselves enter into the consumption or production activity whereas in indirect uses, the environmental function affects the consumption or production activity only through an intermediate good or service. For example, a coral reef that protects a beach from storm surges is an example of an indirect use because the protection affects the consumption of beach-front recreation only because it helps to preserve the beach.[2] In addition to storm and

[1] The word "categories" has roughly the same meaning as the phrase "routes to (benefits)" used elsewhere in this study. Both refer to a type of human reaction to ambient water conditions.

[2] Whether the direct/indirect distinction is relevant depends to a large extent on what the effects are projected to be of the project or policy being valued. To pursue the coral reef example a bit further, if the project is aimed at keeping

Figure 6-1. Sources or Types of Values of Environmental Goods and Services



sea protection, indirect use values include groundwater recharge, flood control, sediment and nutrient retention, and the support of ecosystems.

Nonuse values are those that people place on natural resources even though they do not use the resources, either directly or indirectly. These values come in two types: existence and bequest. An existence value accrues when someone is willing to pay to preserve a natural resource even though they have no plans to ever make use of it; they simply value its existence. Bequest values accrue when someone is willing to pay to preserve a natural resource so that it will be around for the benefit of future generations. Although a third form of value—option or quasi-option values—has traditionally been included in this taxonomy, these values are omitted from our discussion because they are no longer viewed as a separate category of value.[3]

Many authors have employed the term "total economic value" (TEV) to describe the sum of these different values. This description can be misleading because the different values are not necessarily additive. Some uses of a resource may preclude other uses or values, so the values can only be additive to the extent that they are consistent with any one management strategy for the resource. A closely related problem is that there is often an imperfect fit or overlap between the valuation methodologies and the benefit categories, so that even if the values are additive, our estimates of these values often are not. Thus, attempts to generate TEV by summing estimates of individual values obtained using several methodologies may encounter problems with inconsistency and nonadditivity, either because the resource cannot physically generate all of the different values purported to be measured or because of overlaps or inconsistencies

coliform bacteria counts at the beaches down at safe levels, the reef is just part of the background information. But if the project is to protect the reef ecosystem from, say, pesticide runoff from farms behind the beach, then the fact that the reef protects the beach produces one route to the benefits of protecting the reef. Other routes could be direct use values (via snorkeling and diving) and nonuse values revolving around species preservation and the bequest motive.

[3] "A special category of value is *option value*, which arises because an individual may be uncertain about his or her future demand for a resource and/or its availability as a wetland in the future. There is a general consensus in the economics literature that option values are not a separate form of value but represent a difference between *ex ante* and *ex post* valuation. If an individual is uncertain about the future value of a wetland but believes it may be high or that current exploitation and conversion may be irreversible, then there may be *quasi-option* value derived from delaying the development activities. Quasi-option value is simply the expected value of the information derived from delaying exploitation and conversion of the wetland today. Again, there is consensus that quasi-option value is not a separate component of benefit but involves the analyst properly accounting for the implications of gaining additional information." (Barbier, 1994, p. 156)

"I think it is time to expunge option value from the list of possible benefits associated with environmental protection." (Freeman, 1993, p. 264) in the measurement of these values. Conversely, estimates of TEV are not in general decomposable into underlying categories. A survey that asks for WTP to preserve a resource generates a TEV for that resource, but this TEV cannot be distributed among individual values (Cummings and Harrison, 1995).

Summing estimates of the value of a mangrove forest for both timber production and fisheries support provides an example of policy inconsistency because the fishery would likely be damaged by the harvesting of timber. Combining these estimates with the estimates from a WTP survey on the value of coastal, water-based recreation provides an example of overlap, if the respondents consider timber production or fishery support in formulating their response to the WTP question. (See Chapter 2 for an example of a likely overlap of this sort between beach recreation and earnings from beach-related tourism.)

Valuation Methodologies

The general methodologies for environmental resource evaluation are likely to be familiar ground for readers, but a brief review anticipating the longer discussion in Chapter 7 may be useful. The methodologies employed to measure the benefits from wastewater treatment usually depend on the values (or routes to benefits) that are identified with the situation. Goods and services that are traded in the market can be valued through various market-based mechanisms. These methods are well established and have been around ever since Dupuit first pondered the value of a new bridge. The measurement of nonmarket values, on the other hand, is a much more recent phenomenon and is much less settled. There are basically two kinds of nonmarket valuation methodologies: indirect and direct.[4] Indirect methodologies are so named because they value nonmarketed goods or services through the action of consumers of them in related markets—markets for complementary or substitute goods. The three most common forms of indirect estimation are travel cost (TC; including variants such as random utility and hedonic travel cost), hedonic analysis, and averting behavior. Averting-behavior methods value nonmarketed functions through the costs of private and public efforts to substitute for poorer than desired environmental quality. Although these last methods are not employed with the frequency of the others, they are of particular importance for our purposes because they may sometimes be a last resort in the measurement of indirect use values, which in turn may constitute a prime source of value for complex, multifunctional resources such as coastal wetlands. Indirect methodologies are limited to the measurement of use values because there are no goods or services associated with nonuse or option values. On the other hand, direct methods can measure both use and nonuse values through surveys that ask respondents to reveal their total WTP for specific changes in environmental goods and services.

Use Values

Direct

The first type of direct use value—municipal—includes water used for drinking, bathing, and washing. Table 6-1 lists a dozen studies from around the world that deal with aspects of this category. Before commenting on these studies, however, it is worth digressing on the special problem of water for these uses in many parts of the developing world.

[4] There is an unfortunate possibility of confusion here because "direct" and "indirect" are being used for two quite separate items. Earlier, the words referred to the connection between an environmental resource and what is valued by consumers. Methodologically, however, "indirect" refers to ways of inferring values for environmental goods and services from behavior that involves related market behavior while "direct" refers to methods that ask rather than infer.

Table 6-1. Illustrative Estimates of Municipal Use Values

Authors and Date	Country	Method	Numbers
Gilman and Skillicorn (1985)	Bangladesh	Averting	$29.40/yr/household to boil all drinking water for 10 min, representing from 3–8 percent of income
Harrington et al. (1989)	U.S.	Averting	Losses due to giardiasis outbreak estimated at from $12.1 million to $38.5 million
Whittington et al. (1990a)	Kenya	Averting	Authors conclude that value placed on time spent hauling water equivalent to unskilled wage rate
Sadoff (1996)	Morocco	Averting, health effects	NPV for period 1995–2020 of excessive water treatment costs: $350 million–$2.3 billion; health costs, $97 million; fisheries costs, $8 million
Serôa da Motta et al. (n.d.)	Brazil	Health effects	Estimate total health-related damage from water pollution in Brazil at $387.9 million for 1989
Maimon (n.d.)	Brazil	Health effects	Water pollution in Rio de Janeiro estimated to cost $12.5 million annually (medical expenditures, $2.8 million; lost wages, $9.7 million)
Briscoe et al. (1990)	Brazil	CVM	Average maximum WTP was 100 cruzados/mo, about 2.3 percent of mean household income
Whittington et al. (1990b)	Haiti	CVM (availability)	Mean WTP for standposts = $1.14/mo. Mean WTP for private connections = $1.42/month
Whittington et al. (1993)	Ghana	CVM (availability)	Households w/o indoor toilets: WTP for indoor toilets = $1.40/mo. Households with indoor toilets: WTP for sewer = $1.30. Households w/o water: WTP for water = $1.52 and water + indoor toilets = $2.57
Whittington et al. (1991)	Nigeria	CVM (availability)	Authors present frequency distribution of WTP bids for piped water
North and Griffin (1993)	Philippines	Hedonics	WTP for piped water = $1.41–$2.25/mo (about 2.4 percent of income) and $0–$.94 for deep well
Kwak and Russell (1994)	Korea	CVM (quality)	Mean household WTP = $3.34/mo. Aggregate WTP = $106,000,000/yr

When discussing the contamination of municipal water supplies, we have to be careful to understand the connection between wastewater treatment and municipal water treatment at potable water system intakes. There seems to be a tendency among analysts from developed countries to view the problem of inadequate access to potable water as an entirely separate problem from wastewater treatment. This perspective assumes away the problem of what to do with the "used" water. However, in many developing countries, the lack of comprehensive collection and distribution of municipal water supplies and, where such facilities do exist, the lack of universal access to such facilities, results in household water being collected in a number of ways and from a number of different locations. The contamination of local sources of water adversely affects the ability of downstream dwellers to obtain safe drinking and bathing water and thus imposes costs on them. These costs can be in the form of adverse health effects, an increased need for averting behavior, such as boiling drinking water, or an increase in the price of, or effort needed to collect, potable water. These concerns are particularly relevant to communities where the rate of urbanization continues to outstrip infrastructure construction and on small islands that have few sources of fresh water.

Although this use value is a prime candidate for estimation by valuing the averting behavior undertaken to prevent adverse health effects, few studies explicitly do so.[5] Some

[5] Averting-behavior studies are particularly appropriate for valuing municipal use values (or more accurately, a lower bound on such values) because the averting behavior is common, somewhat standardized, and relatively easy to assign a value to (at least when compared with the difficulties associated with valuing pain and suffering). This is not to say, of course, that the resulting values are a conceptually correct measure of the benefits of water quality

studies implicitly utilize averting behaviors by taking the costs of preventive medicine and treatment necessary to prevent the accrual of these health costs as a lower bound on the health costs associated with water pollution. The studies by Serôa da Motta et al. (n.d.) and Maimon (n.d.) employ this technique. A more common approach is to use direct (contingent valuation) methods to derive WTP values for the provision of clean drinking water. However, such studies typically measure more than water treatment because they are generally performed on households that have no existing water supply. The work by Kwak and Russell (1994) is an example of where the authors were able to isolate the benefits associated only with improved quality (freedom from the bad tastes and odors resulting from contamination of the municipal water supply).

Another method of estimating health costs associated with water pollution is to compile or estimate health costs for certain kinds of illness (or value of life in the event of premature death) and attribute the occurrence of these illnesses or deaths to water pollution, either on the scale of an individual outbreak, such as Harrington et al. (1989) do for a giardiasis outbreak in Pennsylvania, or where health risks are more continuous, in some predefined area over a predefined time, as Sadoff (1996) does for the Sebou basin in Morocco, Margulis (1992) does for Mexico, and Maimon (n.d.) and Serôa da Motta et al. (n.d.) do for Rio de Janeiro and for Brazil, respectively. See Annex 6-A for a list of these and other studies that estimate benefits from municipal uses of cleaner or more readily available water supplies.

A second category of direct use values is agricultural and is almost entirely irrigation. The large-scale reuse of untreated domestic sewage water for irrigation is commonplace in many arid and semiarid zones of LAC countries, often without effective controls. The contamination of water used for irrigation can generate adverse health effects among farm workers or when the irrigated crops are consumed; it can also destroy the productivity of land under irrigation.[6] Examples of large-scale wastewater reuse for irrigation, some with raw sewage, include the Santiago metropolitan area (Chile), Mexico City (Mexico), Lima (Peru), and Maracay (Venezuela). The World Bank (1993) has estimated the costs associated with the discharge of municipal and industrial wastes from the Santiago metropolitan area into water that is used to irrigate 130,000 ha of crops, including 7,000 ha of vegetables grown for raw consumption in Santiago. Grandstaff (1986) has estimated the value of reductions in agricultural production due to a loss of irrigation as a result of contaminated water in the Philippines. Although soil productivity can be affected by irrigation with polluted water, no attempts to estimate the value of such effects could be found. Estimates of the economic effects of soil erosion on agricultural productivity are much more common (Veloz et al., 1985; Lutz et al., 1994).

Recreation—fishing, swimming, and boating—constitutes the third category of direct use values. Diminishing water quality can render bodies of water and wetlands unsuitable for recreation by creating health risks from ingestion or contact with the water and by altering or destroying plant, fish, and wildlife stocks that are inputs into the recreational process. Damage to recreational resources takes on added significance when the resource generates foreign revenue through international tourism. Pristine coral reefs are a prime example of a tropical resource that has great potential for attracting tourists (Pendleton, 1994; Maragos et al., 1983)

improvement, only that they are relatively easy to derive. Dixon (1993) and Sudman (1995) describe instances where such averting behavior occurs. Gilman and Skillicorn (1985) and Abdalla et al. (1992) are examples of averting-behavior studies, the former in a developing country and the latter in a developed country.

[6] "In the Tula Basin, Mexico City's wastewater is used to irrigate a variety of crops, including vegetable plots. This practice has contributed to the spread of parasitic diseases like amoebiasis and cysticercosis. It also has polluted good agricultural soils. In one year, as much as 2,300 kilograms of detergents or 750 kilograms of boron find their way into the soil." (Ezcurra and Mazari-Hiriart, 1996, pp. 27–28)

Table 6-2. Estimates of Coral Reef Values

Authors and Date	Country	Value	Method	Numbers
Hodgson and Dixon (1988)	Philippines	Production	Market	Loss in gross revenues of $40 million over a 10 year period
McAllister (1988)	Philippines	Production	Market	Loss from decrease in domestically consumed fish = $160 million/yr. Loss from exports = $11 million/yr
Riopelle (1995)	Indonesia	TEV	Several	NPV of 1 km^2 of reef of more than $1,000,000
Cesar (1996)	Indonesia	Coastal protection	Market	NPV of 1 km^2 of reef for coastal protection ranges from $25,400 to $550,000, depending on extent of coastal development
Cesar (1996)	Indonesia	Tourism	Market	NPV of 1 km^2 of reef for tourism ranges from $3,000 to $503,000 depending on attractiveness of site
Cesar (1996)	Indonesia	Fishery	Market	NPV of 1 km^2 of reef as fishery is $108,900
Mendelsohn et al. (1994)	Belize	Recreation	TCM[a]	Value of $367/trip or aggregate of $78.9 million/yr
Dixon et al. (1995)	Bonaire	Recreation	CVM	Average WTP per diver per year = $27.40

[a] TCM, travel cost methodology.

and that is highly sensitive to wastewater discharges.[7] See Table 6-2 for a sample of studies attempting to value the benefits associated with coral reefs.

However, in discussing the economic benefits associated with the protection of a natural resource that is highly valued by international tourists, one must be careful to note the spatial limitations on such benefits. A benefit estimation study that attempts to measure the consumer's surplus generated among international tourists by a coral reef will capture only those benefits that accrue to people from countries (and often regions) other than the one in which the reef is located. Since, for Bank purposes, relevant benefits are limited to the recreational benefits enjoyed by the country's citizens and to the producer's surplus generated in the country by recreating non-nationals, these studies are typically useful only in that they provide guidance on the amounts that these non-nationals would be willing to pay to gain admission to the various sites (Powers, 1974). However, it might be possible to capture some of these benefits (with an offset for loss of producer's surplus due to fewer tourists) if the study were to explicitly consider not only the preservation of the natural resource but also a change in the pricing policy. In any event, there are a number of studies that specifically examine the willingness of foreign visitors to pay entry fees to environmentally sensitive areas in developing world contexts, including Wells's 1993 study of protected areas in Nepal, Edwards' 1991 study of the Galapagos Islands, Balderes and Laarman's 1991 study of protected areas in Costa Rica, Dixon et al.'s 1995 study of Bonaire Marine Park, and, most recently, Shultz et al. (1998), dealing with Costa Rican parks. This last study provides a good example of an attempt to estimate the tradeoff between environmental quality and elements of the benefit-producer's surplus through market estimation and consumer surplus through a CV survey. See Annex 6-B for a sample of other attempts to estimate benefits from water-based recreation in developing countries.

Industrial (and commercial) uses constitute the fourth category of direct use values. These include not only industrial use of water in processes and cooling, but also commercial fisheries, fuelwood exploitation, and tree products more generally.

As a practical matter, industrial uses do not appear to be much harmed by the lack of domestic wastewater treatment in the developing world. This is because either the use is

[7] "Sewage and siltation linked to human development, for example, are among the most significant causes of coral reef and other ecosystem degradation in much of tropical Asia, Polynesia and the Caribbean." (Turner et al., 1995a, p. 1) See also Maragos et al. (1983), pp. 245–246.

relatively insensitive to quality (i.e., cooling or cleaning) or it is highly sensitive and requires expensive treatment regardless of the ambient water quality (e.g., boiler water or water for the production of foodstuffs). No examples of attempts to estimate benefits to industries in developing countries from cleaner water have been found. A sample of commercial use values involving many different places and specific routes may be found in Annex 6-C.

Finally, the receiving bodies of water are a part of the landscape and their degradation also degrades the sights and smells of the landscape. Individuals can be affected in the midst of near-water recreation, such as hiking or picnicking, at work or at home, or while commuting between the two. There is a paucity of studies estimating benefits solely from aesthetic values. However, d'Arge and Shogren (1989), in a hedonic comparison of housing values near a "clean" and a "dirty" lake, undoubtedly capture differences in the aesthetic values of the lakes, along with other differences.

Indirect

The environmental functions performed by coastal ecosystems such as marshes, river deltas, coral reefs, and mangrove swamps are quite extensive. These functions are a major source of indirect use values. The environmental services identified for coastal ecosystems in LAC areas include waste assimilation and purification, storm and sea protection, flood control, sediment and nutrient retention, support of other ecosystems, and biomass export (see, for example, Maragos et al., 1983; Hinrichsen, 1996; and Barbier et al., 1997).

In LAC coastal areas, mangrove forests in particular are an important source of fuelwood and also provide a large number of traditional forest products.[8] The introduction of wastewater into these ecosystems can render the fish stocks unfit for consumption and create conditions under which valued plants and wildlife cannot survive. The difficulty of estimating the benefits that accrue from ecosystem functions helps explain the scarcity of relevant studies. However, there have been some attempts, generally looking at both direct and indirect use routes, to value wetlands in developing and developed countries. These values imply benefits for protecting the wetlands from destruction, but are less useful when the question is one of more modest degradation. The studies by Bennett and Reynolds (1993), Lal (1990), and Hodgson and Dixon (1988) are all excellent examples of attempts to estimate the wide range of commercial values that can be derived from a complex ecosystem dependent on water quality (mangrove forest) by indigenous people or large-scale commercial applications. See Table 6-3 for a summary of the results of these and other studies that attempt to measure the values associated with wetlands. Other studies concentrating on indirect routes are noted in Annex 6-D.

In this connection, a special word on fisheries is in order, for the fish nursery function of wetlands is often emphasized.[9] Chapter 4 contained a discussion of the problem of taking credit for fishery preservation or enhancement when the fishery is unmanaged (open access). McConnell and Strand (1989) even assert that an improvement in water quality might actually decrease the social returns from an overharvested, open access fishery if improvements in water quality increase demand for the production of the fishery.

Biodiversity as a source of value is difficult to categorize because it contains elements of not only direct and indirect use values but also of existence values.[10] Although much of the

[8] Turner et al. (1995a) list over 70 human activities that are linked to mangrove forests.

[9] See Maragos et al. (1983), pp. 233–244, for a brief explication of the roles of coastal marshes, mangrove forests, deltas, estuaries, and coral reefs in supporting offshore fisheries.

[10] "Biological diversity (or biodiversity for short) plays two central roles in the evolution of ecosystems. First, it provides the units through which both energy and materials flow, thus giving the system its functional properties. Secondly, it provides the ecosystem with resilience." (Dasgupta and Mäler, 1997, pp. 3–4)

Table 6-3. Estimates of Wetland Values

Authors and Date	Country	Value	Method	Numbers
Bennett and Reynolds (1993)	Malaysia	Direct and indirect use	Market	Destruction of mangrove area estimated to cause loss of revenue from fisheries and tourism of up to $24.8 million/yr
Breaux et al. (1995)	U.S.	Indirect use	Market	Discounted cost savings from using wetlands as substitutes for traditional wastewater treatment ranged from $785 to $34,700 per acre
Christensen (1982)	Thailand	Direct and indirect use	Market	Charcoal production = $318/ha/yr. Nipa cultivation = $230/ha/yr. Interestuary fishery = $30/ha/yr. Out-estuary fishery = $100/ha/yr. Oyster culture = $60/ha/yr. Shrimp farming = $206/ha/yr. Rice cultivation = $220/ha/yr
Costanza et al. (1989)	U.S.	Total	CVM, energy analysis	Total present value for an average acre of Louisiana wetlands from $2,429 to $6,400/acre (8 percent discount rate) to $8,977–$17,000/acre (3 percent discount rate)
Farber (1996)	U.S.	Indirect use	Market	Present value of protection from hurricane wind damage afforded by an acre of Louisiana wetlands was $6.82 to $22.94 (1980 dollars)
Farber (1987)	U.S.	Total	Market, CVM, etc.	Total present value in 1990 of an acre of Louisiana wetlands ranged from $8,437 to $15,763
Farber and Costanza (1987)	U.S.	Total	Market, CVM, TCM, energy analysis	Value of annual marginal product/acre for commercial fishing and trapping was $37.46 (1983 dollars); average annual WTP per household to use wetlands for recreation was $103.48; annual value derived from TC was $6/acre; and protection from storm damage, $7.48/acre. Total present value was $586.73/acre. Total present value from energy analysis was $6,400 to $10,602/acre
Folke (1991)	Sweden	Total	Energy analysis/ replacement	Total undiscounted annual monetary replacement cost of $0.4 million to $1.1 million (1989 dollars) for 32 km² of lost wetlands
Lal (1990)	Fiji	Direct and indirect use	Market	NPV of $3,277/ha for forestry ($226/ha) and fishing ($3,051/ha) (1986 U.S.$)
Lynne et al. (1981)	U.S.	Indirect use	Market	Total present value of an acre of marsh in the production of blue crab was $3.00
Navrud and Mungatana (1994)	Kenya	Direct use (Recreation)	TCM, CVM	TCM: average annual consumer's surplus = $114–$120/nonresident and $68–$85/resident. CVM: Average annual WTP = $73/nonresident and $20/resident
Ruitenbeek (1994)	Indonesia	Direct and indirect use	Market	Total present value = $6,250/ha. Commercial fishery = $145.8/ha/yr. Commercial forestry = $83.3/ha/yr. Traditional local uses = $41.7/ha/yr
Thomas et al. (1991)	Tunisia	Direct and indirect use	Market	Annual revenue or cost saved from wetlands was $1.1 million (fisheries and livestock, $0.59 million, water supply $0.34 million, tourism $23,000, and sewage treatment $156,000)

reported research on the value of biodiversity has been focused on tropical forests, it is becoming clear that coastal resources are also rich centers of this attribute.[11] One of the more obvious use values is for medicines, although the enthusiasm for these benefits has faded in light of recent surveys (e.g., Simpson et al., 1996) suggesting that earlier estimates were vastly overstated. Whatever the value, coral reefs may eventually rival tropical forests in the development of new drugs, although no studies attempting to value this aspect of a coral reef have been located.[12]

Nonuse Values

Nonuse values have received much less attention in developing countries than in developed countries. Despite some initial enthusiasm (e.g., Greenley et al., 1981 or Sutherland and Walsh, 1985), it now seems to be widely accepted that for all practical purposes, it is impossible to separate out nonuse values except under unusual circumstances (Cummings and Harrison, 1995). As a result, any measurement of total economic value will likely include an element of nonuse value, but an element that cannot be quantified. Eleven efforts to estimate total economic values for particular resources are summarized in Annex 6-E.

Option Values

Option values are rarely estimated by themselves because of the difficulty in adequately isolating them both theoretically and empirically, but Kaoru (1993) and Desvousges et al. (1987) are examples of studies that have attempted to single them out. In any case, as noted earlier, there is a growing consensus that these are not sources of value that are symmetric with the others discussed, but rather are values for different aspects of the world.

Total Economic Value

See Annex 6-E for studies purporting to generate a TEV for a particular natural resource. Two developed country studies provide specific examples of the two different ways one can proceed in determining TEV. Mazzotta et al. (1997) is a CV survey that derives WTP to conserve different types of land. Farber (1996), on the other hand, attempts to estimate the TEV of Louisiana wetlands by compiling estimates of the values of different environmental goods (components) and services (functions). Although the second approach may avoid concerns

[11] "Biologists consider mangrove forests to be one of the most productive and biologically diverse wetlands on Earth, supplying habitats for more than 2,000 species of fish, shellfish, invertebrates, and epiphytic plants. Their root zones provide sanctuary for sponges, crested worms, crustaceans, and molluscs, as well as algae; their intertidal zones create habitats for a variety of crabs and small animals; their canopies harbor hundreds of species of birds; and their estuaries shelter marine mammals such as dugongs, manatees, and otters, as well as endangered reptiles such as the South American caiman and the Indo-Pacific crocodile." (Hinrichsen, 1996, p. 41)

"It is generally agreed that coral reefs are among the most intensely diversified, scientifically interesting biological systems on earth, in addition to having obvious aesthetic appeal and value as habitats for fish and invertebrate species important to human beings. Coral reefs support tremendous populations of diverse species of plants and animals through biomass production." (Maragos et al., 1983, p. 241)

"Coral reefs are among the most biologically diverse ecosystems on the planet. Although tropical rainforests contain more species than coral reefs, reefs contain more phyla than rainforests. Phyla are large groupings of organisms that are thought to be related. Covering less than 0.2% of the ocean floor, coral reefs contain perhaps 1/4 of all marine species. New studies indicate that biodiversity on coral reefs may be even higher." (EDF, 1997, p. 1)

[12] See Hinrichsen (1996), p. 42, and Maragos et al. (1983), p. 244, for discussions of the potential of coral reefs as a source of new pharmaceuticals.

over hypothetical methods, it risks obtaining inaccurate estimates because of gaps or overlaps in the values elicited or the method employed, as discussed in Chapters 3 and 7.

RANKING ENVIRONMENTAL PROBLEMS IN LATIN AMERICA AND THE CARIBBEAN

The continuing urbanization of LAC countries described briefly in Chapter 1 is largely a result of the economic opportunities that are available in the urban areas.[13] Accompanying these economic opportunities are concentration of industries and in most large cities, severe problems with industrial pollution. This growing concentration of people and industrial plants has already created serious environmental and public health impacts and extreme difficulties in planning and controlling the urbanization process. In fact, most large LAC cities suffer from deficient potable water supplies, inadequate wastewater disposal, limited solid waste collection and disposal services, air pollution problems, housing shortages, lack of other basic municipal services, diminished recreational opportunities, and overutilization of nearby natural resources. As a result of these problems, several measures have been used to determine their severity. They include relative importance, statistics on health effects, and public opinion.

Relative Importance of Problems

One modest goal for the development of the literature that values environmental benefits is to be able to make generalizations about the relative importance of different environmental problems in a country or region. The process of accumulation and analysis necessary to achieve this goal is difficult under the best of circumstances, but with the added complications of rapidly changing social and environmental conditions, considerable scientific uncertainty, and little onsite empirical research, it is virtually impossible. Keeping this caveat firmly in mind, it may still be useful to attempt such generalizations, concentrating on air and water pollution in LAC countries. Unfortunately, there have been few efforts to achieve a comprehensive ranking of problems, and those that have been done are vague and seem to be based largely on conjecture. An example is the United Nations Environment Programme's *Global Environment Outlook,* which ranks the environmental issues of land degradation, forest loss and degradation, atmospheric pollution, and urban and industrial contamination and waste all as critically important issues in LAC countries, while biodiversity loss and habitat fragmentation, freshwater access and pollution, and the degradation of marine and coastal areas are listed as important (as opposed to lower priority or of negligible importance). The publication provides no guidance as to how these priorities were determined.

Health Effects Statistics

A more fruitful approach is to compare the effects of air and water pollution on human health using existing health statistics. Table 6-4 summarizes the results from a study (Murray and Lopez, 1996) that estimated the burden of disease and injury attributable to 10 risk factors, including air pollution and poor water supply, sanitation, and personal and domestic hygiene. In LAC countries, poor water supply, sanitation, and personal and domestic hygiene accounted

[13] "In developed and developing countries alike, coastal zones harbor many of the most rapidly growing towns and cities. Nowhere is this more noticeable than in the Third World, where the rush to cities is accelerating. Coastal cities such as Caracas, São Paulo, Bangkok, Manila, Jakarta, and Lagos have become the 'economic hothouses' responsible for energizing economies and generating the bulk of new jobs." (Hinrichsen, 1996, p. 40)

Table 6-4. Burden of Disease and Injury Attributable to 10 Risk Factors in LAC

Risk Factor	Deaths (000s)	% of Total Deaths	YLLs (000s)	% of Total YLLs	DALYs (000s)	% of Total DALYs
Water, sanitation, and hygiene	135.3	4.5	4,254	7.6	5,183	5.3
Air pollution	33.6	1.1	377	0.7	476	0.5
Malnutrition	135	4.5	4,540	8.1	5,059	5.1
Hypertension	242.5	8.1	1,674	3.0	1,808	1.8
Occupation	97.7	3.2	1,973	3.5	3,681	3.7
Alcohol use	136.1	4.5	3,319	5.9	9,520	9.7
Tobacco use	99.4	3.3	952	1.7	1,340	1.4
Illicit drug use	16	0.5	449	0.8	1,589	1.6
Physical inactivity	117.6	3.9	796	1.4	969	1.0
Unsafe sex	73.9	2.5	2,003	3.6	3,645	3.7

Source: Based on Murray and Lopez (1996, p. 6).

for 4.5 percent of all deaths, 7.6 percent of total years of life lost (YLL), and 5.3 percent of the disability-adjusted life years lost (DALYs);[14] while exposure to air pollution accounted for 1.1 percent of all deaths, 0.7 percent of total years of life lost, and 0.5 percent of the DALYs lost. When compared with all of the risk factors considered, the water pollution, sanitation, and hygiene risk factor was tied for second in terms of the highest percentage of deaths, second in terms of percentage of years of life lost, and second in terms of DALYs. On the other hand, air pollution ranked ninth, tenth, and tenth, respectively. Although these statistics do not provide a complete measure of the damage caused by air or water pollution, they paint a stark picture of the importance of water pollution and associated problems in LAC countries, both absolutely and in relation to air pollution.

Health Costs

A third approach is to compare existing estimates of the health costs attributed to air and water pollution in LAC countries. Although this ignores all the other costs associated with air and water pollution, there are simply not enough data to perform any other meaningful comparisons. One problem with this comparison is that where the figures are considered to be the benefits of improvements, the assumed changes in the ambient environmental conditions are either not stated or may not be comparable because of different beginning or ending ambient conditions.

Keeping these qualifications in mind, it is possible to make comparisons for Mexico; Santiago, Chile; and Rio de Janeiro, Brazil.[15] For ease of comparison, all of the dollar amounts reported in this section have been converted to 1996 U.S. dollars. Perhaps the most complete comparison can be achieved with the data on Mexico, even though this is the product of admittedly rough calculations in Margulis (1992). This study estimates that the annual health costs associated with ambient air concentrations of particulates, ozone, and lead in all of Mexico totaled $1.35 billion. The study allocates $1.06 billion of these costs to particulates (mortality, $607 million and morbidity, $454 million), $126 million to ozone, and $164 million to lead. This same study also asserts that the total annual costs of premature deaths in Mexico in 1992

[14] A DALY is one lost year of healthy life, due either to premature death or disability of specified severity and duration.
[15] See also World Bank (1995), which examines the effects of air and water pollution in Argentina, but fails to generate comparable benefit estimates, although it does emphasize the severity of the water pollution public health problem.

due to water pollution (a total of 28,000) were $4.5 billion. The economic costs of morbidity from water pollution were estimated to be $38 million, for a total of $4.538 billion, or over three times the health costs associated with air pollution. A second attempt to value national health benefits from water pollution control for all of Mexico is provided by the Comisión Nacional del Agua (1996). This study evaluates the economic value of a set of proposed discharge standards establishing the maximum allowable amounts permitted for wastewater discharges into receiving bodies, which are to be implemented over time. The estimate of $2.7 billion in health benefits is based on an assumed 50 percent decrease in morbidity and mortality from gastrointestinal diseases and an unspecified decrease in cancer caused by toxic contamination over 20 years, discounted to the present at a rate of 6 percent. (This implies an annual benefit of about $230 million.) Further information from this study is reported in Annex 6-F. Notice that the implied annual damage avoided here is much less than the annual damage estimated by Margulis.

The data for Santiago are less useful because the estimate of water-related health costs is limited to the effects of using contaminated water to irrigate locally consumed produce. In any event, on the air side, the World Bank (1993) estimates that for the residents of Santiago, Chile, the annual health benefits (avoidance of lost production plus medical treatment costs) of reducing ambient concentrations of four air pollutants from their current levels to existing ambient air quality standards would be about $109 million. The proposed improvement in ambient air quality would reduce ambient levels of particles less than 10 μm in diameter by 8.43 $\mu g/m^3$, ozone by 0.0365 ppm, NO_2 by 0.013 ppm, and SO_2 by 0.16 $\mu g/m^3$. The benefits of reduction per unit of pollutant emissions are $19,775/ton of particles less than 10 μm, $1,511/ton of NO_x, and $538/ton of volatile organic compounds (VOCs). The study also estimates that from 1985 to 1990, the average annual health costs (medical costs and lost earnings) associated with typhoid caused by consumption of fruits and vegetables irrigated with water contaminated by domestic sewage totaled approximately $1.5 million. (The direct medical costs of treating an average of 1,626 cases per year were estimated at $359 per case, with productivity losses from morbidity of $249 per case, for a total of $608 per case or approximately $989,000 per year. The discounted weighted average of lost income for the average 13.5 premature deaths per year was estimated to be approximately $40,500 per premature death, for an additional annual loss of about $547,000.) The study also claims that the health costs associated with a potential cholera outbreak, which occurred in similar circumstances in Peru in 1991, could run as high as $16 million.

The World Bank (1996) has also ranked the seven most serious environmental problems in the state of Rio de Janeiro, Brazil, as follows: (1) water pollution in Guanabara Bay basin, Rio de Janeiro; (2) air pollution in the Basin III airshed in the Rio de Janeiro metropolitan region (RJMA); (3) air and water quality in and around Volta Redonda on the Paraiba do Sul; (4) water pollution in Sepetiba Bay; (5) sewage pollution of lakes and beaches in the lake region; (6) localized environmental problems, such as street-level air pollution from vehicles, localized sewage pollution, and disposal of hazardous wastes; and (7) unsanitary conditions in the household. However, the only environmental cost estimates reported by the study are for the health costs of particulate air pollution in the Basin III airshed in the RJMA. The study asserts that in 1994, particulates resulted in 1,300 additional annual premature deaths (at a cost of $70,800 per death) and 6,500 additional annual cases of chronic bronchitis (at a cost of $58,400 per case) for a total annual cost of $473 million. Although this study did not report health costs associated with water pollution, there are two other studies that do, one for all of the urban populations of Brazil and one for the RJMA only. Serôa da Motta et al. (n.d.) estimated that in 1989, the total health costs associated with water pollution in the urban areas of Brazil were $679 million, of which direct medical costs accounted for $70 million and lost earnings for $609 million. These estimates were based on an estimated 30,306 deaths due to water pollution and an estimated 3.6 million days spent in

Table 6-5. Annual Per Capita Health Costs of Air and Water Pollution (1996 U. S. dollars)

Authors and Date	Area	Air Pollution	Water Pollution
Comisión Nacional del Agua (1996)	Mexico	NA[a]	$28
Margulis (1992)	Mexico	$15	$50
World Bank (1996)	Rio de Janeiro, Brazil	$72	NA
Maimon (n.d.)	Rio de Janeiro, Brazil	NA	$2
World Bank (1993)	Santiago, Chile	$24	$0.35[b]
Serôa da Motta et al. (n.d.)	Urban Brazil	NA	$6

[a]NA, not available in that source.
[b]Only includes health costs of typhoid attributable to irrigation with contaminated water.

hospital. The second study, by Maimon (n.d.), estimates that the total annual health costs of water pollution in the RJMA in 1989 were approximately $17.3 million, of which $3.9 million were direct medical costs; $0.7 million were lost earnings due to morbidity; and $12.8 million were lost earnings due to premature death.

Looking at these results in per capita terms, as reported in Table 6-5, it may be difficult to accept the priorities for Rio de Janeiro. The rationale for the rankings seems to be that the World Bank considers costs other than health costs to be more important in terms of water pollution than air pollution,[16] and accepts that Guanabara Bay has special cultural and social significance within Rio.[17] In fact, the World Bank has asserted elsewhere that "based on our research findings, it appears that water-related illness and death in many cases is the single largest health impact of urban pollution, and one that can often be addressed at reasonable cost." (Dixon, 1993, p. 19)

Public Opinion

Another attempt to explicitly rank environmental problems is provided by a population survey in Costa Rica, sponsored under technical cooperation funding provided by the IDB and administered in May through August 1995. This study (PROGAM S.A. Agosto 1996, *Encuestas de Valuación Contingente: Informe Final*) attempted to discover the relative priorities that citizens of the metropolitan area of San Jose give to broad categories of environmental problems. Over 900 respondents were first asked to identify and rank by degree of severity the eight most critical environmental problems they believed to be affecting the quality of life and the condition of natural resources in the Central Valley. Applying rank-order weights and normalizing, air pollution and solid waste disposal were calculated to be about four times more important than water pollution, which was in third place with a weighted composite priority score of 0.24, compared with 0.89 for solid waste and 0.94 for air pollution. However, the distance between water pollution and the other higher priorities may be an artifact of the way the analysts constructed the final scale from the raw data, which is not clearly explained. Another question asked respondents to order, from most to least important, a set of eight preselected environ-

[16] "The preliminary analysis in Rio confirms the broader lesson from elsewhere that air pollution related health effects are so severe that these effects alone justify very significant control measures. On the other hand, health effects of water pollution are often not as severe unless drinking water supply is affected. Therefore, production losses and environmental amenity values play a more important role in the justification of water quality improvement measures." (World Bank, 1996, p. 11)

[17] "For many Guanabara Bay is a symbol of Rio. It is the reason that the city was founded and grew. Historically, the city's population has lived off and enjoyed their leisure in the Bay. To the outside world the Bay, as seen from the Sugar Loaf, is one of the most potent symbols of Rio." (World Bank, 1996, p. 25)

mental problems (rather than the self-selected problem areas used in the prior exercise). The study, again without fully explaining its procedures, reports priority scores that are much more closely spaced, with water pollution again being in third place with a composite score of 0.62, compared with 0.65 for air pollution and 0.68 for solid waste collection and disposal.

In a companion contingent valuation effort, 881 usable responses were obtained from a referendum question asking whether the household would be willing to pay a preselected amount (ranging from 200 to 4,000 colones per household per month)[18] to achieve simultaneous and rather vaguely explained environmental improvements on a number of fronts, including "protect forests, consolidate and expand national parks, clean rivers, eliminate noise, achieve no industrial pollution, reduce mobile source air emissions, and adequately manage solid waste" (question 40). Evaluation of a logistic function fit to the yes/no data produced an expected value of household WTP over a truncated range (from 0 to 4,000 colones) of 2,924 colones ($17) per family per month. Using these limits of integration does not seem to be a reasonable way to get an expected value (see Ardila, 1993), so the median value of about 4,000 colones ($23.26) per family per month is probably more representative (i.e., the estimated logistic function predicts that the probability of being willing to pay the maximum of the bid range, 4,000 colones, is 50 percent).

Respondents were asked to distribute their total WTP (TWTP) across each of four programs in order to produce a monetary assessment of the relative value of environmental improvements across several media. Consistent with the second set of index scores discussed earlier, each of the four areas, on average, received an almost equal share of total willingness to pay, but in contrast to the index ranking exercise, in these monetary benefit terms, air pollution falls from the top of the list to fourth place, despite the fact that it is combined with noise pollution, which suggests some inconsistencies in the survey or in the perceptions of the respondents. The percentage shares reported in the document and the allocation of the median total WTP are summarized in Table 6-6.

A somewhat similar survey of people in 24 countries, including Brazil (1,414 respondents), Chile (1,000 respondents), Mexico (1,502 respondents), and Uruguay (800 respondents), may shed some further light on perceptions of environmental quality. In this survey, Dunlap et al. (1993) posed a number of questions relating to the environment, including an open-ended question about the most serious environmental problem facing the country (LAC results are summarized in Table 6-7), and asked respondents to rate the seriousness of six different environmental problems in their local community (LAC results are summarized in Table 6-8). In the open-ended exercise, air pollution is first twice and third twice, while water quality is second three times and third once; water pollution is ranked higher in two countries, lower in the other two. When the problems are preidentified, "inadequate sewerage" is always the dominant problem (the highest percentage call it very serious). There is some apparent inconsistency here, especially in view of the vagueness of the "water quality" label in the open-ended exercise and the ratings of the two water categories in Table 6-8.

DIFFERENCES IN POLLUTION COSTS BETWEEN DEVELOPING AND DEVELOPED COUNTRIES

A second goal for the empirical literature review is to compare estimates of benefits across countries or regions.

[18] These figures correspond to $1.16 to $23.26 using the 1995 annual average exchange rate of 179.73 colones per dollar (the second quarter end-of-period rate is almost the same, 179.6 colones per dollar) and inflating to 1996 dollars using the Consumer Price Index.

Table 6-6. Distribution of WTP in San Jose CV Study (1996 U.S. dollars)

Program Area	% of TWTP	Median TWTP/Family/Month
Collect and dispose of solid waste	28.65	$ 6.66
Clean rivers and protect groundwater	24.88	$ 5.79
Protect forests	24.19	$ 5.63
Eliminate air pollution and noise	22.27	$ 5.18

Table 6-7. Three Most Popular Responses to Open-Ended Questions about Most Important Environmental Problem Facing Country

LAC Country	Most Popular	Second Most Popular	Third Most Popular
Brazil	Natural resource loss (53%)	Water quality (9%)	Air pollution (6%)
Chile	Air pollution (33%)	Water quality (22%)	Pollution (general) (21%)
Mexico	Air pollution (41%)	Pollution (general) (12%)	Water quality (7%)
Uruguay	Waste disposal (22%)	Water quality (21%)	Air pollution (11%)

Source: Dunlap et al. (1993, p. 18, Table 4).

Table 6-8. Percentage of Respondents that Consider Certain Environmental Problems in Local Community to be "Very Serious"

Country	Poor Water	Poor Air	Contaminated Soil	Inadequate Sewerage	Too Many People	Too Much Noise
Brazil	43	30	24	49	21	24
Chile	13	18	17	33	9	17
Mexico	25	21	24	39	23	23
Uruguay	7	9	9	25	7	10

Source: Dunlap et al. (1993, p. 20, Table 5).

Health Damage Per Case

There are several reasons to expect the relative importance of environmental benefits from wastewater treatment in developing countries to differ from the benefits in developed countries. For one thing, the direct health effects of polluted water should be more important in developing countries relative to other effects of water pollution. This expectation is based on differences in municipal water supply infrastructure, nutritional status, population demographics,[19] availability and expense of averting measures, access to and quality of health care systems, and housing characteristics. It seems this should be true even though the *per case* health costs should be higher in developed countries because a significant share of the health costs are due to lost productivity attributable to morbidity and/or mortality; and the value of lost productivity should be much greater in developed countries. (There are confounding influences, such as better overall health status and better access to care. More carefully, then, the cost per case with the same outcome should be higher in the developed countries.)

[19] Higher proportions of children in developing countries equate to increased environment-related mortality and morbidity and larger costs because of larger numbers of expected years of life lost to early mortality. Partially offsetting these effects is the fact that life expectancies are greater in developed countries than in developing countries.

Table 6-9. Estimates of Average Direct Medical Costs and Lost Earnings per Confirmed Case of Morbidity Attributable to Water Pollution (1996 U.S. dollars)

Author and Date	Area	Disease	Direct Medical	Lost Earnings	Totals
World Bank (1993)	Santiago, Chile	Typhoid	$359	$249	~$600
Sadoff (1996)	Sebou basin, Morocco	Diarrhea	$5	$1	~$6
Harrington et al. (1989)	Wilkes-Barre, PA, U.S.	Giardiasis	$389	$320–550	~$700–900[a]

[a] The total U.S. costs include diminished productivity during the time spent at work by the average "case," as well as the value of lost leisure.

Some meager evidence in this matter is presented in Table 6-9, where per case costs for three diseases in three different parts of the world are summarized. (However, see Table 6-13 and the accompanying text on the question of differences in average damage across the entire population.)

Relative Costs of Air and Water Pollution

Another way in which benefits will likely differ between developing and developed countries is that in the former, the health costs of water pollution should be relatively more important than the health costs of air pollution. With both air and water pollution, the poor suffer more than the wealthy because of a number of factors, including differences in preexisting health status, access to medical care, household sanitation, and likelihood of exposure. However, a wider range of averting measures are available to avoid the damage caused by water pollution than there are for air pollution, both at the individual (buy a water filter or bottled water) and community levels (treat contaminated water at municipal intake), and also in terms of medical treatment (a large number of deaths from water pollution occur as a result of dehydration, which is easily prevented and/or treated). For this reason, the poor (both within and across countries) tend to bear much more of the health burden of water pollution simply because they cannot afford many of these averting measures.[20]

The Murray and Lopez (1996) estimates of the burden of disease and injury which were the basis for Table 6-4 are also differentiated on the basis of developing and developed countries. These estimates, presented on this basis in Table 6-10, clearly indicate that water quality-related problems are much more important sources of death and disability in developing countries than in developed countries. On the other hand, the percentages indicate that air pollution is a slightly more important health risk in developed than developing countries. Also, it should be noted that although the young and old are more likely to suffer mortality or morbidity as a result of these environmental risks, the tendency is for water pollution to have more of an effect on the very young (less than 1 year old). This distinction also serves to intensify the effects of water pollution in developing countries because the very young typically constitute a larger proportion of the population than they do in developed countries.

Another way of examining relative importance is to look at monetized damage estimates. To this end, the studies used in Table 6-5 plus one done in Morocco by Sadoff (1996) can be compared with the results of a couple of studies that provide estimates of the heath costs of air and water pollution in the United States. The first such study, by Freeman (1982), attempted to estimate the benefits from air pollution control in the United States by estimating the benefits that resulted from a 20 percent reduction from 1970 to 1978 in ambient levels of total suspended

[20] "In Jakarta, households spend more than $50 million per year to boil water for drinking—an amount equal to 1 percent of the city's gross domestic product." (WRI, 1996, p. 24)

Table 6-10. Health Effects of Air and Water Pollution in Developing and Developed Countries

Risk Factor	Deaths (000s)	% of Total Deaths	YLLs[a] (000s)	% of Total YLLs	DALYs[b] (000s)	% of Total DALYs
Water, Sanitation, and Hygiene						
Developing	2,664.7	6.7	85,436	10.4	93,163	7.6
Developed	3.5	0.0	83	0.1	229	0.1
Air Pollution						
Developing	292.9	0.7	3,995	0.5	4,828	0.4
Developed	275	2.5	1,630	1.9	2,426	1.5

Source: Based on Murray and Lopez (1996, pp. 313–315).
[a]Years of life lost.
[b]A DALY (disability-adjusted life year), is one lost year of healthy life, due either to premature death or disability of specified severity and duration.

particulates (TSP) (the annual mean went from 106 $\mu g/m^3$ to 86 $\mu g/m^3$) and SO_2 (the annual mean went from more than 23 $\mu g/m^3$ to 19.4 $\mu g/m^3$), and a 30 percent reduction in CO (although Freeman did not find any empirical basis for attributing significant benefits to reduced carbon monoxide levels). There was no change in the level of O_3 and insufficient basis to calculate a change in NO_2. He concluded that the annual benefits for 1978 were between $13.5 billion and $140.1 billion, with health benefits alone being from $7.65 billion to $99.3 billion. More specifically, he estimated that the reductions in sulfur oxide and particulates resulted in a reduced annual mortality of between 2,780 and 27,800, with 13,900 being the most reasonable point estimate, and that the decrease in morbidity from the reduction in sulfur oxide and particulates avoided 4.6 million days of lost work and 29 million days of restricted activity, and reduced medical costs by $8.06 billion. This study's air pollution results were updated by the Public Interest Economics Foundation (PIEF, 1984) which, among other things, included lead and relied on more and different studies. This study found that in 1981 the total annual health benefits of such reductions were between $34.7 billion and $95 billion.

Freeman also estimated the total annual benefits in 1985 of achieving best practicable technology (BPT) and best available technology/best control technology (BAT/BCT) water pollution control standards in the United States as being somewhere between $8.7 billion and $42.4 billion, with the most likely point to be $21.4 billion (corrected to 1996 dollars). The health part of this total was only $2.3 billion at the most likely point (the estimate range being 0 to $4.6 billion). The results of all of these studies are summarized in Table 6-11 in per capita

Table 6-11. Annual Per Capita Health Costs of Air and Water Pollution (1996 U.S. dollars)

Authors and Date	Area	Air Pollution	Water Pollution
Sadoff (1996)	Sebou Basin	NA[a]	$1.62
World Bank (1996)	Rio de Janeiro	$72	NA
Maimon (n.d.)	Rio de Janeiro	NA	$2
World Bank (1993)	Santiago	$24	$0.32[b]
Margulis (1992)	Mexico	$15	$50
Serôa da Motta et al. (n.d.)	Urban Brazil	NA	$6
PIEF (1984)	U.S.	$147–$402	NA
Freeman (1982)	U.S.	$259	$11

[a] NA, not available.
[b] Only includes typhoid-related health costs attributable to irrigation with contaminated water.

terms. The higher per capita costs in the United States are evident for air, with the estimated water pollution control benefits falling within the range of those found in developing countries.

These results merit a few additional comments. First, the differences among the per capita health costs associated with air pollution in LAC studies is at least partly due to the fact that the World Bank (1996) and to a lesser extent World Bank (1993) studies are narrowly confined to the areas with the worst ambient air quality, but the highest per capita incomes in the countries involved. Fortunately, Freeman (and presumably the Public Interest Economics Foundation) also confined their air pollution analysis to urban areas, asserting that air pollution was, above all, an urban problem. Undoubtedly, if the geographic scope of these studies has been expanded to include less polluted rural areas, as in Margulis (1992), then their per capita estimates would have been significantly lower. Similar effects are also probable sources of differences in water pollution estimates. The residents of the Sebou River basin and the urban areas of Brazil and the United States all most likely have greater access to adequate supplies of clean water than much of the population of Mexico.

RECREATION VALUES IN DEVELOPED AND DEVELOPING COUNTRIES

There seems every reason to expect some nonuse values to be more important in developed countries than in developing countries because they are likely to be luxury goods. Among these are recreation values.

> *One could imagine a hierarchy of values which emerge in response to, and as a result of, economic growth. Initially people are willing to pay for environmental improvements that yield increases in real money income. These are the environmental effects which influence productive capacity. Then they become willing to pay for improvements in quality which enhance the utility of nonmarket activities. Finally, as their expanded budget constraints allow them, they are willing to pay for the presence of environmental quality, regardless of its direct impact on them.* (McConnell and Ducci, 1989, p. 3)

Hearne (1996) takes this argument a step farther, asserting that CV surveys are not appropriate for use in developing countries because questions about nonuse values are not relevant to developing countries. He criticizes efforts to apply the CV and travel cost methodologies (TCM) to developing countries because he believes that in developing countries, environmental functions and components are much more important as inputs into production processes than as environmental amenities generating recreation or nonuse values. A first objection to Hearne's assertion is that this last difference should at least be tempered by considerations of the relative importance of tourism, especially in parts of the LAC area.[21] This tourism factor is magnified by the fact that a substantial and growing portion of this tourism seems to qualify as so-called "ecotourism," and as a result would be highly sensitive to resource degradation.[22]

[21] "Among the Caribbean islands, for example, in 1988 income from tourism as a percentage of GDP was highest for Antigua and Barbuda (69 percent) and for the Bahamas (53 percent), but a dozen Caribbean islands reported that tourism revenues made up more than 10 percent of the GDP." (Turner et al., 1995a, pp. 9–10)

[22] "Even if this [tourism] pie is a little smaller than advertised, ecotourism seems to be the fastest-growing part of it. By the broadest measure (trips with some sort of nature or wilderness component), ecotourism already accounts for perhaps a third of those travelers. . . . Ecotourism is especially prominent in tourism's fastest-growing markets: southern Africa (which has attracted 18% more visits since 1990) and Latin America (which is up by 6%). It even dominates some markets. Kenya estimates that eight out of ten visitors come for the wildlife, as do most of Costa Rica's; these countries, along with Australia, are widely regarded as world leaders in ecotourism." (*Economist*, 1997, p. 48)

Table 6-12. WTP for Water Quality Improvement at Local Beaches

Authors and Date	Area	Mean Annual WTP (per household)	% of Income
McConnell and Ducci (1989)	Uruguay	$14	0.4
Darling et al. (1993)	Barbados	$11	NA[a]
Choe et al. (1996)	Davao, Philippines	$12–24	< 1
Bockstael et al. (1989b)	Baltimore/Washington, U.S.	$9–183	< 1
Hayes et al. (1992)	Rhode Island, U.S.	$80–187	< 1
Georgiou et al. (1998)	East Anglia, U.K.	$21	< 1

[a] NA, not applicable.

A second objection to Hearne's assertion is that it is not supported by the scant empirical literature that exists. For example, the results from six CV surveys regarding improvements in water quality at local beaches, summarized in Table 6-12, seem to indicate that respondents in both developing and developed countries are likely, on average, to be willing to contribute less than 1 percent of their income to such causes. Two points about this table need to be made, however. First, the dollar amounts are not stated in terms of a common base year because few of the studies disclose the year in which the data were gathered. Second, comparability problems related to the degree of change in water quality are even more relevant in CV studies such as these because, while it is difficult to measure the extent of change in water quality (and the size of the area in which the change will occur) envisioned by the surveyors, it is impossible to measure the extent of change envisioned by the respondents. As a result, it is not clear if these studies are comparable in any meaningful way. However, what can be said with some degree of certainty is that there is no indication that the use of CV for measuring environmental improvement in recreational resources is any less useful or appropriate in developing than developed countries. On the contrary, as discussed in Chapter 7, there are reasons for thinking it to be more useful.

BENEFIT ESTIMATES FROM IDB PROJECT ANALYSES[23]

A review of documentation for 27 IDB operations approved since 1989 having water quality improvement as an explicit goal uncovered useful benefit estimates in 18 cases, almost all of which involved CV estimates of WTP by national residents.[24] With only a few exceptions, a referendum question format was employed by Bank analysts, consistent with the NOAA Blue Ribbon Panel's recommendations on CV protocols. The projects being financed usually included potable water supply, household sewer connections, and drainage via collector and interceptor sewers, followed by wastewater treatment or some subset of these components.[25] Given the design characteristics of the projects, two categories of benefit predominate: the neighborhood effects of sewering and the more general ambient water quality improvement in the river or coastal water to which the sewage is discharged. (See Chapter 4, especially Annex 4-A.)

[23] These documents were reviewed in a broader context in Chapter 2.

[24] In two or three cases, producer's surplus estimates were made for tourism affected by improvement in marine coastal waters, but they were not firmly grounded in survey data, and are not discussed. Similarly, estimates of medical costs avoided were undertaken a few times, but they, too, do not appear reliable. One hedonic analysis was done and is included.

[25] Potable water supply benefits are obtained by integrating under statistically estimated demand functions and are not of concern here.

Local Sewers

It was once common IDB practice to measure the benefits of household sewer connections solely in terms of direct use value by statistically determining the shift in the demand curve for potable water allowed by having a connection, and counting as the sewer connection benefit the difference between the integrals of the demand functions with and without the sewer (the difference in total WTP for potable water in the two situations). However, that approach leaves out many other types of benefits which potentially can be captured by CV, as pointed out a decade ago by former Bank economist Jorge Ducci in describing the IDB's first application of CV for the economic analysis of an urban sanitation project in Uruguay [Project Report: Montevideo City Sanitation Project, Second Stage (UR-0023), October 1989; also see McConnell and Ducci, 1989].

Construction of a combined sanitary sewer system (household connections with storm drainage) produces a cost savings for connected residents, who no longer have to maintain and eventually replace more expensive individual wastewater disposal solutions like cesspools and septic tanks. Moreover, it provides a greater level of what Ducci calls "desirability" attached to the absence of clogged piping and foul smells in the vicinity of the home, the avoidance of flood damage to personal property, and the alleviation of transportation delays in rainy periods. Other benefits Ducci does not mention include the reduction of health risks through the elimination of pools of stagnant standing water in the neighborhood, and even some localized improvements in the quality of watercourses (creeks, ravines) that formerly received polluted domestic, commercial, and industrial discharges and storm runoff flows that sanitary sewer projects collect and channel elsewhere, usually to a consolidated downstream outfall. CV estimates of the WTP for sewer connections (sanitation and drainage) undertaken since Ducci's innovative effort appear in Table 6-13. On average, households in this small sample of projects are willing to pay 3 percent of their income each month (a recurrent, not a one-time, charge) to have a sewer connection and drainage services. The large average WTP of almost U.S.$20 per month (U.S.$240 per year) is influenced by the underlying distribution of income levels across the project sample. The log-log regression of WTP against income produces a positive and highly significant income elasticity of WTP of 0.54.[26] While one should not make too much of the magnitude of the elasticity, the fact that it is positive and significant across a sample of independently produced expectations of WTP in different projects provides at least a consistency check on the pattern in the data generated via CV. The existence of no relationship between WTP and income would be cause for serious concern about the plausibility of the method. The income levels in the sample are generally low, however, since IDB projects are often focused on low-income beneficiaries in the client countries, so the WTP elasticity cannot be taken to be a global estimate across all income levels in all countries.

This also suggests that simple WTP averages are not likely to be particularly useful for benefit transfers. Moreover, if the value of the income elasticity is truly less than 1, transfer exercises that arbitrarily assume a value of 1 for convenience may seriously overstate the magnitude of transferred benefits in higher income settings. At least one IDB project analysis has explicitly recognized the dangers in the average benefits or unitary elasticity transfer as-

[26] The regression model that included an intercept produced an adjusted R^2 of 0.58 and the following parameter estimates (absolute value of t statistic in parentheses):

$$\ln WTP = -0.33 + 0.54 \ln INC$$
$$(0.34) \quad (3.49)$$

The intercept term is statistically insignificant, which is reassuring since households with no income cannot make a positive payment.

Table 6-13. IDB CV Estimates of WTP for Local Sewer and Drainage (1996 U.S. dollars)[a]

Country-Project No.	Mean WTP (per household)	Household Monthly Income	WTP as a Percentage of Income
AR-0130	$21.01	$721	2.9
AR-0130	$47.27	$1,471	3.2
BA-0036[b]	$16.82	NA[c]	NA
BR-0067	$28.96	$1,094	2.6
BR-0073	$15.95	$399	4.0
BR-0186 (excludes drainage)	$12.70	$343	3.7
BR-0186 (sewer plus drainage)	$16.36	$343	4.8
BR-0190	$16.75	$558	3.0
CO-0082	$2.32	NA	NA
CO-0227	$15.60	$233	6.7
EC-0025	$12.15	NA	NA
UR-0023	$26.69	$348	7.7
UR-0089	$21.50	NA	NA
Average:	$19.54	$612	3.2
Standard Deviation:	$10.67	$416	NA
Median:	$16.75	$399	4.2

[a]Converted using the exchange rate of the period and the U.S. Bureau of Labor Statistics Implicit Price Deflator.
[b]Includes sewer benefits plus benefits of ensuring water at beaches is safe for contact recreation.
[c]NA, not available.

sumptions and apparently tried to correct for it [Loan Proposal: Metropolitan Montevideo Sanitation Program Stage III (UR-0089), July 1996].

