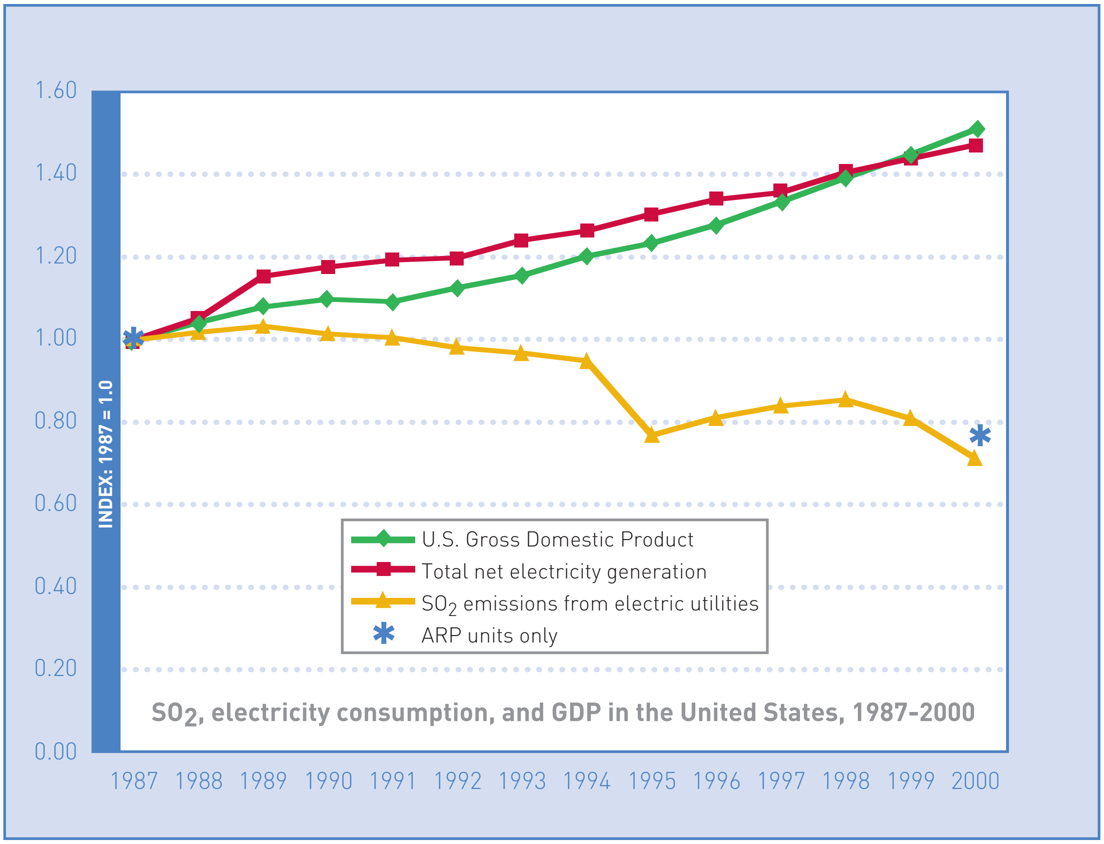
Environmental Protection in Transition Economies



THE NEED FOR ECONOMIC ANALYSIS

ENVIRONMENTAL DEFENSE

finding the ways that work

Environmental Protection in Transition Economies

THE NEED FOR ECONOMIC ANALYSIS

AUTHORS
Alexander Golub
Daniel Dudek
Elena Strukova

WITH CONTRIBUTION FROM
Anil Markandya

environmental defense
finding the ways that work

Cover illustration source: DOC (U.S. Gross Domestic Product), DOE (total net electricity generation), and EPA (SO_2 emissions from electric utilities)

The authors wish to acknowledge Hallie Katarski for her help in preparing this report and USEPA for providing the grant which supported this study.

Our mission
Environmental Defense is dedicated to protecting the environmental rights of all people, including the right to clean air, clean water, healthy food and flourishing ecosystems. Guided by science, we work to create practical solutions that win lasting political, economic and social support because they are nonpartisan, cost-effective and fair.

Contents

CHAPTER 1

Introduction

Until recently, the countries of Central and Eastern Europe (CEE) and The Former Soviet Union (FSU) operated with centrally planned economies. As they began to make the transition to a market economy and to build market institutions in the late 1980's and early 1990's, their economies came under severe pressure for a number of reasons:

- The Soviet Union collapsed in 1991, and the economic ties between countries in the former Socialist bloc were destroyed;
- Far-reaching market reforms including price liberalization started in 1992;
- The globalization of the world economy continued to be rapid.

It soon became apparent that planned economies were not appropriate for the new market realities. A ten year adjustment period brought about an economic crisis that has only gradually turned into a slow recovery, first in the CEE countries and subsequently in the Newly Independent States (NIS) that succeeded the FSU. The main government concerns in this period were abysmal economic growth, high rates of unemployment and inflation, and severe social problems.

Environmental concerns seemed to be less pressing due to sharp production declines which resulted in reduction of all types of pollution. However, even at that point, it was clear that these reductions would prove temporary. Pollution was not reduced proportionately to the decrease in GDP. The anticipated future economic growth in this region now makes the situation more urgent.

This paper makes the case that environmental factors must be taken into consideration in the economic decision-making process. Delaying appropriate action could worsen the situation. The theoretical background of environmentally friendly decision-making lies in understanding the difference between forward-looking and backward-looking approaches. This leads to a shift in the Environmental Kuznets Curve which measures the willingness of society to address environmental concerns.

It is critical to build market and environmental institutions simultaneously. The new entrepreneurs must understand and recognize their environmental responsibilities. The ideal time to build a strong sense of responsibility for environmental protection is precisely during the transition from state to private ownership of property.

This paper presents the results of environmental valuations derived from the US experience, and proposes an estimate of the environmental benefit-cost analysis for the EU accession countries (those countries now in line to join the EU). The economic benefits of an environmentally friendly approach are demonstrated. In addition, the case is made that the cost of environmental protection can be dramatically reduced if policy makers select proper instruments: the US SO_2 trading program is an excellent example of a very ambitious environmental target being met using a least-cost approach.

CHAPTER 2

Characteristics of transition economies

Countries with transition economies present certain specific problems. While there is a regulatory framework, it is different from the one found in developed market economies. Standard microeconomic analysis can be applied to developed markets and outcomes explained through that analysis. A transition economy, on the other hand, is by its nature, unstable. In essence, it is moving from one temporary and stationary economic state, to another.

Transition economies have certain specific features, previously unknown to researchers. These features influence the behavior of managers responsible for the economy. Their reaction to various management tools is frequently different from the reaction of managers in developed markets. It is therefore impossible to apply knowledge acquired in the environmental protection field in a similar manner.

Almost all countries with transition economies are still passing through a profound economic crisis. Only in the last two years have they been experiencing the beginnings of economic growth. The nature of this crisis arises from a fundamental readjustment of the technological infrastructure that was developed for a planned economy. The former structure was inefficient with respect to the sustainable use of natural resources, and with respect to policies with a less destructive environmental impact. At present, countries with transition economies have a real opportunity to replace an old obsolete structure with a new, more environmentally friendly structure.

Concurrently, a serious crisis of investment limits the capital available for restructuring. Government investment has been either cut back or stopped. Enterprises do not have sufficient resources to invest. The banking system is still embryonic, and foreign investors are cautious in this environment of high uncertainty and are in no hurry to invest in developing economies.

Moreover, most countries are experiencing current account as well as capital account deficits. The result is a net outflow of capital that would otherwise have been reinvested in the domestic economy. Some countries are experiencing export growth of natural resources that is resulting in their overexploitation. For example, in Russia, the annual capital account deficit is about 30% of total exports (Bureau of Economic Analysis, 2000, p. 358).

High levels of inflation and inflationary expectations are also common in transition economies. Clearly, the inflation rate has gradually been decreasing. However, even in countries that are quite successfully restructuring their economies, it is still higher than in more developed countries. High inflation decreases a country's investment attractiveness, which thereby reinforces another feature of a transition economy; namely, high investment risk and a low propensity for technological innovation.

Transition economies typically have high discount rates. This fundamentally influences the behavior of those responsible for the management of the economy. Environmental benefits are rarely considered even in the medium term, and willingness to pay (WTP) for environmental services generally is low. Given an uncertain future, the decision-making horizon for individuals and companies tends to be very short. Society does not seem to want to take into account that government policies can have a negative environmental impact in the future.

The problem of disseminating or sharing information is much deeper in transition economies than seems at first glance. Without information, firms would have to pay huge transaction costs that shift their supply curve. However, the market structure is under-developed, the advertising culture is poor, transparency is low, and information is not shared the same way as in developed markets.

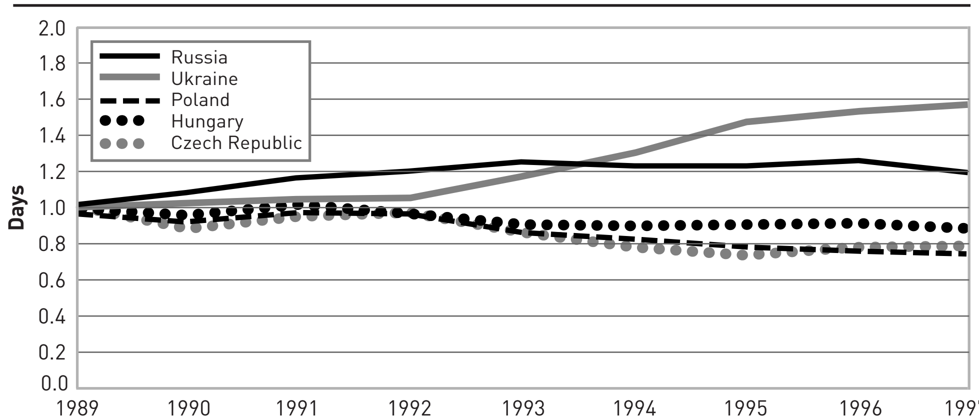
The result is that lack of information significantly increases transaction costs and hides the underlying environmental costs. The absence of environmental information could be balanced out by investors who wish to avoid unexpected liabilities in the future. Special agencies or private firms could create a proprietary information base for green technologies that will help enterprises allocate capital in a more efficient manner. However, this will not occur unless fundamental environmental obligations are assigned to firms.

From an environmental viewpoint, a drop in GDP is positive in that it generally goes hand in hand with a reduction in hazardous materials pollution. However, if one were to analyze the dynamics of emissions per unit of production, these ratios would increase in tandem with GDP growth in almost all transition economies. The trend is dangerous because economic development will almost certainly be accompanied by higher emission of pollutants. At the same time, while the absolute totals rise, pollution per unit of production amounts may actually decline. In sum, the correspondence between pollution and units of production is not clear-cut and depends a great deal on the government's environmental policy.

The "Environmental Performance Review of the Russian Federation" (OECD, 1999, pp.29, 45) stresses that total atmospheric emissions have decreased considerably in Russia, mainly as a result of the decline in economic activity. However, the decreases of emissions of SO_2 (16% decline) and NOx (29% decline) were smaller than the actual decrease in GDP (about 48% in 1999). Water-effluent discharges were reduced by only 5.7%.

In Ukraine, due to the acute decline in energy and fossil fuel usage, SO_2 emissions were reduced by almost 60%, and NOx emissions were down by more than 50%. At the same time, Ukrainian GDP declined by more than 60%. In general, the trend is negative in Ukraine, as the energy intensity of its GDP has increased dramatically.

FIGURE 1
GDP energy intensity: comparison of select countries



 http://helio.interserver.net/Helio/Reports/2001/Ukraine/Figure1_2.html

To summarize, the specific macroeconomic features of a transition economy are:

- Deep economic crisis;
- A crisis of investment;
- High inflation and high inflationary expectations;
- High discount rate;
- High investment risk and a weak culture for technological innovation;
- Uneven distribution of information;
- An increase in emissions per unit of GDP.

As mentioned previously, environmental considerations seem less important when GDP drops, as regulatory authorities have a tendency to pay attention to the absolute numbers in reduction of pollutants. However, economic growth is inevitable in the near term. In the last two years, GDP in both Ukraine and Russia has started to pick up and it seems this trend will continue. At the same time, a weak culture for technological innovation remains. In a risky economic environment, it is cheaper to grow GDP based on old technologies. The result is that environmental degradation becomes inevitable in the absence of a robust environmental management system. Unfortunately, this is the case in most transition economies.

Environmental concerns have not been a primary concern for these countries over the last few years. Environmental authorities have become weak, and the former environmental regulatory system has all but disappeared. Some authorities have been reorganized; others have fallen victim to industrial lobbyists. For example, the State Environmental Committee in Russia was abolished in 2000. Its functions were transferred to the Ministry of Natural Resources, whose goal is to ensure profitable use of natural resources. Environmental protection is not its first priority. The Ministry was so inattentive to environmental management that industrial lobbyists in 2001 convinced lawmakers to invalidate the last aspect of environmental regulation—pollution fees. Moreover, ecological courses were taken away from secondary school programs, indicating the public's general loss of interest in environmental problems. With economic growth in place, but with no regard for the environment, the impact can be immediate and far-reaching.

The behavior of companies in transition economies also contributes to environmental negligence. Enterprise behavior is defined by the parameters and constraints imposed on it. Classical economists believe that companies seek to maximize the difference between the benefits and costs to achieve growth. In transition economies, environmental costs and benefits are not calculated, while in developed economies, revenue from the sale of goods and services is the main source of income, but environmental costs are also considered.

Environmental goods and services in transition economies are often not taken into account due to their non-immediate nature and due to the absence of reliable valuation methodologies. Economic growth and job creation is undoubtedly a first priority. Some countries with transition economies grew faster than others but the cost was a severe impact on their environment. In part, this was out of ignorance of the damaging consequences for human health and for the livelihood of those who depended on the use of natural resources. In part, it was due to a lack of effective environmental regulations. In hindsight, however, it has been estimated that society in these countries will end up paying a higher cost in terms of damages and required corrective action, than if these regulations had been in place at the time rapid growth started[1]. Furthermore, since many companies, if not society as a whole, are operating at under-capacity, it is possible to improve environmental quality through improvements in operating efficiency.

The problem of taking into account an environmental benefit-cost analysis is a major priority for decision-makers in emerging market economies. It is essential if one is to achieve efficient resource allocation and encourage rational decision-making by enterprises.

CHAPTER 3

Environmental valuation in the decision-making process

Transition economies are at a development stage in which the greatest improvement in environmental performance is a result of better management of scarce resources on the one hand, and of technological modernization brought about by new investment on the other. If a proper environmental regulatory framework is in place at the beginning of the modernization effort, and if enforcement is credible, environmental considerations will be incorporated into investment decisions. Businesses can consequently avoid expensive retrofitting at a later point in time. It is precisely economic valuation that can help direct resources to those activities, which yield environmental benefits at the least cost.

These components of total value lend themselves to environmental valuation methodologies, and this makes it possible to express at least some of them in monetary terms.

In the framework of central planning, environmental valuation was not necessary on a macro level. Instead, costs were calculations to achieve certain environmental standards. Environmental standards themselves were based solely on epidemiological data, and no economic considerations were taken into account.

Institutional constraints of the centrally planned economies prevented the organic development and incorporation of benefit-cost analyses on the regional and enterprise level. No incentive existed to use correct methodologies in estimating environmental damage and valuing natural resources. At the household level, no institutional preconditions were present to measure willingness to pay for environmental protection and natural resource supply. Technically and institutionally, these were considered "public goods."

The essence of economic values

Economists assume that, for practical purposes, value is assigned as a factor of consumption rather than as a factor of production. In other words, nothing has a value unless it serves some human need either directly or indirectly as a factor of production. Even though economic values are derived from satisfying human needs, we need to consider other factors.

Economists recognize that there are several components of value. The total economic value attributable to the use of the environment and to the use of natural resources is obtained by adding value estimates for the following five major categories:

- **Direct use of extracted physical resources:** examples include consumption of goods provided by biological resources such as timber, fibers, food, and medicinal herbs as well as fossil fuels and other mineral resources.
- **Direct use of non-extracted physical resources:** examples include consumption of services derived from natural resources such as tourism, recreation, education and scientific research.
- **Indirect use:** society benefits from ecological functions that support economic activity and human welfare, e.g.: waste dissipation/assimilation, climatic functions, and water retention provided by forests.
- **Optional use:** reflects known and hypothetical future uses of any type listed above, e.g.: preserving bio-diversity as an insurance policy against various events and keeping society's options for future use open.
- **Non-use (or passive use):** reflects the satisfaction of the existence of a resource such as a mineral deposit, an organism, species or ecosystem, as well as the satisfaction of passing on that resource to future generations.

Source: Using Valuation of Environmental Impacts in Decision-Making. Paper for the NIS Ministerial Conference in Almaty, Kazakhstan; World Bank, 2001.

Due to a closed economy, distorted price mechanisms, and hidden energy and raw material subsidies, there was no need to evaluate natural resources and assess the sustainability of their use from the perspective of exhaustibility. However, between 1970 and 1990, some important preconditions for environmental valuation were in fact created:

- A system of environmental protection institutions was established, reflecting growing public concern about environmental quality;
- Information and monitoring systems were set up;
- Training programs were developed to conduct engineering studies and cost-effectiveness analyses in the environmental sphere;
- It became common practice to conduct Environmental Impact Assessments for large-scale projects.

In transition economies, environmental valuation can provide a solid basis for economic analysis of environmental policies and projects. Incorporating these external factors would reveal the total social costs of projects and policies on a local, regional and macro level. The most important categories of goods and services provided by the environment that need to be valued are health protection, natural resources, and environmental amenities including bio-diversity. Different environmental valuation methodologies are designed to capture different elements of the total project cost. All these methodologies have been successfully applied to pilot studies in the countries of the CEE and the NIS. Using them would strengthen the decision-making process in the NIS because the history of environmental valuation in the NIS was mostly an artificial calculation of pollution damage. The pollution charges system in the NIS reflected the shortcomings of this environmental damage calculation.

At present, environmental impact is checked for compliance with Maximum Allowable Concentrations of pollutants in air and water (the so-called "PDK"). The base for PDK is the "no impact on human health" approach. New valuation methodologies, i.e. health risk analysis, would be a more standard approach, and would optimize environmental policy by targetting a reduction in human health risk from decreased environmental pollution at the least cost. Engineering cost assessment, and comparative emission and risk reductions, should also include an estimate of the willingness-to-pay for improvements in environmental goods and services.

New regulatory policies should be developed to gauge the impact of industrial activities on different environmental components. Health risk reduction could be an important criterion in this process as well. All technical decisions about industrial improvements should at the very least take into account the benefit of health risk reduction. It is also advisable to consider ecosystem damage. However, a scientific approach to this measure is still difficult and should be analyzed based on available data in each case.

Willingness-to-pay for environmental improvement, or willingness to accept compensation for the deterioration of environmental services, should be the major focus of environmental valuation. This will allow health risk reduction characteristics to be valued, and other estimates of the total economic value of environmental resources to be developed.

Economic valuation of environmental impacts: a brief overview of methodologies

CLASSIFICATION OF METHODOLOGIES

Economic valuation methodologies can be broadly categorized as involving either a physical or a behavioral linkage between indicators of environmental quality and its observed effects on health, productivity, or on natural resource assets. Physical Linkage Methodologies, known also as dose-response methods, focus on the technical relationship between environmental degradation and physical damage but do not account for the subjective preferences of the affected population. Behavioral Linkage Methodologies assume that the value of environmental goods and services should be based on people's willingness to either pay for improved environmental quality or to prevent its deterioration.

Behavioral Linkage Methodologies can be subdivided further depending on whether the preferences are revealed indirectly, through market behavior, or directly, through a statement. The Revealed Preference Approach (also known as a Surrogate Market Approach) derives the demand for environmental goods on the basis of individual demand for complementary goods that happen to be traded on the market. The Stated Preference Approach assumes that people would respond to hypothetical market situations as if they were in an actual market situation.

PHYSICAL LINKAGE METHODOLOGIES

Four valuation techniques are used to value the health and safety effects of environmental degradation, and to measure the costs of material damage and productivity losses. These are the Cost-of-Illness Approach, the Human-Capital Approach, the Cost-of-Productivity-Loss Approach, and the Replacement-Cost Approach. All four methodologies require estimating a damage function or a dose-response function that relates exposure of environmental impacts to changes in health and productivity. Thus, physical linkage studies are usually interdisciplinary tasks.