Ambient Water Quality in Receiving Waters

Contingent valuation estimates of WTP for ambient water quality improvement are even more of a mixed bag than sewer benefits, mainly because the no-project AEQ baseline and the extent of AEQ improvement to be provided by the project is often unclear in the official IDB documents reviewed, including technical support documents in some cases (see Chapter 2).[27] Table 6-14 provides 15 estimates culled from 11 different project documents. Since very few of the documents clearly report the average income of the beneficiary population, the table does not risk stating an income level,[28] and no income elasticity estimate for the value of ambient water quality improvements is possible with these data. However, it is possible to hazard a guess about what level of improvement is being valued. Based on the current AEQ status reported, it is safe to assume a very poor initial level of water quality, with only a few exceptions (e.g., BA-0036). Ignoring differentiated levels of AEQ achievement and taking a crude sample average in Table 6-14 suggests that, in contrast to Table 6-10, households assign to the more amorphous and distant (in both time and space) AEQ improvements in major watercourses only about 30 percent of the value ($5.78) that they assign to the more concrete and immediate utility gains from having sewers. It does not seem at all implausible to find this relationship among respon-

[27] This information is often available from the economic analysis feasibility studies, which were not reviewed because of time and budget constraints. The objective of this study is not the meta-analysis of benefit estimates.
[28] It is dangerous to presume that the expected value for general AEQ improvement reported in Bank documents is independent of income, or that the same average income applies that was used for sewerage (if both elements are present). The scope of the beneficiary population for AEQ benefit evaluation is often the population of the entire city, not just a neighborhood. Usually no details are provided about the logit specification of the probability of acceptance function estimated from the referendum bid data or the way the function was evaluated (truncated mean over bids yielding a positive estimated probability of acceptance versus untruncated expected value). (See the discussions in Chapter 2 and Chapter 7, Annex 7-A.)

Table 6-14. IDB Project Estimates of WTP for General Ambient Water Quality Improvements in Rivers, Lakes, and Coastal Waters (1996 U.S. dollars)

Country-Project No.	Method	AEQ Target	Mean WTP/ Household/Month
BA-0036	CV	Beaches-swimmable water	$1.04
BR-0072	CV	River pollution control-no specifics	$7.85
BR-0072	CV	Beaches-swimmable water	$7.74
BR-0073	CV	Beaches-swimmable water	$5.40
BR-0073	CV	Beaches-swimmable water	$7.28
CO-0082	CV	Odor elimination	$3.24
CO-0208	CV (Direct)	Odor elimination	$3.28
CO-0208	CV (Direct)	Odor elimination and aesthetics	$7.14
CO-0208	CV (Direct)	Swimmable water	$11.34
CO-0227	CV	Swimmable water	$3.72
EC-0161	Hedonic	Property "affected" by pollution	$4.20
ME-0056	CV	Clean river-no specifics	$6.30
NI-0027	CV	Odor elimination and aesthetics	$4.00
PR-0064	CV	Improvement in water quality	$13.38
UR-0023	CV	Beaches-swimmable water	$0.74
		Average:	**$5.78**
		Standard Deviation:	**$3.50**
		Median:	**$5.40**

Note: WTP is expressed in constant 1996 U.S. dollars by application of the U.S. Bureau of Labor Statistics Implicit Price Deflator. CV is a referendum question format unless otherwise noted.

dents of limited means, who, of necessity, give greater weight to providing a higher relative portion of use to nonuse values.[29]

CONCLUSION

This chapter has delved into the empirical literature on environmental benefit estimation to see what insight it might provide and what perspective it might give for addressing the issue of wastewater treatment in LAC countries. This effort has accomplished a couple of things. First, the chapter has presented a taxonomy to help organize discussions of these benefits and the empirical literature that aims to estimate their magnitude. Second, it has assessed the state of the empirical literature in terms of what it does and does not contain. The overriding impression created by this brief examination of wastewater treatment in LAC countries and the more detailed examination of the empirical literature of water quality benefits in developing countries is that there is much left to be done. For one thing, it would be useful to be able to make and empirically support a few broad generalizations about the importance of water quality benefits relative to other environmental benefits and about the differences in benefits in developed and developing countries. However, the empirical literature does not now contain a sufficient number of studies (for LAC and developing countries in general) to confidently make these or any other generalizations. For example, an attempt to compare the seriousness of air and water

[29] It would probably be difficult to try to read anything more into the limited information in Tables 6-13 and 6-14. In particular, no test is done of the statistical significance of the roughly $14/household/month difference between average WTP for sewerage and improved AWQ. Nor is any effort made to translate the description of the AWQ target in likely categories of improvement "size" in order to see if there is any relation between that size and the stated WTP.

pollution in these terms could only conclude that their relative importance seems to vary from site to site because the sample of studies is too small to make statements about averages that are even modestly believable.

However, the glass can also be seen as half full. The field of benefit studies in developing countries is growing very rapidly, and the latest techniques are being successfully applied there. From the IDB's point of view, it appears that there is substantial methodological and practical support for the necessary task of estimating the benefits of water quality improvements, although no actual effort will ever be easy, and it is always possible that further research will identify new problems. In a preliminary examination of these conclusions, the next chapter focuses on the specific valuation methodologies and their promise for supplementing these existing studies.

Annex 6-A

Municipal Use Values

Authors and Date	Title	Country	Resource
Abdalla et al. (1992)	Valuing Environmental Quality Changes Using Averting Expenditures: An Application to Groundwater Contamination	U.S.	Drinking water
Altaf et al. (1993)	Rethinking Rural Water Supply Policy in the Punjab, Pakistan	Pakistan	Drinking water
Bohm et al. (1993)	Sustainability of Potable Water Services in the Philippines	Philippines	Drinking water
Briscoe et al. (1990)	Toward Equitable and Sustainable Rural Water Supplies: A Contingent Valuation Study in Brazil	Brazil	Drinking water
Edwards (1988)	Option Prices for Groundwater Protection	U.S.	Drinking water
Gilman and Skillicorn (1985)	Boiling of Drinking Water: Can a Fuel-Scarce Community Afford It?	Bangladesh	Drinking water
Grandstaff (1986)	Tongonan Geothermal Power Plant Project in Leyte, Philippines	Philippines	Rivers
Harrington et al. (1989)	The Economic Losses of a Waterborne Disease Outbreak	U.S.	Drinking water
Kwak and Russell (1994)	Contingent Valuation in Korean Environmental Planning: A Pilot Application to the Protection of Drinking Water Quality in Seoul	Rep. of Korea	Drinking water
Lovei and Whittington (1993)	Rent-Extracting Behavior by Multiple Agents in the Provision of Municipal Water Supply: A Study in Jakarta, Indonesia	Indonesia	Drinking water
MacRae and Whittington (1988)	Assessing Preferences in Cost-Benefit Analysis: Reflection on Rural Water Supply Evaluation in Haiti	Haiti	Drinking water
Maimon (n.d.)	Health Impacts of Chemical Pollution and Domestic Wastewater in Rio de Janeiro Metropolitan Region	Brazil	Rivers
McConnell and Ducci (1989)	Valuing Environmental Quality in Developing Countries: Two Case Studies	Caribbean Country	Sewage
North and Griffin (1993)	Water Source as a Housing Characteristic: Hedonic Property and Willingness-to-Pay for Water	Philippines	Drinking water
Sadoff (1996)	The Price of Dirty Water: Pollution Costs in the Sebou Basin	Morocco	Drinking water
Serôa da Motta et al. (n.d.)	Environmental Damages and Services Due to Household Water Use	Brazil	Drinking water
Whittington et al. (1990b)	Estimating the Willingness to Pay for Water Services in Developing Countries: A Case Study of the Use of Contingent Valuation Surveys in Southern Haiti	Haiti	Drinking water
Whittington et al. (1993)	Household Demand for Improved Sanitation Services in Kumasi Ghana: A Contingent Valuation Study	Ghana	Sanitation services
Whittington et al. (1991)	A Study of Water Vending and Willingness to Pay for Water in Onitsha, Nigeria	Nigeria	Drinking water
Whittington et al. (1990a)	Calculating the Value of Time Spent Collecting Water: Some Estimates for Ukunda, Kenya	Kenya	Drinking water

Annex 6-B

Recreational Use Values

Authors and Date	Title	Country	Resource
Agnello (1989)	The Economic Value of Fishing Success: An Application of Socioeconomic Survey Data	U.S.	Fishery
Agnello and Han (1992)	Substitution Effects within a Multiple Site Travel Cost Model with Application to Recreational Fishing	U.S.	Fishery
Bell (1992)	Actual and Potential Tourism Reaction to Adverse Changes in Recreational Coastal Beaches and Fisheries in Florida	U.S.	Beaches, fisheries
Bell (1989)	Application of Wetland Valuation Theory to Florida Fisheries	U.S.	Wetlands
Bell and Leeworthy (1990)	Recreational Demand by Tourists for Saltwater Beach Days	U.S.	Beach
Bell and Leeworthy (1986)	An Economic Analysis of the Importance of Saltwater Recreational Beaches in Florida	U.S.	Beach
Bell et al. (1982)	The Economic Impact and Valuation of Salt Water Recreational Fisheries in Florida	U.S.	Fishery
Bergland and Brown (1988)	Multiple Site Travel-Cost Models and Consumer Surplus: Valuation of Oregon Sport-Caught Salmon	U.S.	Fishery
Bergstrom et al. (1990)	Economic Value of Wetlands-Based Recreation	U.S.	Wetlands
Bockstael et al. (1987b)	Estimating the Value of Water Quality Improvements in a Recreational Demand Framework	U.S.	Beach
Bockstael et al. (1989a)	A Random Utility Model for Sportfishing: Some Preliminary Results for Florida	U.S.	Fishery
Bockstael et al. (1989b)	Measuring the Benefits of Improvements in Water Quality: The Chesapeake Bay	U.S.	Ocean bay
Cameron (1992)	Combining Contingent Valuation and Travel Cost Data for the Valuation of Non-Marketed Goods	U.S.	Fishery
Cameron (1988)	A New Paradigm for Valuing Non-Market Goods Using Referendum Data: Maximum Likelihood Estimation by Censored Logistic Regression	Canada	Fishery
Cameron and James (1987)	Efficient Estimation Methods for "Close-Ended" Contingent Valuation Surveys	Canada	Fishery
Caulkins et al. (1986)	The Travel Cost Model for Lake Recreation: A Comparison of Two Methods for Incorporating Site Quality and Substitution Effects	U.S.	Lakes
Daubert and Young (1981)	Recreational Demands for Maintaining Instream Flows: A Contingent Valuation Approach	U.S.	River
Dixon et al. (1995)	Ecology and Microeconomics as "Joint Products": The Bonaire Marine Park in the Caribbean	Netherlands Antilles	Marine park
Duffield et al. (1992)	Recreation Benefits of Instream Flow: Application to Montana's Big Hole and Bitterroot Rivers	U.S.	Rivers
Feenberg and Mills (1980)	Measuring the Benefits of Water Pollution Control	U.S.	Beach
Garrod and Willis (1996)	Estimating the Benefits of Environmental Enhancement: A Case Study of the River Darent	U.K.	River
Gramlich (1977)	The Demand for Clean Water: The Case of the Charles River	U.S.	River
Green and Tunstall (1991)	The Evaluation of River Water Quality Improvements by the Contingent Valuation Method	U.K.	River
Hayes et al. (1992)	Estimating the Benefits of Water Quality Improvements in the Upper Narragansett Bay	U.S.	Ocean
Hundloe (1989)	Measuring the Value of the Great Barrier Reef	Australia	Coral reefs

Annex 6-B. *continued*

Authors and Date	**Title**	**Country**	**Resource**
Huppert (1989)	Measuring the Value of Fish to Anglers: Application to Central California Anadromous Species	U.S.	Fishery
Judge et al. (1995)	Valuing Beach Renourishment: Is it Preservation?	U.S.	Beach
Kahn (1991)	The Economic Value of Long Island Saltwater Recreational Fishing	U.S.	Fishery
Kahn and Kemp (1985)	Economic Losses Associated with the Degradation of an Ecosystem: The Case of Submerged Aquatic Vegetation in Chesapeake Bay	U.S.	Fishery
Kaoru (1995)	Measuring Marine Recreation Benefits of Water Quality Improvements by the Nested Random Utility Model	U.S.	Fishery
Kaoru et al. (1995)	Using Random Utility Models to Estimate the Recreational Value of Estuarine Resources	U.S.	Fishery
Leeworthy (1991)	Recreational Use Value for John Pennekamp Coral Reef State Park and Key Largo National Marine Sanctuary	U.S.	Marine reserve
Leeworthy and Wiley (1994)	Recreational Use Value for Clearwater Beach and Honeymoon Island State Park, Florida	U.S.	Beach, terrestrial reserve
Leeworthy and Wiley (1993)	Recreational Use Value for Three Southern California Beaches	U.S.	Beaches
Leeworthy and Wiley (1991)	Recreational Use Value for Island Beach State Park	U.S.	Marine reserve
Loomis (1988)	The Bioeconomic Effects of Timber Harvesting on Recreational and Commercial Salmon and Steel Head Fishing: A Case Study of the Siuslaw National Forest	U.S.	Fishery
McConnell (1979)	Values of Marine Recreational Fishing: Measurement and the Impact of Management	U.S.	Fishery
Mendelsohn et al. (1994)	Ecotourism and Conservation: A Study of Marine Ecotourism in Belize	Belize	Coral reef
Milon (1991)	Measuring the Economic Value of Anglers' Kept and Released Catches	U.S.	Fishery
Milon (1988)	Travel Cost Methods for Estimating the Recreational Use Benefits of Artificial Marine Habitat	U.S.	Fishery
Milon et al. (1994)	Recreational Anglers' Valuation of Near-Shore Marine Fisheries in Florida	U.S.	Fishery
Morey et al. (1993)	A Repeated Nested-Logit Model of Atlantic Salmon Fishing	U.S.	Fishery
Morey et al. (1991)	A Discrete-Choice Model of Recreational Participation, Site Choice, and Activity Valuation When Complete Trip Data Are Not Available	U.S.	Fishery
Navrud and Mungatana (1994)	Environmental Valuation in Developing Countries: The Recreational Value of Wildlife Viewing	Kenya	Wildlife flamingoes
Norton et al. (1983)	$tripers: The Economic Value of the Atlantic Coast Commercial and Recreational Striped Bass Fisheries	U.S.	Fishery
Parsons and Kealy (1992)	Randomly Drawn Opportunity Sets in a Random Utility Model of Lake Recreation	U.S.	Lakes
Patrick et al. (1991)	Estimating Regional Benefits of Reducing Targeted Pollutants: An Application to Agricultural Effects on Water Quality and the Value of Recreation Fishing	U.S.	Fishery
Pendleton (1994)	Environmental Quality and Recreation Demand in a Caribbean Coral Reef	Honduras	Ocean
Peters et al. (1995)	Influence of Choice Set Considerations in Modeling the Benefits from Improved Water Quality	U.S.	Fishery
Ralston and Park (1989)	Estimation of Potential Reductions in Recreational Benefits Due to Sedimentation	U.S.	Lake
Ribaudo and Young (1989)	Estimating the Water Quality Benefits from Soil Erosion Control	U.S.	Rivers
Rowe et al. (1985)	Valuing Marine Recreation Fishing on the Pacific Coast	U.S.	Fishery
Sanders et al. (1991)	Comparable Estimates of the Recreational Value of Rivers	U.S.	Rivers

Annex 6-B. *continued*

Authors and Date	**Title**	**Country**	**Resource**
Seller et al. (1985)	Valuation of Empirical Measures of Welfare Change: A Comparison of Nonmarket Techniques	U.S.	Lake, river
Silberman and Klock (1988)	The Recreation Benefits of Beach Renourishment	U.S.	Beach
Smith et al. (1993)	Marine Pollution and Sport Fishing Quality: Using Poisson Models as Household Production Functions	U.S.	Fishery
Smith et al. (1991)	Combining Farrell Frontier and Hedonic Travel Cost Models for Valuing Estuarine Quality	U.S.	Fishery
Smith et al. (1996)	Marine Debris, Beach Quality and Non-Market Values	U.S.	Beach
Tay and McCarthy (1994)	Benefits of Improved Water Quality: A Discrete Choice Analysis of Freshwater Recreational Demands	U.S.	River
Vaughan et al. (1985)	The Estimation of Recreation-Related Water Pollution Control Benefits: Swimming, Boating, and Marine Recreational Fishing	U.S.	Rivers, lakes, ocean
Vaughan and Russell (1982)	Freshwater Recreational Fishing: The National Benefits of Water Pollution Control	U.S.	Rivers and lakes
Wegge et al. (1988)	Site Quality and the Demand for Sportfishing for Different Species in Alaska	U.S.	Fishery
Wegge et al. (1986)	An Economic Assessment of Marine Recreational Fishing in Southern California	U.S.	Fishery
Willis and Garrod (1991)	Valuing Open Access Recreation on Inland Waterways: On-Site Recreation Surveys and Selection Effects	U.K.	Rivers and lakes

Annex 6-C

Commercial Use Values

Authors and Date	Title	Country	Resource
Arnsdorfer and Bockstael (n.d.)	Estimating the Effects of King Mackerel Bag Limits on Charter Boat Captains and Anglers	U.S.	Fishery
Barbier et al. (1991)	Economic Valuation of Wetland Benefits: The Hejia-Jama're Floodplain, Nigeria	Nigeria	Wetlands
Bennett and Reynolds (1993)	The Value of a Mangrove Area in Sarawak	Malaysia	Mangroves
Christensen (1982)	Management and Utilization of Mangroves in Asia and the Pacific	Thailand	Mangroves
Evenson (1991)	Genetic Resources: Assessing Economic Value	India	Biodiversity
FAO (1985)	Mangrove Management in Thailand, Malaysia and Indonesia	Thailand, Malaysia	Mangroves
Grandstaff (1986)	Tongonan Geothermal Power Plant in Leyte, Philippines	Philippines	Fishery
Hodgson and Dixon (1988)	Logging Versus Fisheries and Tourism in Palawan	Philippines	Coral reef
Hufschmidt (1986a)	The Nam Pong Water Resources Project in Thailand	Thailand	Reservoir
Hufschmidt (1986)	Systematic Analysis of Water Pollution Control Options in a Suburban Region of Beijing, China	China	River
Hufschmidt and Dixon (1986)	Valuation of Losses of Marine Product Resources Caused by Coastal Development of Tokyo Bay	Japan	Fishery
Kim and Dixon (1986)	Economic Valuation of Environmental Quality Aspects of the Upland Agricultural Projects in Korea	Rep. of Korea	Reservoir
Knowler et al. (1997)	The Effects of Pollution on Open Access Fisheries: A Case Study of the Black Sea	Turkey	Fishery
Lal (1990)	Conservation or Conversion of Mangroves in Fiji	Fiji	Mangroves
Lynne et al. (1981)	Economic Valuation of Marsh Areas for Marine Production Processes	U.S.	Wetlands
Macmillan and Ferrier (1994)	A Bioeconomic Model for Estimating the Benefits of Acid Rain Abatement to Salmon Fishing: A Case Study in South West Scotland	Scotland	Fishery
McAllister (1988)	Environmental, Economic and Social Costs of Coral Reef Destruction in the Philippines	Philippines	Coral reef
Norton et al. (1983)	$tripers: The Economic Value of the Atlantic Coast Commercial and Recreational Striped Bass Fisheries	U.S.	Fishery
Ruitenbeek (1994)	Modeling Economy-Ecology Linkages in Mangroves: Economic Evidence for Promoting Conservation in Bintuni	Indonesia	Wetlands
Thurman and Easley (1992)	Valuing Changes in Commercial Fisheries Harvests: A General Equilibrium Derived Demand Analysis	U.S.	Fishery

Annex 6-D

Indirect Use Values

Authors and Date	Title	Country	Resource
Breaux et al. (1995)	Using Natural Coastal Wetland Systems for Wastewater Treatment: An Economic Benefit Analysis	U.S.	Wetlands
Costanza et al. (1989)	Valuation and Management of Wetland Ecosystems	U.S.	Wetlands
Farber (1996)	Welfare Loss of Wetlands Disintegration: A Louisiana Study	U.S.	Wetlands
Farber (1987)	The Value of Coastal Wetlands for Protection of Property Against Hurricane Wind Damage	U.S.	Wetlands
Farber and Costanza (1987)	The Economic Value of Wetlands Systems	U.S.	Wetlands
Folke (1991)	The Societal Value of Wetland Life-Support	Sweden	Wetlands
Thomas et al. (1991)	Use Values and Non-Use Values in the Conservation of Ichkeul National Park, Tunisia	Tunisia	Wetlands
Turner et al. (1995b)	Wetland Valuation: Three Case Studies	Sweden, U.K.	Wetlands

Annex 6-E

Total Economic Value

Authors and Date	Title	Country	Resource
Choe et al. (1996)	The Economic Benefits of Surface Water Quality Improvements in Developing Countries: A Case Study of Davao, Philippines	Phillippines	Rivers, lakes, and beach
Darling et al. (1993)	The Question of a Public Sewerage System in a Caribbean Country: A Case Study	Caribbean Island	Coral reef, beach
Greenley et al. (1981)	Option Value: Empirical Evidence from a Case Study of Recreation and Water Quality	U.S.	River
Loomis et al. (1991)	Willingness to Pay to Protect Wetlands and Reduce Wildlife Contamination from Agricultural Drainage	U.S.	Wetlands
Loomis et al. (1990)	Toward Empirical Estimation of the Total Value of Protecting Rivers	U.S.	River
Mazzotta et al. (1997)	Valuing Estuarine Resources: A Contingent Choice Study of the Peconic Estuary System	U.S.	Estuary
McConnell and Ducci (1989)	Valuing Environmental Quality in Developing Countries: Two Case Studies	Caribbean and Latin America	Coral reefs, beaches
Oster (1977)	Survey Results on the Benefits of Water Pollution Abatement in the Merrimack River Basin	U.S.	River
Sanders et al. (1990)	Toward Empirical Estimation of the Total Value of Protecting Rivers	U.S.	Rivers
Silberman et al. (1992)	Estimating Existence Value for Users and Nonusers of New Jersey Beaches	U.S.	Beach
Wattage (n.d.)	Contingent Valuation: Estimation of Benefits of Water Quality Improvement	U.S.	Rivers

Annex 6-F

Benefits and Costs of Water Pollution Control in Mexico

The text in Chapter 6 concentrates on the health benefits estimated by the Comisión Nacional del Agua. The Mexican study also estimates other benefits, including (a) an increase in revenue in the agricultural sector due to the substitution of 70 percent of the crops in the basins where the source of irrigation is untreated wastewater away from cereal grains and toward higher-value fruits and vegetables ($249 million), (b) a decrease in the operating and maintenance costs for potable water plants when the source is surface water ($139 million), (c) cost savings resulting from not having to substitute away from water sources that become too polluted to more costly alternative sources, (d) an increase in the value of some 41,293 ha of land bordering bodies of water receiving treated wastewater ($316 million), (e) an increase in exemptions from use rights payments for irrigated water withdrawals (which are granted if treated wastewater is used) because of the increase in treated wastewater capable of being used for irrigation ($483 million). It should be noted that the benefit estimates are largely based on apparently arbitrary, unjustified assumptions and include some effects that should be characterized as transfers instead of benefits (on a national level). Ignoring these problems, Table 6F-1 summarizes these benefits by category.

Table 6F-1. NPV of Benefits from Water Pollution Control by Category and Discount Rate (millions of 1996 U.S. dollars)

Benefit Category	NPV @ 0% Discount Rate		NPV @ 6% Discount Rate		NPV @ 12% Discount Rate	
Health						
Gastrointestinal disease: mortality	3,322		1,800		1,073	
Gastrointestinal disease: morbidity	1,512		882		569	
Toxics: mortality	40	4,874	22	2,704	14	1,656
Crop substitution		4,271		2,491		1,606
Water treatment						
O&M at potable water plants	238		139		89	
Maintaining current water sources	1,395	1,633	813	952	525	614
Increase in land value		358		316		283
Exemptions from water use rights payment		828		483		311
Totals		**11,964**		**6,918**		**4,432**

The Mexican study also estimates public costs (investment costs incurred by municipalities to treat wastewater discharged into public receiving bodies) and private costs (those derived from the proposed regulation that are incurred by the private sector/companies), but it does not estimate costs corresponding to the promulgation of the standards and subsequent efforts at monitoring and enforcement. The total costs and benefits are compared in Table 6F-2.[30]

Table 6F-2. NPV of Total Costs and Benefits of Standards (millions of 1996 U.S. dollars)

	NPV @ 0% Discount Rate	NPV @ 6% Discount Rate	NPV @ 12% Discount Rate
Total Benefits	11,964	6,157	3,533
Costs	8,003	5,122	3,673
Net Benefits	**3,961**	**1,035**	**(140)**

[30] Converted from Mexican pesos using an exchange rate of 7.5 pesos/U.S. dollar. The total benefits displayed in these two tables (for NPV at 6 percent and at 12 percent) do not agree because the corresponding numbers in the underlying documentation do not agree.

Chapter 7

Benefit Estimation Techniques

The goal of this chapter is to review the state of the art of nonmarket benefit estimation relevant to LAC countries. The focus of this review is on the major problems that plague the estimation of benefits from ambient water quality improvement (as distinct from potable water provision) by the traditional approaches to benefit estimation, and on the alternative approaches that might solve, or at least avoid, these problems. After this brief introduction, a section is devoted to typology and definitions. Then the traditional, revealed-preference methods are examined, with a focus on their applicability, special problems associated with them, and the extent to which these methodologies have been applied in developing countries. Readers interested in a more detailed analysis of the theory and practice of nonmarket benefit estimation are directed to more exhaustive reviews. The fourth section turns to the stated-preference techniques, assessing the state of the art, and again focusing on applications in developing countries. The estimation of health benefits is discussed separately in the next section. Two special topics are then explored: the combination of revealed- and stated-preference methodologies, and the possibility of so-called "benefit transfer." These are considered in terms of their general potential as valuation techniques and their specific promise for water quality benefit estimation in LAC countries. A brief conclusion suggests the major lesson of the review.

OVERVIEW OF VALUATION TECHNIQUES

The traditional methodologies are often categorized in a taxonomy similar to the one provided by Freeman (1993), which differentiates the methodologies on the basis of whether they are based on *observed behavior* or on responses to *hypothetical questions*, and on the basis of whether monetary values are generated *directly* or must be inferred, "through some indirect technique based on a model of individual behavior and choice." (Freeman, 1993, p. 23)[1] The methodologies relying on observations of behavior are often referred to as revealed-preference methodologies, while those relying on answers to hypothetical questions are often referred to as stated-preference methodologies. The interaction of these two characteristics generates four categories in which the different methodologies may be grouped, as shown in Table 7-1.

The first category, *direct observed*, can only be used to value market benefits because these methods require observations of behavior that can be directly translated into willingness-to-pay measures. Although the direct observed methodologies (particularly the approaches utilizing dose-response functions) are often used to value environmental benefits, they can only provide a partial measure of these benefits because they are incapable of measuring the nonmarket aspects of improvements in environmental quality. As a result, our analysis of these methodologies is limited to a brief examination of their deficiencies where they constitute the state of the art.

[1] This four-way schematic is an elaboration of the older "direct" vs. "indirect" division, where the former involved direct questioning of a sample of people and the latter meant that the method sought indirect evidence of willingness to pay via data from related markets.

Table 7-1. Benefit Estimation Methods

	Observed Behavior		Hypothetical	
Direct Path to Values	Market Simulated markets	(1)	Open-ended contingent valuation Bidding game contingent valuation	(3)
Indirect Path to Values	Hedonic wage analysis Hedonic property analysis Travel cost Averting behavior Voting in actual referenda	(2)	Referendum contingent valuation Conjoint analysis Contingent choice (between two vectors) Contingent ranking or rating (of three or more vectors)	(4)

Source: Adapted from Freeman (1993, p. 24).

The second category, *indirect observed*, infers values for a nonmarket good from observed behavior in some market by recognizing the link between this behavior and the nonmarket good to be valued. While there are many ways to categorize the methods falling under this label, this report concentrates on three examples that seem most relevant: hedonic markets, travel cost, and averting behavior.

The third category, *direct hypothetical*, is composed of surveys that elicit maximum WTP by asking direct questions about WTP. This box contains only techniques that allow more or less direct inference of WTP. These techniques may include open-ended questions ("What is the maximum amount that you would be willing to pay for this benefit?") or questions or bidding games composed of a series of questions ("Would you pay this amount for this benefit?"). The methods in this category are referred to as contingent valuation because the value elicited is contingent upon the details of the hypothetical market created to elicit the value, i.e., how the benefit is to be achieved, how the WTP is to be paid, and so on.

The last category, *indirect hypothetical*, consists of methods that infer WTP through a series of exercises in which respondents are asked to choose between, rank, or rate sets of hypothetical options, essentially the various forms of what is often called "conjoint analysis" (CA). Overall, these methodologies apply unevenly to the major routes to environmental benefits (or damage), as shown in Table 7-2.

It hardly need be said that there is an extensive literature on the theory of the economic valuation of nonmarket goods and services. Freeman (1993) provides what is still the most complete single source for such theory. However, there are numerous other useful sources, including two commissioned by the Inter-American Development Bank (McConnell, 1995, and

Table 7-2. Environmental Impacts and Valuation Methods

Impact	Valuation Methods
Productivity	Market Averting behavior
Health	Averting behavior Hedonic wage and property analysis Dose-response coupled with hedonic human capital or contingent valuation-based values "Pure" contingent valuation Conjoint analysis
Amenity/recreation	Hedonic market Travel cost Contingent valuation Conjoint analysis
Existence and bequest values	Contingent valuation Conjoint analysis

Source: Adapted from Winpenny (1995, p. 43, Table 3.2).

Markandya, 1991); two practitioner-oriented works from the OECD (Pearce et al., 1994) and the much less technical Winpenny (1995); one work that specifically addresses water quality benefits (Smith and Desvousges, 1986); an excellent compilation of chapters written by experts on the various types of benefit estimation (Braden and Kolstad, 1991); and, finally, an evaluation of these methodologies in the developing world context (Georgiou et al., 1997).

INDIRECT OBSERVED OR REVEALED-PREFERENCE METHODS

As indicated in Table 7-1, the indirect observed methods for benefit estimation include hedonic analyses, travel costs, and measures of averting behavior.

Hedonic Market

Hedonic market valuation is used to estimate direct and indirect use values, typically the effect of changes in environmental quality on housing values or of risk of morbidity or mortality on wages. The hedonic methodology is based on the premise that a good (or service) can be defined as a bundle of characteristics or attributes that together determine the price of the good via the demand for and supply of the characteristics in the good market. For example, the price of a residence can be viewed as a function of its size, age, physical condition, proximity to schools or places of work, the quality of the environment in which it is located, and so on. Similarly, the wage paid for a job can be expressed as a function of, among other things, the risk of injury associated with the job. These functions are called "hedonic price functions." Once estimated, these functions can be used to generate implicit marginal prices for each of the individual characteristics of the good.[2]

Under certain assumptions, these implicit marginal prices can be deemed to represent marginal WTP for the characteristic. Unfortunately, some of these assumptions are problematic. For one thing, the market is assumed to be in equilibrium. Taking the example of a housing market, this assumption requires full information, no moving or transactions costs, and instantaneous price adjustments to changes in supply and demand. Although no existing housing market can be expected to meet these stringent requirements, a reasonably well-functioning market will not, in general, be unduly burdened by its imperfections.[3] However, where the market is relatively inactive, controlled, or distorted, the application of hedonic analysis becomes much more problematic. Second, market actors must be aware of and appreciate the effects of the environmental change to be valued. If there is little information on environmental risks or the risks are not fully appreciated by individual actors in the market, hedonic analysis will not generate a valid estimate of the WTP to avoid these risks. A third problematic assumption is that a sufficiently wide variety of the good is required to approximate a continuous spectrum of choices among characteristics, allowing consumers to choose their optimum bundles without being constrained by the unavailability of some bundle. (That is, they are not forced into a corner solution.) Concerns over this assumption have been partially addressed with discrete choice or random utility models.[4]

[2] If the hedonic price function has been estimated for an entire market, then the marginal implicit price for any characteristic is simply the partial derivative of the hedonic price function with respect to that characteristic.

[3] It should be noted that although the errors introduced by these imperfections are generally random, nonrandom errors will be introduced by the continuous movement, or the expectation of movement, of market forces in any one direction, resulting in biased marginal implicit prices (see Freeman, 1993, especially p. 383).

[4] Random utility models are extensively used in travel cost analysis and are discussed in more detail in the section dealing with that methodology. For a discussion of hedonic random utility models, see Freeman (1993), pp. 412–415, and Palmquist (1991), pp. 116–119.

If these assumptions are accepted and if the change in the characteristic of interest is to be measured ex post, where the accumulated data allow calculation of before and after marginal implicit prices, then the difference in aggregate marginal WTP provides a measure of the value of the environmental change. However, the measure derived from the hedonic function will generally overstate the "true" value of the change.[5] On the other hand, if the analysis is to occur ex ante, or does not have access to before and after data, and the change is nonmarginal (as most environmental changes that result from the implementation of a policy or project would be), then a second, more problematic, stage of the analysis is required in order to derive a marginal WTP or inverse demand function. These functions are needed to map nonmarginal changes in the characteristic to changes in total WTP. The problem is identifying (in the technical, econometric sense) these functions from the observations of marginal implicit prices and quantities. Since the hedonic price function is typically nonlinear,[6] the marginal implicit price for any characteristic is not parametric to the individual, but is a function of the individual's choice of bundle. If all individuals had identical incomes and utility, then this would not be a problem because all individuals would have the same inverse demand curve, which would simply be the marginal implicit price schedule, i.e., the locus of equilibrium for marginal implicit prices and quantities. However, if individuals have different inverse demand functions, then each marginal implicit price observation represents, in general, a point on only one person's inverse demand function and conveys no information about anyone else's inverse demand function. As a result, distinguishing between the marginal implicit price schedule and the consumers' marginal bid functions requires additional data from other markets or the imposition of structure on the consumers' utility functions.

Although researchers are still trying to come to grips with exactly what is needed to validly perform this second stage of the analysis, the most reliable approach seems to be to use additional data from what is essentially a segmented or entirely separate market.[7] However, the practical problems and complications encountered in attempts to apply hedonic analysis in developing country contexts render even first-stage estimation tenuous at best. The satisfaction of additional data requirements for a second stage is generally well beyond possibility.[8] For instance, application to developing countries will only make the three "problematic" assumptions described earlier even less likely to be valid. Developing country markets are less likely to be free of substantial imperfections or rapid change; individuals are less likely to be fully informed about environmental conditions and their effects; and housing or job choices are less likely to be sufficiently broad to allow consumers to make choices unconstrained by the lack of alternatives. Most important, even where markets are well functioning and individuals well informed, data collection may be much more difficult in developing countries. A major difficulty in applying this technique to developing countries is the relative scarcity of monetary land and property transactions in many cultures. To the extent that they occur, they may not be recorded in a fashion required for economic analysis (Bojö et al., 1992).

[5] See Vaughan (1987) for a concise explanation of why this is so.

[6] A linear hedonic price function implies that consumers can "arbitrage" among goods by varying characteristics. "For example, if individuals are indifferent between owning two two-door cars and one four-door car, other things being equal, they can create equivalents of four-door cars by repackaging smaller units. If both sizes exist on the market, the larger size must sell at twice the price of the smaller one, and the hedonic price of a car will be a linear function of the number of doors." (Freeman, 1993, pp. 128–129; see also Rosen, 1974, pp. 37–38)

[7] See Palmquist (1991), pp. 95–102, and Freeman (1993), pp. 387–391, for overviews of the literature addressing this problem.

[8] Vaughan (1987) makes a similar assertion.

Table 7-3. Studies Applying the Hedonic Property Pricing Method

Area of Application	U.S.	Europe	Developing Countries
Agriculture	Few		
Air quality	Many	Some	
Health risks	Few		
Hunting	Few		
Noise	Many	Many	
Parks, nature reserves, and wildlife	Many	Some	
Water quality	Few		
Water supply and sanitation	Few	Few	Few

Source: Georgiou et al. (1997, p. 106, Table 4.3).
Note: "Few" is not defined by Georgiou et al., but it seems safe to take it to mean 1 to 3 or 4. "Some" is defined as up to 10 and "many" as more than 10.

Although these are certainly not all (or even most) of the practical problems associated with hedonic analysis,[9] they are the ones most relevant to developing country applications and should adequately convey the difficulties of applying the methodology in that context. Last but by no means least, hedonic models can provide only a partial measure of a change in environmental quality because they capture only one aspect of the effect of a change in environmental quality; i.e., hedonic property models only measure WTP for the attributes of quality that are "captured" by choice of residence, while ignoring "willingness to pay for improvements in environmental amenities at other points in the urban area—for example, in the work place, shopping areas, or parks and recreational areas." (Freeman, 1993, p. 416)

Given these practical problems, it is little wonder that few hedonic studies have actually been carried out in developing countries, as is indicated by Table 7-3. Similarly, there are few examples of hedonic wage studies, for many of the same reasons.[10] Nonetheless, the partial derivative of a first-stage hedonic price function can be used to produce an approximate benefit number for neighborhood or local effects of environmental change, such as those introduced by the construction of a sewer: "[S]imple versions of the technique may be useful (indeed have been) in establishing the effects on property values of improvements in neighborhood amenities such as water supply, rubbish collection, street lighting, etc. providing there is data on property values before and after the changes, and so give rough estimates of benefits." (Pearce et al., 1994, p. 147)

However, as pointed out in Vaughan (1987), a "correct" application of first-stage estimation can generate estimates that are two to three times as large as the "true" benefits. As a result, even the reduced effort needed to generate first-stage estimates seems unlikely to be merited, given the quality of the results.[11]

Travel Cost Methodology

A second indirect observed method of measurement of nonmarket benefits is the travel cost methodology. TCM values a recreational site or characteristic by using the time and cost incurred

[9] Other problems include determining the appropriate measure of environmental quality (see Freeman, 1993, pp. 377–379, and Palmquist, 1991, pp. 92–93) and choosing a functional form (see Freeman, 1993, pp. 379–381; Palmquist, 1991, pp. 87–89; and Vaughan, 1987 and 1988, pp. 1–4).

[10] "Labour markets in developing countries are likely to be highly imperfect, often having an excess supply of labour. People may even disregard risk in the search for a job and income if they are poor. Furthermore risk perceptions are unlikely to be high, and the returns to a job may depend on caste, class, etc. Data requirements make the approach prohibitively expensive and so few uses in developing countries are envisaged." (Pearce et al., 1994, p. 150)

[11] See Vaughan (1987) for a summary of some of the arguments for and against such efforts.

Table 7-4. Studies Applying the Travel Cost Method

Area of Application	U.S.	Europe	Developing Countries
Fishing: Recreational	Many	Many	
Parks, nature reserves, and wildlife	Many	Many	Some
Water quality	Many	Some	Few
Water supply and sanitation	Few	Few	

Source: Georgiou et al. (1997, p. 105, Table 4.2).
Note: "Few" is not defined by Georgiou et al., but it seems safe to take it to mean 1 to 3 or 4. "Some" is defined as up to 10 and "many" as more than 10.

in visiting the site as a proxy or implicit price for what a visitor would be willing to pay to visit the site. The most basic version of TCM is a continuous-demand model for a single site, in which individuals maximize utility by choosing the number of visits to the site subject to monetary and time constraints. This maximization generates the individual's demand function for the site, from which a consumer's surplus can be calculated and aggregated across individuals.

Although there have been some applications of TCM in developing countries, as shown in Table 7-4, the focus of the TCM studies that value recreational sites is generally on international visitors and therefore of limited use.[12] Unlike hedonic analysis, the essential problem in applying the methodology in developing countries is not so much an absence of external sources of data as it is an absence (or at least alteration) of the relationship between environmental quality and the recreational market:

> *Applications are numerous in developed countries where motor cars enable easy access to sites, and where time has significant opportunity costs. This will not often be the case for developing countries. Recreation areas will often be close to urban areas (due to limited transportation and low incomes) and so travel costs will be very small. Valuation of non-work time is also more crucial since there will be more people who are non-producers than in developed countries. Also, visitors will often use a recreation area to seek a break from work. Access to sites may be subject to constraints, and so observed travel costs may not accurately reflect actual willingness to pay.* (Pearce et al., 1994, p. 142)

As a result, TCM may be of limited help in estimating the benefits of water treatment projects as they accrue via water-based recreational sites because so few nationals incur the kinds of travel costs needed for TCM to generate valid WTP estimates, and because these values, if estimated, are likely to be small in relation to other project benefits. In addition, the great majority of travel cost studies value a recreational site *as it exists*. Valuing the introduction of a new site or proposed changes to an existing site, as would be required for analysis of a water quality improvement project, requires more sophisticated versions of TCM, and these are necessarily more data intensive and assumption sensitive.

All TCM models face a number of common problems requiring simplifying assumptions. Perhaps the most fundamental of these is the assumption that "individuals perceive and respond to changes in the travel-related component of the cost of a visit in the same way they would respond to a change in the admission price." (Freeman, 1993, p. 446) This assumption is particularly important because it requires the accurate monetization of the opportunity costs

[12] In this instance, a measurement of the producer's surplus generated by international tourism would be more appropriate. Another application of TCM in developing countries is to fuelwood and drinking water supply where consumers of these goods must spend substantial time and effort to collect and transport them.

of the time spent visiting and traveling to and from the site in order to generate an accurate demand schedule for it. However, choosing an appropriate method of valuing this time is by no means a straightforward process and is further complicated by severe restrictions on the types of data that can be gathered.[13] Because of these difficulties, most TCM studies use some fraction of the average wage rate—a solution that strikes many as somewhat arbitrary, especially given the importance of this value in affecting the outcome of the study.

In general, the problems that plague TCM are related to the theoretical and practical difficulties of modeling the recreation decision-making process. Sample selection issues illustrate the practical difficulties engendered by this process. Most travel cost data are collected either on site or through the identification of user groups on the basis of some publicly available criteria, such as hunting or fishing licenses or boat registration. Since not everyone uses a recreation site or sites, the sample is either truncated, if nonparticipants are not included in the sample, or censored, if they are.[14] Samples collected on the basis of criteria that suggest the respondent is a likely user of the resource face an additional problem because the availability of the site being studied most likely influenced the decisions that led to the respondents satisfying the criteria. Although statistical techniques can account for the systematic influences that affect the decision to visit the site or meet the criteria and therefore should be included in the survey, choosing to apply one of these techniques imposes a structure on individual decision making in the absence of theoretical guidance.

Other important problems or complications include incorporating the effects of substitute sites in the model, choosing a functional form, defining the appropriate choice set, allocating travel costs when trips are made for more than one purpose, accounting for differences in the amount of time spent on site, ensuring that choice of residence was not determined by recreational preferences, choosing and measuring the appropriate environmental variable, and dealing with the possibility that there is utility or extra-disutility derived from the time spent traveling to and from the site (in which event the travel time would have to enter into the utility function as well as into the time constraint and travel cost amount).

Randall asserts that as a result of these and other persistent problems, the idea that researchers can "define a typical trip and specify its cost is *prima facie* implausible." (Randall, 1994, p. 91) Granting that it is plausible to assume that travel costs increase with distance and hence are "ordinally measurable," he concludes that the best that TCM can ever hope to achieve are ordinally measurable recreation benefits unique only up to a monotonic transformation.

In addition to these problems, TCM is rather limited in applicability. For instance, it is useful for measuring direct use values[15] only where users expend significant time or money in getting to the resource and can be differentiated on this basis. In addition, measuring a change in some qualitative characteristic of the site (such as air or water quality), as opposed to simply valuing the existence of a site, requires the extension of the basic continuous-demand model, causing additional problems in approximating consumer's surplus.[16] As a result, attempts to value individual site characteristics led to the development of hedonic TCM, which attempts

[13] The difficulties of calculating a value of time for travel cost studies has spawned an extensive literature that includes Smith and Kaoru (1990), Bockstael et al. (1987a), Smith et al. (1983), McConnell and Strand (1981), Wilman (1980), and Bishop and Heberlein (1979).

[14] In the context of a recreation visit model, censoring and truncation may be understood as follows: If one samples randomly from a population, the number of visits for each respondent will either be 0 or an integer greater than 0. But the dependent variable of interest is "probability of visiting," and no observations of that, other than 0 or 1, are available, even though the model rests on the assumption that for each person in the sample such a probability exists. That sample is censored. If sampling is done at a site where by definition only visitors are encountered, and interest is in the number of visits per user, the sample will be truncated at one visit, no zero-visit observations being available.

[15] See Chapter 6 for the distinction between direct and indirect use values.

[16] See Freeman (1993), pp. 459–460, and Bockstael et al. (1991), pp. 249–253, for discussions of these problems.

to integrate into TCM the methods of determining implicit prices for attributes of hedonic analysis in order to estimate values for site characteristics. Since its development in Brown and Mendelsohn (1984), hedonic TCM has been heavily criticized (largely as a result of studies that found negative marginal prices for attributes that should have been "goods"). These criticisms focus on the absence of a market mechanism to adjust the price of site characteristics to their equilibrium price (so that marginal implicit prices will necessarily reflect marginal values) and on whether the idea of paying more to acquire a higher level of some qualitative characteristic—which clearly makes sense in a one-time purchase—makes sense where the choice is number of trips to a particular site. Although there have been some attempts to salvage the methodology, it generally receives little attention. Since it is not particularly useful for project analysis anyway because it only generates general marginal implicit prices not specific to any one site, it will receive no further mention.

On the other hand, a third type of TCM, the random utility model (RUM), has become the preferred modeling strategy precisely because it can generate values for the attributes of a particular site or sites. RUMs enjoy an additional advantage over continuous-demand models in that they are able not only to value the losses from eliminating an existing site but also to value the benefits of introducing a new site. The RUM label encompasses a range of econometric models that can be used to analyze preferences for different site characteristics by comparing the characteristics of the site chosen with the characteristics of possible substitutes (accounting for the characteristics of the respondents).[17] Because RUMs focus on the choice among substitute sites for any given trip as a function of the characteristics of the available sites, they are especially suitable when participants are choosing among a number of sites differentiated by quality, and they are generally employed to value changes in specific site characteristics, such as water quality, congestion, or fish catch rates. The distinction between RUM and continuous-demand models arises from the ways in which they regard time. Continuous-demand models examine demand for recreational sites over some period of time, such as a summer or a year, where people choose how many trips they are going to make to the different sites. RUMs examine demand for the sites each time a choice is made, so that people are effectively choosing which site to visit, if any (Freeman, 1993). Another implication of this difference is that continuous-demand models can directly explain the total number of trips an individual takes to a given site in a season, while RUMs cannot. As a result, a continuous-demand model is often appended to a RUM, with the latter explaining the allocation of trips over sites.

Although RUMs may be superior to continuous-demand models in their ability to deal with site characteristics and multiple sites, they are not immune to the difficulties inherent in modeling the complex recreation decision-making process. Three extensions of RUMs illustrate recent attempts to overcome these difficulties. The first such extension, nested models, restricts the nature of the relationships among alternative sites to avoid the assumption of the independence of irrelevant alternatives generally implied by RUMs. Nested models assume that individuals select their recreational experience by making a series of choices, such as the length of trip, region for the trip, and finally, a particular site within the region.[18] A second extension of RUM attempts to expand the analysis beyond the single-trip framework through the development of repeated and sequential choice models.[19] Finally, a number of researchers have attempted to address the failure of RUMs to generate quantity data by incorporating seasonality issues into the RUM framework.[20] In any event, RUMs are widely viewed as the state of the art in TCM

[17] See McConnell (1995), pp. 10–15; Bockstael et al. (1991), pp. 256–264; and Kolstad and Braden (1991), pp. 32–35, for detailed treatments of the theory and estimation of RUMs.

[18] Kling and Thompson (1996) and Kaoru (1995) are recent examples of nested RUMs.

[19] See Kling and Thompson (1996) for an example.

[20] Feather et al. (1995), Hausman et al. (1995), and Parsons and Kealy (1995) are a few of these attempts.

and they would be the model most applicable to evaluating water treatment projects because they can focus on a change in water quality at the site:

> *Despite the fact that estimates of benefits from all versions of the RUM framework appear quite sensitive to strategic modeling decisions, enthusiasm remains high for continuing to base policy analyses on some type of RUM. The framework is attractive to practitioners because it consistently incorporates diverse sources of heterogeneous site characteristics. These characteristics provide the primary means to describe the status of the environmental resources supporting recreation uses. For example, in the case of recreational fishing, historic catch rates, emission loadings, contamination notices, as well as other proxy variables have served as the indicators of the quality of specific lakes, rivers, or areas along the coast that support this recreation. These would be difficult to include within [continuous] demand models.* (Smith, 1997, pp. 15–16)

Averting Behavior

The third type of indirect observed benefit estimation technique to be discussed here, averting behavior, infers a value for an improvement in environmental quality from changes in spending on ways to reduce the impact of the lower quality. Examples of averting behaviors used for studies include boiling drinking water (Gilman and Skillicorn, 1985), staying indoors during times of heavy air pollution (Bresnahan et al., 1997), and the purchase of an air conditioner (Dickie and Gerking, 1991b) or bottled water (Abdalla et al., 1992). Averting-behavior techniques provide a relatively straightforward route to estimating indirect use values when the environmental threat or harm is known, and efficacious averting behaviors exist. Unfortunately, the results are often imprecise because the averting measures generally cannot exactly compensate for the loss. For example, the purchase of an air conditioner is unlikely to provide health benefits so much as improved ambient air quality. Even worse, the direction of the bias cannot be known with certainty because the air conditioner provides other benefits beyond air filtration. In addition, people may employ more than one behavior to address a specific effect, such as wearing sunscreen *and* staying indoors to avoid an increased risk of skin cancer.

Attempts to value environmental quality changes using averting behavior suffer from at least four other problems or complications. First, if conditions get too bad, some people are apt to simply relocate, thoroughly complicating the sampling and valuation processes. Second, the model depends upon people's subjective perceptions of the environmental deterioration and of the risk of harm to themselves, which do not necessarily directly correspond to changes in environmental quality. Third, the behavior often involves some form of discontinuous choice, such as a capital investment in the purchase of an air conditioner or water filter. Presented with this investment decision, people tend to initially resist the purchase, but when the decision to purchase is made, it is guided by long-term expectations as to future environmental conditions and therefore is difficult to interpret in relation to current conditions. Fourth, market imperfections, such as credit rationing, can constrain behavior. As a result of these problems, while averting-behavior values for changes in environmental quality may be relatively easy to obtain, they may only serve as upper or lower bounds for the true values.[21]

[21] Whether these values serve as an upper or lower bounds depends on whether the behavior fully compensates for the environmental degradation, whether the behavior generates ancillary benefits, and whether it is the only averting behavior employed. See Abdalla et al. (1992) for a brief summary of the theoretical literature establishing averting-behavior values as WTP bounds. Bresnahan and Dickie (1995) examine the use of averting-behavior values as bounds for WTP in policy analysis.

Given these problems, it is not surprising that the averting-behavior method has not been used with anywhere near the frequency of the first two methods in either developed or developing countries. However, the one area where the methodology may have the most to contribute is in the analysis of water quality improvement projects. There are a number of reasons for this assertion. First, the perceived risks from poor water quality and the behaviors that can avoid these risks are relatively easily known and understood. Second, these behaviors are often within the means of developing-country residents. Third, the method may offer an inexpensive way to get at indirect use benefits that are difficult to estimate with any other method. For example, the costs of efforts to prevent beach erosion can serve as an approximation of the loss of the storm or sea protection functions from the destruction of a coral reef.[22] Fourth, the results may be more easily communicated to, and related to, by decision makers than estimates generated by more esoteric or hypothetical methods. Finally, it may often be the cheapest of the available methods.

HYPOTHETICAL OR STATED-PREFERENCE METHODS

The hypothetical methods for benefit estimation include contingent valuation and conjoint analysis.

Contingent Valuation

Just a few years ago, the inclusion of a section on hypothetical methods in a review of benefit estimation in developing countries would have seemed pointless, even though the methods were being widely applied in the United States. However, the difficulties encountered in applying the data-intensive, revealed-preference methodologies and the successes achieved in using hypothetical methods—predominantly in calculating demand for water supply and sanitation—have so completely reversed this perception that hypothetical methodologies are now seen to enjoy a comparative advantage over revealed-preference methodologies in developing countries:

> *Ten years ago only a handful of very rudimentary [CV] studies had been conducted in developing countries; at the time the conventional wisdom was that it simply could not be done. The problems associated with posing hypothetical questions to low-income, perhaps illiterate respondents were assumed to be so overwhelming that one should not even try. Today we have come full circle; it is now assumed by many environmental and resource economists working in developing countries that CV surveys are straightforward and easy to do. . . . Bilateral donor agencies and the international development banks are increasingly putting [CV] techniques to use in project and policy appraisal as part of their everyday operations work. . . . Moreover, in light of the controversy over the use of [CV] in the United States, most future applications of [CV] are likely to be in developing countries.* (Whittington, 1998, p. 29)

This preference is also reflected in the analysis of sewer and water treatment projects funded by the IDB and reviewed in Chapter 2, with benefit estimates summarized in Chapter 6. All but one of these analyses relied at least in part on CV estimates.

The following list of requirements may be taken to define the essentials of a CV study. All CV studies (a) define the good or service of interest, (b) inform the respondents of a hypothet-

[22] See Cesar (1996), pp. 22–23. In this sense, the common practice of crediting wastewater treatment plants with reducing either the costs of downstream intake treatment of municipal drinking water or the costs associated with treating water for reuse in irrigation are aggregate applications of the averting-behavior method.

ical change in the availability or level of the good or service to be valued, (c) describe the institutional structure by which the change is to be accomplished, along with the method of payment of the stated WTP, (d) ask respondents to reveal their WTP for this change, and (e) ask socioeconomic and demographic questions in order to relate WTP to these characteristics as an internal consistency check.

The first two requirements, defining the good or service and informing respondents about the change that is to be valued, can be easier or harder, depending on such features of the problem as its familiarity to the respondents and its concreteness. Thus, a study of WTP to improve neighborhood conditions by installing sanitary sewers and getting rid of open sewerage ditches and drains could count on high levels of familiarity and concreteness among the affected residents. Asking those same people, as part of a larger area sample, about their WTP for changes in the quality of a large river flowing through a city might run into lack of familiarity. An even more unfamiliar setting would be evoked by WTP questions seeking to value a national policy that required every discharger to meet certain discharge standards for every part of every surface water body in a region or a nation to satisfy multidimensional ambient water quality standards.

One element of the survey problem this poses is how to make standards expressed in terms of technical measurements (such as the dissolved oxygen and phosphorus requirements discussed in Chapter 5) meaningful for respondents. Lay respondents are unlikely to have any idea of what different levels of DO or P mean. In terms of these measures, you cannot tell much by looking at a water body. And respondents are unlikely to have information on the quality parameters of bodies of water they have actually used recreationally.

In the 1980s, at Resources for the Future, as part of an EPA-funded study of WTP for the existing national policies aimed at improving water quality, this description and information problem was addressed by overlaying a "map" of scientific parameters of quality onto a set of activities, such as boating, fishing, and swimming, that could be carried on in contact with water of particular quality. The fruit of that effort, called the RFF Water Quality Ladder, is described in Annex 7-A. One reviewer of this document has suggested that it could be useful to attempt to define a ladder appropriate to LAC settings, presumably with more emphasis on health and less on recreation.

The different variants of CV can be distinguished on the basis of exactly how they elicit WTP. Some studies ask an open-ended question about maximum WTP. Some engage respondents in a bidding game to determine this maximum. Some ask respondents to choose the value closest to their maximum from a set of values on a payment card. And some simply ask respondents whether they would be willing to pay a certain amount to support a program or policy change. This last type of value elicitation method is commonly referred to as a "single-bounded dichotomous choice" if only one valuation question is asked or "double-bounded dichotomous choice" if two such questions are asked. Dichotomous choice, which is sometimes referred to as the "referendum format," has become the preferred CV approach because it is believed to be incentive compatible and immune to many of the biases attributed to the other methods.[23] However, the tradeoff for this perceived resistance to bias[24] is that respondents never actually state their maximum WTP. As a result, analysts must engage in a complex, somewhat controversial, exercise in econometrics to estimate some maximum WTP for each individual. (See Chapter 8 for a discussion of this problem area.) It may be that the designation of dichot-

[23] See NOAA (1993), the NOAA Blue Ribbon Panel Report, for an influential endorsement of the dichotomous choice format; see Hoehn and Randall (1987) for the conditions under which dichotomous choice CV is incentive-compatible; see Freeman (1993), pp. 165–198, and Hoevenagel (1994a), pp. 201–209, for discussions of the relative merits of the different value elicitation methods.

[24] It should be noted that it is not altogether clear that dichotomous choice CV is any more immune to bias than the other methodologies. For example, see Boyle et al. (1997) for a study of anchoring bias in dichotomous choice CV.

omous choice as the preferred approach is more of a testament to the depth of economists' general distrust of hypothetical methods and lingering suspicions of strategic responses than it is the result of a careful examination of the advantages and disadvantages of this format.[25]

As a result of this distrust and the use of CV to calculate damage in the Exxon *Valdez* oil spill, there has been an extensive and at times bitter debate over the validity of CV.[26] This debate has been fueled by a series of highly critical studies asserting that the results of CV methods lack internal consistency or are inconsistent with economic theory (Diamond and Hausman, 1994; Hausman, 1993; Kahneman and Knetsch, 1992a; 1992b). The response to this two-pronged attack has been either to challenge the assertions on the basis of their "understanding" of economic theory or CV, or to deflect the assertions by pointing out that although they may be valid criticisms of individual CV studies, they are not valid criticisms of the methodology in general (Carson, 1997; Mitchell and Carson, 1995; Carson and Mitchell, 1995; Hanemann, 1994; Smith, 1993; 1992b; and Carson and Flores, 1992). The results of attempts at "validating" (or "invalidating") CV results generally support this response. The validity "tests" can be roughly divided into three categories. Beginning with Bishop and Heberlein (1979), there have been numerous attempts at external validation by comparing the results of hypothetical studies with actual market transactions, the results of stated preference studies, or WTP experiments.[27] The results of these tests are mixed, and perhaps best described as open to interpretation (Smith, 1997). A second approach has been to test the reliability or consistency of CV responses (essentially whether results can be replicated from one period to the next). These tests have generally shown CV results to be consistent (Carson et al., 1995; Hoevenagel, 1994a). Finally, internal validation tests have been used to evaluate CV results against the postulates of economic theory. The results of these tests also generally remain open to interpretation (Carson, 1997; Smith, 1997; Hoevenagel, 1994a; Pearce et al., 1994).

The NOAA Blue Ribbon Panel Report

Another result of this debate is the emergence of efforts to describe what constitutes a state-of-the-art CV study, the most important of which are the recommendations made by the NOAA Blue Ribbon Panel (NOAA, 1993).[28] These recommendations are divided into general and specific guidelines, goals, and fatal problems. (There is also a list of recommendations for future research, which is not discussed here. The interested reader may find it in the report.)

The six general guidelines are the following: employ a professional sampling statistician to conduct probability sampling, minimize nonresponses, utilize face-to-face interviews instead of telephone and mail surveys, test for and assess the existence of interviewer effects, include a specified list of details of the study in the writeup and make the data publicly available, and carefully pretest the survey instrument and test whether respondents understood and accepted the instrument.

The eleven more specific guidelines are the following: where aspects of survey design and response analysis are ambiguous, opt for the choice that will generate a more conservative WTP measure; use WTP instead of willingness to accept; use the referendum format; provide an

[25] See Randall (1997), Carson (1997), and Carson et al. (1996) for reviews of the recommendations made by the NOAA Blue Ribbon Panel.

[26] For an overview of the debate, see Portney (1994).

[27] See Randall (1997), Smith (1997), pp. 31–35, and Hoevenagel (1994a) for discussions of some of these attempts, and Foster et al. (1997), Gegax and Stanley (1997), Kling (1997), and Bohm (1994) for more recent examples.