The procedure of estimating health and productivity effects consists of three steps:

1. Establish a relationship between the exposure associated with different levels of environmental quality to human mortality/morbidity rates, the productivity of bio-resources, and/or material damages. This requires risk assessment studies for health effects or damage function studies for productivity loss.

2. Calculate the magnitude of physical damage for the specific situation using the dose-response coefficients estimated in Step 1.

3. Evaluate the monetary cost of measured damages using market prices for medical expenses and resource costs, and using statistical estimates of the value of life.

A frequent drawback of Physical Linkage Methodologies is the lack of scientific knowledge about cause—effect relationships or the lack of relevant data to establish such a relationship.

BEHAVIORAL LINKAGES METHODOLOGIES

The main techniques are the Hedonic Pricing Method, Travel Cost Method, Averting Expenditures Method, and the Contingent Valuation Method. The first three are revealed preference approaches while the last is based on stated preferences.

Hedonic Pricing Methodologies derive their name from the Greek word "pleasure" and aim to measure the implicit value of environmental quality as revealed by individuals' preferences for related market goods. Commonly used markets are the housing or labor markets. The rationale is that WTP for environmental quality and safety can be inferred from information on price and wage differentials on the one hand, and the environmental characteristics/risks associated with a specific area or job, on the other. The data and econometric knowledge requirements for conducting a hedonic pricing study are quite burdensome. Moreover, in a heavily regulated or otherwise distorted housing and labor market, the available market data will convey the wrong information.

Travel Cost Methodologies are widely used to estimate the amenity and/or recreational value of outdoor recreational sites such as parks and lakes. The underlying assumption is that people's demand for the recreational site is revealed through their willingness to spend money and time traveling to that site. Data on travel costs and other socio-economic characteristics of the users is collected through a site survey to produce the aggregate demand curve. The method is less applicable to urban amenities requiring only short trips.

Averting (or Mitigating Behavior) Expenditures Methodologies study the costs (monetary and opportunity) that people incur in order to avoid adverse environmental impacts. Expenditures are usually for substitute goods (e.g. buying bottled water instead of using tap water) or for activities reducing the associated environmental impact (e.g. cost of soil-erosion prevention). The underlying premise is that an individual's perception of the cost imposed by adverse environment quality is at least as great as the individual's expenditure on goods/activities to avoid the damage.

Contingent Valuation Methodologies (CVM) attempt to measure individual WTP by directly questioning a representative sample of individuals. This method has universal applicability in valuing non-market environmental goods, including non-use values such as option and existence values, and has a very minimal requirement for secondary data. CVM studies are conducted using a survey instrument the design of which is important in interpreting the results. Standard survey design quality assurance procedures need to be followed (such as control questions, pilot of the instrument, representative samples, interviewer training, etc.). CVM studies have received mixed reviews. Major criticism centers on the potential discrepancy between stated behavior in a hypothetical situation and actual behavior. Yet, many researchers feel that conventional biases can be dealt with through better survey design or through appropriate interpretation and qualification of final results.

Source: Using Valuation of Environmental Impacts in Decision-Making. Paper for the NIS Ministerial Conference in Almaty; World Bank, 2001.

Benefit estimates based on willingness-to-pay for environmental goods and services should be used to evaluate proposed projects and policies using the net-benefit criterion (i.e., whether benefits exceed costs). Estimates of willingness-to-pay can be directly obtained from market information, or indirectly from productivity measures (shadow prices), market prices for substitute or similar goods and services, and direct, but expressed valuation, as well as bids. The choice and proper application of these techniques should depend on the particular good or service being valued. At times, multiple methods can be used to compare results. Benefits studies produced should be documented and reviewed by independent experts before acceptance.

Even if all benefits and costs cannot be monetized, they should nonetheless be estimated. For example, the value of health risk reduction has many obstacles, both technical and philosophical. As mentioned above, ecosystem damage is also very difficult to monetize.

In sum, environmental valuation is a very useful tool for decision-making. It can be used for:

- Setting priorities for economic development at the federal, state, and local level;
- Setting of standards;
- Selection of alternative policy options;
- Environmental impact assessment at the project level;
- Establishing priorities for project financing from dedicated environmental funds as well as from state and local budgets;
- Environmental damage assessment for compensation claims in court.

Environmental valuation places environmental pollution in a market framework and reveals its impact at the macro level. It therefore increases the possibility of obtaining funding for improved environmental management, and duly recognizes competition for funding from other sectors. As the costs of environmental projects are also expressed in monetary terms, it becomes possibile to do benefit-cost analysis to demonstrate whether, including environmental benefits and costs, total benefits exceed costs. This comparison is usually conducted in the form of presenting both benefits and costs in terms of present value at various discount rates. The solution is to choose projects with a net benefit or gain from among those under consideration, or to analyze alternative investment options using a benefit-cost approach.

Benefit-cost analysis is generally used when it is possible to present benefits in monetary terms. However, for many projects this is either not possible or of highly questionable value. Another issue concerns projects with excessively broad goals—such as saving human life. The benefit-cost methodology could be used as well to compare the projects' costs when designed to achieve a certain goal, e.g. treating a certain volume of wastewater according to a set standard. The most cost-effective project will be the one with the lowest cost among competing alternatives in terms of present value per unit of physical benefits. This approach is acceptable where target levels have already been selected from the political perspective, and the issue is the cost of reaching the target.

In these circumstances, the policy framework used to reach the environmental goal is an important determinant of cost. For example, if the policy framework requires on-site technical control at each facility, benefit-cost analysis will be the best method. However, if the policy framework is flexible and allows enterprises substantial freedom in the choice of a strategy, then benefit-cost analysis may not offer the best alternative.

Both approaches—environmental valuation and benefit-cost analyses—are widely used in decision-making, and corresponding approaches are presented in the examples below.

In transition economies, the simple formula to calculate net benefits of a given project may be applied in different ways. The issue is the difference between the forward-looking and backward-looking approaches, discussed in the next section.

CHAPTER 4

Conflict of forward-looking and backward-looking approaches in transition countries

The major problem for every transition economy is that the positive results of environmental protection activity are underestimated when compared to economic growth. Developed countries have the same problem, although to a lesser extent. Specific features of transition economies, such as high discount rates and budgetary limitations, partially account for this. Lack of experience with market forces and with using modern valuation techniques reinforce this.

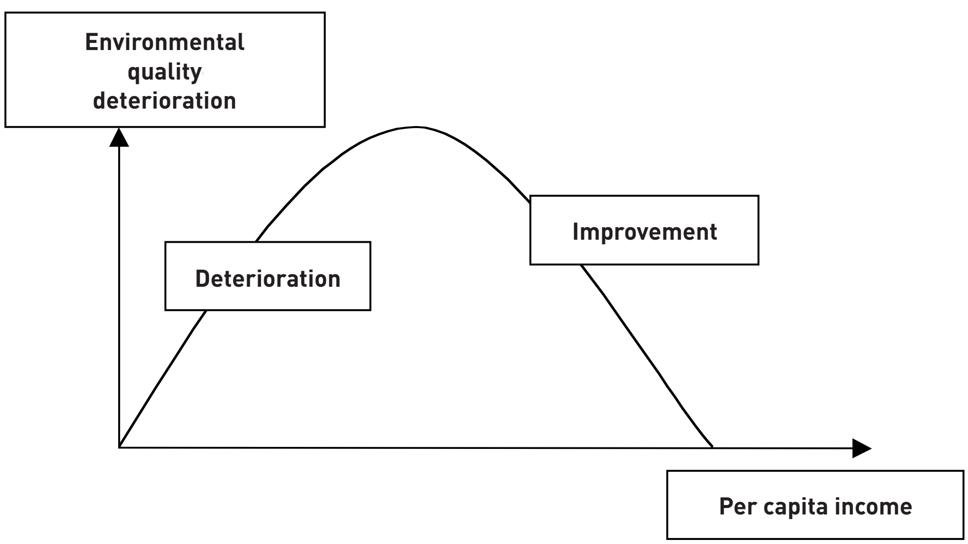
Certainly, the relationship between environmental quality and a country's stage of development is frequently analyzed in economic literature. Initially, environmental quality deteriorates as a result of resource-intensive economic development. Once society reaches a certain level of development, environmental quality starts to improve as demand for quality of life increases. If one presents environmental degradation as a function of per capita income, one usually finds an inverted U-shaped curve which describes the relationship between environmental quality (e.g. concentration of pollutants in the air) and per capita income measured over time. The curve is presented in Figure 2.

There is some evidence (see Yandle, et al. 2002) that it is the same shape as the original Kuznets Curve, which linked per capita income to income inequality (Kuznets, 1955).

The Environmental Kuznets Curve became a standard after the paper written by Gene Grossman and Alan Krueger in 1991 "Environmental Impact of a North American Free Trade Agreement" (see Grossman and Kruger, 1991).

Other authors have tried to quantify these relationships and to estimate the elasticity between income and environmental quality (Khanna, 2002). In some situations, the correlation is high.

FIGURE 2
Environmental Kuznets Curve



Source: Adapted from Yandle, et al. 2002

Some policy makers have used the link between income levels and enviromental quality as a justification for not paying attention to environmental problems when income is low. They argue that one should wait until per capita income reaches a certain point and only then focus on pollution control policies. An analysis of environmental protection policies in certain developed countries demonstrates that such conclusions can be wrong. The main reason lies in the difference between forward-looking and backward-looking valuations of environmental benefits and damages. The difference is due to the following:

- Clean up costs may appear to be much higher than abatement costs;
- Willingness to pay for environmental services may grow faster than per capita income;
- Market forces (e.g. signals) may be weak or absent in transition economies.

Because of this, decision makers may simply miss the turning point of the Environmental Kuznets Curve when investment in environmental protection becomes relevant[2]. Individuals may similarly be affected by a lack of proper information or absence of necessary institutions[3].

In examining the differences in forward-looking versus backward-looking approaches, it is useful to consider environmental valuation results from the US, as well as estimates of environmental benefits and costs for EU accession countries. One can conclude that environmental protection costs could be reduced dramatically if policy makers were to select the right instrument. The US SO_2 trading program is an excellent example of an ambitious environmental target that was met with a least-cost approach.

In transition economies, regulatory authorities and enterprises face the same problem in dealing with the forward-looking and backward-looking approaches. Many decry the negative environmental changes they have witnessed in their lifetime, especially in countries that are still industrializing. Some of the most urgent and difficult choices in these countries regarding conservation have to be made looking forward, rather than backward. Environmental goods and services seem not to be important when confronted with urgent economic problems. One has to look at this from both a forward- and a backward-looking vantage point.