[28] See Carson et al. (1996) for an example of the only study to date to fully comply with the NOAA Panel recommendations. See Bateman and Willis (1998), Mitchell and Carson (1995), Carson and Mitchell (1995), Pearce et al. (1994), Freeman (1993), Mitchell and Carson (1989), and Cummings et al. (1986) for detailed explorations of CV.

accurate description of the program or policy to be valued[29]; pretest the effects of photographs, if any; remind respondents of available substitutes; average values across independently drawn samples taken at different times to reduce time-dependent measurement noise (this is likely to be more of a problem in the case of environmental insults, e.g., an oil spill); provide a "no-answer" option in addition to "yes" and "no" options; ask for explanations as to why the respondent chose the option he or she chose; include questions that help to interpret the responses, such as income and knowledge of problem; and ensure that the instrument is not so complex that completing it is beyond the respondent's ability or interest.

The four relevant goals for value elicitation surveys are the following: remind respondents of their budget constraints; attempt to deflect emotive responses; ensure that respondents distinguish between steady-state and interim losses (again, this is more likely applicable to an oil spill than an IDB project); and ensure that respondents are sensitive to the timing of the program or policy.

The NOAA Panel proposes five problems that, if present, would render the results of a CV study unreliable. These problems are a high nonresponse rate to either the survey in general or to the valuation question in particular, inadequate responsiveness to the scope of the benefits proposed, respondents' lack of understanding of the task, respondents' failure to "believe" the restoration scenario, and no explanations for responses that do not appear to reflect economically rational decision making, as when respondents do not refer to the cost or value of the program in explaining why they chose the option they did.

Although the recommendations have been described as a "thoroughly mixed bag,"[30] their publication effectively constitutes the promulgation of a standard that makes CV prohibitively expensive for many applications. As a result, the qualified endorsement of CV by the NOAA Panel may turn out to be a rather Pyrrhic victory for CV supporters if it ultimately leads to a decline in the use of CV in developed countries. In addition, IDB experience has failed to confirm the NOAA Blue Ribbon Panel's selection of dichotomous choice as the preferred form of CV.[31]

Advantages, Disadvantages, and Applications of Contingent Valuation

Table 7-5 summarizes the key advantages and disadvantages of the CV method. The survival and growth of CV in spite of the attacks on its validity probably has more to do with its advantages in being able to address almost any policy question asked and being able to measure total economic value, than in the effectiveness of the responses to these attacks. Although some authors have discounted the importance of nonuse values in the developing country context and thus the importance of measuring TEV, there are developing country examples where nonuse values are clearly important; the symbolic value accorded to Guanabara Bay in the World Bank report (1996) discussed in Chapter 6 is such an example. In addition to nonuse values, indirect use values can be important and difficult to measure by revealed-preference

[29] This guideline is easy to state but often it is not at all easy to live up to. One big problem is staying within the attention span and technical understanding of respondents. Thus, a table of water quality characteristics, such as dissolved oxygen, fecal coliforms, and turbidity, under with and without project conditions would probably leave lay respondents anywhere both bored and confused. But straightforward photographs are not in this case worth a thousand words (or numbers) because pollution is for the most part invisible. (Hence the reliance on dead fish to make the point in political and fundraising advertisements dealing with water quality.) For a practical effort around this obstacle, see Annex 7-A.

[30] "These guidelines are a thoroughly mixed bag: Some are supported by the conventional wisdom distilled from a substantial accumulation of research, while others accept uncritically certain untested conjectures from recent literature." (Randall, 1997, p. 1490)

[31] See Chapter 8 for a discussion of some of the econometric problems encountered by the IDB in applying dichotomous choice CV.

Table 7-5. Advantages and Disadvantages of CV Method

Advantages	Disadvantages
Applicable to more environmental goods	Based on respondents' stated intentions
Directly (but not separately) measures nonuse benefits	Places respondents in unfamiliar decision setting
Directly estimates correct Hicksian welfare measure if questions correctly phrased	Depends on creation of understandable, plausible scenario
Can incorporate reliability and validity checks	Vulnerable to abuse in survey design

Source: Hoevenagel (1994b, p. 252, Table 1).

methods. In any case, as pointed out in Chapter 6, trying to arrive at TEV by adding up the results of separate revealed-preference methods applied to different routes to benefits is fraught with difficulty because of the good possibility of under- and overlaps. Finally, there is the problem of the "external data" required by the revealed-preference methodologies[32] and the advantages of conducting surveys in developing countries. On the advantages of CV, Dale Whittington, an experienced practitioner, observes:

> *There are some contingent valuation researchers (I count myself among them) that believe it is easier to administer high quality contingent valuation surveys in some developing countries than it is in industrialized countries. For example response rates are typically very high in developing countries, and respondents are often quite receptive to listening and considering the questions posed. Also interviewers are inexpensive relative to prices in industrialized countries. This allows CV researchers to use larger sample sizes and conduct more elaborate split-sample experiments.* (Whittington, 1998, p. 28)

The use of CV in developing countries also presents a number of unique challenges. For one thing, many developing country economies may be only partially monetized, causing difficulties in translating values into monetary terms. This need not be an insurmountable obstacle, as shown by Shyamsundar and Kramer (1993), where WTP is stated in bushels of grain. A second challenge is presented by the translation of the survey instrument and responses into local languages or dialects. Third, considerable attention must be paid to local institutional and cultural issues. The survey designer must be sensitive to the attitudes of local people and their perceptions about local, national, and international institutions. Focus groups are a useful way to learn which payment vehicle, funding, and service delivery mechanism CV survey respondents are likely to trust. Also, respondents who are reluctant to say "no" to a question because of local mores pose interesting challenges for researchers utilizing hypothetical methods. Finally, since developing country applications rely almost exclusively on personal interviews, asking questions about what may appear to respondents or interviewers to be ridiculously high or low numbers in a personal interview is a concern.

Finally, the application of the methodologies to nonmarket benefit estimation in LAC countries may be less a question of whether to use a revealed- or stated-preference methodology than whether the latter methods are worth the effort, given that the revealed-preference methods are not generally available. If so, a general comparison of the methodologies may be of limited use. However, Table 7-6 presents an attempt to provide one in a developing country context. The overall effect of these perceived advantages seems to have been to make CV the

[32] "The [CV], discrete choice and [TCM] techniques all require primary data collection. This perhaps explains their evident success in developing economy applications as, by and large, they avoid the use of secondary data sources." (Georgiou et al., 1997, pp. 114–115)

Table 7-6. Evaluation of Nonmarket Benefit Estimation Methods

Methodology	Validity and Reliability	Comprehensiveness	Completeness	Ease of Implementation
Dose-response	✔	☹☹	✔✔	✔
Hedonic market	✔	✔	☹	✔
Travel cost	✔	✔	☹	✔
Averting behavior	✔	✔	☹	✔
Contingent valuation	✔	✔✔✔	✔✔✔	✔✔

Source: Hoevenagel (1994b, p. 263, Table 3).
Note: ☹☹ represents a very low score, ☹ a low score, ✔ a moderate score, ✔✔ a high score, and ✔✔✔ a very high score.

method of choice, for water quality valuation at least, in developing countries, as illustrated in Tables 7-7 and 7-8.

Conjoint Analysis

Conjoint analysis[33] encompasses a number of indirect hypothetical methodologies that are widely used by market researchers in the evaluation of new products and markets.[34] These multiattribute, preference elicitation techniques are based on the hedonic premise that commodities can be viewed as bundles of various attributes. In CA studies, respondents rank or rate a series of these bundles in which some or all of the different attributes are allowed to vary. From these rankings or ratings of the different bundles, marginal rates of substitution between the different attributes can be estimated. By including price as one of the attributes, these marginal rates of substitution can be translated into WTP for changes in attribute levels.

Four different kinds of multiattribute elicitation formats are used in CA. The first of these is dichotomous or contingent choice (referendum CV), where respondents are simply asked to choose their most preferred alternative from two or more choices with differing levels of attributes. Some contingent choice studies force respondents to choose one of the alternatives and some allow respondents to reject all. Dichotomous choice CV is essentially a special case of dichotomous choice CA, where the study is limited to two alternatives, one of which is the status quo, and only two variables, price and the environmental quality variable, are allowed to vary. Relaxing these restrictions allows CA to emphasize tradeoffs among hypothetical alternatives over the purchase of an environmental amenity.[35] Some have argued that this change in emphasis deflects emotive responses and as a result is less likely to generate protest or symbolic responses (see Mazzotta et al., 1997). Dichotomous choice CA has been used to estimate WTP to preserve different kinds of undeveloped land (Mazzotta et al., 1997), recreational fishing

[33] This terminology is borrowed from the marketing and transportation literatures, at least partly because no alternative has been proposed. The authors that do not use this label generally confine their analysis (and terminology) to specific methodologies within this category, e.g., contingent choice, ranking, or rating.

[34] "Conjoint analysis has become an increasingly popular approach to modeling consumer preferences for multiattribute choices. For example, over a decade ago, Cattin and Wittink (1982) estimated that more than 1,000 [CA] applications had been reported. [CA] has been used extensively in the marketing literature where it has proven especially useful in analysis of new products, market segmentation, or product differentiation." (Gan and Luzar, 1993, p. 37)

[35] However, if the status quo is not one of the posited alternatives in the CA analysis, there may be some theoretical issues: "The contingent pair model differs from the conventional referendum model in that neither card's utility level necessarily matches the status quo utility V[Q, M]. This implies that compensated demands for Q derived from the contingent pairing model may not be well defined since they are not based on any reference utility level (Mitchell and Carson, 1989, pp. 85–86). If utility is homothetic, however, marginal WTP functions can be defined from any indifference curve." (Mackenzie, 1993, p. 594)

Table 7-7. Studies Applying the Contingent Valuation Method

Area of Application	U.S.	Europe	Developing Countries
Agriculture	Some	Few	
Air quality	Many	Many	
Climate change	Zero to few	Few	
Energy	Many	Few	
Fishing: commercial	Some	Few	Zero to few
Fishing: recreational	Many	Few	Zero to few
Forestry	Some	Many	
Health risks	Many	Many	
Recreational hunting	Many	Some	
Parks, nature reserves, and wildlife	Many	Many	Some
Roads/transport	Few	Few	
Water quality	Many	Many	Some
Water supply and sanitation	Many	Some	Many

Source: Georgiou et al. (1997, p. 105, Table 4.1).
Note: "Few" is not defined by Georgiou et al., but it seems safe to take it to mean 1 to 3 or 4. "Some" is defined as up to 10 and "many" as more than 10.

Table 7-8. Studies Applying the Methodologies in Developing Countries

Area of Application	Hedonic	TCM	CV	CA[a]
Fishing: commercial			Zero to few	
Parks, nature reserves, and wildlife		Some	Some	
Water quality		Few	Some	
Water supply and sanitation	Few		Many	

Source: Based on Georgiou et al. (1997, pp. 105, 106).
Note: "Few" is not defined by Georgiou et al., but it seems safe to take it to mean 1 to 3 or 4. "Some" is defined as up to 10 and "many" as more than 10.
[a]CA, conjoint analysis. This is discussed later, but as shown in the table, does not yet seem to have been applied in developing countries.

trips (Adamowicz et al., 1994), preferences for locating landfills (Opaluch et al., 1993), and health risks (Krupnick and Cropper, 1992; Viscusi et al., 1991) (although these two studies blur the distinction between dichotomous choice CV and CA).

A second form of CA, contingent ranking, asks respondents to rank a set of hypothetical alternatives from most preferred to least preferred. Contingent ranking has been used to evaluate the demand for electric cars (Beggs et al., 1981), WTP for improved visibility at national parks (Rae, 1983), and improved water quality (Smith and Desvousges, 1986). A third form of CA, contingent rating, asks respondents to rate a set of hypothetical alternatives on a numerical scale. The difference between ranking and rating is that the latter asks respondents to supply information about how much they prefer one bundle to another. Since the responses to a contingent rating survey contain more information than the responses to a contingent ranking survey, some authors have asserted that contingent rating is the superior exercise.[36] However, these assertions assume that the two methods are equivalent in terms of the "accuracy" of the preferences elicited, and this assumption is unproven. Contingent rating has been used to estimate WTP for different attributes of salmon fishing (Roe et al., 1996) and waterfowl hunting (Gan and Luzar, 1993; Mackenzie, 1992).

[36] "The comparisons of models and WTP measures confirm the hypothesized advantages of the contingent rating approach, and demonstrate its appropriateness for valuing heterogeneous nonmarket goods." (Mackenzie, 1993, p. 602)

A fourth type of CA is graded pair or pairwise rating. In graded pair surveys, respondents are shown two different alternatives and are asked to indicate their preference for one of the products by choosing a number within a set of numbers, say from 1 to 7, where 1 represents the strongest possible preference for one good and 7 the strongest possible preference for the other good. The exercise is then repeated a number of times with different hypothetical alternatives. In addition to estimating the demand for electric cars (Segal, 1995), pairwise rating has been used to estimate WTP to avoid the adverse effects of electricity generation (Johnson et al., 1995), and to achieve health benefits from improved air quality (Desvousges et al., 1996).

CA is perceived as having a number of potential advantages over CV. For one thing, respondents are not required to explicitly monetize environmental goods or services.

> *Finally, [CA] offers some significant practical advantages over [CV]. Respondents are generally more comfortable providing qualitative rankings or ratings of attribute bundles which include prices, rather than dollar valuations of the same bundles without prices. In de-emphasizing price as simply another attribute, [CA] minimizes many of the biases that can arise in open-ended [CV] studies when respondents are presented with the unfamiliar and often unrealistic task of putting prices on non-market amenities.* (Mackenzie, 1992, pp. 175–176; see also Mazzotta et al., 1997)

Stated differently, CA focuses respondents on marginal tradeoffs between attributes as opposed to stating a maximum WTP. A second advantage is that it allows a more detailed evaluation of the alternatives.[37] Like dichotomous choice CV, CA is believed to present a more realistic, familiar setting for respondents[38] and to be free from strategic bias.[39] Finally, if, as some researchers believe (e.g., Johnson et al., 1995), the ability to accurately answer hypothetical questions about unfamiliar goods improves with reflection or examination of preferences, so that responding to a survey can be viewed as a dynamic learning process, then the greater number of elicited responses in CA may allow more room for this dynamic learning process to occur.

However, this last potential advantage is offset by a potential disadvantage—that people will tire of answering questions and as a result the accuracy of their responses will decline as the survey progresses.[40] Another disadvantage, potentially at least, is that CA "necessarily interposes in the environmental valuation process the step of estimating a utility function in

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[37] "[CA] has the advantage of allowing for the valuation of both the product or program as a whole and the various attributes of the product or program." (Johnson et al., 1995, p. 2)

[38] "Contingent referenda and contingent choices among specified attribute-price combinations both mimic familiar consumer decisions processes. While economists view prices and quantities as mathematically dual (and the conventional [CV] approach makes that duality explicit), consumer perceptions are unencumbered by this theoretical framework. Price is simply another attribute of the good in question. Surveys structured in accordance with such perceptions may avoid many of the protest and strategic biases that afflict [CV] studies." (Mackenzie, 1993, p. 596)

"Among alternative [CV] structures, a principal advantage of the paired comparisons approach is that, in many cases, respondents find that choosing among alternative commodities is among the most natural and frequently experienced decision environments, compared to directly evaluating individual characteristics. For example, people have routine exposure to this kind of choice in their purchases of market goods, where they often choose between products that are similar, but not identical." (Opaluch et al., 1993, p. 47)

[39] "There is also some evidence from the transport sector that [CA] diminishes the likelihood of strategic bias in responses, since, when offered trade-offs across a number of attributes, it is less clear how an individual might succeed in influencing policy by contriving answers to achieve certain ends." (Pearman, 1994, p. 240)

[40] "Because conjoint analysis requires answers to several trade-off questions, respondents may become bored or fatigued when answering questions. In these instances, responses to later questions may be of a lower quality." (Mathews et al., n.d., pp. 13–14)

circumstances where the functional form and arguments of the function are not clearly known." (Pearman, 1994, p. 243) Further, the method requires the assumption that "the question sequence presented to each individual in a conjoint study can be treated as a panel of uncorrelated responses." (Smith, 1997, pp. 55–56) Third, anyone attempting to apply or evaluate CA is handicapped by the very paucity of attempts to estimate nonmarket benefits with CA. This disadvantage is somewhat offset by the extensive use of CA in other applications, if for no other reason than that this use has led to the development of commercial software that significantly aids the design, administration, and interpretation of CA surveys.[41]

Although early evidence on CA is mixed, it seems clear that the indirect hypothetical methodologies will be the focus of much of the future development of nonmarket benefit estimation:

> *In addition to willingness to pay, contingent purchases, and contingent policy referendum methods, we are now seeing an explosion of contingent ranking and contingent choice experiments. We can in the near future expect to see contingent resource compensation experiments in which the relative values of different kinds of resource services are compared directly, rather than mediated by money measures. This proliferation of methods is desirable, is being encouraged by the EPA-NSF research program in valuation, and can confidently be expected to continue.* (Randall, 1997, p. 1493)

> *Hypothetical market methods appear to have a substantial role to play in securing a fuller incorporation of environmental considerations into public decision making. For many environmental goods, they seem to offer what is effectively the only way forward. Within this context, [CV] and [CA] should be seen as variations on a theme. They share many of the same strengths and weaknesses; overcoming problems with one will often help overcome equivalent problems with the other.* (Pearman, 1994, p. 244)

Whether this development will lead to or be accompanied by the application of CA methodologies to project analysis in developing countries is less clear. CA would seem to enjoy the same advantages as CV does over the revealed-preference (indirect observed) methodologies. In fact, the multiattribute nature of policy analysis may lend itself more readily to CA than CV. For instance, the description in McConnell (1995) of the typical water treatment project proposal as containing a number of alternatives with differing attributes could just as easily be describing a CA study. The challenge in molding such a scenario into a CA survey is to translate the highly technical or abstract attributes that characterize the different alternatives into attributes that are easily understood and meaningful to survey respondents. Where institutional concerns might bias the results of CV, CA might examine the preferences behind this bias by varying the institutional circumstances in different policy alternatives.[42]

The possibility thus exists that CA methods could allow estimation of a benefit function, making possible direct valuation of alternative possible end points in the environment, with

[41] For example, Sawtooth Software is a software development company exclusively focused on CA, and SPSS has a module dedicated to conjoint analysis.

[42] Whittington describes circumstances where it is common for a substantial portion of the proceeds of World Bank loans to be diverted to local politicians, all of which does not go unnoticed by potential survey respondents. In the course of pretesting a survey, these respondents might well "say that they do not want their government to borrow money from the World Bank because they know that much of the funds will not be used for the intended purposes, and in this regard they are in fact correct. If the survey instrument were implemented in this form, the results could substantially underestimate households' perceived benefits of the new water and sanitation system." (Whittington, 1998, p. 27)

each end point being described by a bundle of attributes achieved. This outcome could be useful to the IDB in at least two settings—project phasing and sensitivity analysis. As discussed in Chapter 4, it is common for water pollution cleanup efforts in LAC countries to be proposed in separable phases, usually owing to finance constraints. In addition to technical questions about the extra costs thus incurred, and practical concerns about whether future phases will actually be completed, such phasing creates a problem for standard benefit estimation. In effect, two phases will require at a minimum two different versions of a CV study that generates WTP for two separate end points (unless those involved in the decision are willing to buy into the concavity assumption and use linear interpolation from the estimate of ultimate benefits as a lower bound on benefits, as described in Chapter 2 for the Guaiba Watershed Management Project). It seems possible that a well-designed CA study could allow for direct valuation of each phase. In a sensitivity analysis, especially where there is uncertainty about both the actual preproject ambient water setting (see Chapter 4) and the operation of the project after its completion, it would be valuable to have a direct estimate of the benefits of different possible ambient water quality outcomes. While these estimates would themselves involve uncertainty, it would be of a different quality from that involved in arbitrarily varying the percent of "full" benefits achieved in a Monte Carlo setting.

Although some assertions have been made as to the relative theoretical and empirical advantages of one group of methodologies over the other, this literature appears too embryonic to be relied upon to any great extent. For example, there are assertions that CA has theoretical and empirical advantages because it generates more information, but there is practically no evidence on the quality of the answers to the additional questions required to generate this information. Essentially the decision to employ CA as opposed to CV will likely turn on the perception of three factors: (a) whether respondents have difficulty in monetizing their preferences within the CV framework, (b) whether the additional information that can be gleaned from CA surveys is useful, and (c) whether the quality of responses is adversely affected by the more involved surveys utilized by the CA methodologies.

MEASUREMENT OF HEALTH EFFECTS AND BENEFITS

As discussed in Chapter 6, despite their potential importance, few studies have been reported that estimate the economic values for the health effects of poor water quality in LAC countries. The few studies that do attempt to estimate such values limit their analysis to costs of treating actual cases of sickness and forgone earnings resulting from morbidity or premature mortality rates that are based on dose-response functions (Sadoff, 1996; Maimon, n.d.; World Bank, 1993; Margulis, 1992; and Serôa da Motta et al., n.d.). This methodology appears not only to be the state of the art in LAC countries, but in developing countries in general, as shown by Table 7-9.

Table 7-9. Health Risk Studies by Methodology and Region

Methodology	U.S.	Europe	Developing Countries
Hedonic property	Few		
Contingent valuation	Many	Many	
Dose-response	Many	Some	Many[a]

Source: Based on Georgiou et al. (1997, pp. 105, 106).
Note: "Few" is not defined by Georgiou et al., but it seems safe to take it to mean 1 to 3 or 4. "Some" is defined as up to 10 and "many" as more than 10.
[a]Including the economies in transition.

However, despite their widespread use, for a number of reasons these studies are clearly inadequate as attempts to measure the economic value of improvements in health. First, the benefits considered are not all of the benefits that result from an improvement in health, although, not coincidentally, they are the only benefits that can be valued without resort to the nonmarket benefit estimation methods discussed in this chapter.[43] Since these benefits are often large enough by themselves to support a project, they are often employed as rough lower bounds on the total health benefits.

Second, the human capital approach to valuing health implicit in this methodology carries with it a great deal of ethical or moral baggage:

> *Some of the implications of the human capital approach are unsettling. Because of discounting and the time lag before children become productive participants in the economy, the human capital approach places a much lower value on saving children's lives than on saving the lives of adults in their peak earnings years. And, because of earning differences by sex and race, the human capital approach places a lower value on saving the lives of women and nonwhites than on saving the lives of adult white males. Furthermore, the human capital approach assigns zero value to persons who are retired, handicapped, or totally disabled.* (Cropper and Freeman, 1991, p. 172)

Finally, perhaps the most important problem with this approach for economists is that it defies microeconomic principles by failing to take into account human preferences.

There are alternatives to the human capital approach that explicitly consider preferences and are less susceptible to moral or ethical criticisms. The first alternative is to measure WTP through one of the observed methodologies. The applicable methodologies include averting behavior and hedonic wage and property models. Although there are a number of examples of studies employing these methods to estimate WTP for improvements in health in developed countries, there are few, if any, in developing countries. The difficulty of obtaining the data necessary to undertake these studies is, as already discussed, the most likely explanation for this absence. This constraint is particularly problematic for those revealed-preference methodologies that require an informational "infrastructure," such as a system for recording property transactions or keeping health care records.

Even if data constraints were relaxed, however, other practical problems cast doubt on the presumption that these methods offer an improvement over those based on dose-response functions. For example, averting-behavior studies often involve discrete decisions, such as the purchase of bottled water (Abdalla et al., 1992), or smoke detectors (Dardis, 1980), or the use of seat belts (Blomquist, 1979), so that for the majority of people, the behavior cannot be pursued to the point where costs equal benefits, making it difficult to derive an average WTP. Joint products remain a problem for averting-behavior applications, for example, when bottled water is purchased both because it offers protection from disease and because it has improved taste. A key for all three methodologies is that individual knowledge and perception of risk (or more accurately, the perceived change in the level of risk on the basis of the averting behavior

[43] "Environmental pollution that impairs human health can reduce people's well-being through at least the following five channels: (1) medical expenses associated with treating pollution-induced diseases, including the opportunity cost of time spent in obtaining treatment; (2) lost wages; (3) defensive or averting expenditures associated with attempts to prevent pollution-induced disease; (4) disutility associated with the symptoms and lost opportunities for leisure activities; and (5) changes in life expectancy or risk of premature death. The first three of these effects have readily identifiable monetary counterparts. The latter two may not. Since reducing pollution may be beneficial to individuals because it reduces some or all of these adverse effects, a truly comprehensive measure of benefits should capture all of these effects. Measures based solely on decreases in medical costs or lost wages are inadequate because they omit major categories of beneficial effects." (Cropper and Freeman, 1991, p. 166)

or choice of job or housing location) is required, as opposed to some more easily measured objective measure of risk.[44] Evidence from developed countries on perceptions of risk compared with objective measures is mixed, generally showing positive correlations, but the perceptions often substantially over- or understate objective measures of risk (Cropper and Freeman, 1991). Finally, it is also difficult to extend indirect examination beyond rather narrow sections of the population, such as a certain class of worker or particular neighborhoods.

On the other hand, the hypothetical methods do not share many of these difficulties and are being used with increasing frequency to measure health benefits in developed and developing countries. (See, for example, for CV: Choe et al., 1996; Alberini et al., 1997; Mitchell and Carson, 1986; Smith and Desvousges, 1987; Gerking et al., 1988; and Jones-Lee et al., 1985. For CA: Desvousges et al., 1996; Viscusi et al., 1991; Krupnick and Cropper, 1992.) Where the link between environmental quality and health problems is direct (such as between water pollution and waterborne illness) and the health problems are severe enough, dose-response methodology will likely continue to be the simplest approach to generating benefit estimates that are high enough to support project approval and that are relatively immune from questions about their "instrumental validity." However, where more precise numbers are needed, dose-response approaches are clearly inadequate and the observed methodologies can provide little additional comfort. In these cases, hypothetical methods will be the methodology of choice. The challenge here will be in constructing accurate but understandable descriptions of the alternative risk situations to be achieved.

SPECIAL TECHNIQUES FOR DETERMINING BENEFITS

The following sections discuss techniques for determining benefits.

Combining Observed and Hypothetical Methodologies

A number of researchers have attempted to improve upon the performance of observed and hypothetical methodologies by combining the two in a single study. These combinations have occurred in two different ways. First, some researchers have attempted to "calibrate" the results from hypothetical surveys using revealed-preference results.[45] The problem with this approach would seem to be that the practical difficulties of implementing a revealed-preference approach constitute a real constraint that is in no way loosened by these combinations.

The second approach to combining the methodologies is termed *contingent activity*:

> *[I]ndividuals can be asked how they would change the level of some activity in response to a change in an environmental amenity. If the activity can be interpreted in the context of some behavioral model, such as an averting behavior model or a recreation travel cost demand model, the appropriate indirect valuation method can be used to obtain a measure of willingness to pay.* (Freeman, 1993, p. 166)

Although this approach overcomes the data difficulties, it does so at the price of incorporating the disadvantages of both types of methodologies with few of the advantages, i.e., it is

[44] "For market prices to convey information about individual preferences for risk reduction, individuals must be informed about the risks being valued. Furthermore, risks, as measured by the researcher, must correspond to individuals' risk perceptions at the time their market decisions were made." (Cropper and Freeman, 1991, p. 187)
[45] Smith (1997), pp. 43–53, provides an overview of the literature exploring the possibility of calibration for improving the results of benefit estimation studies.

hypothetical but requires the theoretic "tricks" of the revealed-preference methodologies to get at the values of interest. However, if a survey is conducted in such a way that stated- and revealed-preference methodologies may be employed to evaluate the responses, then these concerns may be somewhat dissipated:

> *Several of the valuation techniques typically use data from a household survey (for example contingent valuation, travel cost and hedonic property pricing methods). When a technique requires that primary data be collected with a household survey, it is often possible to design the survey to obtain the data necessary to undertake more than one valuation method. This approach is particularly useful in developing countries because reliable secondary data are rarely available for carrying out valuation work.* (Georgiou et al., 1997, p. 107)

However, this combination would add expense to survey design, administration, and interpretation (and might also increase the burden imposed on respondents by the survey, i.e., longer, more difficult surveys) without necessarily improving the estimates obtained and as a result may be difficult to justify.[46]

Benefit Transfer

Another alternative to the traditional nonmarket benefit estimation techniques, benefit transfer, has been suggested as a way around the considerable time and costs involved in performing a study using one of the traditional techniques. Benefit transfer involves the transfer of monetary values or functions obtained from a valuation study or studies at one or more sites (the study site or sites) to an alternative site (the policy site).[47] There are three basic approaches to benefit transfer.

The first approach applies mean unit values estimated for the study site or sites to the policy site. The form of these unit values varies, depending upon the context. For example, the valuation of recreational sites typically generates a consumer's surplus or average WTP estimate that is expressed as the value of a "person-day" for each recreational activity at the site. This value is then multiplied by the change in such days forecasted for the policy site to arrive at an estimate of the aggregate economic benefits of the change in policy. This approach implicitly assumes that the consumer's surplus or WTP experienced on average by individuals at the study site(s) is equal to that which will be experienced at the policy site.

The second approach does not make this assumption, but instead employs adjusted unit values. Adjusted unit values are simply mean unit values that have been systematically adjusted to account for any biases or differences between the sites that might affect WTP, such as differences in user demographics, the nature of the policy change, or the availability of substitute sites or services. The ability to make these adjustments depends on the availability of enough original study information to produce the correction equations.

The third approach is to apply the demand curve estimated for the study site(s) to the policy site or in effect to transfer a benefit function from the study site(s) to the policy site.[48] This is considered the preferred approach because it is believed to better account for differences between the sites.

[46] The IDB has experimented with combining methodologies (see Niklitschek and Léon, 1996).

[47] For general reviews of benefit transfer, see O'Doherty (1995), Brookshire and Neill (1992), and Smith (1992a).

[48] For examples and expositions of benefit function transfer, see Feather and Hellerstein (1997), Parsons and Kealy (1994), McConnell (1992), and Loomis (1992).

The advantages of benefit transfer obscure the extent of the obstacles to achieving believable estimates, i.e., that it is cheap and quick hardly inspires confidence in the results. First, the transferred benefits are only as good as the original study or studies that generated them. Second, applications are limited to the valuation of policies or projects that are closely analogous to existing policies or projects that were evaluated by a transferable study. Third, applications are limited to the valuation of sites that are closely analogous to existing sites that were evaluated by a transferable study. The net effect of these first three problems is that it is generally difficult to find a study that is suitable for transfer to the problem at hand.[49] However, even when a suitable study is found, transferring its results to the policy site is not an easy task because although the policy and study sites may be closely analogous, they will never be identical and the differences are sure to raise concerns about applying results across these differences. In addition, determining the geographical extent of the market at the policy site is difficult in the absence of information about local preferences (see Chapter 4).

The development of benefit transfer as an estimation methodology has proceeded along three tracks. First, researchers have attempted empirical evaluations of the validity of benefit transfer by taking two or more original studies of different sites, transferring the benefits of each to the other, and then testing to see if the transferred estimates are statistically different from the original ones for each site (e.g., Kirchhoff et al., 1997; Downing and Ozuna, 1994; Parsons and Kealy, 1994; Loomis, 1992; Opaluch and Mazzotta, 1992). In general, the result is that they are statistically different, and the conclusion is that benefit transfer should not be relied on to produce precise estimates.[50] More optimistically, the specific results of these studies may constitute the first steps in developing a protocol to make benefit transfer work with some reasonable degree of precision.

The second approach to developing benefit transfer involves the use of reviews or meta-analyses of segments of the benefit estimation literature to develop multistudy unit values or demand functions with some insight into the relationships of these functions or values with a wide range of explanatory variables.[51] These efforts have been hampered by the sensitivity of existing studies to the wide range of choices made by researchers as to model specification and econometric issues.[52] The conclusions of these studies have also been somewhat pessimistic, largely owing to the lack of studies generating the kind of data needed for this exercise.

The third way in which the development of benefit transfer has proceeded is in the implementation and reporting of benefit transfer studies. Although a large number of benefit transfer studies are performed, few are reported.[53] One area that is undoubtedly the subject of

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[49] For examples of the requirements and difficulty of finding a transferrable study, see Boyle and Bergstrom (1992) and Desvousges et al. (1992).

[50] "Our results imply that we ought to be skeptical of many efforts to transfer benefit estimates from one site, resource type, or environmental activity to another. Ultimately, the intended use of the benefit estimate determines whether benefit transfer is appropriate and provides adequate reliability. . . . However, when precision matters in the intended policy application, the appropriateness of benefit transfer is questionable." (Kirchhoff et al., 1997, p. 93)

[51] "A potentially useful approach to the benefit transfer would be to pool the data from existing studies and apply multiple regression analysis. If the basic model specification is complete, that is, if it includes the relevant explanatory variables in the correct functional form, then it could explain the variation in benefits embodied in differences among the explanatory variables. The net benefit estimated for a site lacking data would then be predicted by inserting appropriate values of explanatory variables into the model fitted to data from other study sites." (Walsh et al., 1992, p. 707) See also Smith and Osborne (1996), Freeman (1995), and Smith and Kaoru (1990).

[52] For illustrations of the importance of these effects, see Smith and Kaoru (1990) and Vaughan and Russell (1982).

[53] "Benefits transfer, performed when research funding and time are scarce, is probably the most frequently used method of estimating the benefit of environmental improvement from some proposed policy; it is commonly used for quick analyses by and for U.S. federal and state agencies and on behalf of Indian tribes." (Eiswerth and Shaw, 1997, p. 2381) Unit day values for recreation have long been required in many forms of cost-benefit analysis for U.S. water resources projects.

many of these studies, and in which there have been a couple of reported studies, is the estimation of health benefits.[54] Alberini et al. (1997) and Krupnick et al. (1996) both attempted to transfer air quality benefits, while Kask and Shogren (1994) attempted to transfer water quality benefits. Taken together these studies provide detailed guides to the problems and pitfalls that may be encountered in attempting to transfer health benefits. For example:

> *But transferring the benefits of reduced health risks presents a significant challenge. The challenge arises from the multiple sources of risk, the mortality and morbidity effects indicated by a variety of symptoms, the number of illness days, the long latency period between cause and effect, and an individual's ability to privately or collectively reduce the probability or severity of the risk.* (Kask and Shogren, 1994, p. 2813)

Like much of the other literature on benefit transfer, these studies seem overwhelmed by the obstacles to generating reasonably precise benefit transfers.

As should by now be obvious, the gist of this literature is that, at least at this point, benefit transfer is only suitable for tasks where the need for accuracy is low. However, it can be useful in certain circumstances:

> *If resource constraints are realistically considered, there are likely to be many cases in which benefit transfer methods can actually yield better estimates of economic values than can be obtained from studies specifically tailored to estimate the value of proposed changes at the policy site. This situation is most likely to arise when: (1) the study site is very similar to the policy site; (2) the policy change or project at the study site is very similar to that proposed at the policy site; (3) the valuation procedures used at the study site were analytically sound and carefully conducted; (4) time, financial resources, and (or) personnel available for analysis at the policy site are not sufficient to undertake a high-quality study. Also, benefit transfer methods may be particularly useful in policy contexts where rough or crude estimates of economic benefits may be sufficient to make a judgment regarding the advisability of a policy or project.* (Pearce et al., 1994, pp. 172–173)

Although benefit transfer may not yet be capable of providing precise estimates, it is not clear that it should be discarded. Areas in which benefit transfer might prove useful to the IDB include prefeasibility screening, assessment of environmental damage, and project valuation of global multiple works (Type II). The first steps to utilizing benefit transfer would be the creation of a database from which benefit estimates could be drawn and the adoption of a policy ensuring that original benefit estimates are performed and reported in such a way as to facilitate the future transfer of these estimates:

> *Research using primary data is the basis of benefit transfer via the provision of data and models needed to extend study site values to one or more policy sites. Thus, original investigations using primary data must not simply focus on the end result of estimating a value for the policy issue at hand. Original analyses using primary data,*

[54] "This category [health benefits] is probably the easiest for making credible benefit transfers across locations, given comparable economic circumstances (comparison across affected populations with very different income levels or other socioeconomic circumstance is, in contrast, more difficult). Once atmospheric or other natural processes are taken into account (say, in the estimation of the effect of emission reductions on ambient air quality), one can presume to a first approximation that the health effects and the values that people place on avoiding them are reasonably similar across locations." (Kopp et al., 1997, p. 17)

and reporting of these analyses, must reflect their future use as data for benefit transfer studies. These investigations of statistical relationships will help identify key variables and relationships for determining the suitability of a study site value for transfer, and perhaps, for adjusting study site values at the policy site to reduce potential biases in transfer estimates. (Boyle and Bergstrom, 1992, p. 662)

CONCLUSION

The difficulties in applying the revealed-preference methodologies are well documented. Attempts to apply these methodologies in developing country contexts exacerbate these difficulties and create additional problems where data are scarce and related markets are, for one reason or another, less useful than in developed countries. The difficulties in applying the hypothetical methods are also well documented. However, these difficulties are to some extent ameliorated in developing countries. Perhaps more important, the hypothetical methods are, potentially at least, much more powerful techniques because they can generate an estimate of WTP for the exact policy or project envisioned. Still, the validity of the hypothetical methods is not accepted by all, and for some, the oft-repeated phrase "if you ask a hypothetical question, you get a hypothetical answer" remains the most apt summary (Bohm, 1994).

Even for converts, the hypothetical methods remain a difficult enterprise, requiring significant expertise, particularly in terms of survey design and administration. There is always the temptation to say that while one specific CV method may be best for one instance, other methods may be better for other instances, and to recommend the cultivation of an understanding of the characteristics that determine which methodology works best in which instances. However, the resources required to develop the necessary expertise to design and administer "valid" CV studies may be so great as to exclude the development of expertise in the other methods, rendering this recommendation fine in theory but unrealistic in practice.

Perhaps what is needed is a broader view of the problem. While the goal is often to obtain the best "number" in the cheapest fashion, this framing may be too narrow to properly judge the contributions of the methodologies. In the end, the contributions of the hypothetical methods to an understanding of preferences for project alternatives may prove to be the deciding factor.

[C]ontingent valuation studies serve several functions. While the focus in this volume has been on the estimates of values produced, it is worth remembering that contingent valuation also reveals considerable amounts of information about what local people want. Contingent valuation can therefore be used as a vehicle for public participation in decision making. This is a factor in favour of contingent valuation, especially as we are recognising that many investments "fail" because of a lack of consultation and assessment of local wants and needs. The high cost of a contingent valuation study therefore needs to be compared to the multiple benefits of the surveys carried out. (Georgiou et al., 1997, p. 115)

Annex 7-A

The Water Quality Ladder

Water quality can be described either in terms of the uses for which a particular body of water is suitable or in terms of the objective characteristics of the water itself. In turn, objective characteristics traverse a continuum from those that are readily perceptible to those that can only be detected by scientific measurement. In certain dimensions [e.g., visible phenomena such as the extent of algal growth; the clearness of the water; and the existence of suds, foam, or debris (David, 1971) and smells such as hydrogen sulfide from anaerobic conditions], people at large find it easy to perceive changes in water quality. However, some more important characteristics, such as dissolved oxygen content, cannot be detected by human sight and smell. Thus it is not surprising that people's ratings of water quality levels are likely to exhibit a less-than-perfect degree of association with any one or a combination of the several scientific measures of quality conditions (Binkley and Hanemann, 1978). This poses a problem for benefit estimation because the existence of a positive willingness to pay for water quality improvement depends upon the ability of people to perceive water quality changes when such changes do in fact occur.

This problem has led previous investigators either to attempt to engineer the marriage of an objective water quality index (based on some weighted combination of scientific quality parameters) and a subjective index of publicly perceived quality (Bouwes and Schneider, 1979), or to link subjective indices of public perception and expert perception (Dornbusch, 1975).

In the early 1980s, researchers at Resources for the Future developed a water quality ladder that describes water quality primarily in terms of the uses for which the water is suitable, and secondarily in terms of a few obvious quality conditions (clearness, odor, debris, etc.). The use-based levels were located by indexing a set of five objective scientific water quality parameters, using a variant of the National Sanitation Foundation's Water Quality Index (WQI) (Booth et al., 1976; McClelland, 1974) along with informed judgment.

A number of sources were consulted to ascertain the minimally acceptable concentration levels of five measurable quality characteristics associated with five potential uses of natural watercourses. These characteristics were fecal coliforms, dissolved oxygen, maximum 5-day BOD, turbidity, and pH.[55] The five quality measures were the only ones for which numerical values could be obtained across all use classifications, a requirement dictated by the index approach. Particular attention was given to state water quality standards (North Carolina Environmental Management Commission, 1977; Dorfman et al., 1972), because they report specific critical water quality parameters associated with a set (usually four or five) of descriptive water quality classifications. The consensus results for each quality level are summarized in Table 7A-1.

[55] Sources consulted include Thomann (1971), USGS (1978), Pickle et al. (1973), Davis (1968), Economics Research Associates (1979), Katz (1969), Dorfman et al. (1972), North Carolina Environmental Management Commission (1977), APHA, AWWA and FSIWA (1955), National Technical Advisory Committee (1968), NAS-NAE (1972), EPA (1976), Davidson et al. (1966), National Planning Association (1975).

Table 7A-1. Consensus Water Quality Characteristics of Five Water Quality Classes

	Measurable Water Quality Characteristics				
Water Quality Classification	Fecal Coliforms (No./100 ml)	Dissolved Oxygen (mg/1)[a]	5-day BOD (mg/1)	Turbidity (JTU)[b]	pH
Acceptable for drinking without treatment	0	7.0 (90)	0	5	7.25
Acceptable for swimming	200	6.5 (83)	1.5	10	7.25
Acceptable for game fishing	1000	5.0 (64)	3.0	50	7.25
Acceptable for rough fishing	1000	4.0 (51)	3.0	50	7.25
Acceptable for boating	2000	3.5 (45)	4.0	100	4.25

[a]Percent saturation at 85°F in parentheses.
[b]JTU, Jackson turbidity unit.

In order to associate each of the five possible sets of scientific measures with a single-valued ordinate of the quality ladder, a truncated version of the National Sanitation Foundation Water Quality Index was used:

$$\mathrm{WQI} = \prod_{i=1}^{5} q_i^{\hat{w}_i} / 10$$

where:

q_i = The quality of the i[th] parameter; a number from 0 to 100 obtained from the transformation functions for water quality measures in McClelland (1974);

$\hat{w}_i$ = The weight assigned to the i[th] parameter. The original weights (w_i) reported in McClelland (1974) cover nine quality measures and

$$\sum_{i=1}^{9} = 1.00$$

The RFF adjusted weights cover a smaller number of measures, but were also required to sum to 1.0 via the transformation

$$\hat{w}_i = w_i \left(\sum_{i=1}^{9} = w_i / \sum_{i=1}^{5} w_i \right)$$

The resultant ladder appears in Figure 7A-1. It has been used in several CV studies of water quality benefits, including those reported in Mitchell and Carson (1981) and Smith and Desvousges (1986).

For example, the index value for the "acceptable for rough fishing" classification was developed as shown in Table 7A-2. Similar calculations for the remaining four classes yield the water quality ladder shown in Figure 7A-1.

Figure 7A-1. The RFF Water Quality Ladder

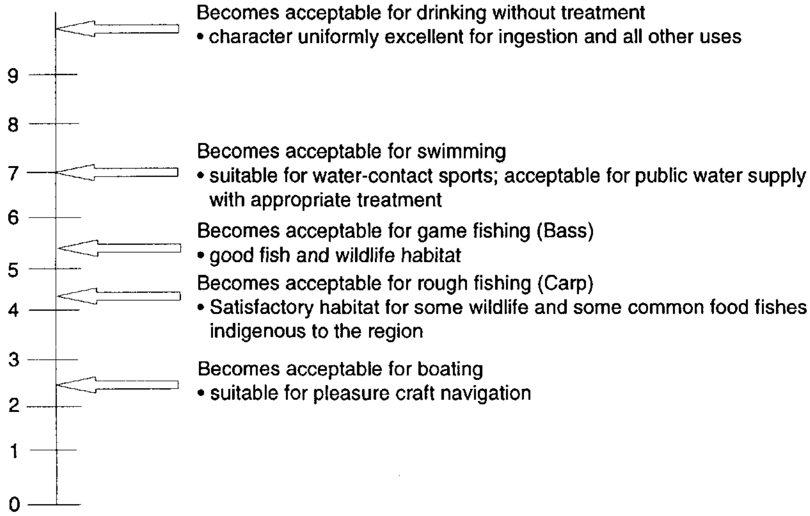


Table 7A-2. Index Value for Rough Fishing

Characteristic	Value	Scaled Value (q_i)	Weight ($\hat{w}_i$)	Weighted Scale Value ($q_i^{w_i}$)
Fecal coliform	1,000/100 ml	20	0.242	1.985
Dissolved oxygen	51%[a]	44	0.274	2.820
Max. 5-day BOD	3 mg/l	74	0.161	2.000
Turbidity	50 JTU	38	0.129	1.599
pH	7.25	93	0.194	2.049
Index $(\prod_{i=1}^{5} q_i^{w_i}/10)$				4.5

[a]Percent saturation at 85°F.

Chapter 8

Uncertain Willingness-to-Pay Measures from Referendum Contingent Valuation Surveys

Previous chapters have remarked on the difficulties encountered in comparing willingness-to-pay estimates for specific categories of benefits across a number of published studies. When the method of referendum contingent valuation is used, this is an inherently uncertain enterprise if the original data are not available because any number of central tendency measures can be extracted, and it is not always possible to determine which one was used and reported.

This chapter explains how to obtain parametric and nonparametric estimates of mean or median benefits from dichotomous choice contingent valuation data. Several approaches are explained and illustrated with examples, including evaluation of the survival function produced by econometric estimation of Hanemann's random utility model (Hanemann 1984, 1989), three nonparametric estimators of the mean, and Haab and McConnell's bounded probit mean and median (Haab and McConnell 1997b, 1998b, 1999). The wide range in per household benefits produced by these various techniques is illustrated using survey data collected for a major water quality improvement project being planned in Brazil. The chapter demonstrates the effects of uncertainty about the project's gross benefits under alternative scenarios, and shows that the project's net present value and the decision to invest can be significantly affected by the benefit measure chosen.

While no "true" and unambiguous benefit measure exists, nonparametric approaches to extracting mean willingness to pay from referendum CV data have the advantage of simplicity over parametric approaches, which impose a preconceived distributional assumption on the data, are more computationally intensive, and are more prone to errors in calculation and interpretation. In particular, the variance of the mean estimate cannot be calculated from analytical formulas using a parametric approach, nor can some variants of expected value. Numerical or approximation techniques are required instead (Ardila, 1993; Hazilla, 1999). This makes it more difficult to represent the degree of statistical uncertainty in survey sample estimates of mean WTP.

The principal message of this chapter is that reasonable upper and lower limits on mean willingness to pay can be conveniently calculated using two nonparametric estimators that are analogous to Laspeyres and Paasche measures of welfare change (Boman et al., 1999). These nonparametric estimates cover the likely range in mean WTP, making the calculation and choice among a host of parametric measures unnecessary. The nonparametric mean estimates not only provide bounds on our subjective uncertainty about which central tendency measure is best, they also permit the randomness around those bounds to be captured by simple formulas for the statistical variation in the means. Chapter 9 takes this issue of subjective and statistical uncertainty about benefits one step further by applying a Monte Carlo risk analysis to produce a probability distribution for the net present value using the Brazilian water quality improvement project as a case study.

OVERVIEW

Cost-benefit analysis of proposed projects is an inherently uncertain enterprise because it involves the future, which we can never know. Costs and project performance can be different from our expectations. The economy in which the project is embedded may change, and the tastes, incomes, and preferences of the population affected by the project may change as well in ways that are hard to predict. When the proposed project involves environmental public goods, such as improved air or water quality, another widely recognized source of uncertainty is the behavior of the natural system involved. Completing this familiar list is uncertainty about project benefits, the issue of concern here.

Previous chapters argued that an increasingly respectable and common way of estimating the benefits of environmental projects is to use the contingent valuation method, which involves directly asking people about their willingness to pay for the environmental effects to be provided by the project. As indicated earlier, two broad alternative ways of asking the valuation question are available: *open-ended*, in which the respondent can name any amount he or she wishes when asked some version of "What are you willing to pay?"; and *dichotomous choice* (referendum or yes/no) in which the respondent is asked: "Are you willing to pay (at least) B (per period)?"

Most relevant to this chapter, the NOAA Blue Ribbon Panel (NOAA, 1993) recommended, among other things, that CV studies be done using yes/no referendum format questions.[1] This recommendation has been adopted by many practitioners who deal with real-world program or project evaluations rather than development of methods. For example, at the Inter-American Development Bank, contingent valuation has become the method of choice for estimating the benefits of investment projects aimed at improving water quality. As noted in Chapter 2, over the past decade the Bank approved 18 projects with sewer and/or wastewater treatment components, and 13 of them employed cost-benefit analysis whose benefits came at least in part from CV estimates. Most of the stated-preference CV surveys used the referendum format (Ardila et al., 1998).

The referendum CV approach opens up a new and substantial source of uncertainty in benefit estimation. That source is the choice of econometric technique and subsequent calculation rules used to translate yes/no responses into mean or median WTP numbers. In project analysis this source of uncertainty is easily overlooked; almost none of the projects reviewed by Ardila et al. (1998) addressed it. Moreover, most analyses appear to have used an estimation formula that understates benefits.

This chapter is organized as follows. First, the nature of the inference problem is described in what we hope is an intuitively appealing way. A simple stylized example shows how ambiguity about the correct measure of central tendency can arise, specifically the theoretically inconsistent phenomenon of a negative mean willingness to pay for a utility-improving intervention. Moving closer to reality, alternative central tendency measures are proposed and illustrated using referendum contingent valuation survey data collected for a recent IDB project appraisal. The water quality impacts of the case study project that respondents were asked to value in a referendum CV survey are briefly described. Then 12 different versions of a central tendency measure of per household benefits are produced from the project data using methods suggested in the literature. Six come from the economist's customary route of econometrically estimating a binary choice

[1] See Chapter 7. There is a vast and rapidly growing literature on the CV method, especially the problems that arise in creating successful survey instruments, obtaining satisfactory response rates, and interpreting responses. See, for example, the relatively early seminal work by Mitchell and Carson (1989); and the exchange in economic perspectives occasioned by the huge damage-estimation efforts done by both sides in the Exxon *Valdez* case (Portney, 1994; Cummings and Harrison, 1995; and Loomis et al., 1996). Survey validity and design issues are outside the scope of this study.

model relating the respondent's acceptance probability to the bid offered and socioeconomic characteristics, using a logit specification of the inverse distribution function.[2] The rest are alternatives that involve either nonparametric measures that can be easily obtained without econometrics from the pooled data (i.e., the marginal rather than conditional distribution), or a more complex method that imposes lower and upper bounds on median WTP in econometric estimation. Finally, the effect that uncertainty about "actual" WTP has on the discounted net benefits of the case study project is explored and some general lessons are drawn.

INTRODUCTION TO THE PROBLEM

Contingent valuation approaches to estimating the benefits of a project necessarily involve surveying samples of the population of interest. If the sample is representative of the population, the sample mean of willingness to pay per capita (or per household) can simply be attributed to everyone in the beneficiary population of size N, so total project benefits are typically obtained as the product of N and per capita WTP.

Implications of the Referendum Format

In the early years of CV, the method of payment elicitation was direct and open ended. People were asked to reveal the specific monetary amount they would be willing to sacrifice for the provision of a nonmarketed good such as an improvement in ambient environmental quality. Obtaining a measure of central tendency from this kind of data was as simple as calculating the mean or median of the WTP values provided by the survey respondents. The econometric analysis involved was minimal, usually being confined to plausibility checks undertaken by split sample comparisons or by regressing the payment amounts on income and other socioeconomic variables to see if the signs on the parameter estimates in the relationship were consistent with prior expectations (e.g., WTP increasing with income).

All of this changed with the advent of the referendum format, which only asks if the respondent would or would not be willing to pay a specific preselected amount. Under this format, it is not possible to know the true WTP of any individual directly.[3] Because those who answer in the affirmative might actually be willing to pay even more, and those who answer in the negative might be willing to pay something less, econometric techniques have to be brought to bear to somehow interpolate and infer an expected value or other central tendency measure from the dichotomous choice information.

Simplicity of data analysis was sacrificed in the referendum method in order to construct what many felt was a more realistic choice game. The upshot of this change has been that the central tendency measure is no longer independent of manipulation by the analyst because, of necessity, it is the outcome of a sometimes complex process of survey design, choice of model specification, model estimation, and function evaluation (Duffield and Patterson, 1991).

In consequence, the notion that contingent valuation experiments of the referendum type can reveal a unique number that accurately and unambiguously represents individual willing-

[2] The emphasis throughout this chapter is on function evaluation to extract a measure of central tendency, not on the prior steps of CV survey design or choice of model estimation. Instead, the survey is taken as a given. The specifications of the functional form and arguments in the econometric choice model follow the selections made by the Brazilian consultant who initially analyzed the data.

[3] The discussion leaves aside prior questions about whether a "true" value exists previous to the survey process, whether respondents try to discover their own WTPs, and whether, even if they know them, interviewees try to conceal their true preferences by providing misleading answers that reflect strategic bias.

ness to pay for improved water quality is unrealistic. Rather, there are several possible numbers, each dependent upon the way the initial survey was designed and administered and the way the resulting raw data were passed through the summarizing econometric sieve and reconstituted in the form of a central tendency measure. In short, such estimates are always uncertain when we acknowledge the existence of many routes that could be taken to obtain them and the several decision alternatives present at each step along the way. This is not a counsel of doom, or a suggestion that cost-benefit analysis based on referendum CV not be undertaken. But it is a fact that any benefit estimate to a greater or lesser degree is always a product of the analyst's protocol and judgment, something careful analysts recognize and communicate to the users of their results.

A Hypothetical Example

In a dichotomous choice referendum survey, a group of i=1...n different payment or bid levels is preselected and the total sample is split up into n groups or subsamples. For each bid, B_i, dichotomous choice information can be summarized by the fraction of subsample respondents offered a given bid amount and saying "No, I am not willing to pay B_i for the public good" relative to the total number of respondents offered B_i. At each bid level surveyed, there will therefore be a fraction, F_i, rejecting the offer, and a fraction, $(1 - F_i)$, accepting it.[4]

If the project being investigated is a good idea, one would expect that everyone would be willing to pay some positive amount to have it, or at least would not require a payment to accept it (that is, not have a negative WTP).[5] However, with a dichotomous choice survey instrument, there will be uncertainty on this score unless a bid level low enough to produce $F_j = 0$ is offered. And, similarly, there will be uncertainty at the upper end unless a bid level high enough to produce $F_j = 1$ is offered.[6] At a simple intuitive level, the first kind of uncertainty can lead to the sorts of difficulties involving negative mean willingness to pay sketched in Figure 8-1.

Consider three different set of yes/no responses to three bid levels: $2.50, $5.00, and $7.50. The data are then used to infer linear cumulative distribution functions over bids, as shown in the figure.[7] For function (1), for example, 30 percent of those offered $2.50 say "yes"; 20 percent of those offered $5.00 say "yes"; and 10 percent of those offered $7.50 say "yes." With the hypothetical sample points indicated in the figure, the equations for the three cumulative probability functions for positive responses $[1 - F(B_i)]$ are

$$(1)\ \text{prob(yes)} = (1 - F_1) = 0.40 - 0.04\ \text{(bid)}$$
$$(2)\ \text{prob(yes)} = (1 - F_2) = 0.50 - 0.05\ \text{(bid)}$$
$$(3)\ \text{prob(yes)} = (1 - F_3) = 1.00 - 0.10\ \text{(bid)}$$

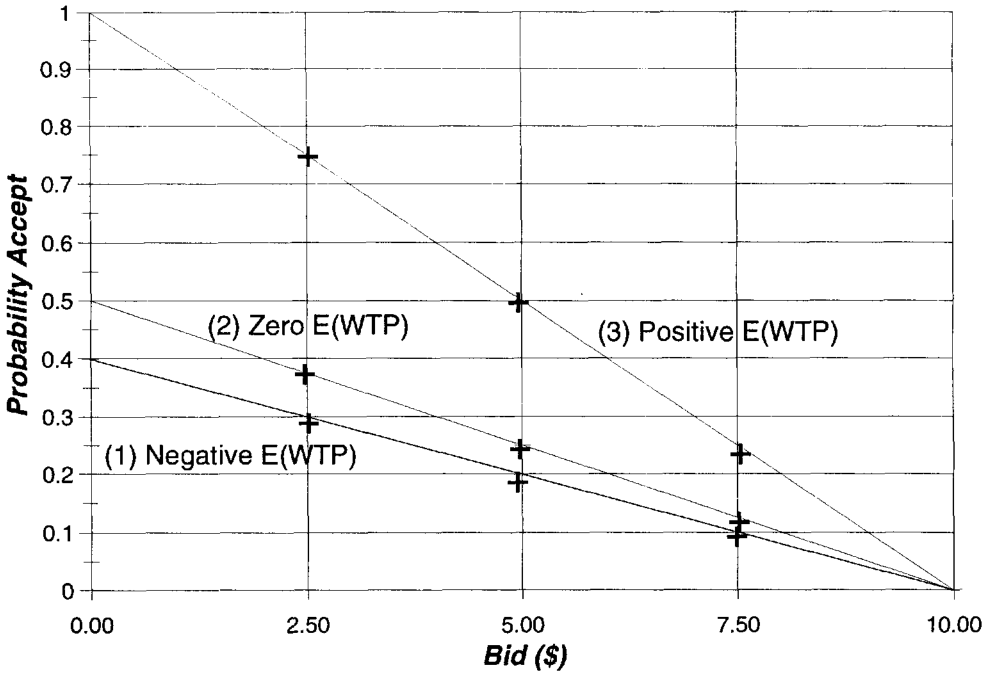
[4] There is an entire literature on the issue of bid design. For a brief and useful review of the implications of different designs, see Creel (1998).

[5] Probably for every project there are some people who will see themselves as losing something because of it. Thus, the prospective neighbors of a wastewater treatment plant might prefer the status quo because of the plant's local negative externalities, especially if they are upstream of the current outfall. In an actual survey setting, some negative WTPs were stated for an improvement in drinking water quality in a study undertaken by Kwak and Russell (1994). These respondents turned out to be vendors in the vicinity of springs that were heavily visited because of the existing perception of a potable water quality problem.

[6] Even with a survey design including very low and very high bid levels, there will be uncertainty because of sampling, but at least a simple plot of the approximate cumulative probability of acceptance or rejection will be defined. See the discussion on nonparametric methods.

[7] Strictly speaking, the relationship between bid and the probability of rejection is the familiar upward-sloping cumulative distribution function F(bid), and the downward-sloping relation between bid and the probability of acceptance shown in the figure is the inverse cumulative distribution function (also called the survivor function), or 1 – F(bid).

Figure 8-1. Three Possible Linear Inverse Cumulative Distribution Functions for WTP



Note that in the figure only function (3) covers the allowable probability range of 0 to 1 for bids greater than or equal to 0. The other two functions have to be extended into the negative bid quadrant (not shown) to yield acceptance probabilities greater than 0.5 and 0.4, respectively. The specifications of $(1 - F_2)$ and $(1 - F_3)$ imply acceptance probabilities of 1.0 only at negative bid levels of –\$10.00 and –\$15.00, respectively. The linearity of the three inverse cumulative distributions means that they represent uniform probability density functions over the intervals they cover such that[8]:

(1) $f_1(x) = 0.04$ from x = –\$15.00 to \$10.00
(2) $f_2(x) = 0.05$ from x = –\$10.00 to \$10.00
(3) $f_3(x) = 0.10$ from x = \$0.00 to \$10.00

The corresponding means or expected values of WTP are equal to the integral over the bid range of the product of the density and x. In the linear case the bid range is between the maximum bid driving $F(x)$ to 0 and the minimum bid that sets $F(x)$ to 1.0[9]:

$$E(\dot{WTP}) = E(\text{bid}) = \int_{\min}^{\max} x f(x) dx \qquad \text{where bid} = x$$

[8] The probability density over x, $f(x)$, is obtained by taking the derivative of the cumulative distribution function $F(x)$ with respect to x. In this case x is the bid and the slopes of the three linear cumulative density functions yield the probability density.