Let us take an environmental asset with a high potential value T years from now but under pressure. The asset may become scarce exactly when future demand increases. Even in the US, one can cite examples such as free flowing rivers suitable for recreation. Protection for the next T years requires an outlay today, when it is difficult to generate much interest in the project. The following variables apply:

V_0 = Value of asset in period 0
V_{T0} = Expected value of asset in period T as judged in period 0
V_{TT} = Value of asset in period T as judged in period T
r_0 = Discount rate in period 0
r_T = Expected discount rate in period T
C_0 = Cost of conservation to be incurred in period 0.

Forward Looking (FL)
A traditional benefit-cost analysis determines the value of the project as FL, where

$$FL = \frac{V_{T0}}{(1 + r_0)^T} - C_0 \tag{1}$$

If (1) is positive, conservation is justified; otherwise not.

Backward Looking (BL)
A backward looking assessment estimates the value BL, where

$$BL = \frac{V_{TT}}{(1 + r_T)^T} - C_0 \tag{2}$$

As in the forward-looking case, a positive value would indicate that a decision to preserve would be the correct one.

It is frequently observed that (1) is negative whereas (2) is positive. The reasons for this may be the following:

(i) V_{T0} underestimates V_{TT}
(ii) R declines over time, as the economy develops
(iii) A shortage of fiscal resources in period zero results in the application of a shadow price applied to C_0 that is greater than one.
(iv) The FL valuation is made when environmental assets are available but many are judged to be expendable; by the time BL valuation is made, fewer are left, raising their value considerably.

Each of these shall be considered in turn:

V_{T0} underestimates V_{TT}.
Economists tend to underestimate the future value of environmental assets, and fail to appreciate the full extent of the increase in values that comes with economic development. Examples comparing environmental asset unit values in transition and developed economies show differences are greater in the latter. This implies an elasticity of WTP for the asset with respect to real income significantly greater than one. More information on this is needed but the little that is available, suggests that assets that are to be set aside for conservation will experience a sharp rise in future value coinciding with economic development.

R declines over time as the economy develops.
Real rates are higher in less developed countries than in developed countries. With changing real rates over time, the correct version of the FL assessment expression is:

$$\frac{V_{T0}}{\prod_{i=0}^{i=T}(1 + r_i)^i} - C_0 \tag{3}$$

An analogous expression holds in the BL case. If real rates fall uniformly over time, it is obvious that (3) will be above (1), implying that the true value is greater than the estimated constant real rate value.

A shortage of fiscal resources in period zero results in the application of a shadow price applied to C_0 that is greater than one.
When there is a shortage of financial resources in the public sector, and in the absence of access to private sector or outside resources, the funds C_0 have an economic value greater than their financial value. The ratio of the economic to the financial value is defined as the shadow price. Although shadow prices are rarely explicitly estimated and applied as such, the decision-making process mimics the presence of these parameters.

The FL valuation is made when environmental assets are available but many are judged as expendable. By the time BL valuation is made, only a few are left, raising their value considerably.
This point is self-explanatory. Each conservation decision undertaken in a micro context results in a decision for each site separately, usually against conservation of that asset. The overall effect, however, is to reduce the number of such sites, thereby raising the value of each remaining site. The BL valuation is made after the fact, and thus shows much higher values in period T.

The problem lies with the fallacy of using a partial equilibrium approach when a more general equilibrium approach is warranted. As demonstrated in the next section, the implications can be quite startling.

Numerical examples

V_{T0} underestimates V_{TT}.
Table 1 gives some estimates of the value of recreational sites in Russia and OECD countries in $/Ha/Person. The average for high income countries is over 50 times higher than for Russia, whereas the real per capita income difference between the two is currently 11.75 (1999 Data, World Bank, 2001). This would suggest an elasticity of WTP with respect to per capita income of approximately 4. We should interpret this with some care, however, as it does not necessarily imply that with a sustainable increase in income in any one country, an elasticity of 4 will apply. But in an environment of economic transition, a shift from a situation in which people are experiencing serious economic difficulties to one in which these difficulties have not only been resolved but real incomes have also increased sharply, a value as high as 4 may indeed be appropriate.

The typical elasticity of WTP with respect to per capita income is in the range of 1-2. In Table 2, we see the impact of underestimating this elasticity for different parameters of a conservation problem.

TABLE 1
WTP for recreational sites in high income countries and transition economies

Study	Area	Year	Units	Value/person
Feather et al	USA	1999	Ha/Year	2.5
Fomenko et al, 1997	Yaroslavl, Russua	1997	Ha/Person	0.25
Fomenko et al, 2000	Kurshkaya Kosa, Russia	1999	Ha/Person	1.68
Bobulev S et al, 1998	Naluchevo, Russia	1998	Ha/Person	0.04

TABLE 2

Regrets as a function of underestimating the true elasticity of WTP

		FL	BL
Cost of protection as % of current value of asset	100	–0.57	0.70
	50	–0.07	1.20
	40	0.03	1.30
Discount rate used	6%	1.22	7.88
	8%	0.27	4.07
	10%	–0.27	1.92
	12%	–0.57	0.70
Ratio of true to estimated elasticity of WTP	4.0	–0.57	0.70
	3.0	-0.57	0.28
	2.5	-0.57	0.06
	2.0	-0.57	-0.15

Note: Baseline calculations assume that income will rise by a factor of 11.75 in the intervening period of 30 years.

a) The probability of no-regrets increases sharply when the applied discount rate declines.

b) When the elasticity is underestimated by a factor of 2.5, a conservation project that is strongly unattractive becomes positive (i.e. the BL value becomes positive). We say that the FL estimate of the project is strongly unattractive because the NPV (net present value) of the project is negative and equal to 57% of the costs, and the internal rate of return on the baseline case is around 9%, or 3% below the required rate of return of 12%.

c) A reduction in the costs of protection relative to the present value of the asset reduces the regret significantly and offers the possibility that regrets will be removed completely.

R declines over time as the economy develops.

There is a systematic decline in the real discount rate commensurate with economic development. Typically, the 'low risk' discount rate in transition economies or in developing countries is around 12–15%, whereas in developed economies, it is in the order of 4–6%. In Table 3, we show the impact of applying a varying discount rate over a 30-year period, on the assumption that the discount rate in the initial year is 12%, in the final year 6%, and the discount rate declines linearly over the 30 year period. The calculation is carried out using equation (4). The shift from the standard FL valuation to the "True Value" results in many decisions shifting from negative to positive. Even in the case where the FL has an internal rate of return as much as 3% below the required rate of return, the true value NPV calculation becomes positive. The impact is greater when the assumed elasticity of WTP for the asset is lower than when it is higher.

A shortage of fiscal resources in period zero results in the application of a shadow price applied to C_0 that is greater than one.

In countries with a transition economy as well as in developing countries, the process of allocating public funds is under much political pressure. Environmental protection that has a long-term objective is not likely to fit short-term political goals. As a result,

TABLE 3
Impacts of using varying discount rates on regrets

	FL	BL	True value
Elasticity of WTP w.r.t. Income = 2			
Costs as % of initial asset value			
100	−0.57	+1.22	+0.02
50	−0.07	+1.72	+0.52
40	+0.03	+1.82	+0.62
Elasticity of WTP w.r.t. Income = 2			
Costs as % of initial asset value			
100	−0.15	+3.44	+1.03
50	+0.35	+3.94	+0.53
40	+0.88	+4.04	+2.64

Notes:
1. Baseline calculations assume that income will rise by a factor of 11.75 in the intervening period of 30 years.
2. Regrets measure lies between 0 and 90, as the angle θ in equation (3) and in Figure 1.
3. Real discount rate falls linearly from 12% to 6% over 30 years.

public funds available for environmental protection are shrinking. For example, according to the OECD Environmental Performance Review of the Russian Federation (OECD 1998, p. 97), the size of extra-budgetary environmental funds as a percentage of GDP dropped from 0.24% in 1992 to 0.056% in 1997. (For more information on environmental funds in Russia see OECD 1998). Investment in abatement facilities dropped more than three times during the last decade. Those expenses in 1999 were just 30% of the level in 1990 (calculated based on the Annual Russian Statistic Book 2000 p. 48—Environmental Expenditures; and p. 249—GDP Deflator). According to Russian statistics, investment in environmental protection was one half of one percent of GDP in 1990 and one quarter of one percent of GDP in 1999. However, in 2000, environmental expenditures increased by one third in real terms (State Committee Statistical Bulletin #6 (80), 2001, p. 96) while GDP growth did not exceed 7%.

One would anticipate that the share of public funds available for environmental protection would grow faster than GDP, and that the shadow price of that funding would go down. In countries with transition economies and in developing economies, the interest rate is probably higher than the discount rate. Nevertheless, borrowing may still be an effective solution to prevent irreversible damage or catastrophic consequences. The Russian example demonstrates that WTP went down faster than GDP, but also recovered faster. This observation, supported by theoretical analysis, demonstrates that the common strategy in developing and transition countries to solve economic problems first, and only then to address environmental issues, is fundamentally wrong.

Developing countries and countries with transition economies may initially have a rich environment which helps create the illusion that there is no need for conservation and environmental protection activity.

The FL valuation is made when environmental assets are available but many are judged expendable, and by the time BL valuation is made, only a few assets are left, raising their value considerably.

This approach can be used not only to generate resources for conservation but also to support prevention of air or water pollution. One of the most critical problems for

Russia related to air pollution is the potential change in the fuel mix to coal. As a result, the air quality may significantly decrease (see Dudek, Golub, Strukova 2000 and 2002). Substitution of natural gas by coal would lead to an increase in SO_2 and PM_{10}. Mortality and morbidity risk would go up. In Dudek, Golub, Strukova, 2002, this increase is estimated at 35,000 additional cases of mortality risk from PM_{10}.

During the last decade, the share of natural gas in Russian energy has increased continuously. Starting in 2000, the situation radically changed, and the share of coal has increased. Substitution of natural gas by coal is the result of market reforms and the absence of institutions to provide effective environmental protection. Gazprom, the state monopoly, can export natural gas or sell it on the domestic market. Currently, domestic consumption accounts for 65% of the natural gas Russia produces, with 35% going for exports. If Russia increases exports and reduces domestic consumption, natural gas in the domestic market will become more and more expensive. Export opportunities for the coal industry are limited and the coal industry is closely tied to the domestic market. As a result, prices for coal are relatively lower (per BTU) than prices for gas, and the energy sector prefers coal as a less expensive fuel.