[9] More generally with nonlinear cumulative densities that are asymptotic to the upper and lower probability bounds of 0 and 1, the bid range runs from minus to plus infinity. See the discussion of analysis issues in a subsequent section.

Calculation for the $f_1(x)$ case gives

$$E_1(\text{WTP}) = \int_{-15}^{10} x(0.04)dx = (0.04x^2)/2 \Big|_{-15}^{10} = -\$2.50$$

Similarly for the other two inferred bid distributions, E_2(WTP) = \$0 and E_3(WTP) = \$5.

Summing up, in the example the bid level at which all of the respondents would say "yes" was an uncertain number that was not provided directly by the sample data, but had to be inferred by fitting a (linear) inverse cumulative distribution. The sample information available at positive bid levels implied, for two of the exercises, that the bid at the 100 percent acceptance level was negative—in one case –\$15 and in the other –\$10. The available information, used in the simplest way,[10] consequently produced mean WTP estimates ranging from a negative \$2.50 to a positive \$5.00. The first general lesson of this section (which carries over to more sophisticated nonlinear functional forms discussed later) is that if, at a zero bid, the predicted acceptance rate is less than 50 percent and the model is not confined to only positive bids in estimation or function evaluation, the predicted mean willingness to pay will be negative.

Even if the odd "objector" might show up, it seems highly unlikely that a negative mean WTP would be a correct inference. The target population could reject a proposed project because on net for the average person, the investment is not producing a "good," perhaps because negative externalities generated by the investment (the treatment plant) are so severe and widespread that the current without-project situation is preferred. Or, those surveyed could exhibit such a strong case of "status quo bias" that they require a subsidy as well as an environmental improvement to voluntarily move away from the current situation (Adamowicz et al., 1998), although this behavior would seem to be irrational and at odds with the utility-theoretic basis of CV. Finally, a very plausible cause of this result could be that those surveyed do not believe the scenario because they are cynical about the possibility that an actual investment in environmental quality will be made. This is a questionnaire design problem involving an unpopular choice of payment vehicle and project executor that usually can be discovered and addressed through focus groups and pretesting before the final survey is administered.

Besides the causes and cures for negative mean WTP, a second lesson from the example is that there is something to be said for spreading out the range of bids presented to respondents so that the chance is higher of discovering the low bid that drives the acceptance rate close to 100 percent, as is the chance of finding the high bid for which acceptance is near 0 (assumed to be \$10 in the example). Knowledge about the tails of the cumulative distribution of acceptance as a function of bid level can be very useful in constructing a nonparametric estimate of the mean to serve as a check against more complicated econometric density function estimation approaches that impose a prespecified shape and range of support (see later discussion). One way to identify the bid levels determining the upper and lower tails of the density is to do an open-ended CV survey in a pretest and design the bid groups and subsample sizes accordingly (Cooper, 1993).

The inference problem, however, is larger but more subtle than that of occasionally producing a negative mean WTP. Simply stated, it is that there are many ways of attacking referendum CV data; that each will in general produce a different mean WTP; and that none of them is so obviously superior that all the others can be rejected. "True" benefits are then unknowable, even if we believe the referendum CV method to be in principle capable of

[10] In practice, no analysts actually use the uniform density/linear cumulative density assumption anymore. While it is easy to fit a linear equation to binary 0,1 choice data with ordinary least squares (OLS), that approach has several undesirable econometric properties (e.g., heteroskedasticity). The advent of accessible computer software for qualitative dependent variable analysis in the 1980s (logit, probit) made the OLS shortcut unnecessary and irrelevant. However, it provides a convenient and dramatic illustration of the problem discussed here.

eliciting them. The extent of the resulting uncertainty is illustrated using data from the project described next.

AN EXAMPLE PROJECT—CLEANING UP THE TIETÊ RIVER

The parts of the Tietê River and its tributaries that flow through the São Paulo, Brazil, metropolitan area (SPMA), are the most polluted bodies of water in the state. The Tietê enters the metropolitan area with acceptable water characteristics, but in Guarulhos, at the confluence of the Jacu, it becomes anaerobic or close to it (see Map 1). From the Jacu downstream, the large volume of domestic and industrial waste dumped into the relatively small volume of river flow has made the river an open sewer that supports no aquatic life and smells most of the year over a stretch of more than 80 km. The reason for this is that the tributaries of the Tietê in the metropolitan area and the Tietê itself receive waste well beyond the river's natural processing capacity. At present the organic load is predominantly from households (360 tons per day, which is 80 percent of the total), with surface runoff accounting for another 62 tons per day (14 percent of the total) and industry contributing another 30 tons per day (7 percent). The problem is severe all year long and becomes critical in the dry season.

The Project

The proposed project for cleaning up the Tietê involves the extension of sewers to currently unsewered households (and businesses) and the provision of wastewater treatment plants at the discharge ends of those sewers. The major concern is the removal of oxygen-demanding organic materials (measured as biochemical oxygen demand) and disposal or recycling of these materials in ways and places that do not tax the dissolved oxygen carrying capacity of the river. The overall project is divided into three stages. The first stage has already been completed and the beginning of operation of the next two stages is contemplated for 2003 and 2010, respectively (see Map 1). The predicted effects of the project's three stages on water quality, as measured by BOD, DO, and the sewage-related health threat proxy of fecal coliform bacteria counts, are discussed in Chapter 9.

The project has local and general water quality benefits. The local benefits of the overall project involve the "neighborhood" effects created by sewering—that is, the cleaning up of locally offensive and dangerous conditions. The benefits component of interest here is that arising from general improvements in the mainstream and tributaries as summarized in Map 2. This was approached in the project analysis via a dichotomous CV survey asking WTP for the described dissolved oxygen improvements, which were from 0 to at least 1.5 mg/l in the segments asked about. To reflect the quality that would actually result from the proposed works, the questionnaire used Maps 1 and 2 to show what parts of the river would improve and when.

There are also benefits from resuming the use of Tietê water for hydroelectric generation after transfer to a different subbasin. This use had been stopped because the low quality of the Tietê water was damaging the much cleaner reservoir into which the Tietê was diverted. There is a possibility that after completion of the third stage, water from the Tietê will once again be suitable for power generation.

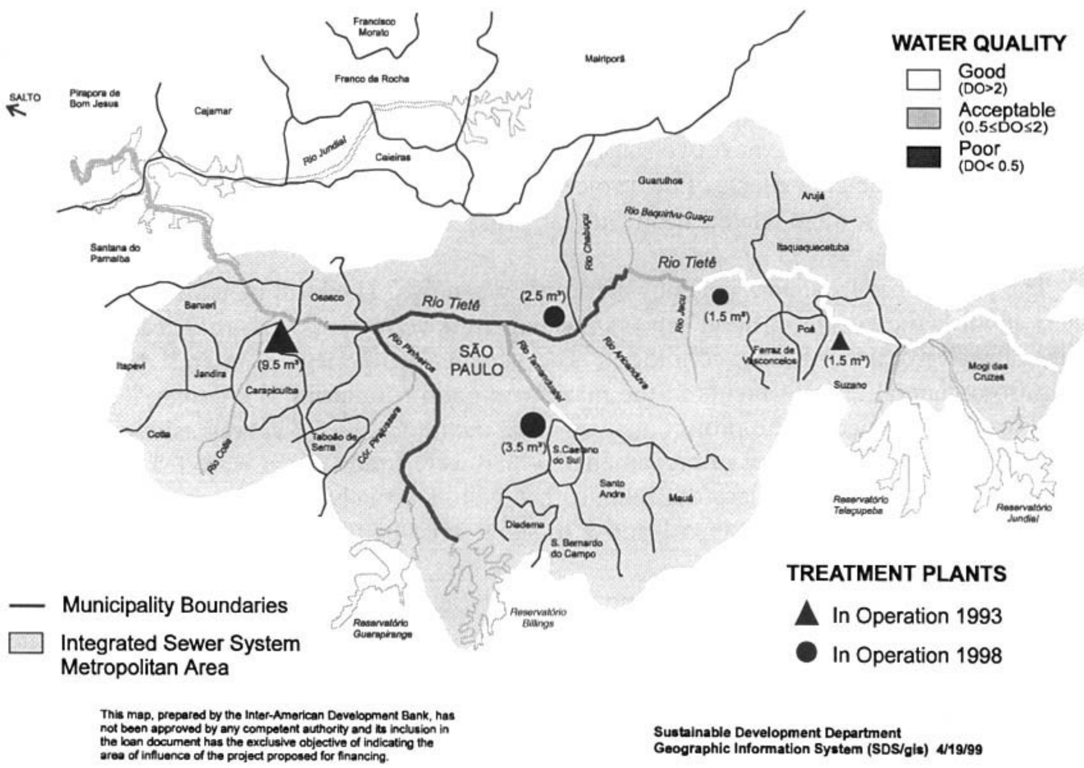
The Survey

The valuation survey indicated that the greatest improvement that could be expected was that the water quality would permit boating and the existence of fish in some segments. It emphasized that it would not be safe to swim in any of the rivers. To take the benefit estimates seriously, regardless of inference method, one would like to be confident that respondents correctly understood the implications of the low-flow assumption and the implications of the DO levels for their lives.[11] But for the purposes of this chapter, any difficulties along those lines are irrelevant, being equally present in all method results.

The survey was conducted with 600 households divided among five subregions of the relevant region. These reflect households both close to and far from the river. This sample was also split another way—into five groups offered different "bid" levels to respond to. The levels, chosen on the basis of focus group comments, were, in reals (R$), 0.50, 2.00, 5.00, 12.00, and 20.00, and were presented as monthly payments that would be made over the 10 years of construction of the second and third stages. A translation of the key valuation question follows directly after the maps.

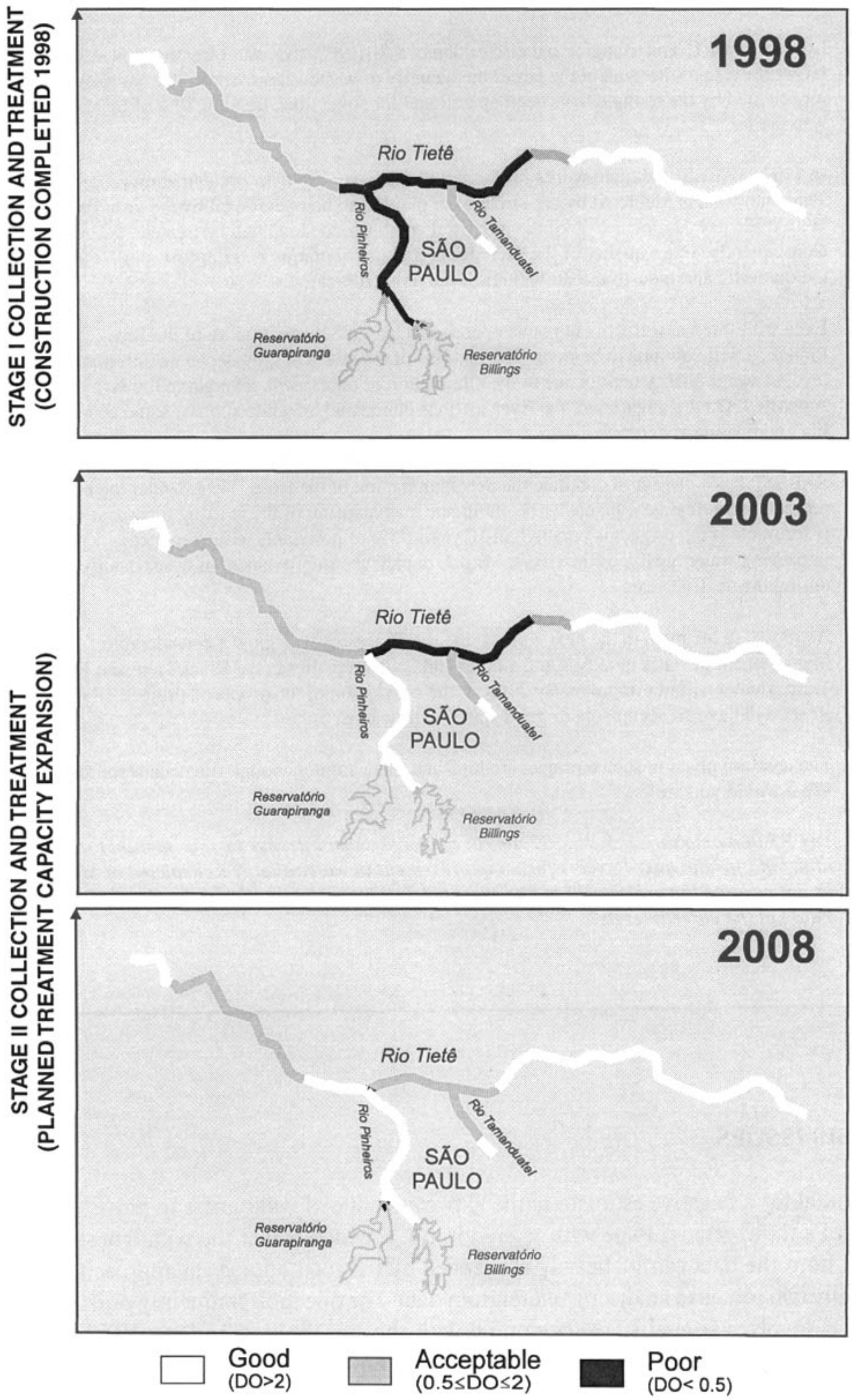
After the survey was completed, it turned out that at the lowest bid level, the proportion of "yes" responses from the far-from-river sample amounted to fewer than 50 percent of the subsample total. This is a warning flag indicating that without restrictions on functional form or the use of limits in calculating the mean, a negative mean WTP will result.

MAP 1. PRESENT AND PROPOSED TREATMENT PLANTS AND 1998 WATER QUALITY IN THE TIETÊ RIVER AND MAIN TRIBUTARIES



[11] The flow assumptions behind the quality predictions are discussed in Chapter 9.

MAP 2. PRESENT AND PREDICTED WATER QUALITY IN THE TIETÊ RIVER AND MAIN TRIBUTARIES



This map, prepared by the Inter-American Development Bank, has not been approved by any competent authority and its inclusion in the loan document has the exclusive objective of indicating the area of influence of the project proposed for financing.

Sustainable Development Department Geographic Information System (SDS/gis) 4/20/99

The Valuation Question

Look at **Map 1**. The triangles and circles depict SABESP's five water treatment plants. The larger the size of the symbol the larger the quantity of wastewater treated. The two plants represented by the triangle have been operational for some time, treating 20% of SPMA wastewater.

In 1993, SABESP initiated works for Stage I of the River Tietê decontamination program. Three new plants (depicted by the circles) are planned to be operational by the year 1998. With these new stations, 40% of the industrial and domestic load will be treated. Consequently, water quality of the Tietê River and its tributaries will improve. Still, 60% of the domestic and industrial load will reach the rivers untreated.

Even with three new treatment plants operational by 1998, water quality of the Rio Pinheiros will continue to be poor. The sections of the rivers in grey depict an acceptable level of water quality mainly due to the elimination of odors; still, no aquatic life is supported. On the other hand, the river sections delineated in white support some aquatic life and boating is permitted.

SABESP has a project to continue the decontamination of the River Tietê. Under the new project, more treatment plants will be built and an expansion of the existing treatment plants is foreseen. If the project is pursued, in 10 years 95% of pollutants will be treated, improving water quality of the rivers. **Map 2** depicts the improvement in water quality during the next 10 years.

As shown in the map, in the next 5 years, the Rio Pinheiros will show a considerable improvement in water quality. On the other hand, water quality in the River Tietê and River Tamanduateí will not improve. By 2008, at the conclusion of the proposed project, all of the rivers will have an acceptable or good water quality level.

The costs involved in such a project are high and there are not enough financial resources. What would you prefer:

Pay R$ (bid amounts: 0.5, 2, 5, 12, or 20) rendered as an increase in your monthly water utility bill for the next 10 years for an improvement in water quality as depicted in Map 2, or not pay and the project will not be executed, leaving water quality of the rivers of São Paulo at the current levels?

ANALYSIS ISSUES

The potential for a negative estimate of the expected value of willingness to pay is only a special example of a more general issue with referendum CV, which is that the willingness-to-pay value extracted from the data can be heavily influenced by the methodological approach taken. There are basically two routes to analyzing referendum data. The one most frequently pursued by project economists involves several steps, beginning with the specification and statistical estimation of one or more probability models of individual choice, employing prior assumptions about the form of the inverse distribution and the covariates belonging in the distribution, which serve to change its location and shape across respondents. This is followed by the evaluation of conditional mean or median formulas derived from the chosen model, which depend on its estimated parameters.

After calculating individual-specific means or medians, averages are taken over the entire sample to produce global central tendency measures. A less frequently traveled but much easier route ignores covariates and does not specify any particular inverse distribution. Instead, it uses all the data in pooled form (i.e., the marginal distribution) to produce nonparametric measures of central tendency. Given its prominence, the next two sections concentrate on the parametric route, followed by a discussion of nonparametric options.

The parametric route can quickly become quite complex, producing a wide array of central tendency estimates. It is not uncommon to find instances where predicted WTP can vary from low to high by a factor of 2, 5, or 10 *with the same data*, depending on the analyst's choice of density function; the specification of the functional form of the indirect utility index and its arguments; and whether a mean, a truncated mean, or a median is used. In short, with referendum data there are a host of possible measures of central tendency of willingness to pay.[12] Gauged by their frequency of use by practitioners, all of them might seem equally legitimate, but this is not a useful criterion. For instance, the untruncated mean extracted from logit estimation of a random utility model has been one of the most popular measures used in IDB project analysis and in the literature more generally, even though it is potentially vulnerable to the problem of negative WTP.

Some Basic Mechanics with Referendum Data and RUM Models

Consider an individual with income Y who must decide whether to answer "yes" or "no" to the following: "Would you vote for a program to increase environmental quality from q^0 to q^1 if it would decrease your annual income by \$B?" Let the indirect utility function be $u(Y,q,X)$ where X is a vector of individual characteristics and the vector of market prices P is omitted since prices are assumed to be constant.

The individual responds "yes" if

$$u(Y - B, q^1, X) - u(Y, q^0, X) + 0 \tag{8-1}$$

and "no" otherwise.

Let $h(\cdot)$ be the observable component of utility. Here, h represents an indirect utility function, which in statistical estimation is often called the "index function" or "utility index," denoted as the summed product of the parameter estimates and the explanatory variables, $X\beta$ (Greene, 1990). The probability of a "yes" response is given by

$$P_1 = P[h(Y - B, q^1, X) + \epsilon_1 > h(Y, q^0, X) + \epsilon_0] \tag{8-2}$$

where ϵ_i (i = 0,1) are independent, identically distributed random variables with zero means and the error term represents influences on utility not observed by the analyst, or just random error in the choice process itself. Assuming the error difference follows a logistic distribution,[13] the

[12] Benefit uncertainty and the influence of analyst choices in econometric estimation is not unique to referendum CV. Similar issues arise with fitting econometric demand or participation models to revealed preference data. Striking examples appear in Smith (1990), Vaughan and Russell (1982), and Ziemer et al. (1980) for recreation demand and in Bachrach and Vaughan (1994) for potable water.

[13] In the literature, only the logit and the probit modeling approaches appear with any frequency, although Hazilla (1999) demonstrates a number of other possibilities. The logit discrete-choice model follows from the assumption that the errors ϵ_0 and ϵ_1 each are independently and identically distributed with Weibull density functions. The difference between any two random variables with (log) Weibull distributions has a logistic distribution Λ, whose cumulative density F(•) equals $e^{(\bullet)} / (1+e^{(\bullet)})$, where the number "e" is the base of the natural system of logarithms and the (•) represents the index function whose parameters are found by estimating the model (Formby et al., 1984).

probability of a "yes" response can be expressed as an estimable random utility (difference) model, or RUM:

$$P_1 = e^{\Delta h} / (1 + e^{\Delta h}) = (1 + e^{-\Delta h})^{-1} \tag{8-3}$$

where $\Delta h = h^1 - h^0$. The linear utility difference index Δh in the "no income effects" RUM is usually specified as a function of the bid level, B, and a set of socioeconomic variables, S, including a constant term but not including income as an argument [i.e., $\Delta h = (\alpha_1 - \alpha_0) + \beta B + \varsigma S$]. This most basic of specifications imposes the assumption of a constant marginal utility of income, which simplifies recovery of an expected value for WTP.

By reversing the sign on the probability difference, we get the expression for the probability of rejecting the offer:

$$P_0 = (1 + e^{\Delta h})^{-1} \tag{8-4}$$

We define the willingness to pay for q^1 by the amount of money that must be taken away from the individual enjoying an improved amenity level, q^1, that leaves them as well off as the initial amenity and income situation:

$$u(Y - WTP, q^1) = u(Y, q^0) \tag{8-5}$$

and

$$h(Y - WTP, q^1) + \epsilon_1 - \epsilon_0 = h(Y, q^0) \tag{8-6}$$

Because of the term $\epsilon_1 - \epsilon_0$, WTP is a random variable. Then the probability of accepting the offer is also the probability that WTP $\geq$ B, and the probability of rejecting the offer is also the probability that WTP < B. This is a cumulative distribution function and can be denoted as F(WTP). As pointed out by Hanemann (1984), the truncated expected value of the random variable (WTP) can be found from the cumulative distribution function as follows:

$$E[WTP] = \int_0^{\infty} [1 - F(WTP)]dWTP \tag{8-7}$$

Here the integration is only over positive values of WTP because if there is utility improvement, WTP theoretically cannot be negative (although it can depend on who you ask and how the question is phrased, as noted in the preceding section and footnote 3). Similarly, the untruncated expected value of the random variable (WTP) can be found from the cumulative density function:

$$E[WTP] = \int_0^{\infty} [1 - F(WTP)]dWTP - \int_{-\infty}^{0} F(WTP)dWTP \tag{8-8}$$

The latter, treating the negative domain of WTP as admissible, will generally be less than or equal to the truncated WTP represented by the first term in expression (8-8) (Johansson et al., 1989) because of the inference described intuitively in the section introducing the problem.

The probability density of the logit, f(•), is just the product of the cumulative density and 1 minus the cumulative, or F(•)[1 − F(•)]. Its closed-form analytical expressions make the logit more tractable mathematically than the alternative assumption of using a normal error distribution, Φ(•) to fit a probit model. Although the logistic distribution is thicker in the tails than the normal, in most cases the logit and probit approaches to binary choice estimation produce similar prediction probabilities and elasticity responses, so the choice between them is largely a matter of convenience (Maddala, 1983; Greene, 1990).

Table 8-1. Formulas for Central Tendencies from the Probability Model

Description	Symbol	Equation
Mean, E(WTP), $-\infty < WTP < \infty$	C+	α/β
Median WTP	C*	α/β
Truncated mean, E(WTP), $0 < WTP < \infty$	C′	$\ln[1 + \exp(\alpha)]/\beta$
Truncated mean, E(WTP), $0 < WTP < B_{max}$ where B_{max} is the maximum bid	C^-	$1/\beta \ln\{[1+\exp(\alpha)]/[1+\exp(\alpha - \beta B_{max})]\}$
Truncated mean, log transform, $E(\exp^{\ln(WTP)})$, $-\infty < \ln WTP < \infty$ (utility difference logit, log of bid, 0, lower limit, no upper limit)	C^+_{ln}	$\exp(-\alpha/\beta)\{(\pi/\beta)/[\sin(\pi/\beta)]\}$ (only applies if $0 < 1/\beta < 1$; otherwise numerical approximation required)
Truncated mean, log transform, $E(\exp^{\ln(WTP)})$, $-\infty < \ln WTP < \ln$ income (utility difference logit, log of bid, 0, lower limit, income upper limit)	C^-_{ln}	No analytic expression—requires numerical approximation
Truncated median, log transform	C^*_{ln}	$\exp(-\alpha/\beta)$

For the logit probability model, Hanemann (1984, 1989) and Ardila (1993) provide the WTP formulas shown in Table 8-1 for the unrestricted expected value, the median, and the truncated expected value that restricts WTP to be positive. The α term in the table is shorthand for an augmented intercept absorbing the estimated constant and the socioeconomic variable influences on Δh [α equals $(\alpha_1 - \alpha_0) + \varsigma S$]. The letter C in the table is shorthand for the central tendency measure of WTP, following the notation of Hanemann (1984, 1989), the original source.[14] For reference, function evaluation formulas are provided in Ardila (1993), Haab and McConnell (1998a), and Hazilla (1999), among others.

Model Assumptions

Given the theoretical underpinnings of the conventional random utility model sketched here, it is necessary to recognize that when the RUM is specified as a logit model with a linear utility difference index, a fundamental contradiction arises because the logit potentially allows predicted willingness to pay to fall between minus and plus infinity, admitting the possibility of negative values. A negative WTP should be ruled out for well-conceived environmental improvements, as should expected payments exceeding actual income.[15] The expedients for guaranteeing satisfaction of one or both of these limits by evaluating the linear utility index model estimated with logit or probit from zero bid to either plus infinity or income (truncated means), or by forcing the estimated density to lie in the positive region by using the logarithm of the bid rather than the untransformed bid in estimation, leave a great deal to be desired. They

[14] The augmented intercept, α, referred to in Table 8-1 is simply the original intercept (for purposes of this note, call it β_0) plus the rest of the $i = 1...n-1$ parameter estimates other than the bid parameter estimate multiplied by the respective sample means of the explanatory variables $-X_i$. The β attached to the bid in Table 8-1 is in this notation equivalent to β_n.

[15] This is strictly true only if the answer supplied reflects an understanding that payments for the good offered are to be taken out of current income without drawing down savings or liquidating other forms of wealth. It is unlikely that low-income survey respondents (who usually dominate CV surveys taken in developing countries) would either have assets to pledge or be willing to pledge them in excess of current income when valuing a non-unique environmental good like water quality improvement. However, the preservation of unique natural assets or irreplaceable historical sites may evoke contributions in excess of income, especially among the upper economic strata, and especially if the question is posed as a one-time payment rather than a series of payments strung out over several years.

are just ad hoc fixes for the conventional random utility model's fundamental specification error of an unrestricted error term.[16]

Although it was originally discussed in the late 1980s (Johansson et al., 1989; Hanemann, 1989), the issue has recently been brought more fully to light by Haab and McConnell (1998a). The latter suggest employing a beta distribution for the density of willingness to pay to consistently hold WTP between 0 and some upper bound, such as income. In an unpublished study, Haab and McConnell (1997b, 1998b, 1999) have proposed an alternative way to achieve a similar restriction by bounded probit (or logit) estimation. Because this method is much simpler to implement than the beta, it is applied to the Tietê project referendum survey data, where it produces reasonable estimates for the median, but curious estimates for the mean.

Central Tendency Measures

A second, related issue is which measure of central tendency to use, once having estimated some probability-of-bid-acceptance model from referendum data. Again, the debate goes back at least 10 years. Hanemann (1989) and Haab and McConnell (1997a, 1997b, 1998b, 1999) argue for the median of individual WTP because in probability models it is less sensitive to distributional misspecification and estimation method. Hanemann (1989) also points out that the median is a more equitable social choice rule for aggregation of willingness to pay across the population for a cost-benefit test, even though it violates the Kaldor-Hicks potential compensation criterion.[17]

Sometimes the discrepancies among the alternative central tendency measures can be large enough to confound a project acceptance or rejection decision using cost-benefit criteria—the project passes the test using some subset of central tendency measures and fails it using others. Put simply, the unbounded expected value measure obtained by using a linear utility index in estimation of a probability model is not generally satisfactory and may understate benefits. But when distributional asymmetry is introduced to correct for this by either truncating the range of expected value function evaluation or by introducing nonlinearity in the utility index, the mean individual WTP extracted from referendum models no longer equals the median and will usually exceed it. In this case using the median as a benefit measure means that project acceptance will not be as strongly influenced by a few extreme observations lying in the tails of the (asymmetric) WTP distribution as it would be using the mean. Experienced analysts know that to get the highest benefits possible and unabashedly seek project acceptance under an NPV or IRR criterion, the mean of an asymmetric distribution can be used, but its median will provide a more cautious, conservative lower bound on project payoff. It seems reasonable to recommend at least taking a look at the latter, or reporting both mean and median.

[16] Creel (1998) sounds a more optimistic note by demonstrating that the marginal expected value of willingness to pay, truncated from below at 0 and from above at a maximum that drives the probability of acceptance to 0, can be consistently estimated from the simplest possible logit model (intercept and price parameters only) providing the bids are spread uniformly between the upper and lower bounds, the upper bound is known a priori, and the acceptance probability is integrated only up to the upper bound in calculating the mean.

[17] The use of a global mean to get an estimate of gross project benefits, which is the focus of this chapter, should not be confused with designing a tariff structure to recover project costs. For rate determination, a global fee based on average WTP would be inappropriate because in aggregate it could induce actual welfare losses among low-income rate payers with a WTP below the mean that offset the net welfare gains accruing to upper-income households whose WTP exceeds the global mean charge. For rate setting, progressive, income-differentiated charges would avoid the equity problem, and calculating them on the basis of referendum WTP data would require a utility index specification that includes income as a regressor.

PARAMETRIC CHOICE MODEL ANALYSIS

To demonstrate the parametric approach, the standard central tendency measures described earlier were obtained by applying a logit choice model to the 600 survey sample observations, coding the dependent variable as 1 if the offer was accepted, and 0 if not. Simple linear and log bid specifications of the utility index were used.[18]

Probability Model Estimation

The independent variables in the statistical logit model included the bid value, the age of the respondent, and a household wealth/social status indicator. A dummy variable was included to distinguish between residents who live close to the river (184 households) and are significantly more affected by its pollution, and those households not residing in close proximity. The estimation results appear in Table 8-2.

All parameter estimates are significant at better than the 5 percent level, and most have signs that are consistent with prior expectations. The positive signs on the "close to river" and "status" variables serve to shift the relation between probability of acceptance and bid to the right, indicating that households close to the river are more likely to be willing to pay than more distant households, as are wealthier households.

Table 8-2. Logit Model Parameter Estimates and Variable Means

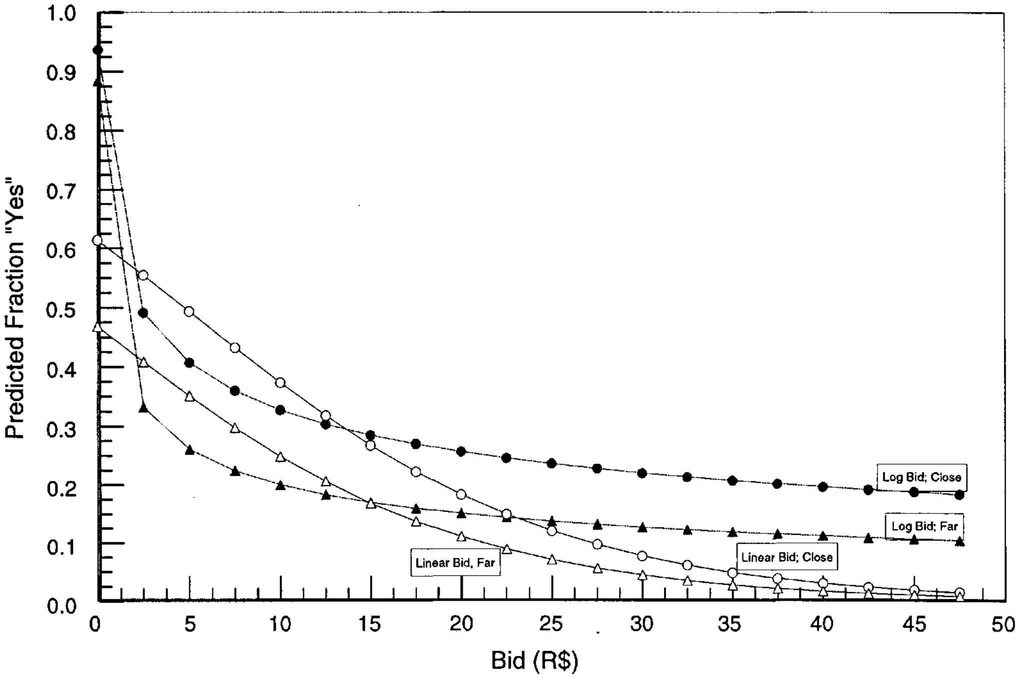
Variable	Linear Bid Model Coefficient[a]	Log Bid Model Coefficient	Means of Variables: Full Sample	Close Sub-sample	Far Sub-sample
Constant	0.7769 *(2.38)*	0.7608 *(2.30)*	—	—	—
Close to river (1 if yes, 0 else)	0.6551 *(3.29)*	0.6629 *(3.33)*	0.3066	1	0
Status (1 if upper, 0 else)	0.8357 *(2.92)*	0.7968 *(2.78)*	0.11	0.1467	0.0938
Age of household head (years)	–0.0221 *(–3.20)*	–0.0227 *(–3.27)*	45.88	49.38	44.34
Bid (R$/household/month)	–0.0978 *(–6.78)*	—	7.9	7.99	7.86
Log of bid (ln R$/household/month	—	–0.4954 *(–6.99)*	1.42	1.43	1.41

Note: For the linear bid index model, the unrestricted log likelihood = –350.00; the restricted log likelihood (intercept only) = –389.08; chi-squared statistic = 78.15; significant at >1% level; and pseudo R^2 = 0.10. For the log bid index model, the unrestricted log likelihood= –350.65, the restricted log likelihood (intercept only) = –389.08; chi-squared statistic = 76.79; significant at >1% level; and pseudo R^2 = 0.098.

[a] Numbers in parentheses are *t* statistics.

[18] Note that the dummy variable specification shifts the function but imposes the restriction that households living near or far from the river share the same regime with respect to the other parameters. The log bid model's expected value could not be evaluated using an analytical formula because its parameters fell outside the limits of the formula's applicability (Hanemann, 1984). Numerical approximation was used to compute the means of the log bid model.

Figure 8-2. Logit Models: Linear and Log Bid



Predictions of the acceptance rates across bid levels for both models, evaluated at their respective subsample means, are displayed in Figure 8-2. Notice that the logarithmic specification confines all of the distribution function to the positive bid quadrant, while the linear specification potentially extends to the left of 0, even though this region is omitted from the figure. The thicker tails of the log bid models suggest arithmetic means that should exceed the arithmetic means of the linear bid models. However, we used geometric means for the log bid models, which explains why they fall below the arithmetic means of the linear bid models in Table 8-3.

Central Tendency Measures from the Parametric Model

Applying the expected value and median formulas produces the WTP estimates in Table 8-3 for the untruncated mean, the mean truncated at 0 but untruncated from above, the truncated mean confined between 0 and the maximum bid (20 reals), and the median.[19]

[19] The unit of currency used throughout this discussion is the Brazilian real (*reais*), denoted as R$. The rate of exchange in March 1998 was 1.14 reals per U.S. dollar. All estimates presented were produced by evaluating the relevant formulas at the means of the explanatory variables rather than calculating individual-specific values and averaging them over the sample to obtain a grand mean. Since the logit is a nonlinear density function, the two routes will not in general produce exactly the same estimate of mean WTP. The two routes would yield the same sample mean if the arguments in the indirect utility difference model were confined to an intercept and the bid, omitting all individual-specific variables involving income and personal characteristics, because then the individual-specific means would all be the same. This reduced model is likely to suffer from biased coefficients caused by omitted variables, so the mean, which is a function of these parameter estimates, will be biased as well to an unknown extent. Creel (1998) shows that the impact of misspecification bias on E(WTP) can be controlled by arraying bids uniformly over the span from 0 to an upper limit (like income), although he does not recommend fitting a simplistic and deliberately misspecified model.

Table 8-3. Parametric Central Tendency Estimates

Central Tendency Measure		Household Willingness to Pay per Month (1998 reals) Close to River	Far from River
Median = untruncated mean, E(WTP), $-\infty < WTP < \infty$	C+	4.74[a]	−1.27
(utility difference logit, linear in bid)	C*	(S.E.= 1.66)	(S.E.= 1.56)
Truncated mean, E(WTP), $0 < WTP < \infty$	C′	9.73	6.16
(utility difference logit, linear in bid)		(S.E.= 1.29)	(S.E.= 0.75)
Truncated mean, E(WTP), $0 < WTP < B_{max}$	C~	7.66	5.03
(utility difference logit, linear in bid)		(S.E.= 0.71)	(S.E.= 0.44)
Truncated mean, log transform, $E(\exp^{\ln(WTP)})$, $-\infty < \ln WTP < \infty$	$C^{+}_{\ln}$	4.66	1.46
Truncated mean, log transform, $E(\exp^{\ln(B)})$, $-\infty < \ln WTP < \ln$ income	$C^{-}_{\ln}$	3.49	1.23
Truncated median, log transform	$C^{*}_{\ln}$	2.34	0.61

Note: The augmented intercepts are 0.4634 for close and −0.1246 for far in the linear model. For both cases, β, the coefficient on the bid variable, is 0.09776 (after multiplying by −1 to make it positive). In the log of bid model, the augmented intercepts are 0.4201 for close and −0.2427 for far. For both cases, β on the natural log of bid is 0.49454 (after multiplying by −1 to make it positive). Geometric means were calculated for the log transform models by taking the antilog of the mean log bid found by numerical approximation. Approximate standard errors are reported in parentheses (S.E.=) in cases where an analytical formula for expected value enabled them to be estimated via a Taylor's series approximation (the "delta" method) using LIMDEP's WALD procedure (see Hazilla, 1999; LIMDEP is a software product of Econometric Software, Inc.).

[a] S.E., standard error of the mean.

These results pose two dilemmas. First, the unrestricted mean WTP for households living far from the river is negative. Second, there is a large disparity between the several alternative truncated means using either a linear or log bid specification.

If project justification (rather than analysis) is the goal, it might be tempting to use the truncated mean that gives the highest benefit and ignore the subtleties. Few would ever detect this sleight of hand. However, an honest project appraisal would admit that things are not quite so simple. Hanemann (1989) indicates that the measure C′ unambiguously overstates the true mean in situations where the augmented intercept is greater than 0 (i.e., when the probability of acceptance at a zero bid is greater than 0.5).

Also, it is inconsistent to use an untruncated distributional assumption for estimation and a truncated rule like C′ for function evaluation. In other words, an inconsistency arises because in estimation of a logit model with a linear utility index difference, the domain of the fitted cumulative density is theoretically allowed to include all the real numbers even though the random variable is known a priori to exclude negative values. Then, in function evaluation, a "correction" like C′ or C^{-} is made ex post by using only that portion of the fitted distribution lying in the positive probability/bid quadrant to compute the expected value integral. For instance, return to the didactic linear probability function $f_1(x)$ in the introduction. Evaluating the integral between the limits R$0 to R$10 produces a positive truncated mean of +R$2.00 instead of the original unrestricted negative mean of −R$2.50 obtained by evaluating the integral from −R$15 to +R$10.[20]

[20] The assumption behind this truncated mean calculation is that the negative domain of the cumulative distribution function (CDF) now piles up at the zero bid level, which is like assigning a 0 to every observation in the sample whose E(WTP) is negative, a technique used by Jorge Ducci in the IDB's very first CV experiment (see Ardila et al., 1998). To do even more violence to the estimation results, in function evaluation it could be assumed that instead of clustering at 0, the negative WTP part of the CDF should be reallocated to the positive part. In our example, the probability of a non-negative WTP at zero bid is only 40 percent, but were it 100 percent, then E(WTP) would be R$5.00 (i.e. R$2.00 ÷ 0.4). This calculation, which cannot be recommended, re-normalizes the positive domain of the estimated inverse CDF to include all the probability mass by rescaling the estimated inverse CDF of function 1 in Figure 8-1 to instead look like function 3 [i.e., $1 - F_1 = (0.40/0.40) - (0.04/0.40)$ (bid)]. The negative E(WTP) problem has been solved by simply ignoring the negative domain of the estimated inverse cumulative distribution, even though that domain was not ruled out in the estimation step.

ANALYSIS OPTIONS

There are two simple yet effective alternatives for obtaining a central tendency estimate of WTP that overcome the necessity of arbitrarily truncating WTP at 0 or some upper bound (or both) in discrete-choice referendum models, taking it as given that the unrestricted mean explained at the beginning is undesirable. The first route is the "distribution-free" nonparametric technique for getting lower bound (McConnell, 1995; Haab and McConnell, 1997a), intermediate (Kriström, 1990), and upper bound (Boman et al., 1999) estimates of the mean. The second involves a reformulation of the probit or logit model that automatically guarantees that median WTP will be greater than a lower bound of 0 but will never be greater than income (Haab and McConnell 1997b, 1999). This route works well for the median, but seems to break down for the mean.

Even for those who prefer the parametric route, it is probably a good idea to calculate one or more nonparametric[21] estimates of the mean and median before getting too deeply involved in parametric estimation approaches for WTP, just to have a benchmark. Since the nonparametric measures work with the aggregate marginal survivor or distribution function, it is crucial that the referendum CV survey be representative, as a random sample of the beneficiary population would be. If some groups or geographical areas are over- or undersampled, separate nonparametric measures have to be computed for each one.[22]

Three Nonparametric Measures of the Mean

The logic behind all three nonparametric estimators is the same. The proportion of "no" answers at each bid level x provides a discrete stepwise approximation to the cumulative distribution function. The mean $E(x)$ of a continuous random variable x with a cumulative distribution function $F(x)$[23] and probability density function $f(x)$, which is the first derivative of $F(x)$ w.r.t. x, is given by

$$E(x) = \int_{-\infty}^{+\infty} x f(x) dx \tag{8-9}$$

The problem is to use a discrete approximation to (8-9) to compute

$$E(x) \approx \Sigma_x x f(x) \tag{8-10}$$

where the range of x is from 0 to some upper limit x_{max} that forces $F(x)$ close to 1.0 because the bid is so high that almost all respondents would be unwilling to pay that amount for the environmental improvement.

The fundamental theorem of the calculus tells us that the area under a curve $f(x)$ between the limits x_1 and x_2 is (a) the sum of a number of infinitesimally small subdivisions in x of length n, (b) the definite integral of $f(x)$ between the limits, or (c) the difference between the integral $F(x)$ evaluated at x_1 and x_2:

$$\lim_{n\to\infty} \sum_{i=1}^{n} f(x_i)\Delta x_i = \int_{x_1}^{x_2} f(x)dx = F(x_2) - F(x_1) \tag{8-11}$$

[21] In the context of this chapter, nonparametric means "distribution-free"; that is, the distribution function of the random variable producing the data need not be specified.

[22] The actual 600-observation WTP sample from our case study was unbalanced because it undersampled households living in districts that are contiguous to the river (62 percent in the sample, 82 percent from the metropolitan area census). Since households living in districts bordering the river are willing to pay significantly more on average for improved water quality than households in noncontiguous districts, the mean from the pooled grand sample would produce a biased estimate of the population's average willingness to pay. For this reason, separate means had to be computed for each group.

[23] To obtain the mean from the survival function, 1– F(x), the same reasoning developed later also applies.

We know the value of F(x) for any bid x from the bid group proportions. Therefore, the bid or x range can be split into intervals and the means from each small interval summed to get the grand mean. That is, the contribution to the overall mean from the approximate mean *within* any bid group interval is the product of some x within the interval [i.e., the lower limit, x_1, the upper limit, x_2, or some value of x in between, which Kriström's (1990) method sets at the group midpoint] times the probability that x lies between x_1 and x_2:

$$E(x) \text{ in interval } x_2 - x_1 = \int_{x_1}^{x_2} xf(x)dx = x[F(x_2) - F(x_1)] \text{ for } (x_1 \le x \le x_2) \quad (8\text{-}12)$$

Generalizing, then, the grand mean is the sum of the interval submeans. Symbolically, using the lower limit of each interval for each x_i and repeatedly applying (8-12):

$$E(x) \approx x_1[F(x_2) - F(x_1)] + x_2[F(x_3) - F(x_2)] + x_3[F(x_4) - F(x_3)] \ldots + x_{n-1}[F(x_n) - F(x_{n-1})] \quad (8\text{-}13)$$

where $x_1 = 0$ and x_n equals a large positive number x_{max} when bounding from above at average income or some assumed fraction of average income. Notice that the (unobserved) value of x_n, which represents the bid driving the probability of acceptance to 0 and the probability of rejection to 1, does not figure in the calculation.

Equation (8-13) is Haab and McConnell's (1997a) lower bound Turnbull mean. The intermediate and upper bound means are obtained by simply redefining the point of evaluation, x, in each interval to ½ of the lower plus upper bounds of the interval, or just the upper bound. While Boman et al. (1999) try to put all three measures on a consistent symbolic footing, there are errors in their notation for the means, and unfortunately their variance formulas are conceptually incorrect.[24] In the next section, all three measures are recast in the Haab and McConnell notational framework, which is conceptually correct.

A Lower Bound: Haab and McConnell's Turnbull Estimate

Consider a stylized contingent valuation question. Respondents are asked: "Would you be willing to pay an amount b_j?" The b_j are indexed $j = 0,1 \ldots M+1$ and $b_j > b_k$ for $j > k$, and $b_0 = 0$. Let p_j be the probability that the respondent's WTP is in the bid interval b_{j-1} to b_j. This can be written as[25]

$$p_j = P(b_{j-1} < w \le b_j) \text{ for } j = 1, \ldots, M + 1 \ldots \quad (8\text{-}14)$$

Alternatively, the cumulative distribution function is written

$$F_j = P(w \le b_j) \text{ for } j = 1, \ldots, M + 1, \text{ where } F_{M+1} = 1 \quad (8\text{-}15)$$

[24] The Boman et al. (1999) variance formulas incorrectly treat the bid, not the cell proportions, as a random variable and are inconsistent with the respective expected-value formulas because they were not derived from them using the fundamental rules pertaining to the variance of a sum of random variables. Instead, an inappropriate textbook formula was forced to stand in. We discovered this discrepancy by comparing the variances of the lower bound means produced using the Haab and McConnell formula and the Boman formula. The variance from the latter was roughly double the former. We then ran 20,000 trials of a Monte Carlo simulation in Crystal Ball, letting each cell proportion at each trial involve a draw from a binomial distribution with parameters defined as the number of observations in each bid cell and the probability of refusal. The empirical results independently confirmed the correctness of the Haab and McConnell variance formula. Our formulas for the variances of the intermediate and upper bound means were derived by extending the Haab and McConnell formula to these situations and were also successfully validated by Monte Carlo simulation. See Vaughan and Rodriguez (2000).

[25] This section is an abridged version of the presentation in McConnell (1995). A complete treatment is available in Haab and McConnell (1997a).

For reasons already discussed, one aims to have b_{M+1} high enough that $F_{M+1} = 1$. That is, b_{M+1} is effectively infinite in the problem setting. Then

$$p_j = F_j + F_{j-1} \tag{8-16}$$

and $F_0 \equiv 0$. The Turnbull can be estimated by treating either the F_j, $j = 1 \rightarrow M$ or p_j, $j = 1 \rightarrow M$ as parameters.

The p's can be estimated quite simply. Let N_j represent the number of "no" responses registered in each bid group j. If $[N_j /(N_j + Y_j)] > [N_{j-1}/(N_{j-1} + Y_{j-1})]$ for all j between 1 and M, then $p_j = [N_j /(N_j + Y_j)] - [N_{j-1} /(N_{j-1} + Y_{j-1})]$. The probability $N_j/(Y_j + N_j)$ represents the proportion of respondents who say "no" to b_j. As such, it is a natural estimator of F_j.[26]

Hence, the estimator of p_j could be written:

$$p_j = F_j - F_{j-1} \text{ where } F_j = (N_j/N_j+Y_j) \tag{8-17}$$

Expected willingness to pay can be written as

$$E(WTP) = \int_0^{\infty} WTPdF(WTP) = \sum_{j=1}^{M+1} \int_{b_{j-1}}^{b_j} WTPdF(WTP) \tag{8-18}$$

Replacing willingness to pay by the lower bound of each interval produces a lower-bound estimate of the expected value of willingness to pay:

$$E(LB_{WTP}) = 0 \times P(0 \le w < b_1) + b_1 P(b_1 \le w < b_2) + \ldots + b_M P(b_M \le w < b_{M+1}) = \sum_{j=1}^{M+1} b_{j-1} p_j \tag{8-19}$$

where $p_{M+1} = 1 + F_M$. The variance of the lower bound mean is

$$V\left(\sum_{j=1}^{M+1} p_j b_{j-1}\right) = \sum_{j=1}^{M+1} b_{j-1}^2 [V(F_j) + V(F_{j-1})] - 2\sum_{j=1}^{M} b_j b_{j-1} V(F_j) \tag{8-20}$$

where the variance of each proportion $V(F_j)$ is equal to $F_j(1- F_j) / (N_j + Y_j)$.

This too can be calculated rather easily from a simple table of proportions of yes's or no's and the total number of respondents in each grouping. The results of applying these formulas are displayed in Annex 8-A, Tables 8A-1 and 8A-2.

Notice in the annex tables that b_M is the highest bid actually offered respondents and is the lower bound of the final interval running from b_M to infinity. In the expected-value formula in Eq. (8-19), b_M is used with no attempt to guess at an appropriate value to apply to the portions of the two subsamples having WTPs greater than R$20 (24 and 11 percent, respectively). This is what produces the lower bound label and distinguishes the Turnbull approach from Kriström's method.

An Intermediate Measure: Kriström's Nonparametric Mean

Kriström's (1990) nonparametric method originally suggested arraying the frequency of affirmative responses in each bid class in monotonically descending order with ascending bids, connecting the points by linear interpolation, and approximating the integral under the result-

[26] The estimate of F_j assumes that the proportion of no responses increases as the bid increases across all bid classes. If not, McConnell and Haab (1997a) show how to join bid groups to achieve monotonically increasing proportions. This was not necessary with the Tietê survey data, except for the first two bid groups in the far-from-river subsample.

ant empirical cumulative density to get the mean. Formalizing and using the distribution function rather than the survivor function, the intermediate mean is

$$E(INT_{WTP}) = \sum_{j=1}^{M+1} bmid_j p_j \tag{8-21}$$

where $bmid_j$ is the midpoint in each bid interval, or ½ $(b_j - b_{j-1})$.

The variance of the intermediate mean is

$$V\left(\sum_{j=1}^{M+1} p_j bmid_j\right) = \sum_{j=1}^{M+1} bmid_j^2 [V(F_j) + V(F_{j-1})] - 2\sum_{j=1}^{M} bmid_j bmid_{j+1} V(F_j) \tag{8-22}$$

Unlike the Turnbull, the bid that drives the probability of acceptance to 0 must be specified by the analyst if the survey does not reveal it, so Kriström's mean depends in part on this arbitrary value of b_{M+1}. To construct the empirical cumulative densities, a conservative upper limit of R$40 for b_{M+1} was assumed, which is approximately 3 percent of average household income (see Ardila et al., 1998). Tables 8A-3 and 8A-4 in Annex 8-A show the calculation steps.

The influence of the final interval between the last posited bid and the assumed bid driving acceptance to 0 is evident from the entries in the penultimate row and last column of the annex tables, just above their average WTP cells. In the close-to-river case, this value accounts for nearly 75 percent of the overall mean value, and in the far-from-river case, 45 percent of the mean value is due to the last interval. If the upper limit driving the acceptance rate to 0 were set to R$30 rather than R$40, the close and far means would fall by about $0.50 and $0.75, respectively, illustrating their sensitivity to this assumption. The nonparametric estimates of location would probably be better if the sample had included more bid intervals spanning a wider bid range.

An Upper Bound: The Paasche Mean of Boman et al.

The upper bound mean involves a straightforward reindexing of the bid in the lower bound formulas and is given by

$$E(UB_{WTP}) = \sum_{j=1}^{M+1} b_j p_j \tag{8-23}$$

Again, like the intermediate mean, an arbitrary value of b_{M+1} must be specified by the analyst. The variance of the upper bound mean is

$$V\left(\sum_{j=1}^{M+1} p_j b_j\right) = \sum_{j=1}^{M+1} b_j^2 [V(F_j) + V(F_{j-1})] - 2\sum_{j=1}^{M} b_j b_{j+1} V(F_j) \tag{8-24}$$

Tables 8A-5 and 8A-6 in Annex 8-A supply the calculation details.

Summary of the Nonparametric Means and their Variances

The three nonparametric measures of average WTP, along with the standard errors of the means, appear in Table 8-4. The upper bound mean is more than double the lower bound.

Table 8-4. Nonparametric Means and Standard Errors of the Means (1998 reals per household per month)

	Close to River		Far from River	
Method	Mean	Standard Error	Mean	Standard Error
Lower bound	6.07	0.80	4.51	0.47
Intermediate	9.42	1.19	7.09	0.66
Upper bound	12.77	1.61	9.67	0.87

The Bounded Probit or Logit of Haab and McConnell

Rather than starting from a RUM model specification as we did in Eqs. (8-1) through (8-6) and then backing out the expression it implies for the median or mean WTP, Haab and McConnell (1997b, 1998b, 1999) start at the other end with an expression for WTP that represents the amount of income the individual is willing to pay, expressed as the product of income and a proportion of income lying between 0 and 1. Somewhat analogous to the conventional RUM, the proportion is estimated as a function of the bid amount and other socioeconomic variables [see Eqs. (8-26) and (8-27)], but the bid-related variable disappears when predicting the median proportion [see Eq. (8-28)].[27]

While this approach makes no claim to being consistent with any theoretical indirect utility function, it solves the practical problem of finding a non-zero WTP that at the same time will not exceed income. Haab and McConnell suppose that WTP lies between 0 and some upper bound, A, such that:

$$\text{median}\,(\text{WTP}_i) = \frac{A_i}{1+e^{-X(i)\beta-\epsilon(i)}} = p(\epsilon_i)A_i \tag{8-25}$$

where $p(\epsilon_i) = 1/(1+e^{-X(i)\beta-\epsilon(i)})$ falls in the (0,1) interval, $\epsilon_i \sim N(0, \sigma^2)$, $X_i\,\beta$ is the inner product of the J covariates ($X_i = X_{i1}\ldots X_{iJ}$), and a vector of coefficients β and A_i is a known constant for individual i, such as income, which is assumed to be a reasonable upper bound on willingness to pay. When A_i is interpreted as income, Eq. (8-25) shows that WTP goes to 0 for very large negative errors or $X_i\beta$ and to income with very large positive errors or $X_i\beta$.

If the i[th] respondent is asked: "Would you pay B_i for a proposed water quality improvement?" the probability of a "no" response is the probability that willingness to pay would be less than B_i. Haab and McConnell write this as

$$P(\text{WTP}_i < B_i) = P\left(\frac{A_i}{1+e^{-X(i)\beta-\epsilon(i)}} < B_i\right) = P\left(\frac{\epsilon_i}{\sigma} < \frac{-\ln[(A_i - B_i)/B_i] - X_i\beta}{\sigma}\right) \tag{8-26}$$

When ϵ_i is distributed N(0,1), the last expression on the right-hand side is the contribution to the likelihood function for a standard probit model, where the probability of a "no" response is modeled with the covariates X_i and $\ln[(A_i - B_i)/B_i]$. Similarly, the probability of a "yes" response becomes

$$P(\text{WTP} < B_i) = P\left(\frac{\epsilon_i}{\sigma} < \frac{\ln[(A_i - B_i)/B_i] - X_i\beta}{\sigma}\right) \tag{8-27}$$

[27] The balance of this section is drawn directly from parts of Haab and McConnell's papers.

Combining Eqs. (8-26) and (8-27) results in a standard probit model with X_i (including a constant) and ln $[(A_i - B_i)/B_i]$ as covariates. The estimated coefficient on X_i will be an estimate of β/σ and the estimated coefficient for $\ln[(A_i - B_i)/B_i]$ will be an estimate of $1/\sigma$. The unscaled βs can be recovered by dividing the estimates of β/σ by the estimated parameter $1/\sigma$ attached to the constructed variable $\ln[(A_i - B_i)/B_i]$. The median WTP for each individual is then obtained by setting ϵ_i in (8-25) to 0 because that is the value that splits the symmetric error distribution in half:

$$\text{median}\ (\text{WTP}_i) = \frac{A_i}{1+e^{-X(i)\beta}} = p(\epsilon_i)A_i \qquad (8\text{-}28)$$

Application of the bounded probit estimator to the Tietê data leads to the median calculations demonstrated in Tables 8-5 and 8-6, using individual household income for the upper limit.[28]

The first two columns of each table refer to estimation of a probit probability model for each of the two subsamples (close, far) where the dependent variable is 0 if the respondent rejected the survey offer (a "no") and 1 if it was accepted (a "yes"). The probit parameter estimates are reported in the third column. In general (Maddala, 1983) they are measurable and estimable only up to a scalar ($1/\sigma$), but the model specification in this particular case provides

Table 8-5. Bounded Probit Median: Close-to-River Subsample
Limit = 100% of Income (Mean Income = 1,524.39 Reals/Household/Month)

Variable	Variable Definition	Original Probit Parameter Estimates (β/σ)	Unscaled Parameter Estimates[a] (β)	Variable Means (X)	Variable Means × Unscaled Parameters ($X\beta$)
Constant		–1.3089	–5.5886*	1	–5.5886
Status	1 if upper; 0 else	0.2715	1.1592	0.147	0.1704
Age	Age of household head, years	–0.0108	–0.0459*	49.38	–2.2677
Btrans[b]	ln [(income – bid)/bid]	0.2342	0.2342*	5.324	NA[c]
Barrio	1 if close to river; 0 else	0.3569	1.5237*	1	1.0000
				$X\beta$ = Column sum	–6.1622
				Fraction of income = $1/[1 + \exp(-X\beta)]$	0.0021
				Median = Share × income	R$3.21

[a]This is the bounding variable whose parameter estimate, $1/\sigma$, is used to unscale the rest of the βs. A * denotes significance at the 1 percent level or better.
[b]Original parameter estimates divided by $1/\sigma$, the parameter attached to Btrans.
[c]NA, not applicable.

[28] At the request of a referee, similar mean and median calculations were done based on estimation of a bounded probit model imposing an upper limit on WTP at 20 percent of household income. Bounds much less than 20 percent could not be imposed using the full sample since in some cases the bid offered was around 18 percent of income, so going below that would involve a negative sign on the variable (A–B)/B, which has no logarithm. For the medians, not much was gained or lost by imposing the limit. The bounded median under a 20 percent of income constraint was R$3.25 for the close-to-river group and R$0.60 for those far away.

Table 8-6. Bounded Probit Median: Far-from-River Subsample
Limit = 100% of Income (Mean Income = 1,148.97 Reals/Household/Month)

Variable	Variable Definition	Original Probit Parameter Estimates[a] (β/σ)	Unscaled Parameter Estimates[b] (β)	Variable Means (X)	Variable Means × Unscaled Parameters ($X\beta$)
Constant		−1.3089*	−5.5886	1	−5.5886
Status	1 if upper; 0 else	0.2715	1.1592	0.094	0.1090
Age	Age of household head, years	−0.0108*	−0.0459	44.34	−2.0363
Btrans[c]	ln [(income − bid)/bid)]	0.2342*	NA[d]	5.324	NA
Barrio	1 if close to river; 0 else	0.3569*	1.5237	0	0.0000
				$X\beta$ = Column sum	−7.5159
				Fraction of income = $1/[1+\exp(-X\beta)]$	0.0005
				Median = Share × income	R$0.63

[a]An * denotes significance at the 1 percent level or better.
[b]Original parameter estimates divided by $1/\sigma$, the parameter attached to Btrans.
[c]This is the bounding variable whose parameter estimate, $1/\sigma$, is used to unscale the rest of the βs.
[d]NA, not applicable.

an independent estimate of that scalar (see the Btrans variable row in the tables) that allows unscaled parameter estimates to be recovered. They are reported in the fourth column. The summed product of the untransformed parameters and the explanatory variables gives an estimate of the average value of the index function $X\beta$. Inserting that index function value in the expression $1/(1+e^{(-X\beta)})$ in Eq. (8-28) produces a median estimate of the fraction of income that would be offered to get the water quality improvements provided by the project (0.0021 for beneficiaries close to the river and 0.0005 for those living farther away). Multiplying the fraction by average income [A in Eq. (8-25)] produces a bounded probit estimate of median WTP. The results of this exercise are reported in the summary table in the next section, where all the WTP estimates are collected.

There is no closed-form analytical solution for the expected value of WTP in the bounded probit or logit formulation, so it must be found by numerical integration (Haab and McConnell, 1997b). The general form of expected willingness to pay is given by

$$E(WTP_i) = \int_{-\infty}^{\infty} WTP(X_i\beta,\epsilon)f(\epsilon)d\epsilon \qquad (8\text{-}29)$$

The integral in (8-29) can be approximated by

$$E(WTP) \approx \sum_{k=1}^{n}(1/\sigma)\phi\left[\frac{\epsilon_k}{\sigma}\right]WTP(X_i\beta,\epsilon_k)(\epsilon_k - \epsilon_{k-1}) \qquad (8\text{-}30)$$

where $\phi(\bullet)$ is the standard normal probability density function, ϵ_k are points on the distributional support of ϵ, and n is large enough so that the approximation is smooth. We used 5,000 points to apply (8-30), approximating $\phi(\bullet)$ by successive differences in the standard normal CDF, $\Phi(\bullet)$, a technique explained in Vaughan et al. (1999).[29]

The bounded probit means (R$140.10 and R$60.46 for households close to and far from the river, respectively) are completely inconsistent with all that has come before, being more than a factor of 10 greater than the highest of all of the preceding estimates, and 45 and 100 times larger than their respective close and far-from-river subsample bounded probit medians.[30]

UNCERTAINTY IN COST-BENEFIT ANALYSIS: A COMPARISON OF RESULTS

Uncertainty about WTP need not translate into uncertainty about a project's net benefits. If the analysis decision is unambiguous because a project's net present value is either consistently positive or consistently negative across the plausible range of possible WTP estimates, then any one of them will suffice. But it is impossible to establish in advance, without actually doing the exercise, that a given project analysis decision will be impervious to variations in the central tendency measure of WTP, thereby making the choice of measure a matter of indifference. On the contrary, uncertainty about benefits is a project-specific issue, and has to be handled on a case-by-case basis. Robustness is likely to be the exception rather than the rule.

The second and third columns in Table 8-7 collect all of the central tendency measures calculated from the Tietê project's WTP survey data. Sorting them from high to low in the near-to-river category confirms rather dramatically the introductory warning that a wide range of plausible estimates can be extracted from referendum data. Even disregarding the bounded probit mean, the highest near-to-river WTP exceeds the lowest by a factor of 5, and the factor is 16 for the far-from-river estimates.

The next three columns of the table show, in deterministic sensitivity fashion, the effect that using each of the alternative WTP measures would have on the economic feasibility of the project at issue, expressed in terms of net present value using a 12 percent interest rate. In general, under optimistic assumptions about execution timing and the earliest possible manifestation of energy benefits[31] (the "best case" scenario), the project decision is not severely affected by the wide variety of per household benefit measures available to appraise it in this particular case. Under the most optimistic of assumptions, the project as a whole (Stages I, II, and III) is not viable except under the bounded probit and upper bound nonparametric mean benefit measures, while the incremental project (Stages II and III) that treats Stage I costs as sunk is economically justified for all but the lowest WTP measure. Said otherwise, if the initial conditions were set optimistically and the problem posed to different analysts, each using a

[29] Haab and McConnell (1997b) provide a quick numerical approximation technique based on a few point estimates of the PDF that dispenses with setting up a large number of points, n, but is less smooth and hence less accurate than (8-30). Although we applied it to test whether our more exact approximation worked, it is not discussed here because the shortcut can be fairly imprecise if the range in the standard normal deviate, ϵ/σ, and the number of evaluation points are not properly chosen.

[30] While this phenomenon might be an artifact of one or more mistakes in setting up the approximation, we were able to replicate all of the examples given in Haab and McConnell (1997b) successfully. In addition, in the example in Table 7 of their paper, the bounded probit mean exceeds the median by a factor of 38, which is similar to what happens with the Tietê data. Imposing a bound on median WTP at 20 percent of income brought the near and far means down to R$50.05 and R$25.54, which are only slightly more plausible. Some doubt about the usefulness of the bounded probit mean (but not the median) in cost-benefit analysis is probably warranted.

[31] These benefits arise, as mentioned earlier, from resuming the use of water from the Tietê for hydroelectric generation after transfer to a different subbasin. This use had been suspended because the low quality of the Tietê water was degrading the reservoir into which the Tietê was diverted.

Table 8-7. Cost-Benefit Comparisons

Central Tendency Measure	WTP per Household per Month (1998 reals): Close	WTP per Household per Month (1998 reals): Far	Net Present Value (million reals), Scenario and Project Stages[a]: Best Case, Stages I, II, and III	Best Case, Stages II and III	Worst Case, Stages II and III
Bounded probit mean, limit =100% of income	140.10	60.46	10,016	10,748	6,647
Boman et al. nonparametric upper bound mean	12.77	9.67	225	957	223
Truncated mean, E(C), $0 < C < \infty$ (utility difference logit, linear in bid, C)	9.73	6.16	−101	631	9
Kriström's intermediate nonparametric mean	9.42	7.09	−83	650	21
Truncated mean, E(C), $0<C<B_{max}$ (utility difference logit, linear in bid)	7.65	5.03	−273	459	−104
Turnbull nonparametric lower bound mean	6.07	4.51	−390	342	−181
Untruncated mean, E(C)= median, $-\infty < C < \infty$ (utility difference logit, linear in bid, C)	4.74	−1.27	−652[b]	81[b]	−354[b]
Truncated mean, log transform, upper limit of infinity (∞ UL) (utility difference logit, log of bid)	4.66	1.46	−598	134	−318
Truncated mean, log transform, income UL (utility difference logit, log of bid)	3.49	1.23	−679	54	−371
Nonparametric median (linear interpolation)	3.33	1.83	−664	68	−361
Bounded probit median, limit =100% of income	3.21	0.63	−720	13	−398
Truncated median, log transform (utility difference logit, log of bid)	2.34	0.61	−734	−41	−433

[a]The best case sets the construction period to 6 years for Stage II and 5 years for Stage III, and has energy benefits online in the first year after Stage II is built. In the analysis of the incremental project (Stages II and III), the operating costs of Stage I are treated as sunk costs. The worst case sets the execution period to 10 years for Stage II and 5 years for Stage III, and assumes no energy benefits come online over a 30-year horizon. In the analysis of the incremental project (Stages II and III), the operating costs of Stage I are treated as avoidable costs.
[b]Far-from-river WTP arbitrarily set to 0 to compute NPV.

different WTP measure, the final conclusion would be nearly unanimous and unaffected by the measure chosen.

The apparent absence of a gray area or zone of ambiguity in the incremental project appraisal decision vanishes when the initial conditions are set less favorably (the "worst case" scenario in the table). While the project as a whole gets even worse and is consistently rejected, the once-favorable decision on the configuration of Stages II and III becomes cloudier if the execution period is extended over 15 years rather than completed in 11, if energy benefits do not materialize at all, and the operating costs of running Stage I are avoidable.[32]

Then, the final column of the table shows that the incremental project only looks economically feasible for four of the measures, all means, and is infeasible (negative NPV) for the other eight, which are mainly truncated parametric means or medians of one sort or another. This result demonstrates another remark made early on about the implications of using the mean rather than the median—the former will generally produce a more favorable outcome with WTP distributions that are skewed to the right.

[32] In the counterfactual "without project" case, if Stage I must be operated whether or not Stages II and III are built, the Stage I operating costs do not figure in the calculation of the net benefits of the incremental project because they will appear in the cost streams of both the "with" and "without" project cases and hence cancel out. In contrast, if the "without project" case postulates abandonment of Stage I as the alternative to the incremental Stage II and III project because the net benefits of Stage I alone are negative, the operating costs of Stage I must appear in the cost stream of the incremental project. The assumption of sunk operating costs for Stage I reflects political and contractual reality, but the assumption of avoidable operating costs is more faithful to the principles of cost-benefit analysis.

However, the median measure only indicates the price at which a project proposal would be accepted by a majority vote under a one-person, one-vote rule. If the project's NPV is negative using the median, that does not necessarily imply it is not worth doing from an aggregate social welfare standpoint. Aggregating up by using the mean to get total benefits is more consistent with standard cost-benefit practice where the "votes" are in monetary units, and outliers with high willingness to pay count in the calculation of the ability of the winners to compensate the losers and still come out ahead (McFadden and Leonard, 1993).

There is no golden rule for resolving ambiguities about project approval brought on by uncertainty about the central tendency measure of willingness to pay except, perhaps, to be aware of this source of uncertainty and to explicitly acknowledge it rather than ignore or conceal it. At a minimum, a search for the existence of a gray area should be conducted. If the project is either economically unjustified using the highest of all legitimate benefit measures or justified using the lowest among the candidates, this is all the better because benefit uncertainty is demonstrably not an issue.

If, on the other hand, the project acceptance decision is reversed somewhere along the spectrum of possible measures, there are several simple decision rules that could be applied, including picking the greatest WTP to push the project ahead and avoid controversy, choosing a measure somewhere in the middle of the range to impart some balance to the final recommendation, or taking a conservative posture by selecting a measure at the low end. A more sophisticated approach would be to fold all of the empirical distributions of the expected-value measures together, either with equal probability of drawing from each (which is akin to picking something in the middle) or with unequal weights reflecting the analyst's judgment or confidence.[33] Finally, one could try to argue for a specific choice on theoretical or econometric grounds, although abstruse technical explanations are unlikely to be popular with decision makers who are ultimately responsible for financing multimillion dollar projects.[34]

In cost-benefit analysis, the conventional investment criterion rests on whether the *expected value* of discounted benefits exceeds the *expected value* of discounted costs. Therefore, to be consistent, all of the variables determining benefits and costs, including WTP, should be expressed as means, not medians or most-likely values.

Suppose that one had to choose a single measure from all of the alternatives. Looking at Table 8-7, it would be prudent to discard the bounded probit and the untruncated RUM means; the former is ridiculously high and the latter is theoretically inconsistent and implausibly negative for households that are far from the river. The choice between means and medians is

[33] Ardila (1993) and Hazilla (1999) show how empirical distributions of mean WTP can be generated, given knowledge of the variances and covariances of the statistically estimated parameter estimates that appear in the E(WTP) formulas.

[34] For example, one reviewer suggested that the analyst could legitimately argue for the E(WTP) from the parametric log bid model if in fact the presence of a fat right tail in the distribution is caused by a high percentage of positive responses at high bid levels. A statistical test of the parametric linear versus log bid models, suggested by both reviewers, would be even more rigorous, but more difficult, since these are non-nested hypotheses (see Ozuna et al., 1993; McFadden, 1994; McFadden and Leonard, 1993, for possible tests). As stated at the onset, the specification issue and associated statistical tests which might help narrow the field of candidate benefit measures in parametric approaches that condition average WTP on covariates is beyond the scope of this chapter and has not been pursued. Our point is that while the issue may be worth further thought, simple nonparametric approaches (like the Turnbull and Kriström methods) obviate the need for such testing because they directly produce an estimate of population E(WTP) from the marginal rather than the conditional distribution of WTP. Moreover, they are at least as precise as conditional mean WTP parametric approaches under most circumstances; do not require any prior assumptions about the distribution of preferences; and yield a consistent measure of E(WTP) that is not susceptible to the misspecification errors that at least potentially can plague the parametric distribution-fitting techniques, rendering their E(WTP) estimates statistically inconsistent (McFadden, 1994).

philosophical; choosing a mean is consistent with standard aggregation practice in cost-benefit analysis. Eschewing the medians and moving on to the remaining means, the Boman et al. (1999) and Kriström (1990) nonparametric means can sometimes be influenced by tail value assumptions and may not be desirable. After this process of elimination, the remaining means are all legitimate contenders. If one had to choose a single measure, a reasonable choice would be the Turnbull expected value because it is a conservative lower bound measure that in this case falls in the middle of the pack.

The next chapter demonstrates how the analyst can subjectively assign probabilities to each of the two plausible limiting nonparametric measures of average WTP (the Turnbull lower bound and the Boman et al. upper bound means) and undertake a risk analysis that simultaneously reflects judgmental and statistical uncertainty. This is our recommended approach, if risk analysis is possible. If only a deterministic cost-benefit analysis can be performed, we suggest the Turnbull nonparametric mean. Users of nonparametric means should be aware, however, that they require representative sampling.

CONCLUDING OBSERVATIONS

No mysterious code of silence has been broken here by revealing the uncertainty inherent in referendum CV estimates of WTP—the academic literature, particularly of late, has implicitly covered the issue in some depth and many experienced project analysts are probably well aware of it. Yet that literature is at times inaccessible and hard to understand, and no synthesis exists that emphasizes the implications of using these several CV measures in appraising investment projects. Therefore, the main purpose of this chapter has been to explain, using examples, the nature of the problem and the solutions available to everyday practitioners. That having been done, what practical recommendations can be made? The most obvious would seem to be the following:

- Do an open-ended survey at the pretest stage to get an idea of the bid range that should be used in the full-blown referendum survey and produce a tentative benchmark WTP from the open-ended data for comparison.
- Design the referendum survey to cover the bid range so nonparametric means and medians can be computed reliably. Make sure the sample is representative of the population and does not involve oversampling of selected socioeconomic groups or geographical areas. Monitor the survey results, perhaps executing it in phases, so adjustments in the bid range can be made if coverage deficiencies become apparent.
- Run a battery of central tendency measures, definitely including a nonparametric measure and perhaps including the bounded probit median, rather than arbitrarily picking one or two of the more familiar parametric measures.
- Explore the influence of the several WTP measures on the cost-benefit analysis outcome, looking for the existence or absence of an uncertain gray area.
- Reach a reasoned final recommendation about project feasibility based on the preceding steps, and be able to explain it.

In sum, before becoming completely and inextricably caught up in the fine points of econometric estimation of parametric choice models, it is worth pausing to consider the options available and the point of the exercise. If the primary goal is to explain and understand respondent behavior, verify whether CV survey responses are consistent with economic theory, or estimate WTP for a population other than the one sampled, parametric choice models must be estimated. If all one needs is a benefit measure for cost-benefit analysis, on the other hand, nonparametric estimates of mean WTP have an advantage, which is why we recommend them. McFadden and Leonard summarize the advantages and disadvantages of each route:

> *[D]irect approaches to valuing a resource do not require any parameterization of preferences or the distribution of tastes, and do not require that WTP be related to any consumer characteristics such as age or income, because the final impact of these variations is taken care of by random sampling from the population. . . . The advantages of parametric methods are that they make it relatively easy to impose preference axioms, pool data across experiments, and extrapolate the calculations of value to different populations than the sampled population. Their primary limitation is that, if the parameterization is not flexible enough to describe behavior, then the misspecification will usually cause the mean WTP calculated from the estimated model to be a biased estimate of true WTP.* (McFadden and Leonard, 1993, pp. 167–168)

Annex 8-A

Nonparametric Lower Bound, Intermediate, and Upper Bound Mean WTP

Table 8A-1. Close-to-River Subsample: Turnbull Lower Bound Mean, Variance, and Standard Error of the Mean

Calculation of Mean

				Cumulative Distribution	Probability Density	
Bid Group j	**Bid b_j**	**Total No. of "no" Answers[a]**	**Total No. of Obs.**	**CDF = F_j = $NO_j/TOTAL_j$**	**PDF = P_j = $F_j - F_{j-1}$**	**Estimate of E(WTP): $b_{j-1} \times p_j$**
j=0	0	NA	None	0	NA	NA
j=1	0.50	10	37	0.270	0.270	0.00
j=2	2.00	13	33	0.394	0.124	0.06
j=3	5.00	26	41	0.634	0.240	0.48
j=4	12.00	26	35	0.743	0.109	0.54
j=M=5	20.00	29	38	0.763	0.020	0.24
j=M+1=6	>20	NA	None	1.000	0.237	4.74
	Col. Sum:	104	184		1.000	6.07
					Average WTP:	R$6.07

Calculation of Variance

$V(F_j)$	b_{j-1}^2	$b_j \times b_{j-1}$	$V(F_j) + V(F_{j-1})$	Term 1: $b_{j-1}^2[V(F_j) + V(F_{j-1})]$	Term 2: $-2b_j \times b_{j-1} \times V(F_j)$
0.000000	...	...	...	...	...
0.005330	0.00	0.00	0.005330	0.000000	0.000000
0.007235	0.25	1.00	0.012565	0.003141	–0.014470
0.005659	4.00	10.00	0.012894	0.051574	–0.113173
0.005458	25.00	60.00	0.011116	0.277909	–0.654927
0.004757	144.00	240.00	0.010214	1.470852	–2.283132
0.000000	400.00	...	0.004757	1.902610	0.000000
			Column Sum	3.706087	–3.065702
				Variance of the Mean=Term 1+Term 2	0.640385
				Standard Error of the Mean	0.800241

[a]NA, not applicable.

Table 8A-2. Far-from-River Subsample: Turnbull Lower Bound Mean, Variance, and Standard Error of the Mean

Calculation of Mean

				Cumulative Distribution	Probability Density	
Bid Group j	Bid b_j	Total No. of "no" Answers[a]	Total No. of Obs.	CDF = F_j = $NO_j/TOTAL_j$	PDF = P_j = $F_j - F_{j-1}$	Estimate of E(WTP): $b_{j-1} \times p_j$
j=0	0	NA	None	0	NA	NA
j=1	2.00	93	170	0.547	0.547	0.00
j=2	5.00	57	79	0.722	0.174	0.35
j=3	12.00	62	85	0.729	0.008	0.04
j=M=4	20.00	73	82	0.890	0.161	1.93
j=M+1=5	>20	NA	None	1.000	0.110	2.20
	Col. Sum:	285	416		1.000	4.51
					Average WTP:	R$4.51

Calculation of Variance

$V(F_j)$	b_{j-1}^2	$b_j \times b_{j-1}$	$V(F_j) + V(F_{j-1})$	Term 1: $b_{j-1}^2[V(F_j) + V(F_{j-1})]$	Term 2: $-2b_j \times b_{j-1} \times V(F_j)$
0.000000	...	...	...	...	...
0.001458	0.00	0.00	0.001458	0.000000	0.000000
0.002543	0.25	1.00	0.004001	0.016004	−0.050868
0.002322	4.00	10.00	0.004865	0.121635	−0.278640
0.001192	144.00	240.00	0.003514	0.505956	−0.571959
0.000000	400.00	...	0.001192	0.476633	0.000000
			Column Sum	1.1202280	−0.901468
				Variance of the Mean=Term 1+Term 2	0.218760
				Standard Error of the Mean	0.467718

[a] NA, not applicable.

Table 8A-3. Close-to-River Subsample: Kriström Intermediate Mean, Variance, and Standard Error of the Mean

Calculation of Mean

Bid Group j	Bid Midpoint b_{midj}	Total No. of "no" Answers[a]	Total No. of Obs.	Cumulative Distribution: CDF = F_j = $NO_j/TOTAL_j$	Probability Density: PDF = P_j = $F_j - F_{j-1}$	Estimate of E(WTP): $b_{midj} \times p_j$
j=0	0	NA	None	0	NA	NA
j=1	0.25	10	37	0.270	0.270	0.07
j=2	1.25	13	33	0.394	0.124	0.15
j=3	3.50	26	41	0.634	0.240	0.84
j=4	8.50	26	35	0.743	0.109	0.92
j=M=5	16.00	29	38	0.763	0.020	0.32
j=M+1=6	30.00	NA	None	1.000	0.237	7.11
	Col. Sum:	104	184		1.000	9.42
					Average WTP:	R$9.42

Calculation of Variance

$V(F_j)$	b_{midj}^2	$b_{midj} \times b_{midj+1}$	$V(F_j) + V(F_{j-1})$	Term 1: $b_{midj}^2[V(F_j) + V(F_{j-1})]$	Term 2: $-2b_{midj} \times b_{midj+1} \times V(F_j)$
0.000000	...	...	...	...	...
0.005330	0.063	0.31	0.005330	0.000333	−0.003331
0.007235	1.563	4.38	0.012565	0.019633	−0.063305
0.005659	12.25	29.75	0.012894	0.157946	−0.336689
0.005458	72.25	136.00	0.011116	0.803158	−1.484501
0.004757	256.00	480.00	0.010214	2.614848	−4.566263
0.000000	900.00	...	0.004757	4.280872	0.000000
			Column Sum	7.876790	−6.454091
			Variance of the Mean=Term 1+Term 2		1.422699
			Standard Error of the Mean		1.192769

[a] NA, not applicable.

Table 8A-4. Far-from-River Subsample: Kriström Intermediate Mean, Variance, and Standard Error of the Mean

Calculation of Mean

				Cumulative Distribution	Probability Density	
Bid Group j	Bid Midpoint b_{midj}	Total No. of "no" Answers[a]	Total No. of Obs.	CDF = F_j = $NO_j/TOTAL_j$	PDF = P_j = $F_j - F_{j-1}$	Estimate of E(WTP): $b_{midj} \times p_j$
j=0	0	NA	None	0	NA	NA
j=1	1.00	93	170	0.547	0.547	0.55
j=2	3.50	57	79	0.722	0.174	0.61
j=3	8.50	62	85	0.729	0.008	0.07
j=M=4	16.00	73	82	0.890	0.161	2.57
j=M+1=5	30.00	NA	None	1.000	0.110	3.29
	Col. Sum:	285	416		1.000	7.09
					Average WTP:	R$7.09

Calculation of Variance

$V(F_j)$	b_{midj}^2	$b_{midj} \times b_{midj+1}$	$V(F_j) + V(F_{j-1})$	Term 1: $b_{midj}^2[V(F_j) + V(F_{j-1})]$	Term 2: $-2b_{midj} \times b_{midj+1} \times V(F_j)$
0.000000	...	...	...	...	...
0.001458	1.00	3.50	0.001458	0.001458	–0.010202
0.002543	12.25	29.75	0.004001	0.049012	–0.151332
0.002322	72.25	136.00	0.004865	0.351526	–0.631584
0.001192	256.00	480.00	0.003514	0.899478	–1.143918
0.000000	900.00	...	0.001192	1.072423	0.000000
			Column Sum	2.373897	–1.937038
			Variance of the Mean=Term 1+Term 2		0.436858
			Standard Error of the Mean		0.660952

[a] NA, not applicable.

Table 8A-5. Close-to-River Subsample: Boman et al. Upper Bound Mean, Variance, and Standard Error of the Mean

Calculation of Mean

Bid Group j	Bid b_j	Total No. of "no" Answers[a]	Total No. of Obs.	Cumulative Distribution: CDF = F_j = $NO_j/TOTAL_j$	Probability Density: PDF = P_j = $F_j - F_{j-1}$	Estimate of E(WTP): $b_j \times p_j$
j=0	0	NA	None	0	NA	NA
j=1	0.50	10	37	0.270	0.270	0.14
j=2	2.00	13	33	0.394	0.124	0.25
j=3	5.00	26	41	0.634	0.240	1.20
j=4	12.00	26	35	0.743	0.109	1.30
j=M=5	20.00	29	38	0.763	0.020	0.41
j=M+1=6	40.00	NA	None	1.000	0.237	9.47
	Col. Sum:	104	184		1.000	12.77
					Average WTP:	R$12.77

Calculation of Variance

$V(F_j)$	b_j^2	$b_j \times b_{j+1}$	$V(F_j) + V(F_{j-1})$	Term 1: $b_j^2[V(F_j) + V(F_{j-1})]$	Term 2: $-2b_j \times b_{j+1} \times V(F_j)$
0.000000	...	...	...	...	...
0.005330	0.25	1.00	0.005330	0.001333	−0.010660
0.007235	4.00	10.00	0.012565	0.050261	−0.144697
0.005659	25.00	60.00	0.012894	0.322338	−0.679038
0.005458	144.00	240.00	0.011116	1.600759	−2.619708
0.004757	400.00	800.00	0.010214	4.085700	−7.610438
0.000000	1600.00	...	0.004757	7.610439	0.000000
			Column Sum	13.670830	−11.064544
			Variance of the Mean=Term 1+Term 2		2.606285
			Standard Error of the Mean		1.614399

[a] NA, not applicable.

Table 8A-6. Far-from-River Subsample: Boman et al. Upper Bound Mean, Variance, and Standard Error of the Mean

Calculation of Mean

Bid Group j	Bid b_j	Total No. of "no" Answers[a]	Total No. of Obs.	Cumulative Distribution CDF = F_j = $NO_j/TOTAL_j$	Probability Density PDF = P_j = $F_j - F_{j-1}$	Estimate of E(WTP): $b_j \times p_j$
j=0	0	NA	None	0	NA	NA
j=1	2.00	93	170	0.547	0.547	1.09
j=2	5.00	57	79	0.722	0.174	0.87
j=3	12.00	62	85	0.729	0.008	0.09
j=M=4	20.00	73	82	0.890	0.161	3.22
j=M+1=5	40.00	NA	None	1.000	0.110	4.39
	Col. Sum:	285	416		1.000	9.67
					Average WTP:	R$9.67

Calculation of Variance

$V(F_j)$	b_j^2	$b_j \times b_{j+1}$	$V(F_j) + V(F_{j-1})$	Term 1: $b_j^2[V(F_j) + V(F_{j-1})]$	Term 2: $-2b_j \times b_{j+1} \times V(F_j)$
0.000000	...	...	...	...	...
0.001458	4.00	10.00	0.001458	0.005830	−0.029151
0.002543	25.00	60.00	0.004001	0.100024	−0.305209
0.002322	144.00	240.00	0.004865	0.700619	−1.114561
0.001192	400.00	800.00	0.003514	1.405434	−1.906530
0.000000	1600.00	...	0.001192	1.906531	0.000000
			Column Sum	4.118438	−3.355452
				Variance of the Mean=Term 1+Term 2	0.762986
				Standard Error of the Mean	0.873491

[a] NA, not applicable.

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Chapter 9

Reflecting Uncertainty in the Economic Appraisal: The Case for Project Risk Analysis

Chapter 2 reviewed the state of the art in IDB practice and found that a majority of water quality project analyses undertaken since 1989 have used the contingent valuation method to estimate environmental benefits. However, in Chapter 8 we discovered that there are nearly a score of alternative ways to measure the mean or median of willingness to pay from referendum CV surveys, and that past IDB practice had been using an approach that systematically understated project benefits. These chapters emphasized the uncertainty that is inherent in project analyses that are based on CV benefit estimates. This chapter explains exactly how to take that uncertainty fully into account in economic cost-benefit analysis by using a Monte Carlo risk approach.

THE RATIONALE FOR PROBABILISTIC RISK ANALYSIS

In the terminology of decision analysis, a decision based on a cost-benefit rule is a two-action problem with infinite states of nature (Pfaffenberger and Patterson, 1987). An investment proposal can be either accepted if it is expected to yield a positive discounted net cash flow above the breakeven point of net present value equal to 0, or rejected if it does not. Because the many influences on NPV are random variables, so is NPV. Therefore, at least conceptually, there are an infinite number of possible net discounted cash flow outcomes from a prospective investment, each with its own probability of occurrence.

The risk-neutral investment decision rule (Brent, 1996; Harberger, 1996) is to proceed with a capital investment project if the *expected value* of its discounted stream of net benefits, E(NPV), is non-negative. If the expectation of discounted net benefits is negative, the project proposal is economically infeasible. The conceptually correct way to obtain E(NPV) is not through a deterministic analysis that inconsistently combines extreme value guesses for some variables driving benefits or costs with an assortment of empirically based measures of central tendency (i.e., a mixture of means, medians, and "most likely" modes) for others. Rather, in principle, the "best" estimate of NPV (in the sense of being unbiased, not "most likely"), is obtained by weighting each possible value of every variable that determines NPV by the probability of its occurrence (Squire and van der Tak, 1975). That is exactly what probabilistic risk analysis based on Monte Carlo simulation does.[1]

[1] Standard deterministic cost-benefit analysis uses an expected value criterion for the outcome, E(NPV), so strictly speaking the variables influencing NPV should be specified as averages and not medians or modes. However, because this approach does not force the analyst to think about probability distributions, in practice it is usually impossible to determine whether favorable tail values or central tendency measures were employed for the key variables, let alone which kind of central tendency measure was used. The use of favorable tail values disguised *as if* they were means is a trick that responsible IDB project economists recognize and strongly disapprove of, according to a survey performed by two of the authors in collaboration with other IDB staff (see Vaughan et al., 1993).

In addition to the mean of NPV, if we want to talk about risk, we also need to ask about spread or dispersion. Risk can be viewed rigorously as either the probability of occurrence of an undesirable outcome (Evans and Olson, 1998) or the variance of the expected value of NPV (Squire and van der Tak, 1975). Probability is the best-known formalism for quantifying uncertainty (Morgan and Henrion, 1990), and probabilistic risk analysis gets a firm handle on it (Sang, 1988). Alternatively, risk can be defined more loosely by asking whether an undesirable outcome (negative NPV) might occur over a range of circumstances that are deemed "plausible" a priori. Although this notion of risk is computationally simple to assess using sensitivity analysis, it is not probabilistic.

Sensitivity versus Risk Analysis

Sensitivity analysis only gives a rough indication of how the net present value or internal rate of return of a project might react to changes in the most crucial input variables. In its most familiar form, the technique deterministically modifies the values of selected cost and benefit input variables one at a time by making up and down adjustments (e.g., ±10 percent, ±20 percent) and then calculating the effect on the outcome indicator (NPV or IRR). Unfortunately, sensitivity analysis does not provide any information on the statistical dispersion of the NPVs (Sang, 1988). Even a sensitivity analysis that improves on arbitrary percentage increases and decreases in input values by postulating "what-if" scenarios that employ, for example, three sets of values for each input variable (minimum, best guess, and maximum) is not fully probabilistic because the collection of variables can in fact take any number of values. Although sensitivity analysis incorporates some of the uncertainty in the input variables, it is not adequate for very sensitive or complex projects, such as those in environmental quality improvement or natural resources management. A great deal of uncertainty about outcomes is likely to be present in these projects because of the necessarily imprecise way we measure benefits (Vaughan et al.,1999, 2000b) and the randomness of the natural world setting on and in which the project operates.

Unlike sensitivity analysis, in probabilistic risk analysis, a probability distribution is assigned to each input variable that has a relevant influence on the economics of the project. Then, through repeated simulation, an empirical approximation to the probability distribution of the outcome of interest (e.g., NPV) is obtained. The probability distributions assigned to each input variable can be subjective or objective. Subjective probability distributions are used when empirical information is unavailable and the analyst must form a judgment or draw on expert opinion. Objective probability distributions are often preferable to subjective ones because they replace speculative guesswork with variability that has been observed in the real world, but of course, they depend on the quality and availability of empirical data.[2]

In the probabilistic context of risk analysis, following an expected value decision rule has a quantifiable cost called the "cost of uncertainty." It can be extracted from the empirical probability distribution for NPV that the Monte Carlo simulation produces. The cost of uncertainty is the expected opportunity loss of making the decision determined by the decision rule. That is, if the expectation E(NPV) taken over the entire NPV distribution is non-negative, the investment will be made. But if some portion of the NPV distribution falls below 0, actual losses in specific instances are still possible. The cost of uncertainty can therefore be measured as the mean of that portion of the NPV distribution truncated from above at 0 (the average loss, given that a loss might indeed occur), multiplied by the probability of a negative NPV occurring. If

[2] On assigning probability distributions to random variables in risk analysis, Vose (1996, p. 51) observes that "The precision of a risk analysis relies very heavily on the appropriate use of probability distributions to accurately represent the uncertainties of the problem." Empirically obtained probability distributions, perhaps tempered by some judgment, may provide a more accurate picture than purely subjective distributions in many situations.

the project is not undertaken because the expected value of NPV is negative, the investment will not be made, thus forgoing any possibility of positive net returns. Symmetrically, the prospective loss in this situation is the mean of that portion of the NPV distribution truncated from below at 0 (the average net gain forgone, given that a net gain might occur), multiplied by the probability of a positive NPV occurring.[3]

A number of authors support the use of risk analysis in project appraisal. Clarke and Low (1993, p. 142) conclude that "while sensitivity testing is useful to highlight the most critical parameters, risk analysis provides an estimate of project worth variability that is both more realistic and easier to interpret than that from the more standard sensitivity analysis." Jenkins (1997, p. 41) in his review of the World Bank's economic analysis methodology recommends that "when possible, a Monte Carlo analysis should be undertaken to assess the key variables affecting the riskiness of the project and to assess the probabilities of the project's potential for success or failure." Squire and van der Tak (1975, p. 46) recommend that "Risk analysis should be considered for larger more complex projects or projects having exceptional risks that cannot be adequately appreciated by means of simple sensitivity analysis. The advantages of further study of certain project features or variables and of a more flexible design to cope better with future uncertainties should be part of the normal process of project preparation and appraisal."

To sum up, risk analysis is not new. Rather, it is a venerable decision analysis technique that is enjoying a latter-day revival in both government and the private sector. Its initial promise was hampered by computational limitations, not conceptual flaws (Pouliquen, 1970; Reutlinger, 1970). With modern software packages like Crystal Ball (Decisioneering, Inc., 1998; Hulett, 1999) and the power of the personal computer, a simulation of tens of thousands of trials can be run in a few minutes, while in 1970 a small simulation of 1,000 trials took from 3 to 5 days on a mainframe (Pouliquen, 1970). In short, a strong case can no longer be made against systematically quantifying project risk.[4] Why, then, is the approach not standard practice at institutions like the IDB?

One can only speculate, but Johnson (1985, p. 19) clearly states the source of the reluctance of multilateral development banks to use risk analysis: "The adoption of any technique that slows operational performance is bound to attract the criticism of both conservative line staff and output-oriented managers." Nowadays, a lack of knowledge and understanding of the method in some echelons of management may still hinder its application. It may also remain true that as Johnson suggests, output-oriented managers in lending institutions may be reluctant to learn about the ex ante probability of failure of a proposed project investment if that information might dampen a prospective borrower's enthusiasm, preferring to focus instead on a single (positive) point estimate of NPV.

Irrespective of the viewpoint of the lender, borrowers in developing countries should be concerned about the consequences of having too many risky projects in the public sector portfolio, especially if that portfolio is dominated by a small number of megaprojects whose risks and impacts cannot be pooled or spread widely across space or population (Squire and van der Tak, 1975). They, if no one else, should insist on a proper comparison of all of the risks associated with proposed projects in comparison with other, potentially less risky alternatives.

[3] Only if the probability distribution of NPV lies entirely in either the positive or negative domains will there be no cost of uncertainty because one literally can't go wrong; either an unattractive investment is unambiguously bad over the entire range of NPV outcomes or no losses are possible because there is zero probability that NPV will fall below 0.

[4] Admittedly, the setup of a spreadsheet cost-benefit model can be more complex and time consuming if it has to be structured to work under a risk analysis add-in like @Risk or Crystal Ball. In the case study discussed subsequently, to accommodate delays in project execution, phasing delays between project stages, and alternative energy benefit scenarios, the spreadsheet design was definitely made more intricate than it otherwise would have been for a deterministic analysis, adding more than 1 person-month of effort.

The risk neutrality rule used by multilateral development banks (Harberger, 1996) to approve projects should perhaps not be imposed on borrowers by ignoring information. Spending a few more months in preliminary analysis to set up and carry out a risk analysis may pay off in the long run in terms of the quality of the investment decisions that developing countries make.

The Specific Issue of Uncertain Benefits

Wattage et al. (2000) argue persuasively that contingent valuation is the most all-encompassing way to measure the (largely nonmarket) benefits of water quality improvement investments in order to apply the cost-benefit test. The Inter-American Development Bank has been a leader among multilateral development institutions in implementing this approach (Ardila et al., 1998; Travers, 1999), having performed over a score of water quality project cost-benefit appraisals since 1989 that have been based on CV benefit estimates. However, with a few notable exceptions (Ardila, 1993; Ardila et al., 1998), these analyses have not attempted to quantify the uncertainty surrounding the benefits that have been obtained via CV, and how it transmits into uncertainty about NPV.

There are a number of plausible techniques for extracting benefit estimates from referendum contingent valuation data.[5] The resulting measures of central tendency span a wide range, and economic theory provides no criterion that can be used to unambiguously choose a correct (or best) estimate (Vaughan et al., 1999). All we have are plausible bounds (Boman et al., 1999). Even if there were a way to find a unique point estimate of benefit, uncertainties would not disappear because, being random variables, each of the estimates has a statistical confidence interval associated with it. This phenomenon of statistical uncertainty was first recognized explicitly in IDB project applications by Ardila (1993). In short, subjective and statistical uncertainty always exists about the magnitude of nonmarket benefits, and usually exists for market-based benefit estimates as well. This uncertainty may or may not affect the decision to undertake a project.

If a project is economically feasible (or infeasible) irrespective of the benefit estimate used, the decision maker can be confident in undertaking (or rejecting) it. For example, if a project's NPV is positive using one of the low estimates of average WTP, it will pass the test with an upper bound estimate as well, and can therefore be accepted with a reasonable degree of confidence. Likewise, if a project fails the cost-benefit test using one of the high estimates of mean WTP, it probably is not a good investment, signaling that changes in designs and objectives are needed before any commitment can be made. These are clear and fairly safe choices, but unfortunately many prospective investment projects fall in the gray area where the project is feasible using some benefit estimates but not others, as when it fails the positive NPV test using a lower bound benefit estimate but passes the test when a high benefit estimate is used instead. In these common situations a decision maker should be told: (a) the probability that the project is feasible, (b) the expected losses if it turns out badly, and (c) the value of obtaining additional information to clarify the decision.

This chapter examines the issue of uncertainty in the context of the case study for the Tietê River cleanup project introduced in Chapter 8 and Vaughan et al. (1999). It builds upon the original economic analysis performed as part of the IDB's formal loan approval process, and uses the Monte Carlo technique to simulate the economic results of the project and analyze its risk.[6]

[5] The direct-revelation approach to CV question design is not discussed here. A nontrivial discrepancy between the expected values of WTP produced via referendum or direct revelation can exist for the same problem setting (e.g., Wattage et al., 2000).

[6] For didactic purposes, the analysis described here explores a broader range of assumptions about costs and benefits than the original analysis did, and differs from it in several ways that will be touched upon later. Our conclusions in no way reflect the official position of the Bank or the borrower.

The chapter analyzes the uncertainty both with respect to measures of willingness to pay developed in Vaughan et al. (1999, 2000a, 2000b) and Vaughan and Rodriguez (2000) and the uncertainty with respect to other estimates used in the calculation of feasibility (e.g., project costs, execution period, shadow prices, and future political decisions that affect use of project outputs). It concludes that the analysis of risk provides the decision maker with important information about the possible consequences of his or her decision and the degree of confidence one can have in making it. It also concludes that the conventional sensitivity analysis commonly provided in many cost-benefit analyses is a poor substitute for risk analysis and can sometimes be misleading.

This chapter begins by presenting the problem addressed in the case study. It then describes the multistage project designed to resolve the problem. At the time of analysis of the second stage of the project, the first stage had already been completed and for that reason the analysis concentrates mainly on the economic feasibility of the works yet to be built (an incremental project). The uncertainties about the factors that potentially determine the project's outcome and how they are captured in the risk analysis are explained. Finally, the results and conclusions of the risk analysis are contrasted with those produced by a conventional sensitivity analysis, and conclusions are drawn.

CASE STUDY: THE TIETÊ RIVER POLLUTION PROBLEM

Referring back to the case study introduced in Chapter 8, the city of São Paulo has developed around the Tietê River in a way that adjusts for its extreme pollution. On either side of the river, there are large expressways, which impede access. Land adjacent to the expressways is used predominantly for industry or commercial storage and wholesale activities. Land use has adjusted so that the population's exposure to the river is limited, but the problem of pollution remains. Surveys indicate that people who drive the expressways and work in the areas are aware of the stench of the river. Sections of the expressways frequently flood in the rainy season, exposing people to health risks. The water is too contaminated even for industrial use.

The contamination of the Tietê caused the government of the state of São Paulo to prohibit the pumping of Tietê water to the Billings Reservoir in another river basin where it had previously been used to generate hydroelectric power.[7] The loss of power production costs the power-generating company about R$75.2 million per year in forgone revenue.

In 1985, the state sanitation company, Saneamento Básico de São Paulo (SABESP), began to address the two problems of low sewerage coverage (64 percent) in its metropolitan service area and river pollution caused by not treating 81 percent of the sewage that was collected. It contracted a master plan to study the least costly way to clean up the river. The study focused on the Tietê basin within the São Paulo metropolitan area rather than the whole basin.[8] This appears to be a reasonable simplification since the headwaters of the Tietê are 95 km to the east of São Paulo and the majority of the contaminants enter in the SPMA. The river extends another 1,000 km after the SPMA, but the contaminants that enter thereafter are minor in comparison to those that enter in the SPMA.

The impact on river quality caused by reducing the inflow of contaminants at different points was simulated using the QUAL2E Stream Water Quality Model developed by the U.S. Environmental Protection Agency (see Annex 5-B in Chapter 5). The model is deter-

[7] Pumping is allowed to lessen flood problems in periods of very high flows.

[8] The study is no longer in the IDB's files, so this characterization comes from the project report for Stage I.

ministic and relatively simple. Hydrological variations are determined outside the quality model and the quality is calculated on the basis of a particular river flow and the contaminant loads that enter the river. The model separately accounts for contaminant loads coming in above the SPMA, point discharges of industrial and sewerage outfalls, contaminant loads from tributaries, nonpoint discharges from surface runoff, and river reflows from underground lenses.

Two dams on the river, the Ponte Nova and Edgard de Souza, can control the flow of water in the basin. The dams affect volume and velocity and therefore quality. If the flow control system is operated exclusively for carrying wastes away from the SPMA, all rivers run in their natural direction. If, however, water is to be diverted to Billings Reservoir, the gates at the Edgard de Souza dam are partially closed to raise the level of water and cause the Pinheiros River to flow in the reverse direction. The water quality model simulates two different operating regimes that are relevant to the economic analysis: operation exclusively to carry away wastes and operation in which 60 percent of the water goes to Billings and 40 percent continues downstream for other uses.[9]

With the water quality model, the master plan used a regional least-cost mixed integer programming model to choose the treatment plant capacities and locations and construction timing. The objective, subject to budget constraints, was to minimize the cost[10] of meeting water quality constraints at three points: (a) just above the confluence with Tamanduatei, (b) at the Pinheiros pumping station, and (c) at the discharge of Edgard de Souza dam (see Table 9-1).

The resulting program included connections, collection networks, interceptors, treatment plants, and disposal of residual solids. In addition, it included a program of control of industrial contamination, which was managed by the Companhia de Tecnologia de Saneamento Ambiental (CETESB). Because the pollution problem was enormous, both SABESP and the Bank knew that the solution would be expensive and would require many years to achieve. Taking the technical and financial resources of SABESP into consideration, SABESP and the Bank agreed to divide the project into three stages.

The impact on water quality of implementing the works in the master plan appear in Table 9-1, which shows the quality of water at minimum flow on various segments of the river system at the end of each of the stages. Water quality will be better than the level shown 90 percent of the time (329 days of the year). The results indicate that by the end of the second stage in 2003, dissolved oxygen will exceed the critical level of 0.5 mg/l from the confluence of the Pinheiros downstream (with the exception of the confluence itself, which does not quite reach 0.5 mg/l if the project is operated for hydroelectric generation). By the completion of the third stage in 2010, there will be significant levels of dissolved oxygen in all segments of the Tietê and Pinheiros rivers whether the system is operated exclusively for carrying wastes or for combined waste disposal and generation of electricity.

The investments in Stage I, which has been completed and is in operation, increased the proportion of wastewater treated from 19 percent in 1992 to 45 percent by 1998. The works included interceptors to carry wastes from collection points to treatment plants. The proposed Stages II and III include the expansion of CETESB's industrial pollution control program, and the construction and operation of more interceptors and treatment plants.

[9] This 60-40 division of water was a suggestion of the consulting firm that developed the master plan for state water use. This division of water does not imply optimal operation. It appears to reflect a judgment of what might be politically feasible. Before the constitutional restriction, water volumes were divided 50-50 between the Tietê and Billings.

[10] Investment, operation, and maintenance costs.

Table 9-1. Water Quality in the Tietê Basin at Minimum Flow at the Conclusion of Each Project Stage

	Operation to Carry Away Wastes			Operation to Carry Away Wastes and Generate Electricity		
Point where Quality Is Measured	Dissolved Oxygen (mg/l)	Bio-Chemical Oxygen Demand (mg/l)	Fecal Coliforms (No./100 ml)	Dissolved Oxygen (mg/l)	Bio-chemical Oxygen Demand (mg/l)	Fecal Coliforms (No./100 ml)
Tietê confluence Tamanduatei						
1998	0	33.49	850,200	0	33.49	850,200
2003	0	23.19	547,800	0	23.19	547,800
2010	1.46	13.64	233,300	1.46	13.64	233,300
Tietê confluence Pinheiros						
1998	0	15.22	150,200	0	28.89	777,700
2003	1.33	3.73	20,400	0.43	22.27	563,600
2010	1.14[a]	3.70[a]	22,000	1.98[a]	12.55[a]	246,900
Pinheiros Pumping Station						
1998	0	18.47	156,500	0	32.22	587,300
2003	1.99	7.21	10,500	0.55	16.66	292,800
2010	2.07[a]	7.20[a]	10,900	2.18[a]	11.62[a]	148,500
Edgard de Souza Dam						
1998	0.29[a]	29.16[a]	664,000	0.98[a]	31.95[a]	651,200
2003	1.34[a]	22.88[a]	505,400	2.95[a]	26.10[a]	460,500
2010	2.48[a]	12.65[a]	216,000	4.01[a]	13.09[a]	174,900
Pirapora						
1998	2.24	20.6	8,400	4.35	14.27	2,200
2003	2.6	17.59	10,200	4.5	13.41	2,400
2010	3.03	10.97	3,700	4.27	8.35	800

[a]The two different modes of operation, to carry away waste and to carry away waste and generate electricity, are associated with different flows (volume, depth, velocity, and direction) that result in different dilution and reaeration rates. In dual-purpose operation, the velocity of water is greater at the location indicated, and since the Barueri waste treatment plant is located between the confluence of the Pinheiros and Edgard de Souza dam, there is less water to dilute Barueri's effluent. For this reason, the levels of both BOD and DO are higher with joint waste and hydroelectric operation than they are for waste operation alone.

OVERVIEW OF THE ECONOMIC ANALYSIS

The economic analysis of the works separates the appraisal of the sewerage connections, collection networks, and collectors from the analysis of the river cleanup (interceptors, treatment plants, disposal of sludge, and industrial pollution control). Investments that connect users to the public system and carry the wastes out of local areas have benefits in those areas that are not related to the benefits from cleaning up the river. Thus the connection and collection systems can be treated as independent projects.[11] The original analysis done by the Bank and SABESP did treat these investments separately and that analysis is not discussed here.

[11] Connection and collection are closely related to the river cleanup project only if one considers the cost of the cleanup project as the cost of mitigation. At the time this project was developed, the need for mitigation was moot. The receptor body was dead; direct dumping of additional effluent would not make it deader. This poses a paradox for economic analysis. Connection programs usually generate large benefits sufficient to cover the cost of mitigation (i.e., treatment). However, in the initial stages of sewage collection projects, the discharge of wastes directly into receptor bodies may cause no significant deterioration in water quality and connection is not economically justified. Expansions of sewer collection may finally start to degrade the water, but the surplus of the marginal population (often the poorest) may not be sufficient to cover the cost of cleaning up everything. The willingness to pay for cleanup by the whole population may not be sufficient to justify the cleanup project, but economists do not use the net benefits from connection to sewer systems carried out in the past to justify the cost of cleanup once mitigation (environmental quality) becomes an issue.

The investment in cleaning up the river will produce benefits only when water quality improves enough to affect human behavior. For this reason, from the economist's point of view, the three stages are interrelated, not independent. All are needed to attain any benefits (improvement in water quality that will affect human perceptions and behavior).

SABESP and the Bank did not do a cost-benefit analysis for Stage I of the river cleanup project, presumably because the first stage by itself would not bring about a change in water quality that would change human behavior. Stage I removed 25 percent of organic material of domestic and industrial origin discharged into the Tietê River, and a similar percentage of other pollutants such as inorganic material, toxic compounds, and fecal coliforms. However, despite biochemical oxygen demand reductions from a "without project" level of 86 mg/l to 40 mg/l, the recovery of dissolved oxygen was insignificant, since absolute BOD levels remained well above the 5 mg/l that defines a "clean" river. With the first stage, levels of DO reach between 0.5 and 1.0 mg/l in some months of the year in segments just before and after a long anaerobic stretch. The only benefit is a minor reduction in odors over a short stretch of the river in those months. Under Stage I, dissolved oxygen does not reach levels that support aquatic life.

The discussion that follows develops two cost-benefit analyses: one for the project as a whole (Stages I, II, and III) and one for the incremental project (Stages II and III), which has yet to be built. The analysis of the incremental project is the only one relevant to the decision about whether to continue. To analyze the incremental project, it is necessary to calculate the NPV for Stages II and III together because Stage II by itself is not sufficient to bring any lasting improvement in the quality of the Tietê. The investment costs and benefits of Stage I are not relevant to the investment decision about Stages II and III because the capital costs have already been incurred and cannot be recovered (they are sunk costs), and the benefits are insignificant. The decision to continue depends only on the avoidable costs and attainable incremental benefits of the incremental project. However, the calculation of the NPV for the whole project is also presented for reasons of transparency and to demonstrate the well-known weakness of using cost-effectiveness analysis to justify a project.

Project Costs and Shadow Pricing Adjustments

SABESP provided information on the costs of the investment in the river cleanup and industrial pollution control program for the first and subsequent stages. The costs of investment for the first stage are known with certainty since they have already been incurred. The investment costs in the second and third stages are estimates and are subject to a margin of error. Because the works of the second and third stages are similar to those of the first stage, and because there are no major construction risks (such as geological risks when digging tunnels), it was assumed that the uncertainty about the estimates was relatively small. The cost estimates were assigned a margin of error of 15 percent in either direction under a symmetric triangular probability distribution.[12] This distributional assumption implies that the midpoint estimate is the most likely; small variations are more likely than large ones; and the maximum possible over- or underrun in costs is 15 percent.[13]

SABESP provided estimates of the operating costs of all stages, which also assigned a symmetric 15 percent margin of error and a triangular distribution. The original economic analysis of the incremental project (Stages II and III) did not charge the operating costs or the

[12] The use of a 15 percent margin of error in either direction (symmetric) is neither necessary nor usual. Investment costs are typically underestimated (human optimism or intentional bias to influence decision makers). Usually an asymmetric distribution is used, with the "best estimate" of cost placed toward the lower end.

[13] In the original Bank analysis, a larger margin of error was assigned to estimates of cost of the third stage, since the final designs had not been prepared and conditions might change.

industrial compliance costs of the first stage operation against the benefits of Stages II and III. Rather, these costs were ignored (treated as sunk) in the analysis of the incremental project because it did not seem realistic to assume that the first stage would be shut down if the incremental project were not built.

The appropriateness of omitting the operating costs of the first stage is open to discussion, since the costs are, in principle, avoidable. The treatment plants could be shut down and the industries could be allowed to suspend operation of their treatment facilities. While it is possible to argue that it would not be politically feasible to admit error and stop these operations, this is not an economic argument. Therefore, to maintain consistency with familiar cost-benefit conventions, our analysis assumes that the first stage operating costs are avoidable, not sunk, which means that they must appear in the cost stream of the incremental project.[14]

The investment and operating costs correspond to different types of works, including interceptors, treatment plants, and systems to pretreat industrial effluents. Some of the factors that cause actual costs to deviate from estimated costs are common to all types of works (e.g., the price of cement or steel), while others differ in their effect across types (e.g., change in designs, troubles with a single contractor). Thus, the risks of variation in costs are somewhat, but not perfectly, correlated. The degree of correlation is important in risk analysis simulation. If categories of costs (or benefits) are perfectly correlated, they all take on extreme values at the same time. This tends to increase the variability of the economic results simulated. If they are uncorrelated, a high value in one cost category often offsets a low value in others, and the degree of variability is dampened or even averages out. It is important to take this into account. By lumping all investment and operating costs together, the analysis here implicitly assumes that they are perfectly correlated. This may overstate the variance of the results.

A cost-benefit analysis must take into account all costs necessary to obtain the benefits, not just the costs financed for the project. One of the costs not financed by the project is the cost of industrial compliance with pollution control, which involved 1,168 industries in Stage I and 350 in Stages II and III. The analysis of the first stage did not include private compliance costs.

Before the first stage was carried out, CETESB estimated that it would cost an average of R$342,464 per firm to carry out the investment necessary to control effluents. In retrospect, it estimates R$171,000 per firm. There is little empirical basis for either estimate.[15]

The analysis in this chapter combines the midpoint of the range in estimated private industrial pollution control investment costs, R$256,848, with the public sanitation investment costs discussed previously and uses the same triangular distribution and 15 percent variation to reflect uncertainty. Annual operating costs for the private pollution control effort are approximated as 10 percent of capital costs. This significantly underestimates the possible variation of an important cost.[16]

[14] Our choice of how operating costs are treated has been made largely for didactic reasons. Treating them as sunk produces an NPV distribution for the incremental project that displays no probability of a loss, which is not a very interesting or typical case. Our simplifying assumption of avoidable first-stage costs overlooks the possibility that the abandonment of Stage I would lead to additional deterioration of the river below its current poor quality, with economic consequences caused by the buildup of sludge, more frequent flooding, and an increased risk of episodic threats to human health. In short, the benefits (damage avoided) of the incremental project may be understated by assuming avoidable Stage I operating costs.

[15] A World Bank study of pollution control by 22 industries in a nearby watershed estimated that the average cost of compliance for those industries was R$717,000, but the variance was enormous and it is difficult to extrapolate from this study without knowing the composition and size of the industries regulated.

[16] Alternatively, the private compliance costs could have been treated in more detail by using a uniform distribution with a range from R$171,000 (CETESB's ex post estimate) to R$342,000. The uniform distribution implies much greater uncertainty than our simplification.

The investment and operating costs of all works relevant to project benefits were subdivided into four categories: traded goods, nontraded goods, skilled labor, and unskilled labor. These costs were adjusted to economic opportunity costs (shadow priced) using a study done for a prior project. That study estimated conversion factors for skilled and unskilled labor of 0.79 and 0.48, respectively. The research for the prior study used the reciprocal of the weighted average tariff to estimate a standard conversion factor of 0.91 for nontraded goods, but it did not take into account the impact of high interest rates (tight monetary policy) in maintaining the level of the exchange rate. Therefore, this analysis uses 0.91 (the estimate from the other study) as an upper bound estimate, 0.75 as a modal (most likely) estimate, and 0.67 as a lower bound for the conversion factor for nontradables. The analysis used a triangular distribution to set the maximum variation. The distribution is slightly asymmetric; the mean is 0.78. This implies that simulations will generate a value less than 0.78 less than half of the time. The four shadow price factors are not correlated.

Table 9-2 reports the expected value of total capital and operating costs for each year of the project's life before the application of shadow price factors and discounting (i.e., these are financial costs).

Project Benefits

There are two principal benefits of the Tietê project: the public good benefits stemming from a reduction in odors and aesthetic blight that had to be estimated by a contingent valuation approach, and the private benefits from increased hydroelectric power generation that could be valued through the market for energy.[17]

Improvement in River Water Quality

To calculate total gross project benefits of better water quality, the average benefit per household has to be multiplied by the number of beneficiary households, distinguishing households that are in districts contiguous to the river from those that are not. According to census data, there are 2.46 million households in districts contiguous to the major tributaries and 1.6 million in noncontiguous districts.

The present population of São Paulo is known with certainty, but the rate of growth, which affects total benefits, is not. São Paulo is heavily built up and its expected population growth rate is low: 0.75 percent in contiguous districts and 1.00 percent in noncontiguous areas. The analysis specified a symmetric triangular distribution for each, with possible ranges of growth of 0.5 to 1.0 percent for contiguous districts and of 0.75 to 1.25 percent for noncontiguous districts. Because the contingent valuation question limited the payment period to 10 years starting with the construction period, these benefits were projected for 10 years beginning with the construction of the second stage.[18]

Finding a reasonable average measure of per household benefit is more problematic, and ultimately much more important, than specifying the size of the future population in this case.

[17] In addition, there are other benefits that have not been quantified; these include increased recreation benefits at Pirapora do Bom Jesus and further downstream, the retardation of saline intrusion in the Cubatão River in the Baixada Santista, and the provision of a more economic source of potable water for the Baixada.

[18] The payment period stipulated in the contingent valuation survey was 10 years. Because the principal benefits accrue for 10 years, but operating costs continue past 10 years, the net benefit flow eventually turns negative. Just when or how often negative net benefits appear depends partly on the length of execution delays and the timing of energy benefits. These complexities mean that over the 30-year analysis period, there can be multiple changes of sign in net benefits, so there may be multiple internal rates of return. For this reason, the analysis calculates only the NPV.

Table 9-2. Project Capital and Operating Costs (Stages I, II and III) for Wastewater Treatment (undiscounted thousand 1998 reals)

Time Period and Stage (Year 1=1992)	Total Investment Costs	Total Operating Costs	Total Costs
Begin Stage I Construction 1	18,304	0	18,304
2	33,559	0	33,559
3	333,948	0	333,948
4	60,864	30,000	90,864
5	72,549	30,001	102,550
End Stage I Construction 6	87,370	30,001	117,371
Stage I Subtotal	**606,594**	**90,002**	**696,596**
Begin Stage II Construction 7	42,333	65,716	108,049
8	54,339	66,124	120,463
9	97,432	66,727	164,160
10	200,685	67,845	268,529
End Stage II Construction 11	72,633	79,190	151,822
Stage II Subtotal	**467,421**	**345,602**	**813,023**
Begin Stage III Construction 12	42,761	104,349	147,110
13	38,090	104,864	142,954
14	60,182	105,100	165,282
15	60,333	105,392	165,725
16	59,667	105,627	165,294
17	59,666	105,916	165,582
End Stage III Construction 18	57,563	106,100	163,662
Stage III Subtotal	**378,262**	**737,347**	**1,115,610**
Begin Full Operation 19	0	135,288	135,288
20	0	137,415	137,415
21	0	137,592	137,592
22	0	137,769	137,769
23	0	137,947	137,947
24	0	138,036	138,036
25	0	138,127	138,127
26	0	138,216	138,216
27	0	138,306	138,306
28	0	138,306	138,306
29	0	138,306	138,306
End Analysis Period 30	0	138,306	138,306
Years 19 to 30 Subtotal	**0**	**1,653,612**	**1,653,612**
Grand Total	**1,452,277**	**2,826,564**	**4,278,840**

Note: Costs exclude household sewer connections, the collection system, and the cost of collectors sufficient to carry untreated effluent to the nearest dumping point in the river. These costs were balanced against local household sewering benefits in a separate cost-benefit exercise not reported here. The costs in the table are related to pollution control and include interceptors, wastewater treatment, and the industrial environmental cleanup program. The costs in the table are not shadow priced or discounted because of the uncertainties about timing and pricing that are handled in the risk and sensitivity analyses discussed later.