This changing trend in fuel mix may cause environmental problems for Russia's cities, especially in the European part of the country. Consider, as an example, a pre-feasibility study of the modernization of the energy system in Novgorod. In 2002, it was proposed to replace a substantial part of natural gas used by power plants with coal and to use the savings (about $5 million per year) to recover investment costs. Independent analysis of the environmental consequences of such a proposal (Avaliani et al, 2002), demonstrated a dramatic increase in mortality risk (30 cases per year in a city with a population of 230,000). We can infer $5 million per year as the cost to prevent mortality risk. Average per capita income in Novgorod is $400. The annual spending for electricity and heat is an estimated $25 per year per person. To keep the existing fuel mix, each person would need to double that spending. The city administration may consider doubling the tariffs or using municipal bonds and raising the tariff gradually to follow the WTP for "clean" energy. If WTP annually increases by 15%, then, according to our calculation, a municipal bond program would be in operation for 10 years.

Based on the examples presented above, one can formulate general rules as to when municipal bonds can help resolve the conflict between forward- and backward-looking approaches. The debt associated with the application of municipal bonds (D_t) should go down to zero sometime before the end of period T.

A general rule for borrowing is the following: it is reasonable to borrow if future damage from limited conservation is greater than the future expenses of repayment. However, the assessment should be done in each individual instance.

Thus, in most developing nations and in countries with transition economies, the conflict between backward-looking and forward-looking assessments of environmental assets is possible. In this situation, the rational conservation option will not be implemented. There are several reasons for this. Economists tend to underestimate the future value of environmental assets. Real interest rates are higher in less developed countries than in developed countries, and have a tendency to fall with development. In addition, an initial shortage of fiscal resources results in the overestimation of conservation costs. Each conservation decision undertaken in a micro context results in a separate decision for each site, one usually against conservation. Numerical examples and theoretical models support this proposition.

CHAPTER 5

Examples from developed countries[4]

1. Examples of benefit-cost analysis

Environmental valuation involves placing monetary value on environmental goods and services, as well as on changes in environmental quality, resulting from certain action being taken or not taken. Unlike other goods and services, environmental considerations are not subject to market forces. Their value cannot be directly revealed through market pricing. However, economists have developed ways to estimate these values on the basis of physical or behavioral linkages between the level of consumption (or production) of environmental goods, and the observed effect on health, productivity, or natural resource assets. One recent example of such a study is the report "The Benefits and Costs of the Clean Air Act" (USEPA, October 1997; November 1999). One study presents broad retrospective analysis (1970–1990), while the other presents future analysis (1990–2010) of the costs and benefits of the Clean Air Act (CAA) adopted in the USA in 1970 and amended in 1977 and 1990. Both reports are built on the same methodology of valuing direct health benefits resulting from the adoption of the CAA and comparing them with its direct costs. The analysis is forward-looking, but as it is based on the methodology of environmental valuation in a developed market economy, it avoids the conflict of forward-looking and backward-looking approaches.

The CAA established a framework for the attainment and maintenance of clean and healthy air quality levels. The main provisions of the Act were as follows:

- The EPA established national ambient air quality standards for the major air pollutants, and the states were required to develop implementation plans.
- The CAA included deadlines and enforcement mechanisms for emission limitations for both the state and federal government;
- The best available technology was identified to attain these standards;
- Some additional regulation was developed for hazardous pollutants and automobile exhaust.

The amendments of 1977 and 1990, which strengthened the earlier version of the CAA, were duly taken into account.

Both assessments were based on the analysis and comparison of two scenarios. The first was a "no-control" scenario, which was based on the assumption that the CAA was not adopted and no additional regulation of air pollution was enacted on a local, state and federal level since 1970 for retrospective analysis, and since 1990 for future analysis. The second was the "control" scenario, which was based on actual historic data from 1970-1990 for the retrospective assessment, and emission forecasts for the future analysis. Comparison of incremental benefits and costs between the two scenarios made it possible to assess the benefits and costs of the CAA.

Several important assumptions were made. First, it was assumed that economic distribution of population and economic activity would be the same under both scenarios. Population movement out of polluted areas was also not taken into account. In addition, the analysis assumes no differences in the pattern of economic activity.

TABLE 4

Comparison of major steps in retrospective and future analyses

Cost side	
RETROPECTIVE ANALYSIS • Direct cost estimation • Macroeconomic modeling	FUTURE ANALYSIS • Direct cost estimation
Benefit side	
• Emission modeling • Air quality modeling • Health and environmental effects estimation • Economic valuation • Results aggregation and uncertainty characterization	

The analytical sequence of the two assessments included the following major steps, presented in Table 4.

The difference on the cost side was that retrospective analysis recognized that the CAA had a strong impact on the US economy. Well-developed historical data for the flow of goods and services was available for 1970–1990. This data provided a solid basis for modeling producer and consumer behavior, and estimating net social costs.

For the future analysis, direct compliance costs of the CAA were considered. In other words, only direct compliance expenditures by corporations were taken into account. This method does not account for corporate or market responses, such as adjustment of production levels, or changes in product prices. Changes in production or consumer welfare require either partial or general equilibrium modeling in order to be properly measured. Modeling demand and supply functions is required in the specific economic sector for which the retrospective analysis was performed. For future analysis, uncertainty is high due to difficulty in projecting future economic and technological changes. This direct cost approach provides a good first approximation of the CAA's impact on different sectors of the US economy.

The modeling of direct costs was closely integrated with historical reporting data on compliance expenditures that was provided by sources for the retrospective analysis . These were then used as a source for macroeconomic modeling to estimate "control" and "no-control" economic conditions, and served as a basis for emissions models that were run to estimate emissions levels under both scenarios. For future analysis, direct compliance costs were based on emissions projections under two scenarios. Special models were used in this instance: The Emission Reduction and Cost Analysis Model, and The Integrated Planning Model.

Direct CAA compliance costs include direct costs carried by businesses, consumers and governmental entities. These triggered other expenditures such as governmental regulatory and monitoring costs, as well as expenditures for R&D by both government and industry.

Adjusted for inflation, annual CAA compliance expenditures in the retrospective study were relatively stable (in 1990 dollars), averaging nearly $25 billion during the 1970's and close to $20 billion during most of the 1980's. For the purpose of cost-benefit analysis, costs were annualized to spread the cost of capital equipment over its useful life. A 5% inflation-adjusted discount rate was used to account for the time value of money. In other words, annualized costs for each year are equal to O&M expenditures (including R&D) plus amortized capital costs (i.e. depreciation plus interest costs associated with the existing capital stock). Total annualized costs ranged from $11 billion in 1973 to $26.1 billion in 1990. Discounting the annualized stream to 1990 resulted in a total cost of $523 billion (1990 dollars).

The same procedure was used to calculate direct compliance costs for the future analysis. Total annual compliance costs for the 1990 amendments in the year 2000 were estimated at $19.4 billion (1990 dollars), and the estimate increased to $26.8 billion in the year 2010.

Emission estimates for the future analysis was based on the construction of an emissions inventory for the base year (1990), and projection of emissions for both scenarios based on sector-specific emissions models.

Emissions estimates form the first step in estimating benefits. The next step is to translate them into estimates of air quality conditions in terms of concentrations of criteria pollutants under each scenario. The future analysis used 1990 base-year monitoring data to project ambient pollution levels at monitors throughout 48 states. Concentration data on monitors was then extrapolated to non-monitored areas in order to generate a more comprehensive data set. Special air dispersion and transport model tools were used to get estimates for both scenarios up to the year 2010. For the future analysis, atmospheric chemistry was taken into account to model the pollutant transformation process. This is especially important for secondary pollution that is not emitted directly, but instead forms in the atmosphere through a series of chemical interactions. The best example is ozone that appears as a result of secondary processes from certain classes of volatile organic compounds (VOCs), and NOx. In addition, the greater part of fine particulate matter ($PM_{2.5}$) is not emitted directly, but is a result of atmospheric transformations of gaseous sulfur dioxide and nitrogen oxides to particulate sulfates and nitrates. In all of these processes, meteorology is an important factor and is taken into account in the specific models, thereby adding uncertainty to all results.

In the next step, the two scenario air quality profiles serve as an input to a modeling system that translates air quality into physical outcomes (mortality, morbidity, crop yield losses) through the use of concentration-response functions. This was done for each pollutant and by geographic area. This is a critical step in the analysis since the benefits attributable to CAA were calculated in major part as the avoided incidence of adverse health effects. Such benefits are measured using concentration-response functions specific to each health effect. The list of physical outcomes for each pollutant considered is presented in Table 5.

Control-response functions are equations that relate the change in the number of individuals exhibiting a "response" (an adverse health effect, e.g. a respiratory disease) to a change in pollutant concentration experienced by that population, due to one of the scenarios being realized. In this process, three values are used: the grid-cell specific change in pollutant concentration, population, and an estimate of the change in the number of individuals that suffer an adverse health effect per unit change in air quality. The last factor, as well as the specific form of the concentration-response equation, is derived from published scientific literature for each pollutant and the relevant health effect relationship. In addition, researchers relied exclusively on long-term studies as opposed to short-term studies in the analysis, as they are likely to result in a more complete assessment of the effect of air pollution on mortality risk.