Population growth can be confidently confined to rather narrow bounds because population density in the areas of the city affected by the project is already high and there is not much room left for absorbing additional inhabitants.

Chapter 8 and Vaughan et al. (1999) provide a detailed discussion of how twelve different measures of mean and median water quality benefits could be extracted from a single referendum CV exercise.[19] Now, how can one or more of these measures be used in the cost-benefit analysis of the project? Table 9-3 recapitulates nine estimates of the mean and the standard error

[19] This specification of WTP benefits and their variance is much more detailed than it was in the original analysis that was done for the formal IDB loan approval process.

Table 9-3. Estimates of Mean Willingness to Pay and the Standard Error of the Mean

	WTP per Household per Month (1998 reals)			
Central Tendency Measure	**Contiguous: Close to River**		**Noncontiguous: Far from River**	
Parametric Measures	**Mean**	**S.E.**	**Mean**	**S.E.**
Bounded probit mean	140.10	[a]	60.46	[a]
Truncated mean [C′]	9.73	1.29	6.16	0.75
Truncated mean [C^{-}]	7.66	0.71	5.03	0.44
Untruncated mean ≡ median [C+ ≡ C*]	4.74	1.66	−1.27	1.56
Truncated mean, log transform [C^{+}_{ln}]	4.66	[a]	1.46	[a]
Truncated mean, log transform [C^{-}_{ln}]	3.49	[a]	1.23	[a]
Nonparametric Measures				
Boman et al. (1999) upper bound (Paasche)	12.77	1.61	9.67	0.87
Kriström's (1990) intermediate	9.42	1.19	7.09	0.66
Haab and McConnell's (1997a) lower bound (Laspeyres)	6.07	0.80	4.51	0.47

[a] Approximate standard errors are reported in cases where an analytical formula for expected value enabled them to be estimated via a Taylor series expansion, or where on analytical formula for the S.E. exists (i.e., the nonparametric measures).

of the mean (where computable) from Chapter 8. There are two kinds of uncertainty associated with these measures. The first is *statistical* uncertainty *within any given measure* since each mean is a random variable with its own distribution and standard error.[20] The second is *methodological* or *subjective* uncertainty *across measures*, since none of them can be ruled out a priori, with the possible exceptions of the untruncated parametric and bounded probit means.

The statistical margin of error can be summarized by a confidence interval around the estimates of, say, 99 percent, or roughly ±2.6 standard errors on either side of the mean. These confidence intervals show a percentage variation about the several means of roughly 75 percent for households contiguous to the river and 51 percent for more distant households. The second kind of uncertainty is about which of the several alternative means to use. The statistical margin of error is far less than the relative range between the low and high estimates of the mean using the different estimation methods. Leaving out the very high estimate obtained with the bounded probit model, the range of mean willingness to pay for those in districts contiguous to the river is R$3.49 to R$12.77; the means for those not contiguous to the river range from R$1.23 to R$9.67. For those contiguous to the river, the percentage variation around the midpoint of the range (i.e., the ratio of the range to the mid-point) is 114 percent and ascends to 155 percent for those not contiguous to the river.

Characterizing Uncertainty about CV Benefits

There is no unambiguously correct way to summarize this range of estimates for the Monte Carlo risk analysis, but there are at least three plausible alternatives.[21] The first is to choose one

[20] Ardila (1993) and Hazilla (1999) explain how to compute the distribution of expected values using the delta method, bootstrapping or numerical approximation assuming asymptotic normality of the parameter estimates that appear in the "function of interest," the formula for WTP. Table 9-3 reports standard errors for the Tietê WTP means obtained via analytical formulas for the nonparametric means and via the delta method for most of the parametric means. Vaughan et al. (1999) review the plausibility and deficiencies of the alternative central tendency measures.
[21] The bounded probit means are outliers, being more than ten times greater than the next highest estimate. It is highly unlikely that average willingness to pay is this high. This improbability should be reflected in the risk analysis. It is also worth noting that the untruncated linear model yields an average willingness to pay that is negative. This

measure, such as the Turnbull lower bound mean, as preferred. The advantage is simplicity and the ability to incorporate statistical uncertainty, but the disadvantage is that any subjective uncertainty is assumed away. In contrast, a judgmental distribution for the mean could be formulated based on the gamut of possibilities in Table 9-3 to reflect subjective uncertainty about which is the "best" estimate. The disadvantages of this route are that all of the alternative measures have to be calculated in order to make probabilistic assignments, and that statistical uncertainty is difficult to incorporate. The work of Boman et al. (1999) promises the best of both worlds because the upper and lower bound nonparametric means offer a way to span a good part of the range and simultaneously incorporate statistical uncertainty.

The Subjective Approach. Since the uncertainty with respect to the appropriate central tendency measure is much greater than the statistical uncertainty, this version of the risk analysis uses a judgmental (subjective) distribution of the central tendency values shown in Table 9-3 plus an additional subjective adjustment to allow for error in questionnaire design and the timing of its application.

In addition to statistical error and possible error from choosing the wrong method to estimate central tendency, there is possible error associated with the questionnaire and its implementation. The Tietê questionnaire was not absolutely clear that the improved quality would still produce unpleasant odors from the river 1 month per year. If respondents had known this, their WTP might have been less. Working in the opposite direction, the survey was conducted at a time when Brazil was in deep recession. People were worried about keeping their jobs. In more normal times, average WTP might have been higher than stated at this moment. To reflect these possibilities, it is prudent to keep the ranges wide.

The subjective analysis characterizes the uncertainty with a histogram. For districts contiguous to the river, it establishes a lower range between R$2.00 and R$4.00. The lower bound is less than the lowest mean (R$3.49) and incorporates the possibility that the respondents might have expressed a lower WTP if they had clearly understood that the project would deliver an odor-free river only 11 months a year, or if they had more time to consider the implications of the income burden the elicited monthly payments would impose annually. The upper bound of the low range incorporates the lowest estimated mean (R$3.49). The analysis here assigns a subjective probability of 30 percent to this range of the histogram.

The second range in the histogram goes from R$4.00 to R$7.00. We assigned the second range a probability of 50 percent. This range includes three of the eight means estimated. The relatively high weight indicates our judgment that the true WTP is probably within this range. The third bar of the histogram is for the range R$7.00 to R$10.00 and has a probability of 15 percent. This range includes the three higher estimates of means that we considered possible but less likely. The fourth range runs from R$10.00 to R$50.00. The statistical analysis produced no estimates in this range, but we considered the range more probable than the alternative of leaving a gap and consigning the entire remainder of the distribution to the interval defined by the bounded probit mean, which appeared unreasonably high. We assigned a probability of 4 percent to this range. It allows for the fact that true WTP might be higher than estimated because of statistical error in the means that appear in the earlier range, or because the survey was carried out in a recession. The final range is from R$50.00 to R$140.00. This includes the extreme measure of the mean from the bounded probit, which was assigned a probability of 1 percent, meaning it is highly unlikely but not impossible. A similar procedure was used to characterize the WTP of households in districts far from the river.

result is inconsistent with theory since it implies that the population would have to be paid in order to acquiesce to the cleaning up of the river. It is possible that some people might have to be paid to accept the project. Certain individuals may believe, for example, that they will be inconvenienced more than they are benefitted by works of the project located in their vicinity. The average person, however, should be positively affected by the project and should have a non-negative willingness to pay.

The subjective probability distributions for average WTP are shown in Figures 9-1 and 9-2. The mean WTP for contiguous households implied by the distribution in Figure 9-1 is R$7.09 per month, which is R$1.02 higher than the corresponding Turnbull lower bound measure. The average is R$3.76 per household per month for the noncontiguous distribution in Figure 9-2, which is R$0.75 lower than the corresponding Turnbull mean.[22]

The subjective WTP distributions are positively skewed, and this assumption will be mirrored in the distribution of the outcome, NPV. In contrast, the alternative assumption of a lower bound Laspeyres (Turnbull) mean with all variation coming from normally distributed random (statistical) error would make the outcome distribution of NPV look more normal.

The Nonparametric Limits Approach. Were it not for the insights of Boman et al. (1999), our treatment of uncertainty on the side of public good benefits would have to stop here, with two sharply contrasting and ultimately unsatisfactory approaches that handle either statistical uncertainty or subjective uncertainty, but not both. Fortunately, Boman et al. have linked the choice among benefit measures to economic theory, which opens the door to an interpretation that narrows the extent of subjective uncertainty compared with the judgment-based method above, and simultaneously accounts for statistical variation around mean WTP.[23]

To make the link to welfare theory, suppose the proportions of "yes" responses to a referendum CV survey are plotted against the bids and the points connected by interpolation to produce a picture of the survival function.[24] The acceptance proportions should generally decrease as the bid level increases. Interpreting the proportions as the fraction of individuals who would be willing to buy a fixed amount of a public good (the "quantity") if it were offered at a specific bid price, the survival function is analogous to a demand curve (Johansson, 1995). The bid levels represent marginal willingness to pay, and average willingness to pay is the integral under the survival function (see the derivation in Vaughan et al., 1999; Vaughan and Rodriguez, 2000).

Panel A of Figure 9-3, adapted from Boman et al. (1999), shows the demand for a public good, z, as a function of its hypothetical price. The price is equal to the representative individual's marginal willingness to pay, MWTP. The change in the level of public good provision is

Figure 9-1. Subjective WTP Distribution

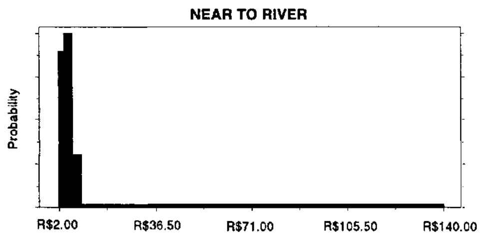
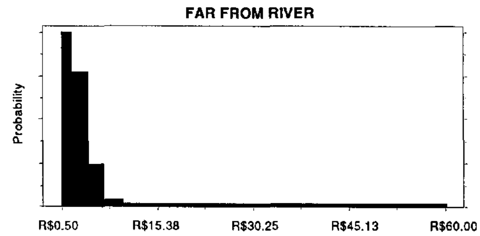


Figure 9-2. Subjective WTP Distribution

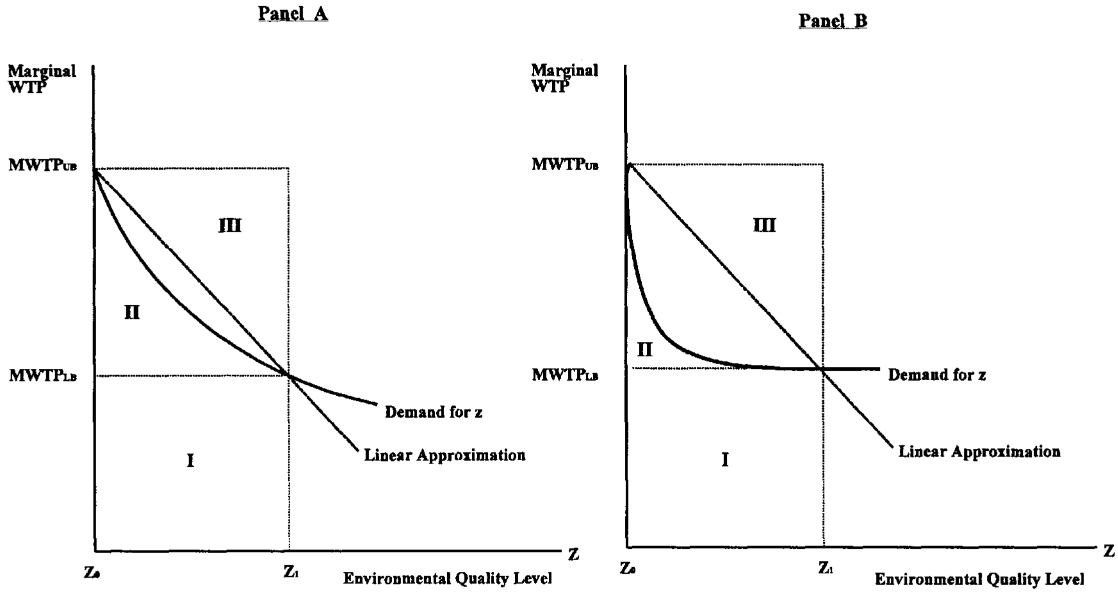


[22] The subjective distributions were not consciously designed to reproduce any of the other means. These numbers are only for purposes of comparison.

[23] In order to pursue the route based on our elaboration of Boman et al.'s arguments, the CV survey data must be reliable. If the analyst has doubts about the survey, the only recourse is to re-do the survey or adjust the mean WTP data ex post, based on subjective judgment.

[24] The analytical formulas for the nonparametric means in Vaughan and Rodriguez (2000) employ the distribution function, where the probability of rejection increases with the bid levels. The acceptance proportion at any bid level is just 1 minus the rejection proportion, yielding the survival function plotted in Vaughan et al. (1999, Figure 3).

Figure 9-3. Welfare Measures (adapted from Boman et al., 1999)



$z_1 - z_0$. The Laspeyres lower bound monetary measure of welfare improvement, L, is represented by the area labeled as I, which is the product of the quantity change and the lower, postchange price, $MWTP_{LB}$. The Paasche upper bound measure, P, is the sum of Areas I, II, and III, or the product of the quantity change and $MWTP_{UB}$. The true welfare change is the integral under the demand curve between z_1 and z_0, or Area I plus Area II.

Adding a linear approximation to the nonlinear demand curve shown in Boman et al.'s original graph provides a rationale for Kriström's nonparametric mean, which falls between the Paasche and Laspeyres means. That is, the true welfare measure, T, is approximately equal to $L + \frac{1}{2}(P - L)$ or $\frac{1}{2}(L + P)$ = Area I + $\frac{1}{2}$ (Area II + Area III). Panel B of Figure 9-3 shows that the Kriström approximation to the true mean gets worse as the true demand curve becomes more convex. At some point the approximation error in Kriström's mean will exceed the approximation error of the lower bound Laspeyres (Turnbull) mean, as it does in Panel B.

From this analogy it is clear that the upper bound mean should *never* be used by itself as a welfare measure in cost-benefit analysis because it always overstates benefits by more than Kriström's intermediate mean, which will also overstate E(WTP) if the demand curve is convex to the origin. It also tells us that the true mean lies somewhere between the Kriström intermediate mean and the Laspeyres (i.e., Turnbull) lower bound. In this sense (unless the demand curve is concave to the origin) the Kriström intermediate mean, not the Paasche mean, is the operative upper bound.

Kriström's intermediate mean is probably a reasonable compromise if a single measure is desired, but so is the Turnbull lower bound for analysts who prefer a more conservative, less forgiving measure. We fall somewhere in between, and prefer to approximate the true mean with a weighted average of the Paasche and Laspeyres means. To do this in the Monte Carlo analysis, a weighting factor, κ, is drawn from a uniform distribution over the interval of 0.50 to 1.0. The factor is multiplied by the Turnbull mean and added to the product of 1 minus the factor times the Paasche mean, where both means are randomly drawn from normal distributions with the standard errors specified in Annex 10-B and Vaughan and Rodriguez (2000). So $T \approx \kappa \times L + (1 - \kappa) \times P$, where the weighting factor, κ, is ≥ 0.5.

Our approximation always gives the Paasche mean equal or lesser weight than the Laspeyres mean in computing a linear approximation to the true mean, rather than the equal weights of 0.5 used by Kriström. In fact, since the mean of the uniform distribution between 0.5 and 1.0 is 0.75, on average our calculation produces a WTP that assigns 75 percent of the weight to the lower bound Laspeyres (Turnbull) mean WTP and 25 percent to the upper bound Paasche mean.[25] The mean WTP for contiguous households implied by our unequally weighted procedure is R$7.75 per month, which is R$0.66 higher than the corresponding subjective measure, but R$1.67 lower than Kriström's intermediate mean. The unequally weighted average is R$5.80 per household per month for the noncontiguous distribution, which is R$2.04 higher than the corresponding subjective mean, but R$1.29 lower than Kriström's mean.[26]

Benefits from Additional Hydroelectric Generation

Until 1992, half of the flow of the Tietê was pumped to Billings Reservoir to generate electricity at the Henry Borden power plant. "Transitory provisions" in the state's 1989 constitution now prevent pumping wastewater from the Tietê to Billings Reservoir. With the treatment plants of Stages II and III, the water to be pumped would no longer be "wastewater" and the environmental authorities might consider the water good enough to pump. Thus the Tietê project may have benefits by permitting additional hydroelectric generation.

It is not certain, however, that pumping will be allowed. While it is true that the water of the Tietê will no longer be wastewater, it will not be good water. Billings Reservoir is classified as a Class II water body. By regulation, water pumped to a Class II body cannot degrade the quality to less than 5 mg/l of dissolved oxygen, more than 5 mg/l of biological oxygen demand, or more than 4,000 fecal coliforms per 100 ml. Billings Reservoir is already out of compliance with Class II standards at the point where the Tietê/Pinheiros water would be injected. Thus, in principle, the only water that could be pumped there would be distilled water.

The water to be pumped from the Pinheiros pumping station will not attain Class II quality in either Stage II or III (see Table 9-1). Billings, however, is a very large reservoir. Its present quality ranges from Class IV (where Pinheiros/Tietê water would be injected) to Class I in the area where water is released for potable uses in the Santos region. The environmental authorities could reclassify sections of Billings to reflect present reality and the fact that the reservoir functions as a natural treatment plant. If the part of Billings where the Tietê water would be pumped were reclassified as Class IV, Tietê water could be pumped. It is highly uncertain, however, whether Billings will be reclassified, if such pumping will be allowed, when it will be allowed, and how much will be allowed.

The Monte Carlo risk analysis simulates a number of scenarios that reflect the political pressures that might be exerted by various institutions and stakeholders. The analysis assigns a subjective probability distribution to various scenarios. It assigns a 50 percent probability to the most adverse case: diversion of Tietê water is never permitted. The analysis assumes that there is a 5 percent probability that pumping will be possible the year after the second stage is completed, a 10 percent probability in the fourth year after Stage II is completed, a 17.5

[25] Rather than drawing from the upper and lower bound means inside the Monte Carlo simulation, an alternative way of accomplishing a similar result would be to calculate the 75/25 weighted mean and variance directly by adapting the calculation of Kriström's mean and variance set out in Vaughan and Rodriguez (2000). The standard errors differ between the two approaches.

[26] No attempt was made to "calibrate" the subjective mean to match the mean based on weighted bound values. On the contrary, the two exercises are totally independent because the subjective distribution was designed several months prior to the publication of the paper by Boman et al. (1999), which inspired our weighted mean. The Monte Carlo results under these two competing approaches can be viewed as if they were undertaken by different analysts.

percent probability the year after the third stage is completed, and a 17.5 percent chance 4 years after the third stage is completed.

Note that in a risk analysis the time lag (pause) between Stages I and II and the length of the Stage II execution period are not fixed, but instead are drawn from probability distributions. The probability that energy benefits will come online in a certain number of years after completion of either Stage II or Stage III is also drawn from a distribution, as noted in the computational steps under base case execution timing:

Scenario	Draw a Uniform Random Number and Choose Scenario If Range Is:	Year Energy Benefit Flows Initiated: Base Case Execution
1	0.000 to 0.050	12
2	0.050 to 0.150	15
3	0.150 to 0.325	19
4	0.325 to 0.500	22
5	0.500 to 1.000	Never

If there are execution delays, in our spreadsheet analysis setup the energy benefits are moved into their correct starting-year position using time indices and Excel's VLOOKUP function.

The acceptance of pumping is not the only uncertainty. If pumping is allowed, the benefits from using Tietê/Pinheiros water will depend on the amount of energy to be generated and the time of day when it will be generated. To estimate the amount of additional energy that might be generated, historical data were obtained on the amount of water processed by the Henry Borden power station before and after the restriction was imposed on pumping from the Pinheiros/Tietê. The average difference is equivalent to a continuous flow of 67.0 m^3/sec (Table 9-4). This flow

Table 9-4. Henry Borden Power Plant Energy Production and Water Use before and after Restriction on Pumping from the Tietê to Billings

Year	Energy Produced (MWh)	Water Used (m^3/sec)
Before Restriction		
1985	3,702,424.5	75.4
1986	4,244,978.9	86.4
1987	5,056,923.8	104.5
1988	4,816,698.2	99.1
1989	5,230,506.9	108.1
1990	3,603,258.4	74.2
1991	4,798,196.6	98.1
Average 1985–1991	**4,493,283.9**	**92.3**
After Restriction		
1992[a]	2,811,472.1	57.2
1993	1,579,454.0	32.1
1994	694,913.7	14.1
1995	1,255,767.6	26.2
1996	1,535,861.1	31.0
1997	1,131,306.8	23.3
Average 1993–1997	**1,239,460.6**	**25.3**
Difference of Period Averages	**3,253,823.3**	**67.0**

[a] In 1992, the restrictions on pumping were imposed.

Table 9-5. Power Stations on the Tietê Downstream from Edgard Souza

Power Plant	Production ($MW/m^3/sec$)
Rasgão	0.1754
Porto Góes	0.1887
Barra Bonita	0.1727
A. Souza Lima	0.1881
Ibitinga	0.1872
Promissão	0.2057
Nova Avanhandava	0.2605
Ilha Solteira	0.3902
Jupiá	0.1982
Porto Primavera	0.1669
Total	2.1336

resulted from a 50-50 division of the Tietê's flows. A master plan for the water resources of the state of São Paulo suggests that a 60-40 division might be possible. If so, it might be possible to pump the equivalent of a continuous 80 m^3/sec to Billings. The amount that can be pumped will be determined after a political negotiation of stakeholders with divergent interests. It is not known with certainty. The Monte Carlo risk analysis uses the range in the calculations (67.0 to 80 m^3/sec).

The incremental energy generated is the difference between the energy that can be generated at Henry Borden with the pumped water less the energy that could be generated with the water on the 10 downstream plants on the Tietê, and the energy used in pumping. Henry Borden has a production capacity of 5.654 MW/m^3/sec. Pumping a cubic meter from the Pinheiros/Tietê to Billings reduces the net production by 0.314 MW/m^3/sec. Thus, Henry Borden's net gain from receiving a cubic meter per second is 5.34 MW/m^3/sec. This net gain is also reduced by the losses of the hydroelectric plants on the lower Tietê.

Table 9-5 shows that the production generated by a cubic meter passing through all the plants is 2.1336 MW/m^3/sec. Thus, the net national gain from transferring a cubic meter of water from the Tietê to Billings is 3.206 MW/m^3/sec (i.e., 5.3400 minus 2.1336). This converts to 27,084.56 MWh of additional energy per cubic meter per year. If the incremental water pumped to Billings is between 67 and 80 m^3, the incremental energy is in the range of 1,881,665 MWh to 2,246,763 MWh per year.

Because Billings is an enormous reservoir with interannual storage, it is possible to produce most of the incremental energy at peak. The plants on the lower Tietê have enough storage capacity to guarantee peak operation with or without the diversion to Billings. The decrease in power on the Tietê will be power offpeak. It is not certain, however, that Borden will be allowed to use all the Tietê water during peak hours, because the power company may be ordered to increase the amount of water that is released at a constant rate to prevent saline intrusion in another river basin (the Cubatão). If such releases are required, they will be offpeak. The value of high-voltage energy during peak demand is between R$37.33 and R$42.69 per megawatt-hour, depending on whether it is the wet or dry season, and the value offpeak is between R$25.67 and U.S.$30.20 per megawatt-hour, again depending on the season. The economic analysis uses a range of R$30.20 to R$42.69 in its calculations. These values are based on the long-run average incremental cost of supply at high voltage.

The risk analysis presented later combined the information about uncertainty with respect to the amount of energy that would be generated each year and its value by multiplying the low extreme of one by the low extreme of the other (1,881,665 MWh × R$30.20) and the high extreme by the high extreme (2,246,763 MWh × R$42.69). It used a symmetric triangular distribution to describe the probability.[27]

Project Timing: Delays in Stage II Execution and Phasing

The economic analysis of the IDB usually assumes that the length of time between the beginning of construction and the beginning of operation and initiation of benefit flows (known as the execution period) is the 4 (or more recently, 5) -year period stipulated in the loan contract. Most analyses do not estimate the impact of slow execution on NPV. Analyses for multistage projects assume no phasing delays between the end of one stage and the beginning of another. Omission of the risk of delays or pauses gives an overly optimistic view of NPV when net benefits are growing at a rate lower than the discount rate (the usual case).

To examine the realism of the 4-year execution period, we reviewed IDB data on 17 sanitation projects undertaken between 1980 and 1992 in Argentina, Brazil, Chile, Mexico, and Uruguay. The frequency distribution displays positive skewness, with an average execution period of 5.75 years, a median of 5.5 years, and a mode of 4.75 years. All exceed the 4-year execution period usually assumed. (Equivalent data on phasing of multistage projects are not available.)

The execution period for the first stage of the Tietê was 6 years (known with certainty). To reflect the uncertainty about the execution period for the second stage, the analysis assumed that it will be similar to that experience cited earlier. The risk analysis draws from the empirical frequency distribution for the execution period for Stage II, while holding Stages I and III fixed at the 6 years actually experienced in Stage I and the 7 years planned for Stage III. The probability of delay is characterized by the empirical histogram with a 61.5 percent probability of an execution period of between 5 and 6 years, a 23.1 percent probability of an execution period between 6 and 6.5 years, and a 15.4 percent chance of delays between 6.5 and 10 years.

Phasing delays between the end of Stage I construction and the start of Stage II are only relevant for the economic analysis of the entire multistage program. In the absence of any hard historical data on phasing delays, the analysis assumes that a hiatus of between 0 and 4 years could occur with equal probability (uniform probability distribution) between Stages I and II, while allowing Stage III to follow without delay.

STANDARD RESULTS AND SENSITIVITY ANALYSIS

Most project analyses report only a single payoff (NPV or internal rate of return) as if it were the only result possible, and analysts almost always choose one point estimate of per unit benefits without exploring or necessarily being aware of other possibilities. In the past the unbounded parametric mean has been the measure commonly extracted from referendum CV surveys, but in this case it breaks down because of a negative willingness-to-pay estimate (see Table 9-3). For illustrative purposes, the Turnbull lower bound mean is a close substitute. If an analysis had been conducted for Stages II and III using only the most likely or midpoint estimates about the factors that determine project payoff, basing benefits on the Turnbull mean,

[27] This simplified treatment was adopted in order to accommodate the complexities of execution timing more easily in the spreadsheet design. It implies that the value of energy and the amount produced are perfectly correlated, which they are not. This leads to an overestimate of the probability of very high or very low energy benefits. Had the risks been specified independently, the distribution of the estimate of energy benefits would have been more compact, as high draws from one distribution would have frequently been offset by low draws from the other.

it would have shown that the NPV of continuing the project is negative. The incremental investment loses R$44 million,[28] so the project does not appear to be economically feasible.

Sensitivity analysis is the method most multilateral financial institutions use to address the uncertainty in the projects they appraise, if they address it at all.[29] In this case, the negative NPV result would halt project approval. If strong pressure for approval were to be exercised by influential stakeholders, it would probably send the analyst searching for factors that, with "small" adjustments, might make the investment appear to be profitable and hence bankable (i.e., eligible for a loan).

A standard sensitivity analysis reflects the uncertainties about project outcome by looking at the impact on NPV (or IRR) of standard but arbitrary percentage increases and decreases of the factors that determine a project's outcome. Occasionally, it presents calculations of the "critical value" of each influence that will drive NPV above 0 if the NPV is negative or below 0 if it is positive, called "switching values." The crudest approach is to increase or decrease gross benefits and costs, while a more refined approach explores the effects of varying the factors driving these aggregates.

Table 9-6 presents the best estimate for NPV of Stages II and III and a sensitivity analysis that is more detailed than those generally presented. It shows the NPV that would result if benefits, construction period, energy benefits, timing, costs, and shadow price factors were 10 or 25 percent higher or lower than their "base estimate" values.[30] It uses the mean of the Turnbull distribution of WTP as the "best estimate," and instead of using the available statistical information about the uncertainty of benefits (the confidence interval), it arbitrarily applies the same percentage change factors to everything. This type of presentation is more detailed than the typical presentation, but shares the feature that both statistical uncertainty (which is, in fact, quantifiable) and methodological uncertainty are totally ignored.

On the basis of Table 9-6, most analysts would probably conclude that the project is not economically feasible because it has an NPV below 0; the project entails a net loss of 44.2 million reals. With the exception of a shorter execution period, small favorable changes on the order of 10 percent in the most important factors that determine the outcome would not make it feasible. If the sensitivity analysis used only aggregated benefit and cost flows (under "Crude" in Table 9-6), even less could be said, such as "An increase of 10 percent in all benefits or a 10 percent decrease in all costs would raise the NPV to about R$14 million." If the sensitivity provided additional detail (under "Detailed" in Table 9-6), the analyst would have more to say, "The sensitivity analysis explored the impact of changes in all the factors important to project success. It found that the project is most sensitive to the WTP of families in districts contiguous to and not contiguous to the river, followed by the shadow price of nontradables, the execution period, and the timing of energy benefits. Public good benefits are a given from the contingent valuation survey, and the shadow price of nontradables has been firmly established by a study. The execution period and the timing of energy benefits depend critically on institutional and legal issues in the host country and are hard to predict. But it would appear that unless the

[28] This calculation treats the capital costs in Stage I as "sunk," so they are not relevant in calculating the incremental net returns from Stages II and III.

[29] Ardila et al. (1998) and Chapter 2 found that of the 18 water quality investment projects approved by the IDB over a 10-year period, 7 reported no sensitivity analysis at all, 8 used standard up/down sensitivity, 3 statistical uncertainty in benefits, and 1 performed a Monte Carlo risk analysis that incorporated statistical and subjective judgments about the uncertainty of the factors that determine outcome.

[30] In this example, the base estimate of each of the factors that influence the NPV outcome is generally a mean value (see the note in Table 9-6, and Table 9-7), so NPV is an expected value. Often, however, project analyses do not clearly distinguish between mean and modal values so their base estimate of NPV may have no clear statistical meaning.

Table 9-6. Sensitivity Analysis, Incremental Project (Stages II and III)

Degree of Detail and Variable	Units	Base Case Mean	NPV with Percent Changes in Variables from their Base Case Means (000 R$) −25%	−10%	+10%	+25%	Range	NPV Switching Value
Detailed								
Water quality benefits, near river[a]	R$ WTP/household	6.07/month	(136,760)	(81,322)	(7,022)	48,406	185,166	6.80/month; 12% increase
Water quality benefits, far from river[a]	R$WTP/household	4.51/month	(89,466)	(62,212)	(26,141)	1,112	90,578	5.61/month; 24% increase
Project execution period[b]	Years	6	NA	20,551	(27,846)	(63,782)	84,334	5 years; 17% decrease
Energy benefit initiation[c]	Project year	19	21,908	(12,871)	(59,909)	(83,070)	70,198	Year 12; 37% decrease
Value of energy benefit	R$/MWh	36.44	(53,900)	(48,066)	(40,287)	(34,453)	13,613	77.98/MWh; 114% increase
Shadow price, nontradables[d]	Factor	0.78	(114,569)	(73,616)	(14,538)	29,920	103,236	0.90; 15% increase
Population growth rate near river	%/Year	0.75	(47,398)	(45,367)	(42,811)	(40,923)	4,444	3.1%/year; 313% increase
Population growth rate far from river	%/Year	1	(46,249)	(45,009)	(43,340)	(42,077)	2,932	5.7%/year; 470% increase
Shadow price, unskilled labor	Factor	0.49	(40,003)	(42,438)	(45,916)	(48,350)	5,913	None. NPV negative at 0
Shadow price, skilled labor	Factor	0.79	(37,598)	(41,545)	(46,808)	(50,755)	9,210	None. NPV negative at 0
Crude								
All costs (investment plus O&M)	000 R$ NPV	313,171	101,388	14,109	(102,403)	(189,741)	203,790	589,370; 7% decrease
All benefits (water quality plus energy)	000 R$ NPV	589,370	(191,519)	(103,114)	14,760	103,166	206,280	633,573; 7.5% increase

Note: Numbers in parentheses are negative (losses). The sensitivity settings are as follows:

Variable	Base Case Mean	−25%	−10%	+10%	+25%
Base case NPV (000 R$)	(R$44,177)				
CV Water quality benefits, near the river/month	R$6.07	R$4.55	R$5.46	R$6.68	R$7.59
CV water quality benefits, far from the river/month	R$4.51	R$3.38	R$4.06	R$4.96	R$5.64
Project execution period	6	4.50	5.40	6.60	7.50
Energy benefit initiation	19	12.00	15.00	22.00	Never
Value of energy benefit	R$36.44	R$27.33	R$32.80	R$40.08	R$45.55
Shadow price, nontradables	0.78	0.59	0.70	0.86	0.98
Costs (investment plus O&M) (000 R$)	R$633,547	R$475,160	R$570,192	R$696,901	R$791,934
Population growth rate near the river	0.75%	0.56%	0.68%	0.83%	0.94%
Population growth rate far from the river	1.00%	0.75%	0.90%	1.10%	1.25%
Shadow price, unskilled labor	0.49	0.37	0.44	0.54	0.61
Shadow price, skilled labor	0.79	0.59	0.71	0.87	0.99

[a]Turnbull lower bound mean.

[b]The lower limit on the execution period is 5 years in the spreadsheet model, so NPV cannot be calculated below that.

[c]Conditional mean assuming energy benefits materialize computed as the normalized probabilities in each year times the alternative initiation years. That is, $0.05/0.5 \times 12 + 0.1/0.5 \times 15 + 0.175/0.5 \times 19 + 0.175/0.5 \times 22$.

[d]This factor is applied to all benefits and nontradable costs, so NPV increases as the shadow price factor increases because shadow-priced benefits increase more than costs.

project yields hydroelectric benefits very early on or can be built more quickly, it is not economically feasible. The population growth rate and shadow prices for labor have a marginal effect on NPV and are inconsequential factors."

Of course, one could never find an explanation like this in the official record, because the IDB only finances economically feasible projects. In this case, the execution period assumption is the most obvious and easiest one to fix. If the analyst managed to shorten the execution period by 1 year, say, through negotiating an accelerated construction schedule with the project's borrower and executor, the project could be nudged into the feasible region and its financing could be approved. Then, similar official conclusions about sensitivity would emerge in mirror image, substituting "shorter" for "longer execution period," "decrease" for "increase of 10 percent in benefits," and so on. And the analyst would probably be right, but perhaps for the wrong reason, unless the revised execution period could truly be said to be consistent with reasonable expectations. The right reasons emerge from a more profound treatment of perceived uncertainty.

What has been overlooked here? Uncertainty about benefits, as it often is. Using the mean WTPs from the subjective distribution and the baseline assumptions for all other variables would have produced a less negative NPV of –R$12.11 million, which grows to a positive R$109.85 million with our unequally weighted average of high and low nonparametric WTPs, and swells to R$263.27 million using Kriström's intermediate means. The project's economic feasibility is so firmly established using the latter WTPs that it becomes impervious to the magnitudes of the changes customarily invoked in the up-and-down elevator economics of standard sensitivity analysis. The trouble with sensitivity analysis is that the real distance the elevator has to travel between the basement and the penthouse and how many stops it makes along the way are unknown.

Sensitivity analysis is better than no analysis, but it has flaws. First, it gives the decision maker no context in which to evaluate the likelihood of a 25 or a 10 percent adverse change. For example, the decision maker has absolutely no idea of the likelihood of a 25 percent adverse change in the willingness-to-pay benefits. Had the analyst used only one estimate of WTP, the Turnbull, he or she would have found that the 99.5 percent confidence interval has a range of more than ± 50 percent, which is much greater than the arbitrarily chosen ±25 percent. The range, considering alternative central tendency estimates, is greater yet. A sensitivity analysis that calculates "switching values" (the last column of Table 9-6) provides more information. It indicates the magnitude of the favorable change necessary to bring the NPV to 0, one factor at a time. This is better than arbitrarily limiting the sensitivity analysis to standard, prespecified percentage changes from the initial conditions, but it provides the decision maker with no information on the probability that the factor could reach the switching value.

A second flaw of sensitivity analysis is that it usually does not look at combinations of adverse assumptions. Combinations can be important. Two of the three most important benefits of the Tietê project are the WTP benefits for groups in districts contiguous and not contiguous to the river. These benefits depend on the same thing, improvement of water quality to a certain minimum standard. They are highly (or perfectly) correlated. If the WTP of those contiguous to the river is lower than expected, so will be the benefits of those who are not contiguous. The calculation of the impact on NPV of an adverse or favorable change in one while holding the other constant is misleading. It will underestimate the adverse impact on NPV. In general, sensitivity analysis provides no indication of the probability or impact on NPV of several adverse or favorable changes occurring simultaneously rather than singly (e.g., cost overruns, low WTP benefits, and execution delays). Thus it is a poor representation of real world uncertainty.

As emphasized throughout the text, almost all of the components of the cost and benefit estimates that go into a cost-benefit analysis have a margin of error. If the probability that all the "best estimates" are simultaneously correct is unknown, the probability that the deterministic outcome predicted with these estimates will occur is also unknown. To accurately reflect what we think we know about the potential return of a project, it is necessary to use all the information that we have about the estimates and their margins of error. The obvious technique for doing this is Monte Carlo simulation.

MONTE CARLO RISK ANALYSIS

Monte Carlo simulation overcomes the limitations of sensitivity analysis. It produces a probability distribution of a project's net return based on either empirical or judgmental probability distributions for the factors that determine the outcome. It does this by repeatedly computing the economic return, each time picking input values from their respective probability distributions, to obtain a frequency distribution of economic returns that is a good approximation of the probability distribution of the outcome (see Vose, 1996; Morgan and Henrion, 1990).[31]

Table 9-7 summarizes the variables simulated, the range of values assumed, and the shapes of the probability distributions used for our risk analysis. Many of the probability distributions are subjective rather than empirical. They reflect a judgment as to whether any number within a range is equally likely (a uniform distribution) or whether some values are more likely than others (a normal, triangular, or custom histogram distribution). The most important judgmental distributions involve the timing of energy generation benefits and the range of WTP.

Monte Carlo Simulation Results: Incremental Project (Stages II and III)

Two Monte Carlo analyses were made. One considers the project as a whole (Stages I, II, and III); the second considers only the second and third stages. The second analysis is relevant for the decision to continue the program, since the first stage has already been carried out and its capital costs cannot be recovered. For now the analysis concentrates on the results for the analysis of Stages II and III under the alternative assumptions that WTP is distributed following either the subjective or weighted bounds models.

NPV under the Subjective WTP Distribution

Figure 9-4 presents the probability distribution of the NPV calculated for Stages II and III based on the subjective distribution of WTP. The distribution of the NPV outcomes provides the decision maker with a rather different picture from that provided by the deterministic calculation with sensitivity analysis. It shows an average NPV of R$4.6 million, which changes the investment decision from "no-go" to "go" because it is positive and higher than the deterministic loss of R$12.1 million mentioned earlier.

The expected NPV is higher because the subjective probability distribution for WTP allows for the possibility that average willingness to pay is very high (its probability distribution is skewed to the right). A few draws of high WTP values pull up the simulated average NPV.

Figure 9-4 indicates that there is more risk than the decision maker would have known about had he or she been given a standard sensitivity analysis. All the sensitivity analysis said was that if the project were undertaken, it would incur a loss unless the execution period could be shortened by 17 percent or unless energy benefits could come online in two-thirds the time

[31] For this exercise, a combination of Microsoft Excel and Crystal Ball from Decisioneering, Inc. in Denver, Colorado, was used. Crystal Ball can exploit a spreadsheet's ability to slide blocks of benefit and cost flows up and down across analysis periods in repeated draws.

Table 9-7. Assumptions about Input Variables in the Monte Carlo Simulations

Variable	Distribution	Measures of Central Tendency and Spread	Remarks
Operating and investment cost variation factor	Subjective, triangular	Most likely = mean=1.0 Range = 0.85 to 1.15	Symmetric. Total costs are multiplied by factor
Skilled labor shadow price factor	Subjective, triangular	Most likely=mean= 0.79 Range = 0.75 to 0.84	Asymmetric. Costs are multiplied by factor
Unskilled labor shadow price factor	Subjective, triangular	Most likely = 0.48 Mean = 0.49 Range = 0.45 to 0.55	Asymmetric. Costs are multiplied by factor
Nontradables shadow price factor	Subjective, triangular	Most likely = 0.75 Mean = 0.78 Range= 0.67 to 0.91	Asymmetric. Costs are multiplied by factor
Execution period, Stage II	Empirical, custom histogram	Mean = 6.0 years Range = 5 to 10 years	<u>Range, Years</u> — <u>Rel. Probability</u> 5.0–6.0 — 0.615 6.0–6.5 — 0.231 6.5–7.0 — 0.077 7.0–10.0 — 0.077
CV Benefit (WTP/household) contiguous, wtd. ave. of Turnbull and Paasche means	Empirical, each normal	Mean = R$7.75/month S. E. = R$1.23	Reflects methodological and statistical uncertainty
CV Benefit (WTP/household) noncontiguous, wtd. ave. of Turnbull and Paasche means	Empirical, each normal	Mean = R$5.80/month S. E. = R$0.86 Correlated with contiguous (r=0.80)	Reflects methodological and statistical uncertainty
CV Benefit (WTP/household) contiguous, expert judgment estimate	Subjective, custom histogram	Mean = R$7.09/month S. E. = R$10.74	<u>Range, R$</u> — <u>Rel. Probability</u> 2.00–4.00 — 0.30 4.00–7.00 — 0.50 7.00–10.00 — 0.15 10.00–50.00 — 0.04 50.00–140.00 — 0.01
CV Benefit (WTP/household) noncontiguous, expert judgment estimate	Subjective, custom histogram	Mean = R$3.76/month S. E. = R$4.95 Correlated with contiguous (r=0.80)	<u>Range, R$</u> — <u>Rel. Probability</u> 0.50–2.00 — 0.35 2.00–4.50 — 0.45 4.50–7.00 — 0.14 7.00–10.00 — 0.03 10.00–25.00 — 0.02 25.00–60.00 — 0,01
Energy benefit factor	Subjective, triangular	Mean = 1.0 Range = 0.7 to 1.30	Symmetric. Total value in R$ multiplied by factor
Energy scenario	Subjective, custom	Not applicable; categorical; 50% chance of no benefits	Asymmetric, discrete. See text for probabilities and timing of the 5 scenarios
Population growth rate, contiguous	Subjective, triangular	Mean = 0.75% per year Range = 0.5 to 1.0%	Symmetric. Small range due to urban space constraints
Population growth rate, noncontiguous	Subjective, triangular	Mean = 1.0% per year Range = 0.75 to 1.25%	Symmetric. Small range due to urban space constraints

originally expected. It consequently advised that project approval should be delayed until a new execution timeline could be worked out with the borrower. Most decision makers would find it useful to know the amount of the average loss they face if in fact there is a loss. This loss can be estimated as the probability of each loss times its size (the mathematical expectation of losses). The decision maker can look at the size of this loss to see if he or she can afford to take the risk.

The expected loss, however, has a more sophisticated interpretation and use. The expected loss is also the *cost of uncertainty*. It reflects the maximum amount of money a risk-neutral decision maker should be willing to pay to eliminate all uncertainty about the factors that determine the outcome. It is the maximum amount because it would be better to face the expected loss than pay more than that to obtain additional information to eliminate uncertainty. With information about the expected loss in hand, the decision maker can choose to (a) undertake the feasible project but risk an expected loss; (b) reject the feasible project, avoid the cost of uncertainty, but face an expected loss arising from the potential gains forgone (called the "cost of irrationality"); or (c) collect more information to reduce the cost of uncertainty.

Figure 9-4 and Table 9-8 show that although approval need not be delayed, since the investment passes the positive NPV test, there is a 73 percent chance that the project will register a negative NPV (is not economically feasible) and only a 27 percent chance it may register a gain. The expected loss that would be incurred by making the investment is the average of the negative NPVs, –R$228.1 million, multiplied by the probability of a loss, yielding –R$167.2 million. The average potential gain from the project is huge, R$643.3 million, and when multiplied by the probability of a gain (27 percent), the expected gain of R$171.8 million slightly exceeds the expected loss. The difference between average gains and losses, both weighted by their respective probabilities of occurrence, is the overall expected NPV of R$4.6 million. The subjective distribution of WTPs tells decision makers that a loss is three times more likely than a gain, but the average payoff is also about three times higher. However, the median payoff that can be expected with 50 percent probability is negative (–R$150.1 million) and far below the mean because the NPV distribution is skewed to the right. Risk-averse investors might be wary of this bargain, even though it has a small positive expected overall payoff.

Figure 9-4. Incremental Project NPV under Subjective WTP Distribution

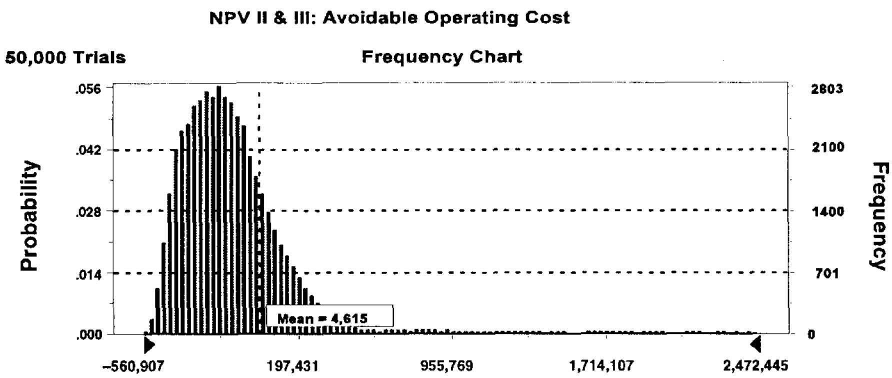


Table 9-8. Monte Carlo NPV Distribution for the Subjective WTP Assumption

Percentile	Million 1998 R$
0	–560.9
10	–384.6
20	–319.5
30	–261.8
40	–206.3
50% (Median)	–150.1
60	–93.2
70	–26.0
80	64.7
90	228.0
100	13,135.4

Distribution of NPV for the Weighted Average of the Upper and Lower Bound WTP Distributions

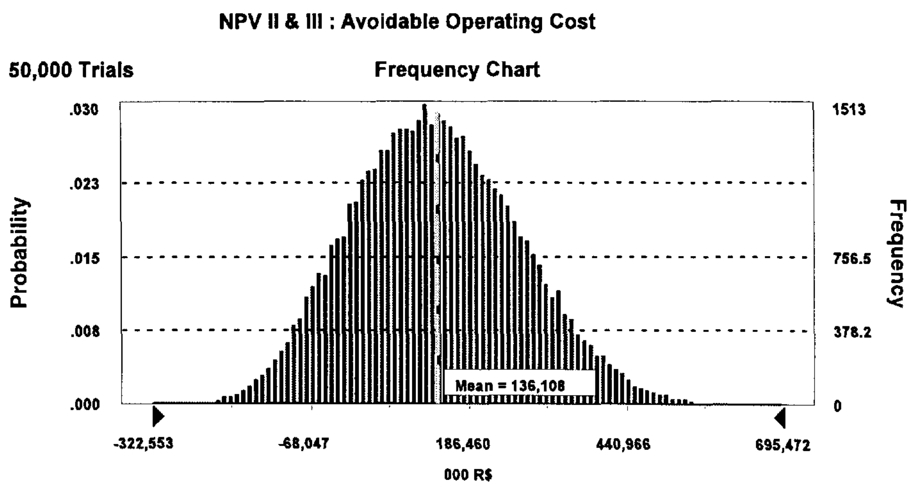
The degree of apparent risk associated with the incremental investment is reduced appreciably when the subjective WTP distribution is replaced by a 75-25 weighted average of the Turnbull lower bound and Paasche upper bound distributions of WTP. In contrast with Figure 9-4 and Table 9-8, the distribution of NPV in Figure 9-5 and Table 9-9 is more symmetric, which reflects the strong influence that the underlying normal distributions of the expected value of WTP have on the outcome.

The global expected value of NPV is more sanguine yet at R$136.1 million, which is almost 1 standard error away from the critical value of 0. The investment no longer looks so borderline because the probability of a loss is just 16 percent, which is almost exactly what would be expected for a normally distributed random variable (from the cumulative unit normal distribution, the probability of a value more than 1 standard error above 0 is 15.87

Table 9-9. Monte Carlo NPV Distribution for the Weighted Average WTP Assumption

Percentile	Million 1998 R$
0	–322.6
10	–36.5
20	17.9
30	59.7
40	96.6
50% (Median)	131.7
60	167.2
70	205.9
80	251.8
90	314.2
100	695.5

Figure 9-5. Incremental Project NPV under Weighted Nonparametric WTP Bounds Distribution



percent). The average loss falls to –R$60.8 million, or only one-fourth of what was predicted under the subjective WTP distribution, so the expected loss that might be suffered by making the investment becomes –R$9.9 million rather than –R$167.2 million. The average of the positive NPVs, R$174.3 million, produces an expected gain of R$146.0 million after multiplying by the probability of a gain of 84 percent. Again, the difference between the expected gains and losses (R$146.0 million – R$9.9 million) equals the global mean NPV. Even a risk-averse investor would probably like these odds.

Summing Up: From Frog to Prince

The standard decision rule for cost-benefit analysis is that the government should undertake all projects with a non-negative expected NPV. This rule is based on the assumption that a government undertakes many projects and can afford to look at the overall average result across its investment portfolio, knowing that some projects will come out better than average and some worse.[32] But when a single project absorbs a large percentage of the investment budget, a large adverse result may prevent undertaking other projects. There may be no chance to let project returns average out. The stages of the Tietê constitute a large project both for SABESP and for the city of São Paulo, so it is certainly worth looking at information other than the expected (mean) NPV. We have presented just two alternative versions about the worth of the investment based on risk analysis. Potentially there are several other interpretations, which are summarized in Table 9-10.

If one were to argue that operating costs from Stage I should be ignored (are sunk), the economic justification for the incremental project is extremely strong.[33] The mean NPV is over 3 standard deviations above 0 for all benefit assumptions except the subjective distribution, which places the mean 0.4 standard deviations away from the breakpoint. Even if Stage I's operating costs are included in the cost stream (are "avoidable"), the incremental investment looks extremely promising if it is evaluated using the intermediate nonparametric distribution of WTP. It still remains fairly promising when a weighted average of the lower and upper bound distributions of the expected value of WTP are used instead.[34] A recommendation against going forward with the project can only be made if the best representation of public good benefits is taken to be the conservative Turnbull lower bound WTP distribution.

What initially looked like an unattractive investment under a deterministic analysis that used conservative (low) estimates of average benefits was transformed into an attractive but risky or moderately risky investment when the nuances of benefit uncertainties and random variations in costs and timing were incorporated via Monte Carlo risk analysis. This transformation is not trickery, but a reflection of how much (or how little) the analyst knows. There is no unique bottom line in this case; it only proves that rigid adherence to a deterministic non-negative NPV investment criterion can be simplistic, dogmatic, and ofttimes misleading. While no actual cost-benefit analysis would ever report this broad spectrum of alternatives for fear of needlessly confusing decision makers, Table 9-10 makes it clear that conclusions about the economic viability of an investment are rarely free of subjective judgment. Risk analysis forces the full disclosure of those assumptions.

[32] This can imply carrying out a project with a non-negative expected NPV even though the probability of non-negative returns is less than 50-50 because the probability distribution of NPVs is highly skewed to the right.

[33] This was the preferred assumption in the official cost-benefit analysis done by three of the authors for IDB loan approval.

[34] To drive the overall expectation for NPV below 0, on average the weight attached to the lower bound mean WTP would have to be 0.93 rather than 0.75.

Table 9-10. Effect of Benefit Assumptions on Risk Analysis Results for the Incremental Project

	Central Tendency			Downside Risk		
WTP Distributional Assumption	Mean NPV (10^6 R$)	Standard Deviation (10^6 R$)	Median NPV (10^6 R$)	Probability of Losses (%)	Average of Losses (10^6 R$)	Expected Loss (10^6 R$)
Operating Costs of Stage I Are Avoidable (Are Charged to Stages II and III)						
Turnbull lower bound	($24.4)	$91.0	($26.9)	61.6%	($81.3)	($50.1)
Subjective	$ 4.6	$822.6	($150.1)	73.3%	($228.1)	($167.2)
Weighted average	$136.1	$134.9	$ 131.7	16.2%	($60.8)	($9.9)
Kriström intermediate	$298.5	$131.6	$ 296.0	0.8%	($37.9)	($0.3)
Operating Costs of Stage I Are Sunk (Are Not Charged to Stages II and III)						
Turnbull lower bound	$295.5	$87.8	$292.9	Negligible	($9.9)	~0
Subjective	$324.6	$822.3	$169.2	19.9%	($68.1)	($13.6)
Weighted average	$456.0	$133.3	$451.7	0.0%	$0.0	$0.0
Kriström intermediate	$618.4	$130.4	$615.5	0.0%	$0.0	$0.0

Note: Numbers in parentheses are negative (losses).

Risk Analysis of the Entire Project (Stages I, II, and III)

The IDB's loan for Stage I of the Tietê project was approved around the time of the heavily publicized United Nations Conference on Environment and Development held in Rio de Janeiro, Brazil, in 1992. The event raised expectations and encouraged countries to pursue sustainable development following the principles of *Agenda 21*. Cost-effectiveness analysis was used to justify going ahead with the first stage of investment, on the unproven assumption that the entire project was economically viable. This decision locked the borrower into contractual obligations for operation of the first-stage facilities of the project, repayment of the first-stage loan, and continuance of the pollution control program. Now, having made that initial commitment, the best choice is to go ahead and complete the program.

Figure 9-6 and Table 9-11 show that even under the highest of the plausible benefit estimates, Kriström's intermediate mean WTP, the entire project is not economically feasible. At best, the expected NPV is –R$230 million, and would be even lower under more conservative assumptions about benefits. This is an example of the rare extreme case mentioned in the

Figure 9-6. Entire Project NPV under Kriström's Nonparametric WTP Distribution

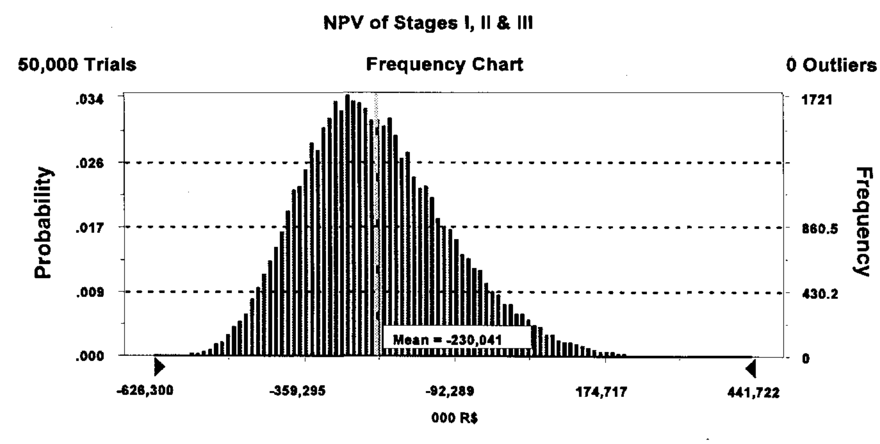


Table 9-11. Monte Carlo NPV Distribution for Entire Project under Kriström's Intermediate WTP Assumption

Percentile	000 R$
0	–626,300
10	–388,368
20	–342,288
30	–306,060
40	–274,227
50% (Median)	–242,367
60	–207,978
70	–170,313
80	–123,140
90	–53,198
100	441,722

introduction: The project fails the deterministic cost-benefit test even under the most favorable set of assumptions. So in hindsight, sophisticated analytical approaches would not have been needed to reach the conclusion that the investment should not be made. However, at the time the first stage was approved, benefit estimates were not available, so the expected payoff was unknown.

The probability distribution in Table 9-11 shows that the chance of losing more than R$230 million is about 56 percent. At best, the probability that the entire project might have a non-negative NPV is only 5 percent, so the probability of incurring a net discounted loss is greater than 95 percent. The cost of uncertainty (the expected positive NPVs forgone by not making the correct decision to reject the project) is only R$3.5 million, and the cost of irrationality (the expected value of losses from undertaking a project that is predicted to be infeasible) is R$233.6 million.

It is worth asking why there is such a difference between the incremental project (Stages II and III) and the project as a whole. First, the cost of the project as a whole includes the investment costs of Stage I, which (although necessary for subsequent stages) had no measurable benefits. These costs weigh heavily against the NPV of the project as a whole, particularly since they occur years before any benefits begin to accrue. Second, the risk of delays between the first and second stages affects the net present value of the project as a whole, but not that of Stages II and III (both costs and benefits of the incremental project are delayed). These two factors have a significant impact on the NPV of the three stages taken together.

Benefit Uncertainty and the Value of Information

The *value of information* is defined as the decrease in expected loss that results from greater precision in estimates about the variables that contribute to expected loss. To determine whether it is worthwhile to collect additional information, it is necessary first to determine which estimates most affect the spread in NPV. Table 9-12 shows that uncertainty about WTP benefits ranks highest among the possible candidates, explaining more than two-thirds of the overall variation in NPV across a variety of circumstances. Having identified the main sources of uncertainty, it is then necessary to estimate the reduction in uncertainty that would result from further study or additional sampling (e.g., what the new probability distribution about the WTP estimate would look like if more data were collected). The value of more precise WTP information is determined by the difference between the expected losses of two simulations: the original risk analysis and the modified analysis that substitutes the probability distribution that would result from further study.

Table 9-12. Contribution of Uncertainty about WTP Benefits to Total NPV Uncertainty

WTP Distributional Assumption	Standard Deviation of NPV for Variation in: WTP Benefits Only (million R$)	All Variables (million R$)	Amount of Total Variation Due to Uncertainty about WTP Benefits [a] (%)
Incremental Project: Operating Costs of Stage I Are Avoidable (Charged to Stages II and III)			
Turnbull lower bound	$64.6	$91.0	71%
Subjective	$803.5	$822.6	98%
Weighted average	$108.2	$134.9	80%
Kriström intermediate	$94.7	$131.6	72%
Incremental Project: Operating Costs of Stage I Are Sunk (Not Charged to Stages II and III)			
Turnbull lower bound	$64.6	$87.8	74%
Subjective	$803.5	$822.3	98%
Weighted average	$108.2	$133.3	81%
Kriström intermediate	$94.7	$130.4	73%

[a]Calculated as the ratio of the standard deviations. When only variation in WTP is allowed, the default settings for the frozen variables are at the means (see Tables 9-6 and 9-7).

To decide what information should be collected and how far to go in collecting it, it is necessary to compare the value of information computed in the manner described above with the cost of achieving the reduction in uncertainty. This kind of analysis is not as simple as it sounds, and the next chapter is devoted to explaining it (Vaughan and Darling, 2000).[35]

CONCLUDING OBSERVATIONS

This chapter demonstrates that any single-point estimate of the net economic returns of a project does not provide a decision maker with much information and is likely to be wrong most of the time. Accompanied by the standard sensitivity analysis that assumes variations that may have nothing to do with the true margin of error, it may mislead the decision maker into being confident about a decision that in fact is risky, or into being skeptical about a promising investment opportunity.

While it is true that many of the probability distributions in the analysis are subjective, they at least reflect a focused and explicit judgment by the analyst. They may not all be based on empirical facts, but they reflect the information at hand. The economic results that come from these probability distributions are also not empirical facts, but they are consistent with best judgments of the analyst and open for all to review and question.

This case study also demonstrates that it is important to carry out cost-benefit analysis of projects, particularly multistage projects, before the first stage is initiated. Least-cost analysis alone is not sufficient. It is true that the requisite cost-benefit information about inputs, values, and timing for projects with long maturation periods is necessarily imprecise at early stages. However, the cost-benefit risk analysis technique provides a method for meaningfully summarizing what analysts know about a project and for helping design a cost-effective strategy to reduce this uncertainty, using the concept of the value of information.

[35] It is not necessarily true that the factor that has the highest value of information is the one that should get the most attention. It may be that other important factors can be made more precise at lower cost. Our emphasis has been primarily on issues involving nonmarket valuation. The next chapter, drawn from Vaughan and Darling (2000), elaborates along this line by looking at the effect that larger CV surveys can have on reducing the uncertainty about WTP and NPV. Strategies for reducing the variance of cost estimates or other variables are not covered.

In the case of the Tietê, it is possible that decision makers, had they known that the expected NPV of the overall project was negative, would have started the project anyway. Economic reasons are not the only reasons for doing things. It is, however, also possible that having seen the expected results, the decision makers might have changed the sequence of investments, with better results. It is clear that economically it would have been better to invest first in connections and collection systems that had high economic returns. Having done that, they could have concentrated the investment in cleanup in a shorter period and increased the NPV of the cleanup project, although it probably still would not have had a positive NPV.

When millions of dollars are at stake in water pollution control program investments, cost-benefit analysis is worth the effort, even though the benefits of water quality improvement are hard to specify precisely (Wattage et al., 2000). However, that very uncertainty means that an honest economic appraisal should probably include a good risk analysis. We economists sometimes know less than we pretend and than others expect. These conclusions are hardly new (see Jenkins, 1997), but unfortunately they seem to be too easily forgotten precisely at the times when their message means the most.

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Chapter 10

The Value of Information: Finding the Optimal Sample Size for a Contingent Valuation Survey

As noted in Chapter 1, like many multilateral financial institutions, the IDB requires that investment project proposals be screened using economic analysis, preferably of the cost-benefit rather than the cost-effectiveness variety. Chapter 2 showed that in recent years, contingent valuation has become the preferred route for estimating the benefits yielded by investments aimed at improving ambient air or water quality (Ardila et al., 1998). Chapters 8 and 9 discussed the uncertainty surrounding point estimates of benefits, how to handle the uncertainity by doing a probabilistic cost-benefit analysis, and introduced the notion of an empirical *probability distribution* for NPV. Now, that idea can be used to answer a question that is repeatedly raised by those responsible for actually doing economic analysis based on CV (staff economists and their counterparts in the executing agencies of prospective borrowing countries). It is: "How many subjects should be interviewed in a CV survey to produce a reliable estimate of per capita or per household benefits?"

It would be hard to disagree in principle with Mitchell and Carson's (1989, p. 223) admonition that "No matter how realistic the scenario, the data collected from even the best contingent valuation survey instruments are useless unless sample statistics of sufficient quality for policy purposes can be obtained." In practice this message is often lost in decision making, even when investments of a quarter of a billion dollars or more are at stake. In testimony to its obscurity, no guidelines on sample size selection exist in the economic analysis protocols of the Inter-American Development Bank, and our conversations with economists at sister lending institutions suggest that they are perplexed as well.[1]

In these days of low overheads and limited budgets in public sector and international institutions involved in grant or loan financing of projects and programs, management is justifiably reluctant to overspend on project preparation. One high-level IDB executive with considerable expertise in economic analysis remarked to the authors that he suspected, without looking deeper into the question, that CV samples in the area of 250 interviews would probably be sufficient for a reliable cost-benefit project appraisal. The literature contains little that refutes this conventional wisdom.

[1] In the only known internal IDB source discussing sample size, Vaughan (1994) looks at the size of a sample of projects needed to confirm the overall economic viability of so-called "multiple works programs" which are designed to finance a group of similar works, of which only a subset is exposed to ex ante economic analysis. The issue of CV sample size is not addressed therein. At a training workshop on CV presented by the authors at the World Bank in June 1999, the sample size question was raised by the audience, but we were unable to provide a wholly satisfactory answer beyond intuitively suggesting that samples of between 500 and 1,000 usable observations probably would be sufficient.

STANDARD SUBOPTIMAL APPROACHES

In their seminal book on CV, Mitchell and Carson (1989) suggest that based on a simple statistical tolerance formula, sample sizes between 200 and 2,500 are probably appropriate (Chapter 10, footnote 13, p. 225), assuming a best guess of 2.0 for the coefficient of variation that drives the calculation. The academic literature on CV has little more to say that is useful. Instead, and perhaps quite realistically, it devotes a great deal of effort to algorithms for figuring out how many bid groups should be used in a referendum CV survey, and how many interviews should be allocated to each, taking the total sample size as exogenously set by the research funding limits (Cooper, 1993).

Given the paucity of guidance, determining an appropriate sample size is often handled loosely by everyday practitioners in government and international agencies, whose work is often underfunded and done under time limits. Budget considerations play a dominant role. However, does it make any sense to divide a predetermined budget amount (net of fixed costs) by the cost per interview to get to a CV survey sample size? What determines the level of the budget for analysis and CV surveys? Yet, is it any more legitimate to challenge any budget ceiling on the basis of a statistical tolerance formula that says "We need N interviews to produce a sample estimate of mean willingness to pay that comes within ± X percent of the true population mean with $(1-\sigma)$ percent confidence?" There is no particularly strong rationale for choosing a percentage difference of, say, 5 percent rather than 10 percent between the true willingness to pay of the population and the sample mean, or any reason other than custom for selecting a significance level, α, of 1 percent rather than 5 percent.

Arguments for augmented survey budgets along these lines are unlikely to be persuasive to those who must allocate limited resources. Without a criterion that balances the value of additional information provided by larger samples with the cost of collecting it, the only recourse is to fall back on rough statistical rules of thumb (Mitchell and Carson, 1989) or do one's best with whatever funds are made available.

SEEKING OPTIMALITY

The immediate objective of much CV work is valuation per se, that is, estimation of the value of natural assets like protected areas, or the benefits of ambient quality improvements. While these CV values may serve as inputs into the assessment of the prospective monetary gains and losses of alternative investment decisions or courses of environmental policy action, the subsequent decision analysis is often someone else's responsibility once the job of the CV experts is done. When the CV exercise is effectively sealed off from the policy or investment decision step, CV researchers working in isolation have no recourse but to satisfice when it comes to sample size, choosing a size N that somehow is defensible or appears reasonable. However, investigators using contingent valuation results directly in cost-benefit analysis have a hidden advantage that can be exploited to reach a much more persuasive conclusion about the optimal CV sample size. This chapter explains how the value of additional sample information can be quantified (either approximately or precisely) and balanced against the cost of obtaining it. That value is closely related to the way the distribution of the investment's expected net present value reacts to the size of a CV sample.

The optimal sample size approaches explained here have their origins in Schlaifer's (1959, 1961) Bayesian decision analysis approach, which is discussed in an accessible way by a number of standard texts on the use of statistics in business decision making (Bonini et al.,1997; Jedamus and Frame, 1969; Pfaffenberger and Patterson, 1987; Lapin, 1994; Winkler, 1972). For a rapid Bayesian analysis, an approximate normal probability distribution for NPV can be obtained by constructing a simple linear relation between it and sample estimates of the mean WTP and its standard error. Larger CV samples reduce the uncertainty about willingness to pay (reduce the standard error of mean WTP) and about NPV as well, where the compression in variance (uncertainty) with increasing sample size is transmitted through the linear relationship. By monetizing variance reduction in this way, the marginal costs of expanding sample sizes can be compared with the marginal benefits of the additional information they contain to reach an optimal sample size decision.

Alternatively, the linearity assumption linking NPV to mean WTP may not hold, and NPV may not be normally distributed even if WTP is, because other random influences on NPV skew the distribution of the outcome. A more precise Monte Carlo risk analysis in the Bayesian mold can be employed to characterize the way the empirically generated probability distribution of NPV reacts to better information on WTP to verify the approximate optimum based on the normality assumption.

OPTIMAL RESULTS WITHOUT PAIN

Anyone can implement the method developed in the remainder of this chapter using a spreadsheet algorithm in Quattro Pro that is available from the authors on request.[2] The optimization routine presumes that an initial small survey sample has already been taken, and asks whether it would be optimal to add to it in a second round of sampling. To run the program, all the user has to do is click on an "Optimizer Macro" button to compute the optimum number of additional observations needed to augment an initial "small" survey, if any. The data entry and results forms are shown in Table 10-1. Annex 10-A to this chapter contains the full set of spreadsheet instructions.

Only six pieces of input information are needed: (a) the size of the initial small CV survey sample, (b) the expected value (mean) of willingness to pay extracted from that sample, (c) the variance of mean WTP, (d) the average (equals marginal) cost of collecting a single survey observation, (e) the intercept of a linear function [called the "conditional value of perfect information" (CVPI) function] relating NPV to WTP, and (f) the slope of the linear function. All but the last two are obvious. The intercept and slope of the CVPI function are also easy to get without doing a complex cost-benefit analysis beforehand. The intercept is just the discounted sum of project investment and operating costs (net of any non-CV benefits, if they exist). The slope is just the discounted sum of the number of beneficiaries to whom mean WTP benefits from the CV survey are attributed. Subsequent sections develop the rationale for the proposed method and explain each step in detail.

ANTICIPATING THE OPTIMAL RESULTS FOR A CASE STUDY EXAMPLE

It is always difficult and perhaps even dangerous to make generalizations about optimal sample size. The best sample size is always case specific because it depends on the cost and benefit flows of the prospective investment under consideration. Yet there is a general pattern: The better the project appears ex ante, the smaller the CV sample size needed to obtain the benefit estimates that justify it. Given the investment cost and return data for the example used later, the total sample size required depends critically on just two pieces of information: the standard-

[2] The sample size template in Quattro Pro Version 9 is available on the Environment Division Web Site at *http://www.iadb.org/sds/ENV/publication/publication_488_2002_e.htm*. The template is not available in Microsoft Excel.

Table 10-1. Quattro Pro Macro: Data Input Form and Optimal Results Summary Output

DATA ENTRY			INSTRUCTIONS:
STEP I. ENTER THE INITIAL SMALL SAMPLE DATA	**Units**	**Data Entry**	ENTER DATA IN BOX AT LEFT
Size of Initial "Small" Sample?	# of Cases	250	
Sample Mean Willingness to Pay?	$/Household/Unit Time	$7.47	
Variance of Sample Mean?	$/Household/Unit Time	0.70	
Sampling Cost per Household Interview?	$/Case	$89.00	
STEP II. SPECIFY THE LINEAR CVPI FUNCTION RELATING NPV TO WTP (NPV = α + β · MEAN WTP)			
Intercept (α) ?	$ Total Discounted Costs [Enter as Negative #]	-$594,653,984.00	
Slope (β) ?	# of Beneficiaries [Total Discounted]	100,988,487	
			AND THEN
RESULTS			
STANDARD ERRORS OF NPV AWAY FROM NPV = 0		**1.89**	
SHOULD A SECOND SAMPLE BE TAKEN TO AUGMENT THE INITIAL SAMPLE?		***Probably Yes***	
IF "Yes" CLICK ON THE BUTTON AT THE RIGHT TO RUN THE OPTIMIZER MACRO		**OPTIMIZER MACRO**	HIT THE OPTIMIZER BUTTON
Approximate Sample Size (Used as a Starting Value for Optimization)		2,793	
EXACT OPTIMUM		**2,378**	PROGRAM RETURNS THE OPTIMUM

Note: This routine assumes the analyst has no prior knowledge about average WTP or its variance beyond what the initial "small" sample reveals. Neither the authors nor the Inter-American Development Bank warranty this program or the methods it employs.

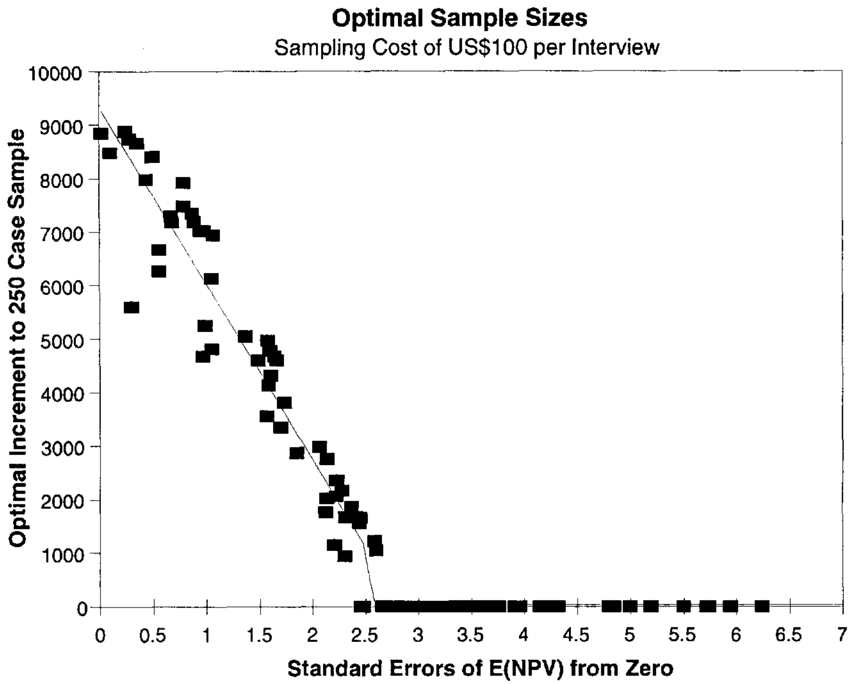
ized distance of expected NPV from 0 and the unit variable cost of collecting a single WTP survey response. In our experience in Latin America and the Caribbean, sampling costs will be high if an interview costs around U.S.$100, and more moderate if interviews cost about U.S.$20 to U.S.$35 each.

The remainder of this chapter uses actual data from our Tietê case study to illustrate the general principle that the optimal sample size for CV surveys is a decreasing function of how good the project appears to be ex ante. The results are based on the benefits and costs of the proposed project to clean up the Tietê River in São Paulo, Brazil (for details about the project, see Chapters 8 and 9 and Vaughan et al., 1999, 2000a, 2000b).[3]

Figure 10-1 shows the results of calculating the optimal sample size for 100 randomly drawn values of mean WTP and its standard error, assuming a high cost of sampling of R$89 (U.S.$100) per interview. The figure shows that investments with a high likelihood of success

[3] The figure assumes an initial small sample of 250 cases has already been taken and shows how many cases should be added to it. To generate the figure, the baseline deterministic cost and energy benefit flows from the case study example discussed in Vaughan et al. (1999, 2000a, 2000b) were combined with mean WTP estimates that were randomly drawn from a uniform distribution covering R$5.89 to R$13.00, which spans most of the range of the estimates discussed in Vaughan et al. (1999, 2000a, 2000b). The variance of each mean was derived from a random draw of values of the coefficient of variation between 0.75 and 6.0, following Mitchell and Carson (1989). No prior information about the mean WTP or its variance was incorporated, in contrast with Table 10-1, which includes some runs with tight priors.

Figure 10-1. Sample Size Depends on Project Prospects



(expected NPV more than 2.5 standard errors from 0 because the mean WTP is high and the variance of the mean is low) require very little sampling effort beyond taking a small sample of 250 or so observations.

An ex ante speculation about the chances that a project will succeed measures the degree of confidence. Extremely promising projects have large positive expected net present values, while marginal projects have NPVs near 0. By dividing the expected net discounted benefits of any project by the standard error of the mean NPV, any investment can be put on a commensurate scale that has a ready interpretation from basic statistics. Assuming approximate normality in the distribution around the estimate of expected NPV, projects whose standardized NPV is more than 2.33 standard normal deviates from 0 have a 99 percent chance of being successful; projects that are 1.28 deviates above 0 have a 90 percent chance of success; and so on down the line. An investment whose standardized NPV is expected to be only slightly above the critical value of 0 will fail half the time so it has only a 50 percent chance of success.[4] A summary of the results of a large number of other Bayesian optimization exercises not shown in Figure 10-1 appears in Table 10-2.

The optimal sample sizes[5] reported for variations on the case study data alone cannot be generalized to other situations. But exact optimal sample size answers can easily be found for any specific investment project using the optimization algorithm supplied in Annex 10-A. The interesting characteristic demonstrated by the results from the case study is that no specific sample size, either large or small, is desirable across the gamut of situations.

[4] To arrive at these percentages, the normal distribution in question must be centered on the expectation of NPV, not on 0.

[5] The minimum sample size is 250 cases because a small initial sample is needed to start the optimization routine. The net benefits of increasing the sample size above 250 cases rise initially at a rapid rate and then become almost flat over a wide span (± 20 to 30 percent) around the optimum. Consequently, the sample sizes in Table 10-2 are termed "near optimal" because the additional cases needed beyond the minimum of 250 have been scaled back to 70 percent of the optimum following Schlaifer (1961, Figure 21.5, p. 337); they are lower bound ranges.

Table 10-2. Range of Nearly Optimal Total Sample Sizes from the Tietê Case Study Data

	Chances of Project Success		
Interview Cost in Latin America	Over 99% (>2.33 S.E)	90 to 99% (1.28 to 2.33 S.E.)	50 to 90% (0 to 1.28 S.E.)
High (Brazil: U.S.$100 per interview)	250 to 1,150	250 to 3,300	1,600 to 8,600
Typical (U.S.$35 per interview)	250 to 2,000	250 to 5,200	3,000 to 14,400

Before getting involved in the details of the optimization approach, several variants of the standard classical method are illustrated using contingent valuation survey data that were collected in 1998 to estimate the water pollution control benefits of the Tietê project. The results of exploring several different sets of assumptions with the classical sample size selection method reveal just how little guidance it actually provides. Small sample means of WTP and their variances are constructed from the data before moving on to explain and apply the principles of sequential determination of optimal sample size using data from a small initial sample as a point of departure.

THE STANDARD APPROACH TO DETERMINATION OF SURVEY SAMPLE SIZE: VARIATIONS ON A THEME

For its referendum CV questionnaire to estimate willingness to pay for improved water quality in the Tietê River, the original project analysis drew a sample based on household characteristics drawn from a 1996 survey of households in the Saõ Paulo metropolitan area. The strategy was to represent the population of São Paulo in terms of the factors thought to have a strong influence on willingness to pay. In theory such factors might include the household's income and its perception of odors from the river, environmental awareness, and education. Of these, the census only had information on income and education, which are highly correlated.

According to the census, the average household income in the SPMA is R$828/month with a standard deviation of R$702. Using a 95 percent confidence interval and a 10 percent sampling error, the original analysts figured that a sample of 276 homes would be required. The necessary sample size was initially calculated based on the amount of tolerable error in the sample estimate of mean income rather than mean WTP (which was unknown), using a standard statistical formula that acknowledges only Type I error (Pfaffenberger and Patterson, 1987).

The standard formula used was

$$N = [z_{\alpha/2}\ \sigma/E]^2 = [(1.96 \times 702) / 82.8]^2 = 276$$

where:

- N = desired sample size
- z = the 95 percent confidence interval statistic (1.96) at significance level α = 5 percent, two-sided test
- σ = standard deviation of income (R$702)
- E = acceptable error (R$82.80) in the sample estimate of the population mean WTP obtained as one-tenth of the census estimate of average household income of R$828 (i.e., a 10 percent error)

Note that the variable of interest is household willingness to pay, not income, so the above application of the standard sample size formula only holds if the mean and standard deviation of WTP bear a fixed proportional relationship to the mean and standard deviation of income, which is unlikely.

In a second line of attack, the analyst might try to formulate explicit prior beliefs about the population mean and standard deviation based on historical experience and proceed from there. For instance, assume a simple distribution for willingness to pay, such as the triangular. The mean and standard deviation can easily be obtained from this distribution given a guess about just three values, the minimum, the most likely, and the maximum WTP (Vose, 1996):

$$\text{Mean, triangular} = (a + b + c) / 3$$

$$\text{Variance, triangular} = (a^2 + b^2 + c^2 - ab - ac - bc) / 18$$

where a is the minimum, b is the mode, and c is the maximum.

Establishing the minimum WTP is easy if the investment improves utility or at least does no harm; so it can be safely set to 0. For the maximum, we know from experience that, on average, people are willing to pay about 3–4 percent of their income for sewer connections and that the willingness to pay for ambient water quality is, on average, less than one-third of that (Chapter 6 and Ardila et al., 1998). So the maximum of WTP could be set to what households are willing to pay for sewer connections—around 3 percent of income. If income is approximately right-triangular distributed, the maximum income is three times the mean of R$828, or R$2,484. Then the highest individual observation of WTP would be 3 percent of R$2,484, or R$74.52. The modal value is more difficult, but Choe et al. (1996) found that willingness to pay for water pollution control in the Philippines was only about 1 percent of income, which is consistent with our experience in Latin America (Ardila et al., 1998), yielding a most likely value of R$8.28. Then, from the triangular distribution formulas, the prior mean WTP becomes R$27.60, the variance R$346.93, and the standard deviation, σ, R$18.63.

As before, if the sample is to come within ± 10 percent of the mean, E = R$2.76. Then:

$$N = [z_{\alpha/2}\ \sigma/E]^2 = [(1.96 \times 18.63) / 2.76]^2 = 175$$

This recommendation for a very small sample is based on ostensibly reasonable guesses. As we will see later, these prior estimates turn out to be extremely poor compared with the mean and standard deviation of the actual sample data.

For another variant of the same game, suppose we believe the *average* WTP per household is 1 percent of income, or R$8.28, and that the most frequent response (the mode) is 0. This yields a right triangular distribution with a = 0 and b = 0. The implied maximum WTP, c, is three times R$8.28, or R$24.84, and the variance is just $c^2/18$, or R$34.28, yielding σ = R$5.85. Again, if the sample is to come within ± 10 percent of the mean, E = R$0.83 and another recommendation for a small sample results:

$$N = [z_{\alpha/2}\ \sigma/E]^2 = [(1.96 \times 5.85) / 0.83]^2 = 191$$

Finally, for a fourth route, Mitchell and Carson (1988) suggest a clever manipulation of the standard formula above that obviates the need to guess about σ or the absolute magnitude of acceptable error in the mean of WTP. Instead, a guess about the ratio of the standard deviation to the population mean (the coefficient of variation, V) is required; Mitchell and Carson suggest a value for V of about 2. At the α =5 percent level, using V=2 and a Δ of 10 percent as an acceptable difference between the true population mean WTP and the sample estimate:

$$N = [(z_{\alpha/2}\ V) / \Delta]^2 = [(1.96 \times 2.0) / 0.10]^2 = 1{,}537$$

The four standard routes to sample size determination illustrated above lead to quite different answers because the first three implicitly assume that the value of V is less than 1.0 and therefore recommend small samples (i.e., the V values are R$702/R$828 = 0.85, R$18.63/

R$27.60 = 0.68, and R$5.85/R$8.28 = 0.71) rather than the value of 2.0 reflecting Mitchell and Carson's review of actual contingent valuation surveys in the 1980s, most of which were undertaken in developed countries.

The guessing game played above could go on indefinitely without ever producing a firm conclusion about the reliable sample size needed for any particular CV survey. Although it would seem to suggest that in developing country applications, small samples will suffice, this is an erroneous generalization. The rest of this chapter shows that small samples sometimes suffice, but the classical method fails to isolate the circumstances under which it is safe to take a small CV survey sample rather than a large one.

In any event, the first result, 276 households, was not used for the Tietê referendum CV survey. Neither were the second or third of 175 and 191 cases or the highest estimate of 1,537 from Mitchell and Carson's route. Instead, 600 interviews were actually undertaken for the project analysis, split between two subsamples to account for the distance effect on WTP (184 households living in districts bordering the polluted river and 416 living farther away and presumably less affected by its noxious odors and health risks). Even though the variable cost of each interview, U.S.$100, was fairly expensive,[6] the available budget permitted an expenditure of U.S.$60,000 to take a larger sample and get more precise results than the lowest estimate of U.S.$17,500 could provide, but not aim for the tighter variances from 1,537 interviews that could be purchased for U.S.$153,700. The question is, which of the sample size estimates, in retrospect, comes closer to the optimal size? The answer lies beyond the quick and convenient, but imprecise and arbitrary classical method.

STARTING TOWARD AN OPTIMAL SOLUTION: SMALL SAMPLE MEANS AND THEIR STANDARD ERRORS

The next step toward an optimal answer begins with the collection of an initial small CV survey sample (say, 250 cases) and the calculation of the mean WTP and its variance. With this information in hand, the researcher can then proceed to ask whether additional sample information would be desirable, following the optimization procedure explained subsequently.

But first, initial estimates that characterize the distribution of WTP are needed. In the case of WTP, the standard error of the mean can, in principle, be reduced by increasing the sample size. However, estimation of this effect using conventional parametric techniques on referendum CV data is problematic, since analytical formulas are generally lacking.[7] However, the nonparametric estimators (McConnell, 1995; Haab and McConnell, 1997a; Vaughan et al., 1999; Vaughan et al., 2000a, 2000b) directly relate the standard errors of lower bound, intermediate, or upper bound mean WTP estimates to sample size, and these formulas can be exploited to help compute the optimal sample size (see Annex 10-B). Alternatively, an initial

[6] This value was equal to $114 Brazilian reals in 1998. For the loss-cost exercise that follows, it must be put on equal terms with the value of information, which is shadow priced in the project analysis. Multiplying it by the shadow price of nontradable inputs of 0.78 gives a variable (equal to marginal) cost per observation of R$88.92. CV surveys in Latin America usually do not cost this much; costs around U.S.$30 per case have typically been quoted (Ardila et al., 1998).

[7] The parametric approach requires that a conditional cumulative density (or survival) function be statistically fit to the data and, subsequently, an expected value extracted using formulas that are functions of the estimated parameters of that assumed density (usually logistic; see Vaughan et al., 1999, 2000a, 2000b). Lacking analytical formulas, the mean standard error must be found either via the delta method (a second-order Taylor series approximation of an unknown variance function which itself depends on the standard errors of the survival function parameter estimates) or by bootstrapping.

Table 10-3. Small Sample Nonparametric Means

Estimator	Mean	Variance of Mean	Standard Error of Mean	Population Standard Deviation [a]
Turnbull lower bound	5.75	0.45	0.67	10.61
Weighted Turnbull (0.75) and Paasche (0.25)	7.47	0.70	0.84	13.23
Kriström's intermediate	9.20	1.02	1.01	15.97
Paasche upper bound	12.66	1.88	1.37	21.68

[a] Approximation from the square root of the product of the variance of the mean and the sample size, 250.

open-ended rather than referendum CV survey could be conducted and the mean and its standard error calculated directly from the explicitly stated WTPs of the respondents.

To demonstrate, a balanced random subsample of 250 observations was drawn from the actual 600-observation grand sample of Tietê CV survey interviews.[8] We chose a size of 250 in order to have a reasonable minimum number of observations in each of the five bid groups in the referendum. Unlike the grand sample, the small subsample was deliberately drawn to be representative of the spatial distribution of the respondents, so the distinction between the WTPs of residents living close to and far from the river can be dropped, which simplifies the problem of selecting sample size. Table 10-3 presents the small sample estimates of the non-parametric means and their variances, and Annex 10-B provides the details.[9] Our prior guesses for mean WTP and the population standard deviation in the section on the standard approach did not turn out to be very prescient, although setting the average WTP at 1 percent of income comes close (R$8.28).

To anticipate a bit, what would the classical method recommend if our guesses for μ and σ were exactly equal to what the actual sample reveals? The Turnbull mean WTP is R$5.75 and the sample estimate of the population standard deviation is R$10.61 (the standard error of the mean multiplied by the square root of 250). Applying the familiar formula, the recommended sample size under these nearly perfect guesses would be large indeed, and very close to what Mitchell and Carson's approach recommends:

$$N = [z_{\alpha/2}\, \sigma/E]^2 = [(1.96 \times 10.61) / 0.58]^2 = 1{,}286$$

Similar calculations for the other means in Table 10-3 also yield sample sizes of around 1,200 cases.

Unfortunately, the recommendation of the classical method, even if it is based on actual sample information, is potentially misleading. The optimal Bayesian decision under the base-line configuration of net project benefits and sampling costs, assuming a mean WTP of R$7.47,

[8] The actual 600-observation referendum CV sample from our case study was unbalanced because it undersampled households living in districts that are contiguous to the river (31 percent in the sample, 61 percent from the metropolitan area census). Since households living in districts bordering the river are willing to pay significantly more on average for improved water quality than households in noncontiguous districts (R$6.07 per household per month versus R$4.51), the mean from the grand sample is a biased estimate of the population's average willingness to pay. We corrected for this by randomly drawing 250 observations from the grand sample using the constraint of the census proportions, which meant that 152 of the 184 available households living close to the river were included in the small sample, along with 98 of the 416 families living in more distant districts.

[9] The variance estimates in the table were independently verified by simulation, drawing from separate binomial distributions reflecting the number of "no" answers in each bid group and repeatedly calculating the mean 5,000 times. The standard errors of the means matched those from the analytical formulas provided in Annex 10-B. For the balance of the discussion, the approximately equal allocation of cases across bid levels is taken as given, ignoring the possibilities for variance reduction at any given total sample size that might be achieved by concentrating the bulk of the sample in the region of bid levels where $F_{j\cdot} = 0.5$.

recommends a sample of over 2,000 observations, as demonstrated later. In short, the standard classical method is no more useful than a dart board. On the other hand, the optimization approach is more useful, but the optimal solution is extremely sensitive to the choice of nonparametric mean, which is ultimately a subjective decision.[10] For the balance of this chapter, our preferred mean is the intermediate nonparametric mean composed of a weighted combination of the Turnbull lower bound mean (75 percent weight) and the Paasche upper bound mean (25 percent weight).

PRELIMINARIES ON PROJECT RISK, THE VALUE OF INFORMATION, AND LOSS-COST MINIMIZATION

The analogs of the decision analysis approach to sample design in general (Schlaifer, 1959, 1961) and in statistical quality control applications in particular (Vaughan and Russell, 1983; Russell et al., 1986) provide the keys to unlocking the optimal CV sample size problem. Somewhat loosely stated, the core concept involves finding the sample size that minimizes the sum of sampling costs and expected losses.

The pure form of the optimal sample size approach involves Bayesian decision analysis and expands on the concept of prior information that was employed in the second and third variants of the standard approach. It combines prior subjective characterizations of the probability distribution of mean willingness to pay with data from an initial small sample of, say, 250 cases to decide whether an additional round of sampling should be undertaken and, if so, how many subjects should be interviewed in that second round. Schlaifer (1961) calls this Bayesian "preposterior" decision making about the desirable sample size because a decision can be reached on the basis of partial information before actually doing any additional sampling.[11]

Expected Gains and Losses

In the terminology of decision analysis, the cost-benefit decision is a two-action problem with infinite states of nature. The investment proposal can be accepted if it is expected to yield a positive discounted net cash flow above the breakeven point of NPV greater than or equal to 0, or rejected if it does not. Because the many influences on NPV are random variables, so is NPV. Therefore, at least conceptually, there are an infinite number of possible net cash flow values, each with its own probability of occurrence.

Cost-benefit risk analysis accommodates the variance in benefits and other variables, so the risk-neutral decision rule (Brent, 1996) is clearer than it would be in a deterministic analysis that inconsistently combines extreme values for some variables with various measures of central tendency for others. The rule is to proceed with a capital investment project if the *expected value* of its discounted stream of net benefits, E(NPV), is non-negative; but if the expectation of discounted net benefits is negative, the project proposal is economically infeasible. In the probabilistic context of risk analysis, following this expected value decision rule has a quantifiable cost called the "cost of uncertainty", a concept that was introduced in preceding chapters.

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[10] See Chapter 8 and Vaughan et al. (1999) for a review of parametric versus nonparametric means.

[11] In Winkler's (1972, p. 297) words, "This type of decision is called a *preposterior decision* because it involves the *potential* posterior distributions following the *proposed* sample." Winkler notes that preposterior analysis can be carried out repeatedly in sequential decision making. Our proposal involves a two-step sequence of taking an initial "small" sample and then doing a preposterior analysis that looks for the optimal number of surveys to add to the initial sample, which can turn out to be either 0 or some positive number. Of course, in some circumstances the initial sample size itself may be suboptimal (too large), but then there will be no need to add to it.

The cost of uncertainty is the expected opportunity loss of making the decision determined by the decision rule. That is, if the expectation E(NPV) taken over the entire NPV distribution is non-negative, the investment will be made. But if some portion of the NPV distribution falls below 0, actual losses in specific instances are still possible. The cost of uncertainty can therefore be measured as the mean of that portion of the NPV distribution truncated from above at 0 (the average loss, given that a loss might indeed occur), multiplied by the probability of a negative NPV occurring. If the project is not undertaken because the expected value of NPV is negative, the investment will not be made, thus forgoing any possibility of positive net returns. Symmetrically, the loss in this situation is the mean of that portion of the NPV distribution truncated from below at 0 (the average net gain forgone, given that a net gain might occur), multiplied by the probability of a positive NPV occurring.[12] The two opportunity loss situations are shown in Figures 10-2 and 10-3.

Figure 10-2. Case I: Project Feasible: Correct Decision Is to Invest

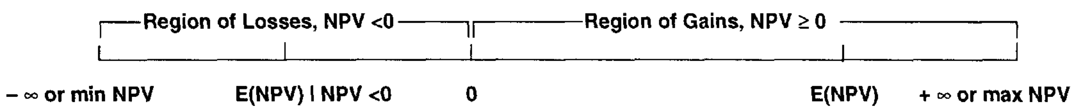
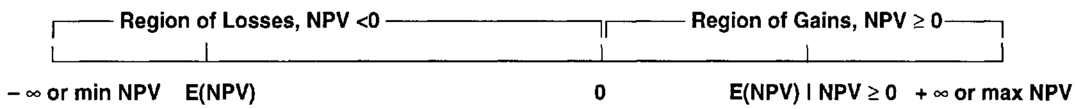


Figure 10-3. Case II: Project Infeasible: Correct Decision Is Not to Invest



If the project is economically feasible, its global mean NPV will be non-negative. The project will be undertaken so the region of opportunity loss is from negative infinity (or the minimum possible NPV) to 0. If the investment's expected NPV is negative, it should not be undertaken, thus forgoing some possible gains lying in the region of opportunity loss from 0 to plus infinity (or the maximum possible positive NPV).

Table 10-4 provides more formal definitions of the decision criterion, the probability of opportunity loss, the truncated mean loss, and the cost of uncertainty. The cost of uncertainty ($E_{loss,\,I}$ or $E_{loss,\,II}$ in the table) is in part a function of the amount of prior subjective and sample information size on hand when the options are weighed to either invest immediately or wait and collect more information. At the point after a small initial sample of size N_0 is taken (or even before, when a prior guess is formed without any sampling at all), it represents the most the investor would be willing to pay to gather more information and eliminate all uncertainty about the project, which is why it is also called the "expected value of perfect information, EVPI."

[12] Only if the probability distribution of NPV lies entirely in either the positive or negative domains will there be no cost of uncertainty because you literally can't go wrong; either an unattractive investment is unambiguously bad over the entire range of NPV outcomes or no losses are possible because there is zero probability that NPV will fall below 0. In either of these extreme situations, case-specific sample estimates of willingness to pay may not even be necessary. If extreme upper and lower limits for willingness to pay can be posited a priori via benefit transfers or other past experience, and the investment either fails the cost-benefit test using the highest possible WTP or passes it using the lowest conceivable non-negative WTP value, the investment decision can be made without incurring sampling costs.

Table 10-4. Fundamental Definitions

	Case I **Correct Decision:** **Invest** **E(NPV) ≥ 0**	**Case II** **Correct Decision:** **Do Not Invest** **E(NPV) < 0**
Decision criterion: global mean NPV	$E(NPV) + \int_{-\infty}^{\infty} NPV_i \times p_i \, d\,NPV$	
Probability of opportunity loss	$F_{loss\,I} = \int_{-\infty \text{ or min}}^{0} p_i \, d\,NPV$	$F_{loss\,II} = \int_{0}^{+\infty \text{ or max}} p_i \, d\,NPV$
Truncated mean loss	$E_{T,I} = E(NPV \mid NPV< 0)$ $= \left\{ \int_{-\infty \text{ or min}}^{0} NPV_i \times p_i \, d\,NVP \right\} / F_{loss\,I}$	$E_{T,II} = E(NPV \mid NPV>0)$ $= \left\{ \int_{0}^{+\infty \text{ or min}} NPV_i \times p_i \cdot d\,NVP \right\} / F_{loss\,II}$
Cost of uncertainty or expected value of perfect information or expected loss of a terminal action (ELTA)[a]	$E_{loss,\,I} = E_{T,I} \times F_{loss\,I}$	$E_{loss,\,II} = E_{T,II} \times F_{loss\,II}$

Note: The probability of occurrence of the i^{th} NPV is represented as p_i in the table.

[a]These terms all appear in the literature and they all mean essentially the same thing. It may seem unnecessarily circular to express the cost of uncertainty as the product of a truncated mean and the fraction of the total probability distribution lying in the region of opportunity loss. However, this is necessary given the way the information is produced by the Crystal Ball Monte Carlo simulation routine we used to verify the approximate solutions in the worked examples.

Additional sampling can never eliminate all uncertainty. However, changes in $E_{loss,\,I}$ (or E_{loss}, II) with increases in sample size beyond the original small sample N_0 provide a measure of the gross benefit of the second stage of a sequential CV sampling scheme. Incremental CV samples with $\Delta N>0$ reduce the standard error of the CV mean estimate of project benefits (WTP), which transmits into a reduction in both the truncated mean loss in project NPV and the cumulative probability of that loss.

Value of Information: Variance Reduction through Sample Size Increases

Given any initial referendum CV survey's sample size and the prospective investment project's NPV estimates based on the survey's mean WTP, decision makers can either finalize the project acceptance/rejection decision or commission further studies to try to reduce the uncertainty about the outcomes. Only information about the factors that can have a significant impact on the project outcome will reduce the cost of uncertainty in a meaningful way; in most cases uncertainty about benefits will be a major influence (Vaughan et al., 1999, 2000a, 2000b). The *value of information* is the change in the cost of uncertainty occasioned by gathering additional information. The value of information must be compared with the cost of information. If the value exceeds the cost, it is worth doing additional sampling to gather more information; otherwise the project should be accepted or rejected on the basis of the information on hand.

To sum up in words, the steps to find the optimal sample size via Bayesian decision analysis in a sequential approach are

1. Postulate an a priori guess about the expected value of WTP per household (or per person) and a reasonable opinion about the range in the expected value.

2. After the survey focus group sessions and the pretest, draw a small initial referendum CV sample (e.g., N_0 of about 250 observations, say 50 in each of 5 bid groups) and administer the final questionnaire. Calculate a nonparametric sample mean WTP per household, the variance of the sample mean, and the standard error of the sample mean. Approximate the population variance, σ^2, as the product of the initial sample size N_0 and the estimate of the variance of the sample mean.
3. Do an initial economic project cost-benefit analysis to estimate the expected value of discounted net benefits, E(NPV), at baseline conditions. Determine whether the opportunity loss follows Case I (project acceptance) or Case II (project rejection) and locate the region of opportunity loss for NPV. Establish the parameters of a linear relationship between the expected value of opportunity loss in NPV and the expected value of WTP.
4. Combine the prior guesses from Step 1 with the sample WTP information from Step 2 following a Bayesian formula to develop posterior estimates of the mean and standard error of WTP.
5. Using the posterior estimates from Step 4 as prior estimates, hypothetically increase the sample size from the base used in Step 2. Repeatedly compute the reduction in the variance of mean WTP that would result after sample augmentation over a range of sample sizes ΔN above the initial base $N = N_0$.
6. Assume that the expected value of NPV is normally distributed. Using the linear relationship between NPV and WTP from Step 3, monetize the reduction in variance in the expected value of NPV losses associated with different degrees of augmentation of the original sample. These reductions in the *expected cost of uncertainty* ($E_{loss,\,I}$ or $E_{loss,\,II}$ as the case may be) from a second round of sampling represent the expected value of additional sample information, EVSI, or the benefits of sample augmentation.
7. Over a range of hypothetical ΔNs above 0, numerically compare the *expected value of information* contributed by an additional sample observation (i.e., successive changes in the *cost of uncertainty* obtained in Step 6) with the marginal cost of a sample interview. Find the sample size where the marginal value of information is approximately equal to the cost of an additional referendum CV interview (for simplicity, this is assumed to be equal to the variable sampling cost and hence constant). The result is the *optimal (additional) sample size*, ΔN^*. The total sampling effort N_T will thus equal $N_0 + \Delta N^*$. The original small sample will be adequate if ΔN^* equals 0.[13]

The first two steps have already been covered. The next section explains the rest of the steps in detail. A subsequent section simplifies the procedure by eliminating the need to formulate priors (Step 1). Then the approach is demonstrated using the case study project data in an example, assuming tight, diffuse, and nonexistent prior judgments. The effects of project cost increases on the optimal sample size are explored and conclusions are drawn.

AN OPTIMIZATION METHOD FROM BAYESIAN DECISION ANALYSIS

This section adapts Schlaifer's (1961) method to the problem of determining CV sample size in the context of cost-benefit analysis.

[13] If a more precise measure of ΔN^* is desired because the normality assumption is in doubt, a full Monte Carlo cost-benefit analysis can be undertaken to compute the EVSI empirically and find ΔN^*.

The Linear Payoff Function

The first key to implementing Schlaifer's (1961) approximate optimization method is the linear payoff function. It describes the relationship between the quantity measured by the sample (mean WTP in this instance) and the payoff decision variable that depends on the sample information, in this case the expected value of NPV. This function is a compact summary of the cost-benefit analysis. Net present value is written as the linear relation $E(NPV) = -\alpha + \beta \times E(WTP)$. If the expected value of WTP from a CV survey is the only source of benefit, the intercept, $-\alpha$, represents the sum of discounted capital and operating costs of the investment. If there are any other sources of benefit (such as energy generation benefits), they can be netted out of the discounted costs to get the intercept. The slope, β, is the marginal contribution to discounted net benefits of an increase in average WTP per household (undiscounted).[14] It too can be easily calculated by simply taking the present value of the number of beneficiaries to whom the mean WTP is applied over the project's lifetime. For our case study, E(NPV in R$) = –594,653,964 + 100,988,487 × E(WTP).[15]

Given this linear relationship between discounted profits and household WTP, if the sample mean WTP is normally distributed, the outcome variable, E(NPV), will also be normally distributed with mean $E(NPV) = -\alpha + \beta \times E(WTP)$ and variance $VAR(NPV) = \beta^2 \times VAR[E(WTP)]$.[16] The breakeven value that sets E(NPV) to 0 is $\mu_b = \alpha/\beta$ = 594,653,964/ 100,988,487, or R$5.89. For any expectation of WTP less than μ_b, opportunity losses in NPV will be incurred. From Table 10-4 it is clear that all of the nonparametric sample means other than the Turnbull lower bound mean WTP are above the breakeven value, so the correct decision is to invest.[17] However, the sample mean is a random variable so there is some non-zero probability that it could be below the breakeven value. For example, the preferred measure, a 75-25 weighted combination of the lower and upper bound means from Table 10-3, is R$7.47, and its standard error is 0.84, putting the sample mean 1.88 standard errors above the breakeven value. The Kriström and Paasche means are even more distant from the breakeven value (R$3.31 and R$6.80, respectively, in absolute terms and 3.28 and 4.96 standard errors, respectively). Under these means the cumulative probability of a loss is clearly lower than it would be using the 75-25 weighted average to measure WTP and predict NPV.

A Normal Approximation to the Distribution of the Benefits of Additional Sampling

This leads to the second key to Schlaifer's approach. It is that, given a successful project on average, each possible NPV loss has a probability associated with it. Centering the net benefits distribution on the most likely value of WTP, the observed sample mean, the loss probabilities

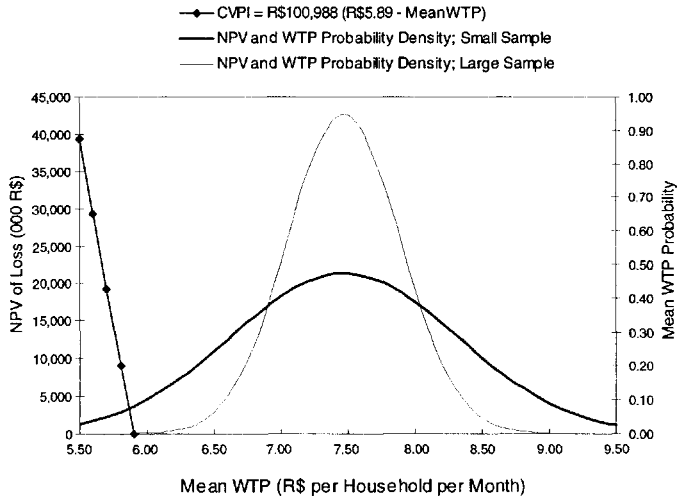
[14] It is assumed that in the cost-benefit analysis of the investment, per household (or per capita) WTP benefits are multiplied by the size of the beneficiary population in every year to obtain aggregate gross benefits. The expected signs for α (negative) and β (positive) are assigned.

[15] The intercept for costs was shadow priced. The slope also incorporates a shadow price factor to allow the WTP to be expressed in terms of the original survey responses, without shadow pricing. Because WTP per household is on a monthly basis, population in every year has to be multiplied by a factor of 12 in addition to the shadow price factor.

[16] From the properties of the expectation and variance operators $E(\alpha + \beta X) = \alpha + \beta E(X)$. This says that the expected value of a constant (α), plus another constant (β) times a random variable (letting X represent WTP) is the constant α plus the constant β times the expected value of the random variable. For example, see Pfaffenberger and Patterson (1987, p. 208) and Little (1978, Chapter 10) on strictly linear relationships between random variables versus error propagation formulas.

[17] Probabilistic cost-benefit analysis not reported here strongly suggests that the investment is justified if a reasonable mean benefit is just slightly higher than the Turnbull lower bound (Vaughan et al., 2000a), so this chapter does not employ the Turnbull mean and its variance in choosing the optimal sample size.

Figure 10-4. Losses and Loss Probabilities for WTP below Breakeven Point



are defined by the tail portion of a normal density function lying below μ_b. The sum of the products of all the possible expected losses and their associated probabilities reveals the cost of uncertainty. Figure 10-4 illustrates the superimposition of the linear relation representing the opportunity loss function (also called the "CVPI function"[18]) on the normal E(WTP) distribution whose standard error is assumed to be known from the first small sample.

The distribution of gains and losses in Figure 10-4 is centered on the sample mean WTP of R$7.47, with an initial spread given by the mean standard error of the small sample, 0.84. Increasing the sample size decreases the amount of spread in the (assumed) normal density, thus decreasing the expected value of a loss, as demonstrated by the probability density function generated by a larger sample and a smaller standard error (the lightly shaded line in the figure). Unlike the the small sample's density, it has an infinitesimal amount of its area to the left of the breakeven point of R$5.89.

Losses in NPV are shown as positive in the figure. The horizontal axis intercept of the CVPI function is the breakeven value, μ_b, given discounted costs equal to $-\alpha$. It can be found by setting E(NPV) to 0 and solving the linear function $-\alpha + \beta \times \mu_b = 0$ for $\mu_b = \alpha / \beta$. The slope, β, measures the decrease below 0 in NPV for any WTP below the breakeven value. So, for WTP $< \mu_b$, the CVPI function's dependent variable equals $\beta \times [\mu_b - E(WTP)]$ and for WTP$\geq \mu_b$ the CVPI function is 0.

All else being equal, higher discounted costs (a larger negative α) shift the CVPI function to the right, raise the requisite breakeven value, and given the sample mean WTP, put more mass of the tail of the normal probability density under the non-zero part of the CVPI loss line. The expected opportunity loss or cost of uncertainty is the sum of the products of the normal density function to the left of the breakeven value and the conditional value of perfect information to the left of the breakeven value. Therefore, higher costs raise the cost of uncertainty, given the sample's mean estimate of willingness to pay.

Developing the Objective Function

To calculate the required loss integrals, Schlaifer (1959, 1961) normalizes the extent of departure of the breakeven point from the sample mean WTP and computes a "unit loss integral"

[18] Opportunity losses are conditional because after the first sample has been taken and the optimal act is chosen [in the case illustrated, invest because E(NPV)>0], they are conditional on that act. See Pfaffenberger and Patterson (1987, p. 1069). For a full discussion that may be more accessible than Schlaifer's original book, see Winkler (1972).

from the standard normal distribution (Table IV in Schlaifer, 1961). Multiplying the value of the unit loss integral by β representing the marginal contribution of the sample measurement (WTP) to the outcome (NPV) yields the expected loss of an (optimal) terminal action, ELTA. As mentioned in the note in Table 10-4, the ELTA is also called the "cost of uncertainty" or the "expected value of perfect information," EVPI, if the decision to invest is made immediately after taking the first small sample without gathering any additional information. That is:

$$\text{ELTA} = \text{EVPI} = \beta \times \sigma_1(\tilde{\mu}) \times L_N(|D|)$$

where:

β = Marginal contribution of WTP to NPV in dollars

$\sigma_1(\tilde{\mu})$ = Standard error of the mean WTP posterior to taking a first small sample. The posterior can either be the standard error of the mean of that sample in the absence of a subjective prior (or under a very diffuse prior) or the posterior combination of a prior guess and the sample standard error. (See the next section for details.) Specifically (Schlaifer, 1961; Pfaffenberger and Patterson, 1987) the information contained in the prior distribution, I_0, is the reciprocal of the prior variance of the population mean, or $1/\sigma_0^2(\tilde{\mu})$, denoting the prior with a "0" subscript. The information in the first small sample, $I_{\bar{x}}$, is the reciprocal of the variance of the sample mean, or $1/\sigma^2(\bar{x})$. The posterior variance of the mean, $1/I_1$, is the sum $1/(I_0 + I_{\bar{x}})$ and the posterior standard error is the square root of that sum, or:

$$\sigma_1(\tilde{\mu}) = \sqrt{1/I_1} = [1/(I_0 + I_{\bar{x}})]^{1/2} = \{1/[1/\sigma_0^2(\tilde{\mu}) + 1/\sigma^2(\bar{x})]\}^{1/2}$$
$$= \{[\sigma_0^2(\tilde{\mu}) \times \sigma^2(\bar{x})]/[\sigma_0^2(\tilde{\mu}) + \sigma^2(\bar{x})]\}^{1/2}$$

$|D|$ = The absolute value of the standardized difference between the breakeven value of WTP, μ_b, and the mean posterior to taking a first small sample, $E_1(\tilde{\mu})$.[19] That is, in Schlaifer's notation:

$$|D| = |\mu_b - E_1(\tilde{\mu})| / \sigma_1(\tilde{\mu})$$

$E_1(\tilde{\mu})$ = The mean posterior to taking a first small sample. It can either be the mean, $\bar{x}$, of that sample in the absence of a subjective prior or the posterior combination of a prior guess about the mean, $E_0(\tilde{\mu})$, and the sample mean. Using I_0 and $I_{\bar{x}}$ from directly above as weights:

$$E_1(\tilde{\mu}) = [I_0 \times E_0(\tilde{\mu}) + I_{\bar{x}} \times \bar{x}] \div (I_0 + I_{\bar{x}})$$

L_N = Unit loss integral, or the expected value of the difference between the normalized random variable of interest, x, and D.[20]

[19] For a profitable investment D < 0.

[20] While Schlaifer is not too clear, Bonini et al. (1997) define L_N as:

$$L_N = \int_{-\infty}^{-D} (-D - x) f_N(x)dx = \int_{D}^{\infty} (x - D) f_N(x)dx$$

where $f_N(x)$ is the standardized normal density function. The expression on the left applies to profitable investments, the expression on the right to unprofitable ones. The expressions are symmetric; for any –D whose absolute value equals D they both produce the same value of L_N. In words, the expression on the left is the integral of the standard normal variate from negative infinity *up to* –D, which is the standardized offset between the sample mean and the breakeven value. Beyond –D, the probability of an opportunity loss is 0, so, although the zero probability is ostensibly omitted from the calculation, L_N is not the mean loss of the truncated distribution, but measures the untruncated mean loss of the entire distribution (Jedamus and Frame, 1969).

By taking a second sample and not acting immediately on the basis of the first small sample, it may be possible to reduce expected losses. The expected value of the new sample information, EVSI, is a function of the monetary value of the reduction in variance due to the second sample, or the reduction in the ELTA. To find the optimal size of a second sample, ΔN^*, the function to be maximized includes the benefit of variance reduction and the sampling costs. The benefits are measured as the expected value of information obtained from a second sample of size $\Delta N > 0$, assuming the population variance of WTP is known or set equal to the variance obtained from the first sample. Analogous to EVPI, the expected value of the sample information, EVSI, is the value of the reduction in losses due to the reduction in variance brought about by taking more observations, ΔN:

$$\text{EVSI} = \beta \times \sigma(\tilde{E}_1) \times L_N(|D_E|)$$

where:

$\sigma(\tilde{E}_1)$ = The preposterior *reduction* in the standard error of the mean attributable to taking a second sample of size ΔN. It is calculated as the square root of an information-weighted average of the posterior variance of the mean from above, $\sigma_1^2(\tilde{\mu})$, and the variance of the mean the new sample is presumed to produce, $\sigma^2/\Delta N$. To get $\sigma^2/\Delta N$, assume the population standard deviation (of individual observations, not the mean) σ, is approximately equal to the standard deviation from the first sample. The value of σ^2 can then be obtained as the product of the size of the first sample, N=250, and the variance of the mean WTP (see Table 10-3), or $\sigma^2 \approx N\,\sigma^2(\bar{x})$. Then the Bayesian preposterior reduction in the standard error of the mean (Schlaifer, 1961; Pfaffenberger and Patterson, 1987; Winkler, 1972; Lapin, 1994) is just the square root of

$$\sigma^2(\tilde{E}_1) = \sigma_1^2(\tilde{\mu})\left[\frac{\sigma_1^2(\tilde{\mu})}{\sigma_1^2(\tilde{\mu}) + \sigma^2/\Delta N}\right]$$

$|D_E|$ = The absolute value of the standardized difference between the breakeven value of WTP, μ_b, and the mean posterior to taking a first small sample, $E_1(\tilde{\mu})$, now using $\sigma(\tilde{E}_1)$ as the *preposterior* estimate of the standard error of the mean WTP at new sample size ΔN. That is, in Schlaifer's notation:

$$|D_E| = |\mu_b - E_1(\tilde{\mu})| / \sigma(\tilde{E}_1)$$

The costs of sampling are assumed to be a linear function of ΔN, with fixed costs K_s and unit variable costs k_s. Then, the full loss-cost function to be minimized with respect to ΔN is

$$£ = \min_{\Delta N}\ (\text{EVPI} - \text{EVSI}) + (K_s + k_s \Delta N)$$

Once N_0 is chosen, EVPI and K_s are constants.[21] Therefore minimization of £ is equivalent to maximizing a concentrated net benefit function £′ where EVSI represents the benefits of taking an additional sample of size ΔN and incurring total variable costs of $k_s\,\Delta N$. The expected net gain from (additional) sampling, ENGS, becomes:

$$£' = \max_{\Delta N}:\ \text{ENGS} = \text{EVSI} - k_s\Delta N = \beta \times \sigma(\tilde{E}_1) \times L_N(|D_E|) - k_s\Delta N$$

[21] Throughout we assume that the fixed costs of taking a second sample, K_s, are 0, because most of these costs (for consulting services, focus groups, questionnaire pretesting and design) would be incurred to obtain the initial sample of 250 cases.

EVSI is a function of ΔN because $\sigma(\tilde{E}_1)$ and $L_N(|D_E|)$ are nonlinear functions of ΔN. The optimum sample size that maximizes $\pounds'$ with respect to ΔN has to be found numerically. In some cases, the EVSI function will be less than the variable costs of sampling for all values of ΔN, so no additional sampling effort is warranted. In other cases, the net gain from additional sampling will be positive for ΔN between a new sample size of 1 and the number of cases where EVSI = $k_s \Delta N$, and should be relatively easy to locate. Finally, the net gain from additional sampling may initially be negative and decrease with ΔN (because EVSI < $k_s \Delta N$ over this range), and only later exceed variable costs in a narrow region of values for ΔN.[22] Finding the optimum in this case may depend on making a good choice of the starting value for the numerical search. Approximations to aid the search are discussed in Annex 10-C.

SIMPLIFYING THE BAYESIAN DECISION ANALYSIS BY ASSUMING TOTAL IGNORANCE

The Bayesian approach is not difficult to implement. Although it looks complicated, the steps involved are relatively simple.[23] The appearance of complexity is misleading, arising mainly from the need for an elaborate system of notation in order to keep track of the several prior and posterior means and variances involved in the several solution steps.

However, things can be made simpler yet by dropping the requirement that priors be formed. Tight priors are desirable because they reduce the required size of the optimal sample, all else remaining equal. However, while many CV studies have been done in developing countries, they almost defy easy summarization (Ardila et al., 1998), so forming reasonable prior beliefs on the basis of fragmented and inconsistent past experience is difficult indeed. In fact, unless priors are reasonably accurate, they will not contribute much information on WTP location and spread beyond what an initial survey sample contains, so the influence of relatively diffuse priors on the optimal decision will be trivial. In this common situation, little can be gained from formulating wildly inaccurate prior estimates; all the information content will be in the first small sample, N_0. The simplified sequential approach suggested in this section mirrors those realities.

Modifying the Bayesian Linear Profit and Normal Loss Distribution Method

Recall the fundamental Bayesian relation between the information content of the posterior (denoted with a 1 subscript, or I_1) and the information contained in the prior (denoted with a 0 subscript, or I_0) and the sample ($I_{\bar{x}}$):

$$I_1 = I_0 + I_{\bar{x}}$$

Equivalently, the posterior information content equals the sum of the reciprocals of the prior and sample variances of the mean:

$$1/\sigma_1^2 = 1/\sigma_0^2 + 1/\sigma_{\bar{x}}^2$$

Under total ignorance there is no information content in the prior because the prior variance is extremely large, so the only useful information comes from the sample itself. Then $I_1 = I_{\bar{x}}$, σ_1^2 is equal to $\sigma_{\bar{x}}^2$, and $\mu_1 = \bar{x}$. This means that the expression for the expected value of

[22] See Schlaifer (1961), pp. 330–331, for a discussion of the behavior of the ENGS function; the explanation is complicated and defies intuitively obvious summary.