TABLE 5
Human health effects of criteria pollutants

Pollutant	Quantified health effects	Non-quantified health effects	Other possible effects
Ozone	• Mortality • Respiratory symptoms • Minor restricted activity days • Respiratory restricted activity days • Hospital admissions • Asthma attacks • Changes in pulmonary function • Chronic sinusitis and hay fever	• Increased airway responsiveness to stimuli • Centroacinar fibrosis • Inflammation of the lungs	• Immunologic changes • Chronic respiratory diseases • Extrapulmonary effects (e.g., changes in the structure, function of the organs)
Particulate matter/ TP/sulfates	• Mortality • Bronchitis—chronic and acute • Hospital admissions • Lower respiratory illness • Upper respiratory illness • Chest illness • Respiratory symptoms • Minor restricted activity days • All restricted activity days • Days of work loss • Moderate or worse asthma status (asthmatics)	• Changes in pulmonary function • Inflammation of the lungs	• Chronic respiratory diseases other than chronic bronchitis
Carbon monoxide	• Hospital admissions—congestive heart failure • Decreased time to onset of angina	• Behavioral effects • Other hospital admissions	• Other cardiovascular effects • Developmental effects
Nitrogen oxides	• Respiratory illness	• Increased airway responsiveness	• Decreased pulmonary function • Inflammation of the lungs • Immunological changes
Sulfur dioxide	• In exercising asthmatics: changes in pulmonary function, respiratory symptoms, combines responses of respiratory symptoms and pulmonary function changes		• Respiratory symptoms in non-asthmatics • Hospital admissions
Lead	• Mortality • Hypertension • Non-fatal coronary heart disease • Non-fatal strokes • IQ loss effect on lifetime earnings • IQ loss effects on special education needs	• Health effects for individuals in age ranges other than those studied • Neurobehavioral function • Other cardiovascular diseases • Reproductive effects • Fetal effects from maternal exposure • Delinquent and anti-social behavior in children	

Source: USEPA, *Benefits and Costs of the Clean Air Act, 1970 to 1990*

Thus, based on Tables 5 and 6, adverse health effects for both scenarios were quantified for 48 states and the District of Columbia. It is important to note that a high degree of uncertainty is present in each step of the analysis. The researchers tried to present all uncertainties accurately. However, the choices selected to overcome the problem of limited data, the inadequacy of currently available scientific literature, and other constraints do not bias the overall results of this analysis.

The next step of the assessment is to aggregate benefits across endpoints. The benefits must be monetized so researchers can add up the realized benefits of the CAA and its amendments, and compare them with associated costs. Two categories of benefits were analyzed: human health effects including mortality and morbidity endpoints, and welfare effects including agricultural and ecological benefits, visibility, and workers' productivity. Valuation estimates were obtained from related economic literature. In essence, the dollar amount required to compensate a person for exposure to an adverse effect is roughly equivalent to the dollar amount a person is willing to pay to avoid this effect. This is the so-called WTP to avoid an adverse effect. For environmental goods and services that are not subject to market forces, this value can be inferred either from observed consumer behavior or through specially designed surveys, whereby a market of environmental goods is constructed and respondents are asked to state their preferences in this

TABLE 6
Summary of table dose-response functions: estimated increment in annual health effects associated with unit change in pollutants

	Pollutants (UNITS)				
Outcome	**PM_{10} (10 μg/m^3)**	**SO_2 (10 μg/m^3)**	**Ozone (pphm)**	**Lead (1.0 μg/m^3)**	**NO_2 (pphm)**
Premature mortality (% change)	0.96	0.48			
Premature mortality/100,000	6.72				
RHA/100,000	12		7.7		
ERV/100,000	235.4				
RAD/person	0.575				
LRI/child	0.016				
Asthma symptoms/asthmatic	0.326		0.68		
Respiratory symptoms/person	1.83		0.55		
Chronic bronchitis/100,000	61.2				
MRAD/person			0.34		
Respiratory symptoms/1,000 children		0.18			
Respiratory symptoms/adults		0.1			0.1
Eye Irritations/person			0.266		
Hypertension/100,000 adult males				7,260	
Coronary disease/100,000 adult males				34	
Premature mortality/100,000 adult males				35	
IQ decrement (100,000) children				97,500	

Notes:
RHA = Respiratory hospital admissions; ERV = Emergency room visits
RAD = Restricted activity days; LRI = lower respiratory illness
MRAD = Minor restricted activity days
PPHM = Parts per hundred million

Source: Ostro, Bart. 1994. *Estimating the Health Effects of Air Pollutants. A Method with an Application to Jakarta.* Policy Research Working Paper No. 1301.

market. Whenever possible, mean estimates of WTP are used. When not available, researchers use the cost of treating or mitigating the effect as an alternative estimate (the so-called "cost of illness" approach). Usually, this approach underestimates the true value of avoiding a health effect, taking into account only direct expenditures to avoid adverse effects.

In the example of the unit values applied to various health endpoints in the *Benefits and Costs of the Clean Air Act, 1970 to 1990*, the values applied to Coronary Heart Disease and Stroke represent "cost of illness" estimates, not WTP. See Table 7.

The table changed slightly for the future analysis, but the basic valuation of one incident of mortality stayed the same.

TABLE 7

Health and welfare effects unit valuation (1990 dollars)

Endpoint	Pollutant	Valuation (mean est.)
Mortality	PM & Pb	$4,800,000 per case
Chronic bronchitis	PM	$230,000 per case
IQ changes		
Lost IQ points	Pb	$3,000 per IQ point
IQ<70	Pb	$42,000 per case
Hypertension	Pb	$680 per case
Strokes*	Pb	$200,000 per case—males $150,000 per case—females
Coronary heart disease	Pb	$52,000 per case
Hospital admissions		
Ischemic heart disease	PM	$10,300 per case
Congestive heart failure	PM	8,300 per case
COPD	PM & O_3	8,100 per case
Pneumonia	PM & O_3	$7,900 per case
All respiratory	PM & O_3	$6,100 per case
Respiratory illness and symptoms		
Acute bronchitis	PM	$45 per case
Acute asthma	PM & O_3	$32 per case
Acute respiratory symptoms	PM, O_3,, NO_2, SO_2	$18 per case
Upper respiratory symptoms	PM	$19 per case
Lower respiratory symptoms	PM	$12 per case
Shortness of breath	PM	$5.30 per case
Work loss days	PM	$83 per day
Mild restricted activity days	PM & O_3	$38 per day
Welfare benefits		
Visibility	DeciView	$14 per unit change in DeciView
Household soiling	PM	$2.50 per household per Pm_w change
Decreased worker productivity	O_3	$1**
Agriculture (net surplus)	O_3	Estimated Change in Economic Surplus

*Strokes are comprised of atherothrombotic brain infractions and cerebrovascular accidents; both are estimated to have the same monetary value.

**Decreased productivity valued as change in daily wages: $1 per worker per 10% decrease in O_2.

Source: USEPA. *Benefits and Costs of the Clean Air Act, 1970 to 1990*

TABLE 8
Total monetized benefits in 48 states (billions of 1990 dollars).

	PRESENT VALUE		
	5th percentile	**Mean**	**95th percentile**
Total	$5,600	$22,200	$49,400

For the retrospective analysis, the present value of monetized benefits from 1970 to 1990 was estimated at about $22,000 billion.

Other welfare effects were considered as well. For example, in retrospective analysis, agricultural effects were quantified. Ozone is the primary pollutant affecting agricultural crops (barley, corn, soybeans, peanuts, cotton, wheat and sorghum). Nationwide crop damage was estimated under the "control" and "no-control" scenarios. Net change in economic surplus (in 1990 dollars) was estimated annually and as a cumulative present value (discounted at 5%) over the period 1976–1990. The present value in 1990 dollars was $23 billion. Other welfare effects were similarly estimated. The combined present value in 1990 dollars did not exceed $200 billion. Thus, welfare effects accounted for about 1% of the total benefits in the retrospective study.

For the retrospective study, total monetized benefits in the 48 states are presented in Table 8 (present value in billions of 1990 dollars, and discounted to 1990 at 5%).

The resulting estimates monetized annual benefits in the period resulting from annual changes in air quality. However, the uncertainties associated with the yearly estimates were not quantified. Thus, the researchers relied on the ratios of the 5th percentile to the mean and the 95th percentile to the mean in the target years to capture these uncertainties. Such estimates were conducted for both studies, underlying the high level of uncertainty in the results obtained. The Monte Carlo technique was then used to aggregate monetized benefits across end-points. In the retrospective study, the pattern of benefits during the 1970–1990 period is related to the difference in emissions between the "control and "no-control scenarios", and is magnified by population growth during this period. Quantified annual benefits increased steadily during the study period. The mean estimate of quantified annual benefits increased from $355 billion in 1975 (in 1990 dollars) to $1,248 billion in 1990.

The same estimates were used for the future study, although in this study, more welfare effects were quantified. Benefits from reductions of freshwater acidification and loss of commercial timber were included. The study included analysis of the benefits of stratospheric ozone protection provisions. The last analysis was

TABLE 9
Present value of monetized benefits for 48 states (billions of 1990 dollars, discounted in 1990 at 5%)[5]

	PRESENT VALUE		
	5th percentile	**Mean**	**95th percentile**
Amentments 1–5	$5,600	$22,200	$49,400
Amendments to limit chemicals depleting stratospheric ozone	$5,600	$22,200	$49,400

TABLE 10
Summary comparison of benefits and costs (in billions of 1990 dollars)

	Retrospective analysis	Future analysis
Monetized direct costs	523	210
Monetized direct benefits		
5th percentile	5,600	260
Mean	22,200	1,200
95th percentile	49,400	2500
Net benefits	21,700	1,000
Benefit cost ratio		
5th percentile	11/1	1/1
Mean	42/1	6/1
95th percentile	94/1	12/1

different from the criteria pollutant analysis. The long-term nature of the program made it impossible to generate an estimated annual benefit that could be reliably linked to emission reductions as well as to the corresponding costs in a single year. The researchers then generated an annualized equivalent of the cumulative net present value of benefits and costs, without taking into account specific results for any given year.

On the other hand, it was possible to generate annualized estimates for health and welfare benefits. The criteria pollutant health benefits in 2010 (in billions of 1990 dollars) were estimated at about $105 billion. Welfare benefits for 2010 were about $5 billion (or about 5% of all benefits). The same procedure of computerized statistical aggregation in the retrospective analysis was used to generate the present value of the estimated total monetized benefits. Results are presented in Table 9.

The last step of the analysis is to compare monetized benefits and costs. This was done through calculation of net benefits and a benefit-cost ratio. Results for both studies are presented in Table 10.

Thus, one can conclude that under these assumptions and with uncertainty specified, the benefits of the CAA and its amendments exceed its costs. Furthermore, the results of the uncertainty analysis imply that it is extremely unlikely that the monetized benefits could be less than the costs.

2. Example of cost-effectiveness analysis and its influence on decision-making

However, even with benefits exceeding costs almost six-fold for the mean case, the costs of the program still appear to be high for business and government. Another way to present the results is to use cost-effectiveness analysis. Cost-effectiveness involves estimates of costs per unit of benefit (e.g. lives saved). This type of analysis is very useful in comparing programs with similar goals. As an illustration, one can recalculate that, due to Amendments 1–5 (from 1990) to the CAA (for the year 2010), total annual direct costs of implementation are approximately $27 billion. Then, in the year 2010, we avoid 23,000 cases of premature mortality, as well as gain estimated non-mortality benefits of about $20 billion. One can generate the net cost per life saved by subtracting the total non-mortality benefits (to account for them properly) from the cost, and then dividing that by the number of lives saved. Thus, a net cost per life saved was approximately $300,000 ($27 billion minus $20 billion divided by 23,000). Although many additional uncertainties exist, the net cost per life year saved for 2010 for the mean case is $23,000 ($7 billion divided by 310,000 life years saved).