[23] For a flowchart that is clear and easy to follow, see Lapin (1994), Figure 26-15, p. 1046.

perfect information, EVPI, from above can be rewritten as a function of the standard error of the mean from the first small sample:

$$\text{ELTA} = \text{EVPI} = \beta \times \sigma(\bar{x}) \times L_N(|D|) = \sigma / \sqrt{N_0} \times L_N(|D|)$$

where now:

$|D|$ = The absolute value of the standardized difference between the breakeven value of WTP, μ_b, and the mean from the first small sample, standardizing with the standard error of the sample mean:

$$|D| = |\mu_b - \bar{x}| \,/\, \sigma(\bar{x})$$

Under this simplification, the prior mean and standard error entering into the preposterior step where the optimal ΔN is sought are just $\bar{x}$ and $\sigma(\bar{x})$. The value of the reduction in opportunity loss brought about by the contribution of any sample size increase of ΔN to variance reduction now becomes a simpler expression. It depends on standard deviation ($s \approx \sigma$) from the first sample, without any adjustment for a subjective prior.

$$\text{EVSI} = \beta \times \sigma^* \times L_N(|D_E|)$$

where:

σ^* = The preposterior degree of reduction in the standard error of the mean contributed by a second sample of size ΔN under total ignorance. Here, σ^* is the amount of revision in the standard error of the mean from the prior to the posterior distribution

$|D_E|$ = The absolute value of the standardized difference between the breakeven value of WTP, μ_b, and the mean from the first small sample, standardizing with σ^* representing the amount of revision in the standard error of the mean between the first small sample of size N_0 and the ultimate preposterior sample of size $N_0 + \Delta N$

As before, the trick is to place the posterior variance of the mean from the previous step in the role of prior in this step, so $\sigma(\bar{x})^2$ from the first sample now plays the role of the prior. From the Bayesian rule $I_1 = I_0 + I_{\bar{x}}$ above, the preposterior variance of the mean after the second sample is taken is the sum of the prior variance from the first sample and the variance of the mean from the second sample. With the population standard deviation $\sigma = s = \sigma(\bar{x}) \times \sqrt{N_0}$ assumed known, the information content of the posterior is greater than the prior because of the expansion in sample size from N_0 to $N_0 + \Delta N$:

$$1/\sigma_1^2 = 1/\sigma_0^2 + 1/\sigma_{\bar{x}}^2 = 1/(\sigma^2/N_0) + 1/(\sigma^2/\Delta N) = (N_0 + \Delta N)/\sigma^2$$

As intuition would suggest, the posterior variance is the variance of the pooled sample $N_T = N_0 + \Delta N$. Taking reciprocals of the preceding:

$$\sigma_1^2 = \sigma^2 / [N_0 + \Delta N]$$

Then (Pfaffenberger and Patterson, 1987; Lapin, 1994; Winkler, 1972) the shrinkage in the standard error of the mean due to sample size augmentation, σ^*, is defined as

$$\sigma^* = \sqrt{\sigma_0^2 - \sigma_1^2} = \sqrt{[\sigma^2/N_0] - [\sigma^2/(N_0 + \Delta N)]}$$

Under total ignorance, σ^* is simply a function of the population variance, the initial sample size, and the addition to it. The rest of the optimization proceeds just like the pure Bayesian case.

Verification: Heuristics of a Monte Carlo Approach

Schlaifer (1961, p. 341) comments that even in "violently non-Normal problems" a number of numerical analyses showed that the approximation performs well, but he also cautions that "In problems where a good deal is at stake it will be well to use the Normal optimum only as a starting point and then use exact methods to trace out expected total loss in the neighborhood of this point."

Let us back up to the beginning and suppose the researcher admits to near total ignorance about mean WTP and its variance before an initial sample is actually taken.[24] Once the first sample of size N_0 is in hand, measures of mean WTP and its variance can be calculated. Using Monte Carlo cost-benefit analysis, the initial NPV distribution and EVPI can be easily obtained, as can the EVSI for sample size increases above the initial base.

The reduction in the standard error of the mean (σ^*) for a range of values of ΔN near the optimum previously found under the normal approximation method can be calculated as a function of the population variance, σ, N_0, and ΔN. Repeated Monte Carlo cost-benefit simulations can then be run using the shrinkage in the standard error of mean WTP associated with each of several ascending values of ΔN as initial conditions. Without having to invoke the normality assumption, empirical estimates of EVSI and ENGS can be extracted from each simulation and the optimum ΔN^* found by trial and error.

This tedious, time-consuming, and computationally intensive Monte Carlo process is hardly operational. A Monte Carlo cost-benefit analysis with the case study data was undertaken to verify the accuracy of Schlaifer's approximate solution. The results confirmed that in practical work Monte Carlo analysis can be safely bypassed by using Schlaifer's linear loss function and normal NPV distribution approximations instead, and maximizing EVSI with respect to ΔN. The next section demonstrates the results of applying these steps to the Tietê investment project data using Schlaifer's approximate solution that assumes linear profits, normal distributions, and a known (or knowable via the first sample) population variance.

AN EXAMPLE OF FINDING THE OPTIMUM SAMPLE SIZE

In review, using actual data involves forming a prior "guesstimate" about the mean WTP and the population variance, choosing an initial small sample, combining the sample estimates of mean and variance with the prior estimates to arrive at posterior estimates, and using those estimates to monetize the potential reduction in expected opportunity loss that might be gained by gathering more data and hence to decide whether a larger sample would be optimal. By invoking the assumption of total ignorance, the sequential optimization approach only requires mean and variance information from a small original sample. If additional sampling would be optimal, the extra observations can be collected in a subsequent round of interviewing.

The case study demonstration follows the structure of the spreadsheet algorithm for finding the optimal sample size provided in Annex 10-A, which documents all of the calculation steps.[25] The subsequent discussion is based on the weighted 75-25 mean and its standard

[24] Before consulting the Bayesian decision analysis literature and Schlaifer's optimization method, we originally took an intuitive Monte Carlo loss-cost minimization approach that was similar, but not identical, to the Monte Carlo routine discussed here. We are grateful to a reviewer of an early version of this chapter who asked for a theoretical justification and generalization of that intuitive brute force method. His comments directed us to the Bayesian decision analysis literature and the approximate solution for optimal sample size assuming linear profits and normal distributions.

[25] The spreadsheet was successfully benchmarked using the example data in Schlaifer (1961). It was also independently replicated by a colleague to verify the cell formulas. The interested reader can safely duplicate the structure

error, but similar calculations using any of the other means (e.g., the Turnbull, Kriström, or Paasche means) can easily be done by following the same structure.

The WTP Distribution

Referring to the spreadsheet algorithm for sample size optimization provided in Annex 10-A, the first two steps have already been touched on. The concise summary that follows uses Schlaifer's (1959, 1961) notation to make it easier for the reader to consult the original sources and the annex.

Priors for the Parameters of the WTP Distribution

When example priors were constructed for the classical method described earlier, a triangular distribution was invoked for ease of use. Now, assume instead that the population mean of WTP, μ, is a random variable having prior probabilities that can be obtained from a normal density function.[26] Since normality is the operative assumption, suppose the prior mean $E_0(\tilde{\mu}) = \hat{\mu}$ is 1 percent of income, or R$8.28. A prior measure of the standard deviation of μ is needed to summarize the a priori variability in possible values of μ.

To guesstimate the variability in mean WTP, μ, in advance of taking any measurements at all, Schlaifer's technique (1961) asks the decision maker to speculate about what interval around the prior mean would give the guess an even (50-50) chance of being correct. Somewhat arbitrarily choosing an error of R$4.00 on either side of the prior says the true mean is as likely as not to fall between R$4.28 and R$12.28. From the standard normal distribution, the standardized value of $[\mu - E_0(\tilde{\mu})]/\sigma_0(\tilde{\mu})$ that demarcates 25 percent of the distribution's area is 0.67, so solving $0.67 = 4.00/\sigma_0(\tilde{\mu})$, the prior for the population standard deviation of μ is 4.00/0.67 = R$5.97. This represents a weak or diffuse prior because the guess about the mean WTP has a relatively broad band of uncertainty and therefore $E_0(\tilde{\mu})$ has very little information content.

Initial Sample Estimates of the Parameters of the WTP Distribution

The expected value of the weighted sample mean is the population mean, μ. That is, $E(\bar{x}) = \mu$ = R$7.47. Recall from Table 10-3 that the variance and standard deviation of the distribution of sample means at $N_0 = 250$ cases are $\sigma^2(\bar{x}) = s^2/n$ = R$0.70; and $\sigma(\bar{x}) = s/n^{1/2}$ = R$0.84. Finally, the sample standard deviation can be obtained from the sample estimate of the standard error (or deviation) of the mean (Schlaifer, 1961) and used as if it were the true population standard deviation. Thus $\sigma \approx s$ and $s = \sigma(\bar{x}) \times N_0^{1/2}$ = R$13.23.

By definition, the mean is asymptotically normally distributed. Therefore, discounted net benefits will be normally distributed if WTP benefits are the only source of gross benefit, if NPV is linearly related to mean WTP, and if costs are either deterministic or normally distributed as well.

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and insert his/her project data to compute an optimal sample size using the Bayesian approach. To get results under total ignorance, a separate spreadsheet is not needed; simply insert a very large number in row 10 for the prior standard error of the mean. This will wash out the influence of the prior in all subsequent calculations.

[26] In the treatment of the standard method, we had to form guesses about the mean WTP and the standard deviation of individual observations in the population. Here, we are speculating about the mean of all possible prior means and the spread in that (normal) prior distribution of hypothetical means. This explains the use of the notation $\sigma_0(\tilde{\mu})$ rather than σ_0.

Profits from Investment and Sampling Costs

Linear Profit Function

The relation between NPV and WTP can be easily extracted from a deterministic cost-benefit spreadsheet model through a simple sensitivity analysis by fitting a linear ordinary least squares model to the NPV data points that result from varying WTP. Or, simpler yet, the shortcuts to finding the intercept and slope of the CVPI function covered previously can be used. In the case of our sample data, the fit is perfectly linear. At baseline cost conditions, E(NPV) = R\$159,730,009 = –R\$594,653,983 + R\$100,988,485 (WTP). The breakeven value of E(WTP) is R\$5.89 per household per month. The intercept represents discounted project costs and some market benefits for energy production that were not estimated via CV.[27]

Linear Sampling Cost Function

The sampling cost function is linear, with a marginal (equals variable) cost per observation of R\$89 in shadow-priced terms, as required by IDB protocols (Powers, 1981). Zero fixed costs for the second round of sampling are assumed.

Results

Under baseline initial conditions, including project costs and the diffuse prior, it is optimal to augment the initial sample size beyond 250 cases. The logit probability formula from Annex 10-C indicates that additional sampling should be done, and brute force exploration reveals that additional sampling can produce positive values for EVSI net of a variable sampling cost. Numerical optimization using Excel's Solver routine returns a solution of 2,243 cases for ΔN^*. The optimal sample size needed in total is 2,493 cases, which is almost 1,000 cases larger than the largest sample size recommended by the standard method. The explanation for this result, while not intuitively obvious, can be uncovered by looking at the empirical net benefits distribution shown in Figure 10-5.[28]

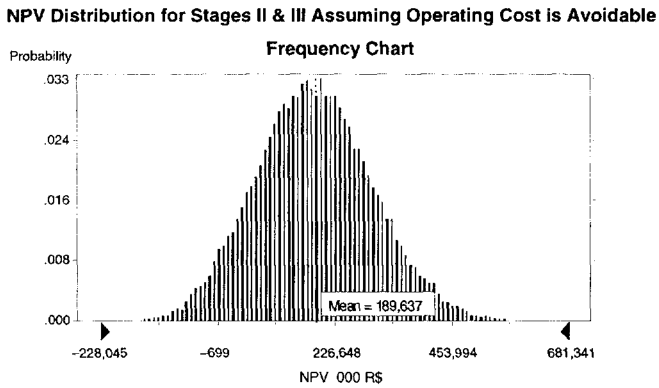
The empirical Monte Carlo NPV distribution in the figure is approximately normal because the influence of the normally distributed WTP benefit estimates dominates other non-normal sources of variability in the model. The baseline expected value of NPV is R\$190 million and the standard deviation of the empirically generated distribution of mean NPV is R\$114 million. Because the grand mean of the distribution of means is 1.67 standard errors above 0,[29]

[27] The energy benefit offset to discounted capital and operating costs is R\$38,892,992, while at baseline conditions total discounted costs are R\$633,546,963. For changes in cost conditions relative to the baseline, the intercept $-\alpha$ in the linear profit function varies according to the linear relationship $-\alpha$ = R\$38,892,992 – R\$633,546,963 (cost level/base cost level), while the slope remains unchanged. For instance, if costs increase by 10 percent, the intercept changes from –R\$594,653,984 at the baseline to –R\$626,331,332. This relationship can be used to explore the effect of decreasing the standardized distance of E(NPV) from 0 on the optimal sample size.

[28] The figure was produced by 50,000 Monte Carlo trials of a risk-based cost-benefit analysis using Crystal Ball. No formal test of normality (e.g., the Komolgrov-Smirnof statistic) was undertaken, but the median, R\$187 million, is very close to the mean, and the measures of skewness (0.12) and kurtosis (2.98) are consistent with approximate normality.

[29] From the linear profit function VAR[E(NPV)] = $\beta^2 \times$ VAR[E(WTP)] or $(100{,}988{,}485)^2 \times 0.70$ and SE[E(NPV)] is approximately $\{\text{VAR[E(NPV)]}\}^{1/2}$ = R\$84 million. This is lower than the empirical result of R\$114 million because it only reflects variation in WTP benefits. Under the linear approximation, the distance of NPV from 0 in S.E. units is R\$160,955,974/R\$84,493,829, or 1.9. Equivalently, the sample mean of WTP, R\$7.47, is the same number of standard errors away from the breakeven value of R\$5.89 because (R\$7.47 – R\$5.89)/0.835 also equals 1.9, with the discrepancy due to rounding. The Monte Carlo mean NPV and its standard error differ from the deterministic linear prediction because they are affected asymmetrically by randomness in cost and timing variables that the deterministic linear function ignores.

Figure 10-5. Empirical NPV Distribution



the (empirical) probability of project success is 95 percent, so there is a 5 percent chance of incurring an opportunity loss. While this is a fairly robust investment, recall from Figure 10-1 that the standardized distance between E(NPV) and 0 has to be above about 2.4 for ΔN to be 0. Therefore, stopping with the initial small sample would have been recommended only if the sample mean WTP were R$7.92 or greater, rather than the R$7.49 that was actually observed (e.g., no additional sampling would be recommended if the optimization were based on Kriström's mean of R$9.20).

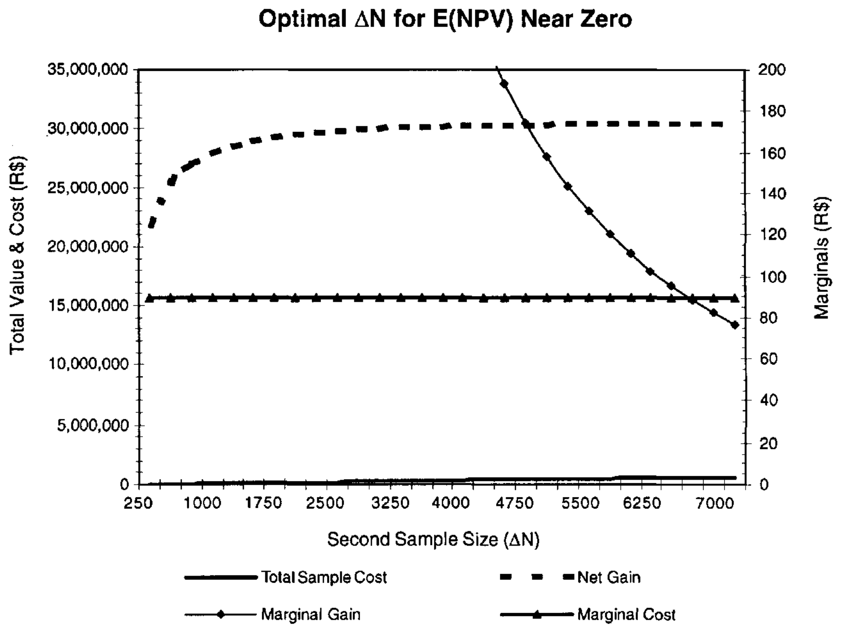
Another relevant issue is how the optir l ample size would behave if the gap between the mean of the E(NPV) distribution and .. ...ı.. .....rowed rather than widened. The optimal sample size is sensitive to the extent of the ai: :ement of the decision variable, the expected value of NPV, from 0, which is equivalent to the standardized offset of E(WTP) from the breakeven value μ_b. Looking back at the EVSI formula, as $|D_E|$ increases with a widening gap between the expectation of WTP and the breakeven WTP, the more certain the decision maker becomes that the optimal decision under the prior information is correct without additional sampling. As $|D_E|$ increases, the expected value of opportunity loss, L_N, falls, lowering the EVSI of any particular ΔN (see Lapin, 1994; Winkler, 1972).

For contrast with improving on the baseline case, looking at the opposite extreme is instructive. Raising discounted costs by 25 percent (i.e., shifting the intercept of the linear net profit function from –R$595 million to –R$753 million; see footnote 27) brings the expected value of NPV very close to 0 and puts half of the distribution of expected net returns into the negative region. Here, the optimal size of ΔN will be at a maximum. Table 10-5 shows the

Table 10-5. Crude Step Search for N* with E(NPV) Near Zero

	Trial Value for ΔN:	5,750	6,750 ≈ Optimum	7,750
Change in S.E. of mean	σ^*	0.809321811	0.811879233	0.81379242
Standardized distance	D_E	0.0410	0.0409	0.0408
Unit normal loss integral	$L_N(D_E)$	0.378772051	0.378834534	0.378881025
Value of sample information	EVSI (ΔN)	R$30,957,867	R$31,060,815	R$31,137,831
Total sample cost	$K_S + k_s \Delta N$	R$511,750	R$600,750	R$689,750
Net gain	ENGS(ΔN)	R$30,446,117	R$30,460,065	R$30,448,081
Marginal gain	Δ EVSI (ΔN)/ ΔN	R$120	R$88	R$67
Marginal cost	k_s	R$89	R$89	R$89

Figure 10-6. The Expected Net Gain from Additional Sampling



elements of a crude trial-and-error search for an optimum, assuming a diffuse prior for WTP, and Figure 10-6 shows the optimum graphically.

The table and figure show that the response surface is very flat. The approximate optimum is 6,750 cases, but the gains to additional sampling diminish quickly after about 2,000 cases. By inspection of the figure, the net gains from an additional sample of 4,000 cases (R$30,303,436) or even less are fairly close to the net gains at the optimum (R$30,460,065). This is consistent with Schlaifer's (1961, Figure 21.5) numerical investigations, which showed that moderate departures from the optimum number of cases (± 20 percent or even ± 30 percent) are likely to be inconsequential.

To find the effect on the optimal N of the location of the E(NPV) distribution relative to E(NPV) equal to 0, the baseline E(NPV) distribution has to be shifted leftward. To shift the NPV distribution, several exercises were conducted, raising the mean of total project operating and investment costs above the baseline by 5 percent through 25 percent, while holding the mean WTP constant at R$7.49.[30] The increase of 25 percent is the extreme discussed above that brings E(NPV) as close as possible in the risk analysis to the breakeven point of 0 while remaining barely positive.

Table 10-6 shows the effect that the standardized distance of E(NPV) away from 0 has on the optimal incremental sample size ΔN* under diffuse priors, tight priors, and total ignorance.

Figure 10-1 and the results in Table 10-6 suggest that in this case small samples suffice when sampling costs are high and the mean of the NPV distribution is over about 2.4 standard errors away from the breakeven point of E(NPV) = 0 because the decision has little downside risk. There is no payoff in taking larger samples to shrink that risk by reducing the variance and further compressing the portion of the NPV distribution lying below 0. However, substantial

[30] Increasing project-costs can be thought of as a proxy for decreasing E(WTP) or reducing the standardized distance between E(NPV) and 0 at the initial sample size of N_0 = 250 cases.

Table 10-6. Optimal Incremental Sample Sizes, ΔN^*, Depending on Priors and Initial EVPI

Costs Relative to Baseline ($\alpha_1 \div \alpha_0$)	1.0	1.05	1.10	1.15	1.20	1.25
Small Sample Standard Errors σ ($\bar{x}$) of E(NPV) from 0[a]	1.90	1.52	1.15	0.77	0.40	0.02
High Sampling Cost of R\$89 per Interview						
Tight prior[b]	0	0	0	1,996	4,393	6,530
Diffuse prior[b]	2,243	3,411	4,600	5,657	6,409	6,715[c]
Total ignorance	2,351	3,507	4,673	5,697	6,413	6,687
Low Sampling Cost of R\$30 per Interview						
Tight prior[b]	0	0	0	3,994	7,458	10,729
Diffuse prior[b]	3,866	5,656	7,506	9,160	10,340	10,821
Total ignorance	4,022	5,799	7,615	9,219	10,343	10,774

[a]Because the prior mean exceeds the sample mean, the posterior standardized distance exceeds the distance using the sample mean alone.
[b]Prior guess of E(WTP) of R\$8.28 with a prior ±50 percent error of R\$0.50 for the tight prior and R\$4.00 for the diffuse prior.
[c]Exact optimum corresponding to the approximate optimum in Table 10-5 and Figure 10-6.

gains to increasing the sample size begin to emerge after the expected value of NPV falls below about 2.4 standard errors from 0. Although the algorithm does not explicitly incorporate Type II errors, the fact that the required sample size increases as the gap between E(NPV) and the breakeven point narrows provide protection against false acceptance of a mean WTP that justifies the project when in fact the true mean WTP would lead to the opposite conclusion.

Table 10-6 convincingly shows that good prior estimates of the mean WTP and its spread can significantly reduce the amount of sampling effort needed to reach an optimum CV survey sample size for investment decisions. Unfortunately, given the state of the art, good prior estimates remain unattainable, especially in developing countries.

CONCLUDING OBSERVATIONS

Small CV survey samples are probably adequate for cost-benefit analysis when the fixed and variable costs of sampling are relatively high and the investment is extremely robust. If the investment has a probability of failure of less than 1 percent, it is not necessary to take large samples. In this sense the common perception is correct. If investment proposals are carefully screened and only the very best of them become candidates for final project analysis, massive CV sampling efforts to measure WTP more precisely are not worthwhile. However, investments with infinitesimal risks are rare.

At the other more common extreme, when the investment is borderline because nearly half of the NPV distribution falls in the negative quadrant even though its mean NPV is barely positive, small samples of 250 observations will always be inadequate. In this situation, which can be easily identified a priori, a search for the optimum number of *additional* cases needed to augment the small sample is recommended. Variations in the size of the second sample of 20 or 30 percent around the optimum number of extra observations needed are not likely to have much effect on expected total loss (Schlaifer, 1961). So even if a second sample is necessary, money can be saved by taking only 70 to 80 percent of the recommended optimal number of additional surveys.

Like our worked example, many prospective investments fall somewhere in the middle ground between "can't miss" and borderline proposals. The existence of this gray area makes it risky to rely exclusively on any particular rule of thumb, be it for small, medium, or large samples. But, in general, given an initial expectation for WTP and a service flow outcome so the time pattern and magnitude of gross benefits are held constant, the more costly the project, the larger the sample that will be needed to justify it. This chapter has tried to show how to make that general rule operational.

The literature on benefit transfer and meta-analysis has been skeptical about the value of using accumulated past experience to estimate the benefits of new projects (Brouwer and Spaninks, 1999). The real value of this kind of information has largely been ignored because researchers have focused mainly on the degree of correspondence between predictions of WTP generated from past studies and the actual mean WTP results from field work, working under the "as if" presumption that prior information would be used to replace new sampling.

This focus might be misplaced. Prior information need not be regarded purely as a substitute for new in situ CV survey sampling. The two are complementary because combining good prior predictions of WTP with actual survey samples can save a good deal of new sampling effort and money. The synthesis of past CV results to make accumulated contingent valuation WTP information transferable to new situations might pay off, but only if the status quo were to involve the systematic use of an optimal Bayesian sample size protocol under the handicap of total ignorance or diffuse priors.

To date, international lending institutions have not systematically followed reliable protocols for selecting CV survey sample sizes in their appraisal of prospective investments, and they are not alone. In this operating environment, new information has little value beyond its immediate contribution to the specific decision at hand, which is to economically justify a given project. The WTP data are used once and then forgotten. However, this information could become more valuable if in the future sample sizes were chosen that take account of the expected opportunity loss the actual investor might incur. Then there would be a good reason to take a longer range view about the value of information.

Annex 10-A

Spreadsheet Formula Layout for the Optimum Sample Size Calculation: Tietê Case at Baseline under Diffuse Prior Information (1998 Brazilian Reals)

Row No.	COLUMN A: Labels	COLUMN B: Labels	COLUMN C: Formulas	Result	Comment
1-3	**I. Form Priors**				
4	*Prior Mean and Standard Deviation of WTP*				
5	Prior Mean	$E_0(\tilde{\mu}) = \hat{\mu}$	0.01*828	$8.28	Guess the mean WTP.
6	Prior error @50%	e	4	$4.00	Guess the variation in the mean covering a ±50% interval.
7	Prior error UL@50%	$\hat{\mu} + e$	C5+C6	$12.28	Find the upper limit of the interval.
8	Prior error LL @50%	$\hat{\mu} - e$	C5-C6	$4.28	Find the lower limit of the interval.
9	Prior Upper Alt. U @+25%	$[\mu - E_0(\tilde{\mu})] / \sigma_0(\tilde{\mu})$	NORMINV(0.75,0,1)	0.6745	Find the standard normal z statistic value for each tail outside the interval (i.e., each contains 25%)
10	Prior Standard Deviation of Population Mean	$\sigma_0(\tilde{\mu})$	(C7-C5)/C9	$5.93	Find the standard deviation of population mean WTP implied by the prior, based on the error limits, e.
11	Prior Variance of Population Mean	$\sigma_0^2(\tilde{\mu})$	C10^2	$35.17	Find the variance of mean WTP implied by the prior.
12-13	**II. Get Posterior Distribution from Normal Prior and Sampling Distributions, Sampling Variance Known**				
14-15	*Sample Data*				
16	Initial (or First) Sample Size	N_0	250	250	Input value. Number of cases in initial small sample.
17	Expected Value of Sample Mean	$E(\bar{x}) = \mu$	7.47	$7.47	Input value. Calculate (nonparametric) mean WTP.
18	Sample Variance	s^2	C19^2	$174.40	Calculate variance of WTP from sample estimate of standard deviation of WTP immediately below.
19	Sample Standard Deviation	s	C16^0.5*C21	$13.21	Calculate sample estimate of standard deviation of WTP using standard error and square root of sample size, 250.
20	Variance of Sample Mean	$\sigma^2(\bar{x}) = s^2 / N_0$	0.449	$0.70	Input value. Calculate variance of sample mean WTP.
21	Standard Error of Sample Mean	$\sigma(\bar{x}) = s / N_0^{1/2}$	0.670	$0.84	Input value. Calculate standard error of sample mean WTP as square root of variance of mean WTP.
22	*Posterior Calculation*				
23	Posterior Mean	$E_1(\tilde{\mu})$	(C5*1/C11+C17*1/C20) / (1/C11+1/C20)	$7.49	Compute posterior mean as a weighted average of prior and sample means based on quantity of information provided by each. See Rows 27 through 29.
24	Posterior Standard Error of Mean	$\sigma_1(\tilde{\mu})$	(1/(1/C11+1/C20))^.5	$0.83	Compute posterior standard error of mean as a weighted average of prior and sample standard errors based on quantity of information provided by each. See Rows 27 through 29.

Row No.	COLUMN A: Labels	COLUMN B: Labels	COLUMN C: Formulas	Result	Comment
25	Posterior Variance of Mean	$\sigma_1^2(\bar{\mu})$	C24^2	$0.68	Compute as square of posterior standard error.
26	*Quantity of Information*				
27	In Sample Mean	$I_{\bar{x}}$	1/C20	1.43	Relative Information content.
28	In Prior Mean	I_0	1/C11	0.03	Relative Information content.
29	In Posterior Mean	I_1	1/C24^2	1.46	Pooled information content.
30	Check	$I_1 = I_0 + I_{\bar{x}}$	C27 + C28	1.46	
31-32	**III. Expected Profit after First Small Sample (i.e., Based on Posterior from II above)**				
33	Linear Profit Function Intercept	α	-594653984	-$594,653,984	Input data for intercept of linear relation between NPV and WTP, i.e., NPV=α+β*WTP. Calculate outside from data generated by deterministic cost-benefit analysis model.
34	Linear Profit Function Slope	β	100,988,487	$100,988,487	Input data for slope of linear relation between NPV and WTP. Calculate outside from data generated by deterministic cost-benefit analysis model.
35	Expected Profit (NPV)	$\alpha + \beta \bullet E_1(\bar{\mu})$	C33+C34*C23	$161,738,697	Expected NPV at posterior baseline mean WTP of $5.83.
36	Breakeven Value of WTP	$\alpha \div -\beta$	C33/-C34	$5.89	Value of WTP that sets expected NPV to 0, given posterior mean WTP.
37	Standardized Loss	$\lvert D\rvert = \lvert\mu_b - E_1(\bar{\mu})\rvert / \sigma_1(\bar{\mu})$.	ABS(C36-C23)/C24	1.94	Standardized distance between breakeven WTP and posterior baseline mean.
38	Unit Normal Loss Integral	$L_N(D)$	NORMDIST(C37,0,1,0) -C37*(1-NORMDIST(C37,0,1,1))	0.010051	Unit normal loss integral factor (Schlaifer, 1961, Table IV, p. 370).
39	Expected Loss of Optimal Terminal Action	ELTA = EVPI = $\beta \bullet \sigma_1(\bar{\mu}) \bullet L_N(\lvert D\rvert)$	C34*C24*C38	$839,505	Expected loss of making an optimal "go" or "no-go" investment decision at this point <u>without</u> any additional sampling (i.e., based only on the initial priors and the original small sample N=250 cases).
40-41	**IV. Optimal New Sample Size (Depends on Data in Rows 46 to 64)**				
42	Optimal New Sample Size	ΔN	Insert Approximate Trial Size to Initialize Optimization from C83 (2683) and then Optimize	2243	Size of a hypothetical second sample to augment the initial sample of N=250. To find an optimum, iterate over alternative values of ΔN to find the sample size that maximizes the expected net gain from a second sample ENGS(ΔN) in Row 64.
43-45	**V. Expected Value of Information from a New Sample vs. Cost of Sampling**				
46	*Current Prior Set Former Posterior=New Prior*				
47	Current Prior for Mean=Posterior From III	$E_1(\bar{\mu})$	C23	$7.49	Repeat of previously computed posterior value for new set of calculations. It now becomes a <u>prior</u> value in this step.
48	Current Prior for Standard Error of Mean	$\sigma_1(\bar{\mu})$	C24	$0.83	Repeat. Former posterior in III now a prior.
49	Current Prior for Variance of Mean	$\sigma_1^2(\bar{\mu})$	C25	$0.68	Repeat. Former posterior in III now a prior.

Row No.	COLUMN A: Labels	COLUMN B: Labels	COLUMN C: Formulas	Result	Comment
50	Variance of (Population) Mean at New Sample Size, Given Population Sigma Assumed Known and ≈ s	$\sigma^2(\bar{x}) = \sigma^2 / \Delta N^2$	C19^2/C42	$0.08	Key step. Standard error of the mean of the new sample of arbitrary size ∅N=50. Used below to get revised posteriors in Rows 52 and 53.
51	*New Posteriors*				
52	Preposterior Reduction in Variance of Mean from ∅N	$\sigma^2(\bar{E}_1)$	C49* (C49 / (C49+C50))	$0.61	Calculate updated change in variance due to a second sample of size ∅N using posterior from first sample as a prior and the new sample estimate from Row 50.
53	Preposterior Reduction in Std Error of Mean from ∅N	$\sigma(\bar{E}_1)$ or σ^*	C52^0.5	$0.78	Square root of change in variance in Row 52.
54	lDl Absolute Value of Prior Standardized Loss from above	$\|D\| = \|\mu_b - E_1(\tilde{\mu})\| / \sigma_1(\tilde{\mu})$	C37	1.94	Repeat from above. Uses posterior 1 as new prior with 0 subscript
55	$\|D_E\|$ Absolute Value of Change in Standardized Loss due to ΔN	$\|D_E\| = \|\mu_b - E_1(\tilde{\mu})\| / \sigma(\bar{E}_1)$	ABS(C36-C47)/C53	2.04	Uses new posterior standard error of mean to calculate standardized difference between breakeven WTP and mean posterior to first sample of N=250.
56	Unit Normal Loss Integral	$L_N(D_E)$	NORMDIST(C55,0,1,0) - C55*(1-NORMDIST(C55,0,1,1))	0.00755	Unit normal loss integral factor for D_E (Schlaifer, 1961, Table IV, p. 370).
57	Expected Value of Sample Information	EVSI (ΔN)	C34*C53*C56	$597,647	Expected value of information in new optimal sample of ΔN^* = 2243.
58	Unconditional Expected Terminal Loss	UETeL(N)	C39-C57	$241,859	Terminal loss after a new sample of ΔN=2243 is taken. Equal to ELTA before an additional sample (i.e., at ΔN=0) minus the EVSI(∅N).
59	*Sampling Costs*				
60	Fixed Cost	K_s	0	0	Set to zero to simplify. Include actual value here.
61	Marginal=Variable Cost	k_s	89	$89	Input data. Example estimate is in 1998 Brazilian reals(R$) .
62	Total Sample Cost		C60+C61*C42	$199,680	Multiply marginal sample cost by N and add to fixed cost.
63	*Expected Net Gain from a Second Sample*				
64	Expected Net Gain from a Second Sample of Size ∅N	ENGS(ΔN)	C57-C62	$397,967	EVSI(ΔN) minus the total cost of taking an additional sample of size (ΔN). This is the objective to optimize over alternative values of (ΔN). Use a grid search (see text) or, more efficiently, Solver in EXCEL, setting the target cell as C64, equal to max; by changing the (ΔN) cell, C42, subject to the constraint that C42 is ≥ a small positive number (e.g., 0.001)

Row No.	COLUMN A: Labels	COLUMN B: Labels	COLUMN C: Formulas	Result	Comment
65-67	**VI. Addendum: Decide if Additional Sample is Necessary and Compute Approximate ΔN for Optimization Starting Value (see Annex 10-C)**				
68	*D Value Standardized Loss*	D	**C54**	**1.94**	**The first essential parameter.** From above.
69	*Intermediate Components for Calculating Z_0*				
70	*First Component*				
71	$\sigma_1(\tilde{\mu})$ From Small Sample N_0		C48	$0.83	From above.
72	σ Guess from Sample Data = s		C19	$13.21	From above.
73		$\sigma_1(\tilde{\mu}) / \sigma$	C71/C72	0.06	
74	*Second Component*				
75		k_t	C34	$100,988,487	From above
76		k_s	C61	$89	From above
77		$(k_t\sigma/\ k_s)^{1/3}$	((+C75*C72)/C76)^0.33	245.19	
78	*Z Value*	Z_0	**C77*C73**	**15.36**	**The second essential parameter.**
79		η*	((1/C78*0.5)*(NORMDIST(C68,0,1,0)))^0.5	0.0446	Crude approximation of the optimal ratio of: $n/[(k_t\sigma/\ k_s)^{1/3}]^2$ from Schlaifer.
80		$((k_t\sigma/\ k_s)^{1/3})^2$	C77^2	60,116	Denominator in ratio of $\eta = \Delta N / [(k_t \sigma / k_s)^{1/3}]^2$ See Row 83.
81	Probability an Additional Sample Should be Taken	Logit Probability Model Approximation	+1/(1+exp(2.206+C78*-1.1255+C68*-4.6102+C78^2*0.006596+C68^2*4.8539)	0.99	Logit function fit to data from Schlaifer's sample decision Figure 21.4.
82	Take an Additional Sample before Acting?		**IF(C81>0.5,"YES","NO")**	**Probably**	**Yes. Result from evaluating logit model.**
83	Quick Approximate Optimal ΔN (Ignore if Answer to "Sample Before Acting?" is "No")		**C79*C80**	**2683**	**Approximate solution for ∅N. Use in C42 above to initialize grid search or Solver optimization.**

Annex 10-B

Data and Formulas for Nonparametric Estimates of Mean WTP and Its Variance

Table 10B-1 displays the total number of cases and the number of rejections in each bid group in the balanced random subsample drawn for the 600-observation grand sample.

Table 10B-1. The Small Sample Data

Bid Group j	Bid R$ $[b_j]$	Total "No" Answers $[N_j]$	Total Cases $[Total_j]$
j=0	0	NA[a]	None
j=1[b]	2	35	98
j=2	5	34	54
j=3	12	39	51
j= M =4	20	37	47
j=M +1=5	> 20	NA	None
Column Totals:		145	250

[a]NA, not applicable.
[b]The first two bid groups (R$0.50 and R$2.00) were pooled at R$2.00 to preserve monotonicity.

THREE NONPARAMETRIC MEASURES OF THE MEAN AND THE VARIANCE OF THE MEAN

From Chapter 8, recall that there are three nonparametric estimators of the mean: (a) a lower bound measure that understates mean WTP (the Turnbull mean; see Haab and McConnell, 1997a), (b) an intermediate measure (Kriström's mean; see Kriström, 1990, and Boman et al., 1999), and (c) an upper bound measure that overstates mean WTP (the Paasche mean; see Boman et al., 1999). The logic behind all three nonparametric estimators is the same. The proportion of "no" answers at each bid level b_j provides a discrete stepwise approximation to the cumulative distribution function. The mean E(b) of a continuous random variable x with a cumulative distribution function $F(b)$[31] and probability density function $f(b)$, which is the first derivative of $F(b)$ w.r.t. b, is given by

$$E(b) = \int b\, f(b)\, db \tag{10B-1}$$

The problem is to use a discrete approximation to compute

$$E(b) = E(WTP) \approx \sum_j b_j\, f(b_j) \tag{10B-2}$$

where the range of b is from 0 to some upper limit b_{max} that forces $F(b)$ close to 1.0 because the bid is so high that almost all respondents would be unwilling to pay that amount for the environmental improvement.

[31] To obtain the mean from the survival function, $1 - F(x)$, the same reasoning developed later also applies.

Haab and McConnell's lower bound Turnbull mean sets each b_j to the lower bound of the bid interval (i.e., the first interval runs from 0 to the lowest bid offered, so b_j at j equals 0 is set to 0, etc.). The intermediate and upper bound means are obtained by simply redefining the point of evaluation, b, in each interval to some fraction κ times the lower bound plus $(1-\kappa)$ times the upper bound of the interval, where $0 \leq \kappa \leq 1$. Kriström's intermediate mean sets κ to $1/2$ (the midpoint of the interval), while the upper bound mean sets κ to 0. While Boman et al. (1999) try to put all three measures on a consistent symbolic footing, there are errors in their notation for the means and, unfortunately, their variance formulas are incorrect.[32] In Table 10B-2, all three measures are recast in Haab and McConnell's notation, which is conceptually correct.

Table 10B-2. Formulas for Nonparametric Means and their Variances

Measure	Mean[a,b]	Variance of Mean[b,c]
Lower bound	$\sum_{j=1}^{M+1} b_{j-1}p_j$	$\sum_{j=1}^{M+1} (b_{j-1})^2[V(F_j)+V(F_{j-1})]-2\sum_{j=1}^{M}(b_j b_{j-1})V(F_j)$
Intermediate	$\sum_{j=1}^{M+1}[\kappa b_{j-1}+(1-\kappa)b_j]p_j$	$\sum_{j=1}^{M+1}[\kappa b_{j-1}+(1-\kappa)b_j]^2[V(F_j)+V(F_{j-1})]-2\sum_{j=1}^{M}[\kappa b_{j-1}+(1-\kappa)b_j]\times[\kappa b_j+(1-\kappa)b_{j+1}]V(F_j)$
Upper bound	$\sum_{j=1}^{M+1} b_j p_j$	$\sum_{j=1}^{M+1}(b_j)^2[V(F_j)+V(F_{j-1})]-2\sum_{j=1}^{M}(b_j b_{j+1})V(F_j)$

[a]The probability density in bid group j, p_j, equals the difference between the estimates of cumulative density in the current and preceding bid groups, $F_j - F_{j-1}$, where, letting N_j represent the number of "no" responses and Y_j the "yes" responses in group j, $F_j = N_j / (N_j + Y_j)$. There are $j = 1 \ldots M$ distinct bids specified in the survey. The bid $j = M+1$ is the ultimate bid level that the researcher must *assume*. It presumably drives F_j to 1.0.
[b]The parameter κ is assumed by the researcher to form a weighted average of the lower and upper bound bids in any interval. Kriström's mean uses $\kappa = 0.5$, but any value of κ between 0 and 1 is admissable. If $\kappa = 0$, the Turnbull lower bound mean results, and $\kappa = 1$ returns the Paasche upper bound mean of Boman et al. (1999).
[c]The variance of each proportion F_j is equal to $[F_j \times (1 - F_j)] \div (N_j + Y_j)$.

Our preferred measure of the mean is the one used in the main text that sets κ to 0.75, which is more conservative (lower) than Kriström's intermediate mean. Table 10B-3 provides the mechanics of our intermediate mean and variance calculation. Calculation of the rest of the means and variances proceeds analogously.

[32] The Boman et al. (1999) variance formulas incorrectly treat the bid, not the cell proportions, as a random variable and are inconsistent with the respective expected value formulas because they were not derived from them using the fundamental rules pertaining to the variance of a sum of random variables. Instead, an inappropriate textbook formula was forced to stand in. We discovered this discrepancy by comparing the variances of the lower bound means produced using the Haab and McConnell formula and the Boman formula. The variance from the latter was roughly double the former. We then ran 20,000 trials of a Monte Carlo simulation in Crystal Ball, letting each cell proportion at each trial involve a draw from a binomial distribution with parameters defined as the number of observations in each bid cell and the probability of refusal. The empirical results independently confirmed the correctness of the Haab and McConnell variance formula. Our formulas for the variances of the intermediate and upper bound means were derived by extending the Haab and McConnell formula to these situations and were also successfully validated by Monte Carlo simulation.

Table 10B-3. Calculation of Intermediate Nonparametric Mean and Variance of Mean Assuming $\kappa = 0.75$

Bid Group j	Bid	Weighted Bid bwt_j $\kappa = 0.75$	Total "No" Answers	Total Cases	Cumulative Distribution	Probability Density	Bid Group j Variance	Square of Weighted Bid	Product of Adjacent Weighted Bids	Product of Adjacent Group Variances	Variance Term 1	Variance Term 2	E(WTP)
	$[b_j]$	$[\kappa b_{j-1} + (1-\kappa)b_j]$	$[N_j]$	$[N_j + Y_j]$	$[F_j = N_j / Total_j]$	$p_j = F_j - F_{j-1}$	$V(F_j)$	bwt_j^2	$bwt_j \times bwt_{j+1}$	$V(F_j) - V(F_{j-1})$	$bwt_j^2 \times [V(F_j) - V(F_{j-1})]$	$-2 \times (bwt_j \times bwt_{j+1}) \times V(F_j)$	$bwt_j \times p_j$
j=0	0		NA[a]	None	0.000000	NA	0.000000	...	...	...	...	...	
j=1	2	0.50	35	98	0.357143	0.357143	0.002343	0.2500	1.3750	0.002343	0.000586	–0.006443	0.18
j=2	5	2.75	34	54	0.629630	0.272487	0.004318	7.5625	18.5625	0.006661	0.050375	–0.160322	0.75
j=3	12	6.75	39	51	0.764706	0.135076	0.003528	45.5625	94.5000	0.007847	0.357506	–0.666802	0.91
j=4	20	14.00	37	47	0.787234	0.022528	0.003564	196.0000	350.0000	0.007092	1.389995	–2.494630	0.32
j=M+1=5	40	25.00	NA	0	1.000000	0.212766	0.000000	625.0000	0.0000	0.003564	2.227348	0.000000	5.32
		Column Totals:	**145**	**250**		**1.000000**					**4.025811**	**–3.328198**	**7.47**

Mean:	**7.47**
Variance of the Mean (Term 1 + Term 2):	**0.697613**
Standard Error of the Mean (Square Root of Variance):	**0.835233**

[a] NA, not applicable.

Annex 10-C

Approximations to Indicate Whether More Sampling Is Needed and the Size of ΔN

Schlaifer (1961) relates the need for more sampling to the values of his *essential parameters of the problem of sample size*, labeled Z_0, and the previously defined D_0, and provides a nomogram (Figure 21.4, p. 332) that indicates whether it is worth taking a second sample, depending on the values of these parameters. The essential parameter Z_0 is a function of the marginal contribution to NPV of a change in WTP (i.e., β), the marginal costs of sampling (i.e., k_s) the population's standard deviation of WTP (i.e., σ, approximated by the standard deviation, s, from the first sample), and the standard error of the mean WTP after taking a first small sample [i.e., $\sigma_1(\tilde{\mu})$]:

$$Z_0 = [\sigma_1(\tilde{\mu}) / \sigma] \times [\beta\sigma / k_s]^{1/3}$$

Since many readers may not have easy access to Schlaifer's book and the decision nomogram, we fit a logit probability model with a second-order index function to 197 pairs of D_0 and Z_0 points read from his Figure 21.4, coding the dependent variable as 1 if the nomogram recommended "sample before acting," and as 0 if it recommended "act without sampling." The model fit was reasonably good (pseudo R^2 of 0.80) with 182 correct predictions and 15 false predictions. As a substitute for Schlaifer's figure, a decision to take an additional sample should be made if the predicted probability from the model is equal to or greater than 0.5:

$$\text{probability}_{\text{sample}} = 1/\,\{1 + \exp\,[2.2061 - 1.1255\ Z_0 - 4.6102\ D_0 + 0.0066\ (Z_0)^2 + 4.8539\ (D_0)^2\,]\} \geq 0.5\ ?$$

If the answer from evaluating the probability model (or Schlaifer's Figure 21.4) is "sample before acting," it will be necessary to search for the optimum, ΔN^*. A good starting value for the grid search over ΔN can be found from a rough approximation to the optimum, ΔN_{approx} using another simplification from Schlaifer (1961, pp. 334–335):

$$\Delta N^* \approx \Delta N_{\text{approx}} = [(\beta\sigma / k_s)^{1/3}]^{1/2} \times [\tfrac{1}{2} Z_0 / P_N(D_0)]$$

where $P_N(D_0)$ is the probability density of the standard normal density function evaluated at D_0. The optimum size ΔN^* of the addition to the original small sample ($N_0 = 250$) can be found either through trial and error by constructing a crude fixed-step size grid search in the neighborhood of the initial guess, ΔN_{approx} or by calling an optimization routine like Excel's Solver after specifying ΔN_{approx} as the starting value.

Chapter 11

Conclusions and Recommendations

This book has covered a lot of ground, even though it stops well short of being a straightforward manual for doing cost-benefit analysis of water quality investments. For example, it has tried to clarify the range of applications of the cost-benefit idea itself, from optimizing over a complete set of possible investments, to choosing the best of a limited set of possibilities, to offering a test of the (aggregate economic efficiency) performance of a particular project proposed for possible funding by some technical process. This process at one extreme might be an ad hoc consulting engineering report, the charge for which was to design a sewage treatment plant for a particular location, raw wastewater load, and desired effluent quality. At the other extreme, the project might be one component of a basinwide plan found by solving the problem of minimizing the costs of meeting a given standard of in-stream water quality everywhere in the basin. The study argues and tries to demonstrate that the latter approach is preferable if an accommodating regulatory environment exists and enough lead time can be allocated for data collection and analysis.

Chapter 2 was devoted to a review of IDB experience with cost-benefit analysis in the context of water quality investments. Here, the analysis used in the justification of 27 projects was described and characterized on a number of dimensions. This was intended to set the stage for further conceptual discussion by illustrating the methodological challenges that have to be dealt with.

An effort was also made to identify and warn against some common pitfalls in economic analysis, such as the use of "stemming" benefits; the challenge of identifying optimal solutions when functional forms are uncertain or known to produce nonconvex feasible spaces; and the distinction between cases in which it is and is not safe to ignore sources upstream of the would-be project. There was even a discussion of when concern about spending on future maintenance and repair on the completed project pushes the analysis into the realm of a truly dynamic problem, so that myopic solution methods are no longer acceptable.

Economic analysis of water quality improvements cannot be realistic unless it is integrated with modeling of natural systems. Because the water quality context is, in general, important, considerable space and effort was devoted to a discussion of publicly available water quality models. Two such models were contrasted in terms of whether or not the water quality change attributable to a particular change in the discharge of pollution depends on the level of water quality existing just upstream of the discharge in question. A decision support system having at its heart a quite simple water quality model was exercised to illustrate how answers to basinwide cost-minimization questions vary with location of pollution loads.

Two chapters were devoted to benefit or damage estimation. The first of these concentrated on the relative size of different benefit categories in developed and developing nations (air vs. water, health vs. recreation, for example). The second went into methodology, at least to the extent of characterizing frequency of application of particular methods in the developing country context. That chapter came down on the side of the so-called "direct" methods as currently and potentially most useful in Latin America, largely because of the combination of data unavailability and conceptual difficulties with obtaining all and only all the benefits via the so-called indirect methods.[1] This conclusion is consistent with IDB practice.

The next three chapters dealt with the critical problem of uncertain benefits, which has received less attention than it deserves in most economic project analyses done by the IDB. The value of the services provided by environmental quality improvement projects is subject to large margins of error. These services are not sold in markets and have no established prices, and often have to be valued through stated-preference surveys (i.e., contingent valuation). These uncertainties are necessarily transmitted into uncertainties about the net present value of environmental investments. The NPV distribution can, and, we argue, should be quantified and evaluated during the project design and approval process.

Chapter 8 demonstrates that the statistical techniques for imputing benefits using referendum contingent valuation data are sensitive to the analyst's assumptions. Equally justifiable parametric and nonparametric evaluation routes to average benefits may produce very different mean estimates, estimates that are outside the 99.5 percent statistical confidence intervals generated by any given route. In short, average (and by implication, total) benefit estimates based on referendum contingent valuation data are fraught with *statistical* uncertainty. This source of uncertainty can be measured by the standard error of the mean, computed using any particular approach. However, the more important *methodological* uncertainty about which approach to extracting mean benefits is best has no such easy resolution, and both types of uncertainty about benefits matter.

In addition to uncertainty about benefits, environmental investments often are accompanied by uncertainties about execution timing as a result of institutional obstacles, divergent interests of stakeholders, and the behavior of the natural world the project operates on and in, as well as the more familiar uncertainties about costs and economic prices. To reflect all of these uncertainties, the economic cost-benefit analysis demonstrated in Chapter 9 employs Monte Carlo simulation, which permits their effect on the distribution of a project's net present value to be quantified.

Chapter 9 argues that Monte Carlo risk analysis offers a more comprehensive and informative way to look at project risk ex ante than the traditional (and often arbitrary), one-influence-at-a-time sensitivity analysis approach customarily used in IDB analyses of economic feasibility. The case for probabilistic risk analysis is made using data from a project for cleaning up the Tietê River in São Paulo, Brazil. A number of ways to handle uncertainty about benefits are proposed, and their implications for the project acceptance decision and the consequent degree of presumed project risk are explained and illustrated. This exercise not only changes the terms of the "go or don't go" investment decision, it also identifies areas of uncertainty where it would be most valuable to improve the available information.

One of the most obvious ways to reduce uncertainty about benefits is by taking larger CV survey samples. Traditionally, the sample size question has been answered in an unsatisfactory way by either dividing an exogenously fixed survey budget by the cost per interview or by employing some variant of a standard statistical tolerance interval formula. Neither of these approaches can balance the gains to additional sampling effort against the extra interviewing costs. Chapter 10 explains and illustrates with an example the rationale for and mechanics of a sequential Bayesian technique for finding the optimal CV survey sample size, which is only applicable when there is some monetary payoff to alternative courses of action that can be linked to the sample data. In this sense, unlike pure contingent valuation studies that are unconnected to a policy decision, investigators who use contingent valuation results directly

[1] We also discussed at some length a problem with the dichotomous choice method of doing direct willingness-to-pay surveys (Chapter 8).

in cost-benefit analysis have a hidden advantage that can be exploited to optimize the sample size. The advantage lies in exploiting the link between willingness to pay and the decision variable, the net present value of the prospective investment.

All this material might fairly be characterized as the sort of technical background that is useful once it has been decided to do a cost-benefit analysis. However, intuitively it does not make good economic sense to require the same level of analytical effort for every possible project. Indeed, it seems likely that some possible projects will involve such small stakes that there will be no point in doing even a very modest cost-benefit analysis. A question worth some consideration in this last chapter, then, is whether it is possible to go much beyond that simple intuition without actually doing a cost-benefit analysis for every project during some "experimental" period, observing the sizes of the largest possible losses avoided, and using those data to construct a rule or set of rules for when the analysis does and does not have to be done. A related question is whether there is any evidence that when cost-benefit analysis is done it actually improves the observed performance of projects in the real, developing world.

A first observation is that the question of when and whether to do a cost-benefit analysis is as old as the technique. Two answers are common and useful in practical settings. One is to adopt rules of thumb that use one easily determined project dimension as the key to the decision about analysis. In the case of water quality improvement projects, two obvious (and closely related) possibilities are cost and flow. That is, roughly estimating the flow that a possible future treatment plant and sewer system would have to handle is largely a matter of counting families and businesses and multiplying by easily available rules of thumb. Then a second-level rule of thumb can be invoked that requires further analysis when an estimated cost or flow is greater than some chosen level. How to choose the magic level? One possibility is to ask how low the benefits might be for such projects as a ratio to costs—how badly chosen the site or level of treatment might be. Then an upper limit on acceptable possible "loss" (negative net expected present value of benefits) implies a level of cost at which such losses *could* be unacceptable. At and above that level, analysis should be done. (The risk analysis in Chapter 9 can be seen as the prototype for deciding how large a potential "failure" can be.[2])

A second answer is to adopt different levels of analysis at different levels of estimated cost or flow. This might be called "the screening approach," and could be combined with threshold rules. For example, at some cost level, C_1, a very rough cut at analysis could be made. This rough cut could take into account the current condition of the receiving water, the size of the affected population, and that population's income level. Ad hoc adjustments could be made for special features of the situation, such as important resources threatened by poor water quality (a reef or inshore fishery important to a regional economy). The results of the screening analysis

[2] If a rule is devised that limits the performance of cost-benefit analysis to projects with "large enough" potential negative net benefits, a simple game can be constructed within which doing the analysis is the risk-averse strategy. Thus there are three possible strategies for the "agency": Do the project without any analysis; don't do the project and don't do the analysis; do the project if the cost-benefit analysis supports it. Assume the decision maker is not in general ready to abandon any project without some investigation, so the strategy of doing neither analysis nor project is irrelevant. There are two possible outcomes after the project is done: It is a good project, so that $B_G - C_G > 0$; or it is a bad project, so that $B_B - C_B < 0$. The cost-benefit analysis discriminates perfectly ex ante over the ex post outcomes. Then the payoff matrix is as follows, where C (CBA) stands for the costs of doing cost-benefit analysis.

	Good	Bad	Worst Outcome
Do project/no cost-benefit analysis	$B_G - C_G$	$B_B - C_B$	$B_B - C_B$
Do project if cost-benefit analysis says it is "good"	$B_G - C_G$	–C (CBA)	–C (CBA)

A rule for doing a cost-benefit analysis that keys off maximum possible losses would ensure that $|C(CBA)| < |B_B - C_B|$. So doing a cost-benefit analysis is the risk-averse choice.

could be the basis for a decision about the project or about further analysis.[3] (For example, a result within some range around zero net benefits could trigger a closer look, while sufficiently large positive or negative numbers could trigger a decision.) A conceptually related problem—the choice of how many individual projects in a multiple-works program proposal should be subjected to the net benefit test—is addressed by Vaughan (1994). A translation from a sample-size question to an analytical-effort question could provide useful insights.

It is important to recognize, however, that getting from where the IDB is today to such a graduated decision world would require an up-front investment in analysis and methodological development. The thresholds for any analysis and for closer looks at questionable projects would have to be determined. Much more challenging, the screening methods would have to be agreed on. It also seems clear, however, from the review of IDB experience in Chapter 2, that a range of methods are already being employed in analysis. The choices are not necessarily being made on the basis of the size of potential losses to be avoided. Indeed, it is not clear that any rationale lies behind the across-project variation in method.

The above suggestions are based on the notion that the primary purpose of cost-benefit analysis is to "stop bad projects" (Jenkins, 1997, p. 40). This same notion suggests that an answer to the question "Does analysis do any good?" ought to be based on differential failure rates. But this brings the discussion up against several logical and practical difficulties.

The first of these is that you cannot observe the results for projects not built. So no information relevant to the ex post value of analysis is generated by projects that die at the analysis stage. The second observation is that project "failure" in the water quality investment area cannot be observed by looking at standard financial accounting data. Failure can only be observed (and then only probabilistically, as demonstrated in Chapter 9) via ex post cost-benefit analysis. This puts a rather substantial price tag on finding an answer to the question about the value of analysis. A third observation, related to the second, is that financial success is not a guarantee of economic success in this area. This is true despite the expectation that the revenue extractable, via some uniform (nondiscriminating) charge scheme, ought to be lower than the affected population's willingness to pay for the service in question. The problem is that once a sewer system, with or without attached treatment plant, is built and households and business are attached to it, it will generally be difficult or impossible for a household (or business) to choose not to continue paying for the project even if, on ex post reflection, the benefits do not seem to be worth it. This is just another reason that a full cost-benefit analysis will be necessary to separate the actual failures from the actual successes.

What about evidence, however? The most useful and relevant evidence found for this report comes from analyses of World Bank data on resources devoted to analysis at various levels and on measures of project success, as reported in Jenkins (1997) and Deininger et al. (1998).

Jenkins reports on work that classified approved project outcomes in four "grades," from highly satisfactory to highly unsatisfactory, and correspondingly classified the economic analyses of those projects according to their quality (also categorically). A logit model was estimated that related the quality of the analyses to the probability of unsatisfactory results. After 3 years of life, a project for which the a priori appraisal was of poor quality was found to be 7 times more likely to be performing unsatisfactorily than a project for which the analysis was of good quality. After another year, that probability differential had grown to 16 times (Jenkins, 1997).

While the Deininger et al. paper is quite long and complex, and does not appear to include results from the environmental "sector," its lessons also seem likely to be applicable. Briefly

[3] The World Bank's decision support system for integrated pollution control amounts to the discharge and cost-of-reduction side of a screening model. Adding a rough benefit side to it would produce a complete screening model (World Bank, 1997). See also Chapter 5 of this document.

stated, those results are that background research, which might be seen as setting the stage for specific project planning and preparation, is reflected in higher rates of returns to projects and in lower effort required in specific project planning and supervision. In the context of this report, it seems reasonable to identify their background research (economic and sector work) with basinwide cost-minimization efforts.

This line of argument brings the discussion back to a conclusion reached on a tentative basis in Chapter 1. There it was suggested that in the current state of the benefit estimation art, a practicable and potentially very useful procedure for analyzing water quality investments is to begin with basinwide planning based on minimizing the costs of meeting given ambient water quality constraints. Then, the projects identified through that exercise can be held up to the standard of the cost-benefit test. That is, it can be required that each potential project pass a cost-benefit test as a condition of approval.[4] This is essentially the strategy recommended on the analytical side by Mariño and Boland (1998) although they embed this within an elaborate structure of ex ante public consultation and ex post monitoring and evaluation.

There is no guarantee that the resulting set of projects is the "best" set in cost-benefit terms, since nothing about their benefits was used in identifying them. But the method does combine conditional acceptance of a nation's choice of an ambient quality standard with a check on whether those standards are supported by the benefits and costs that can be attributed to projects that help attain the standards. It is possible to do considerably worse by ignoring the analytical tool, and the state of the art does not allow for doing better in general.

[4] When the cost-benefit analysis is done for each project, something must in general be assumed about the status of the other projects. The only fully symmetric assumption would seem to be that the other projects all exist. In fairness, some experts do not appear to agree with our position that cost-benefit analysis, when wed to basinwide decision support modeling systems, is both possible and useful (Ongley and Booty, 1999; Ongley, 1998). Opponents would argue that in the developing country context, any effort along the lines of model-based cost-benefit analysis described in this document is doomed from the start because of the lack of data and of ecological knowledge about the specific situations encountered. In this view, such evaluations enrich donor country consultants without leaving the necessary skill behind in the country with the problem.

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Note on references. The following groups have been cited by acronym in the text: American Public Health Association (APHA), American Waterworks Association (AWWA), Environmental Defense Fund (EDF), Environmental Protection Agency (EPA), Economic Commission for Latin America and the Caribbean and Inter-American Institute for Cooperation on Agriculture (ECLAC/IIACA), Food and Agriculture Organization (FAO), Federation of Sewage and Industrial Wastes Association (FSIWA), Inter-American Development Bank (IDB), National Academy of Sciences, National Academy of Engineering (NAS/NAE), National Oceanographic and Atmospheric Administration (NOAA), National Resources, Environment and Rural Poverty Division (NRERPD), U.S. Geological Survey (USGS), World Resources Institute (WRI).

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