This way of presenting results dramatizes the expensive nature of the program. The costs to achieve this reduction in morbidity and mortality seem high. For transition economies, additional costs could be simply too high. The US experienced the same difficulty when the national plan to reduce SO_2 emission was put forward prior to the amendments of 1990. It included provisions for emissions trading to reduce total costs and achieve the target SO_2 reduction. Since political compromise based on benefit-cost analysis was reached in the form of an emission cap (to reduce SO_2 emissions by almost 50%), the major analysis was shifted to cost-effectiveness, i.e. how to reach the target at the lowest possible cost.

Under the program, each SO_2 source got a certain amount of pollution permits that could be freely exchanged in the US. Indeed, any individual could buy such permits. However, they had to cover real emissions. Actual allowable emission levels for each source gradually decreased. Unused pollution permits could be saved for future use. Thus, a new type of security to reduce costs was created in order to achieve the SO_2 reduction target.

Investors soon realized that this new market not only helped reduce the costs of reaching the SO_2 reduction target, but could also be a source of profit for cleaner firms. This led to the development of a stock exchange to trade such permits, facilitated by intermediaries such as brokers.

In this manner, a target can be reached at the least possible cost. All firms have set annual emission levels which are constantly monitored. At the same time, firms have flexibility in how to reach the target. An Administration Authority keeps track of program implementation at all times and technological innovation is encouraged. The focus is on the final emission reduction, not on the means of getting there. The penalty for SO_2 emissions not covered by pollution permits was $2,000 per 1 ton of SO_2 plus a requirement to reduce additional emissions in the following year.

Since 2000, all power stations whose capacity exceeds 25 MWt have received pollution permits. In 2000, pollution sources participating in the program had annual SO_2 emission budgets of 9.2 million tons. In 2010, it will be reduced to 8.95 million tons. When the program started in 1990, the price of pollution permits per ton of SO_2 was assumed to be in the $300-$1,000 range. Actual prices varied from $69 to $212 per ton of SO_2. The lower price was possibly due to the many technologies that were developed to achieve SO_2 emission reductions. Moreover, by 1999, the actual reduction of SO_2 emissions by power stations exceeded target levels by 20%. A substantial supply of unused pollution permits currently exists.

The main lesson was that lower implementation costs lead to more rigid emission targets. Permanently resetting environmental targets and therefore associated costs leads to an optimal situation from the perspective of protecting the environment.

As a result, new and more rigid SO_2 emission targets are being introduced in the US. This demonstrates the importance of the Environmental Kuznets Curve.

Below, we propose to decision makers some practical recommendations on how to use economic valuation techniques to improve environmental regulations.

It is critical for those decision makers who have begun to implement economic analysis methods to shape environmental policy to understand the opportunities and limitations of each policy tool. A benefit-cost analysis can be useful in one stage of the policy making process, but inappropriate in another. The CAA sheds light on the proper use of benefit-cost and cost-effectiveness analyses.

A description and evaluation of the economic benefits of environmental policy is a useful way to start a discussion on new regulation. However, the lessons learned from the experiences of other countries with market economies suggest that it is only a first step.

For transition economies, with vast potential for no-regret and low-cost options to decrease pollution, an initial assessment of the benefits would definitely help set the policy direction. Environmental protection authorities and NGOs could offer several examples of investment projects with substantial economic returns as well as quantified environmental benefits. Not all benefits have to be calculated in order to make the case. The valuation of economic benefits or damages is a good tool for focusing attention on an environmental problem, and for contributing to a proper debate on the subject.

However, after a certain point, another approach is warranted. It has been demonstrated that emission targets are established as a result of political compromise, and not of economic calculation. Using benefit-cost analysis in isolation could affect the decision-making process. In a transition economy, one could underestimate avoided damage (the contradiction between forward- and backward-looking approaches) and overestimate cost (through a high discount rate). The example of the CAA suggests that benefit-cost analysis cannot be used by itself to justify emission targets.

Scientists first established acceptable levels and political leaders agreed on appropriate targets. The next step was a debate on implementation. Decision makers were looking for the most appropriate tool to reach the environmental goals at the lowest possible cost. The task for economists was to compare implementation costs. Using cost-effectiveness analysis, emission trading was chosen as the instrument which would let industry minimize costs.

The final step was to set the business strategy. The emission target was in place. A low initial allocation was established, business units received their permits, and the rules of the game were determined. Since then, business has been looking for a cost-effective solution. Emission reduction projects (unless they are an end-of-the-pipe technology) have multiple benefits . Emission trading creates the value for emission reduction, so business is able to aggregate those benefits together with others. Benefit-cost analysis reappears later at the micro level.

The US experience could be useful for decision makers in countries with transition economies. The following requirements are necessary in order to set goals for environmental regulation and for emission targets:

- Reach the environmental goal at the lowest cost;
- Build a compromise among the different interest groups;
- Create realistic and achievable goals;
- Build an effective enforcement mechanism.

Economic analysis can be a powerful and useful tool if its application meets the following criteria for choosing between a benefit-cost and a cost-effectiveness analysis:

* Benefit-cost analyses define the scope of the political discussion;
* Emission targets are always the result of political processes and compromise;
* Cost-effectiveness analyses help the selection of the regulatory strategy;

Benefit-cost analyses at the corporate level help enterprises choose the appropriate compliance strategy.

CHAPTER 6

Example of EU Accession countries

A number of studies have been conducted estimating the environmental benefits and costs of entering the EU for CEE countries. In Anil Markandya's "Expenditure Policies in the Context of EU Accession. Dealing with the Challenges of Accession: The Environmental Dimension" (prepared for the World Bank/IMF Conference on "EU Accession—Developing Fiscal Policy Frameworks for Sustainable Growth," September 2002), the author compared the benefits and costs of meeting environmental requirements for accession. Table 11 provides the initial estimate of investment costs. These costs are large, about $100 billion over a period of 20 odd years, amounting to between 23% and 134% of the country's present GDP (1997 estimates). The incremental annual operating costs are estimated in the order of $7–$10 billion per year. Dividing expenditures equally over 20 years yields annual investment of between 1.2% and 6.7% of current GDP. Taking the annual investment and operating costs together suggests that about 5% of GDP will have to be devoted to environment-related expenditures, or more than double the current level.

The author noted that the total cost is valuable only to give an idea of the overall size of the task. It is more important for these countries to prepare detailed 3–5 year plans to ensure compliance with the environmental agreements, and to do so in a way that both minimizes costs and ensures sustainable funding levels. Costs can be reduced substantially by following a least-cost investment strategy, especially with respect to energy-related investments. Economic instruments such as bubbles, permit trading, and the like can be very effective and reduce costs substantially. Moreover, they can lead to the setting of more stringent environmental targets and, correspondingly, to higher economic benefits.

TABLE 11

Estimated environmental costs of accession for the CEE (billions of Euros unless otherwise indicated)

	WATER			AIR	WASTE		TOTAL INVESTMENT			
	Water supply	**Waste water**	**Total**	**Total**	**Min.**	**Max.**	**Min.**	**Max.**	**Per capita (Euros)**	**As % of 1999 GDP**
Bulgaria	2.20	2.70	4.90	5.10	1.80	5.10	11.80	15.10	1668	134%
Czech Republic	2.20	1.10	3.30	6.40	8.00	3.80	17.70	13.50	1427	32%
Hungary	3.50	3.10	6.60	2.70	2.10	4.40	11.40	13.70	1306	32%
Poland	4.40	13.70	18.10	13.90	2.20	3.30	34.20	35.30	927	26%
Romania	3.80	6.30	10.10	9.10	1.00	2.70	20.20	21.90	943	72%
Slovak Republic	1.00	0.90	1.90	1.90	0.30	1.60	4.10	5.40	760	23%
Slovenia	n/a	n/a	n/a	0.69	1.15	1.15	1.84	1.84	n/a	n/a
Baltics										
Estonia	0.13	1.38	1.51							
Latvia	0.11	1.60	1.71	8.45	0.45	0.85	8.90	9.30	1148	48%
Lithuania	0.11	2.27	2.38							
Total	17.45	33.05	50.5	48.24	17	22.9	110.14	116.04		
As % of total	14	27	42	40		19		100		

Sources: EDC (1997),

Per Capita & GDP Percentage Numbers are for averages of Min. & Max. Investment.

Another consideration relates growth to the capacity of these countries to meet the environmental requirements for accession. With a higher rate of growth, the overall costs increase, as demand for some services such as energy and transport, increases. However, other services, such as water, do not increase proportionately, and so the total increase is less than the growth in national income. Hence, if countries pursue a successful growth strategy, they will be able to meet future costs, once the strategy has been realized.

The next step is to estimate benefits. These include health, resource and ecosystem protection. The EC directives covered air, water and solid waste. Results were measured as the present value of benefits, discounted at a real rate of 4%, and are given in Table 12. Estimates range quite widely within each category and country. The highest estimate is as much as an order of magnitude greater than the lowest estimate, reflecting the substantial uncertainty surrounding environmental benefit estimates in general. The greatest uncertainties are for waste, followed by air and water. In the ranking of benefits, air was generally seen as more important than water, especially if one takes the higher estimate into account (the exceptions are Bulgaria and the Czech Republic).

These benefits can be compared with the investment and operating costs, discussed previously, and shown in Table 13.[6] The ratios are generally, but not always, greater than one. In particular, for waste directives, estimates are less than one in most cases of "low" benefits. In the case of water, the ratios are less than one for some countries in terms of the low benefits (Bulgaria, Hungary, Romania, Estonia and Latvia). In the case of the air directives, there is only one case of benefits less than one (Bulgaria).

The author's main point is that these estimates are useful in showing the overall benefits and costs and could be the point of departure in any discussion with the Commission on decreasing or accepting temporarily lower standards. They must, however, be interpreted with some care. First, an overall ratio of greater than one does not imply that: (a) the net benefits of all projects will be greater than one;

TABLE 12

Total benefits over the period 2005–2020, NPV at 4% discount rate (billions of Euros)

	WATER		AIR		WASTE		TOTAL BENEFITS		
	Low	**High**	**Low**	**High**	**Low**	**High**	**Low**	**High**	**Per capita (Euros)**
Bulgaria	1.58	4.20	1.07	11.00	0.20	6.62	2.85	21.82	3.01
Czech Republic	15.23	24.05	7.10	35.10	0.93	11.20	23.26	70.35	9.09
Hungary	2.72	10.49	5.74	39.92	1.12	18.50	9.58	68.91	7.77
Poland	13.59	31.96	25.80	149.90	1.60	26.30	40.99	208.16	6.44
Romania	3.96	12.15	7.59	56.95	0.83	26.30	12.38	95.40	4.79
Slovak Republic	3.00	6.61	3.40	21.90	0.29	4.28	6.69	32.79	7.31
Slovenia	1.47	3.44	0.68	4.62	0.24	2.82	2.39	10.88	6.64
Baltics									
Estonia	0.26	0.99	0.39	2.05	0.09	1.75	0.74	4.79	3.95
Latvia	0.38	1.34	0.49	3.12	0.05	1.07	0.92	5.53	2.69
Lithuania	1.23	2.75	1.56	7.98	0.06	2.00	2.85	12.73	4.21
Total	43.42	97.98	53.82	332.54	5.41	100.84	102.65	531.36	6.06
As % of total	42%	18%	52%	63%	5%	19%	100%	100%	

(b) the ratio cannot be increased by delaying implementation of some of the directives; and (c) the social costs of implementing the directives may outweigh the assessment based on the ratios presented. Indeed, it is likely to be so in a number of cases. It does point, therefore, to the need to carry out a careful benefit-cost analysis of the implementation for each investment.[7]

From Table 13, it is clear that even in the worst case (low estimate of benefits), the benefits of meeting EU environmental standards will be only slightly lower than costs (benefit/cost ratio is equal to 0.73). As for the high estimate, the benefits will be more than 3 times higher than the costs. For Poland, Hungary, Romania and the Slovak Republic, the ratio is close to 10. This implies substantial potential benefits from meeting the environmental standards necessary for EU accession.

Thus, the major recommendation for CEE countries is for them to propose efficient ways to meet new environmental standards. This will allow them to reach higher benefits and lower potential costs. If they do not agree, their losses could be extremely high, and these will increase with renewed GDP growth.

Table 13 demonstrates that the ratio of benefits to direct costs for water clean-up in CEE countries varies by more than 10 times. There are several factors for this difference such as the different amounts of water resources to clean up in each country, as well as cross-boundary issues (e.g. the Czech Republic gets relatively cleaner water from Western Europe). Another factor could be density of population and industry. The higher the density, the more businesses and households benefit from improvements in water quality. In addition, the deviation mentioned above could be explained by the difference in environmental protection policies these countries have implemented. Saving on past abatement expenses turns into high current clean-up costs, at a time when CEE countries are looking for ways to meet EU standards. Some of those costs could have been avoided if CEE countries had had different environmental protection policies in place.

TABLE 13
Ratio of benefits to costs for EC directives in CEE countries

	WATER		AIR		WASTE		TOTAL BENEFITS	
	Low	**High**	**Low**	**High**	**Low**	**High**	**Low**	**High**
Bulgaria	0.73	1.94	0.26	2.64	0.08	0.40	0.20	0.85
Czech Republic	10.43	16.48	1.36	6.70	0.08	0.91	1.06	2.92
Hungary	0.93	3.59	2.60	18.07	0.36	1.30	0.63	9.16
Poland	1.70	3.99	2.27	13.18	0.49	2.46	0.98	13.12
Romania	0.89	2.72	1.02	7.65	0.56	3.01	0.51	9.11
Slovak Republic	3.57	7.87	2.19	14.09	0.66	0.83	1.37	10.55
Slovenia	n/a	n/a	1.20	8.18	0.14	0.76	1.06	4.45
Baltics			8.08	49.71	0.30	3.85	0.28	1.27
Estonia	0.39	1.48						
Latvia	0.50	1.77						
Lithuania	1.17	2.61						
Total	43.42	97.98	53.82	332.54	5.41	100.84	102.65	
531.36	6.06							
As % of total	42%	18%	52%	63%	5%	19%	100%	100%

CHAPTER 7

Conclusion

In conclusion, this paper demonstrates that, although transition countries face difficult choices in allocating limited resources to improve welfare, promote economic growth, and reduce poverty, environmental conditions can greatly influence people's welfare. This is particularly true in terms of their health, and the sustainability of economic growth. However, the value of the environment for society is often underestimated due to institutional constraints, as well as currently low WTP for environmental services. As it is reflected by the environmental Kuznets curve, initially, environmental quality deteriorates as a result of resource-intensive economic development. Once society reaches a certain level of development, environmental quality starts to improve as demand for quality of life increases. Environmental goods and services seem not to be vital when confronted with urgent economic problems. Both the forward- and backward-looking points of view are quite important in this respect.

It is not easy to express environmental value in monetary terms. By contrast, costs related to improving environmental quality in terms of investment and expenditures are more easily measured, and thus dominate the thinking of decision-makers. As a result, fewer resources are being allocated to improve environmental quality, and the cost of inaction is being ignored. In terms of economic valuation, the main problem determined by the specific characteristics of transition economies is that forward-looking estimates do not adequately take into account the value of natural resources or of environmental services. The forward-looking valuation is made when environmental assets are available, but many are judged as expendable. By the time backward-looking valuation is made, only a few assets are left, raising their value considerably.

Several examples were presented for both developed countries (USA: benefits and costs of Clean Air Act and the least cost SO_2 reduction program) and transition countries (EU accession countries). They demonstrate the relevance of benefit-cost analysis and cost-effectiveness analysis for decision-making purposes. Both analyses provide necessary information to encourage the adoption of environmentally friendly laws and regulations, and the implementation of environmental projects, and serve as justification for environmentally friendly investments. Moreover, they transform environmental regulation, resulting in stricter environmental targets and a reduction of the relevant costs of meeting those goals.

Also, we used the example of the Clean Air Act to suggest that benefit-cost analysis cannot solely be used to justify emission targets. The US experience could be useful for decision makers in transition economies. Focusing on minimizing costs, building compromises between interest groups, and instituting an effective enforcement mechanism, will help decision makers set realistic goals for emission targets and environmental regulation.

The most important factor in choosing between economic analysis methods to shape environmental policy is to understand the benefits and drawbacks of each method. The valuation of economic benefits or damages remains a good tool for focusing attention on an environmental problem and contributing to the initial debate. However, different analyses are useful for different reasons. Benefit-cost analysis alone could misguide the decision-making process, especially in a transition economy. The analysis could underestimate avoided damage (the contradiction between forward- and backward-looking approaches) and overestimate cost (through a high discount rate).

With regard to economic analysis, we recommended the following for choosing when to use a benefit-cost or cost-effectiveness analysis:

- Benefit-cost analyses are useful in defining the scope of the political discussion;
- Emission targets are always the result of political processes and compromise;
- Cost-effectiveness analyses help select the regulatory strategy

References

1999 Data, World Bank, 2001.
Annual Russian Federation Statistic Book, 2000.
Avaliani S., D. Dudek, A. Golub, G. Safonov, D. Shaposhnikov, E. Strukova. 2002. Co-benefits of greenhouse gas emission reduction related to conventional air pollution in Veliki Novgorod (Russia), Environmental Defense.
Bureau of Economic Analysis, Survey of Economic Policy in Russia in 1999. Moscow, TENC, 2000.
Bobylev S. & A. Stetsenko. 2000. Total economic cost of Natural Park Naluchevo (Kamchatka)/Environmental Valuation by J. Dixon, L. Scura, R. Carpenter, & P.Sherman. Vita, Moscow, 270, p.242–243.
Lipton, D., K. Wellman, et al. June 1995. Economic valuation of natural resources. A handbook for Coastal Resources Policymakers. NOAA Coastal Ocean Program.
Dudek, D., A. Golub, and E. Strukova. 2000. Economies in transition: Pathways for sustainable economic development and climate protection. Washington, DC: Working Paper, Environmental Defense.
Dudek, D., A. Golub, and E. Strukova. 2002. Ancillary Benefits of Reducing Greenhouse Gas Emissions in Transitional Economies. Washington, DC: Working Paper, Environmental Defense.
Ecotech. 2001. The Benefits Of Compliance with the Environmental Acquis for the Candidate Countries. European Commission: DG Environment, Brussels.
EDC. 1997. Compliance Costing for Approximation of EU Legislation in the CEEC. European Commission: DG Environment, Brussels.
Environmental Defense. September 2000. From Obstacles to Opportunities. How acid rain emissions trading is delivering cleaner air. Environmental Defense.
Environmental Performance Review of the Russian Federation. OECD, 1999.
European Commission. 1997. Guide to Cost Benefit Analysis of Major Projects in the Context of EC Regional Policy. European Commission, Brussels.
Feather P., D.Hellerstein, and H. LeRoy. April 1999. Economic Valuation of Environmental Benefits and the Targeting of Conservation Programs: The Case of the CRP. Agricultural Economics Report No. 778.
Fomenko, G. et al. 1997. Environmental Management for Sustainable Development. NPP "Cadastr," Yaroslavl.
Fomenko G., M. Fomenko, K. Loshadkin. 2000. Monetary valuation of natural resources and services of ecosystems in the development of territories: adjustment of the UN approach to Russia. NPP "Cadastr," Yaroslavl, pp. 128-81.
Golub, A. and E. Strukova. 2000. "Russia at GHG Emission Market." Cambridge, MA: Draft paper.
Grossman, G. and A. Krueger. 1997. Environmental Impact of a North American Free Trade Agreement. Working Paper 3914. National Bureau of Economic Research, Cambridge, MA.
Hager, W. 2000. Environmental Investment in the CEEC Preparing for Accession Mimeo, World Bank.

Hughes, G. and J. Bucknall. 1999, Poland: Complying with EU Legislation. ECSSD Working Paper 14, World Bank.
Hughes, G. et al. 1997. Can the Environment Wait? World Bank.
Khanna, N. 2002. Is air quality elastic? Revisiting the Environmental Kuznets Curve Hypothesis.
Kuik, O. J. and F. H. Oosterhuis 2001. Lessons from Southern Enlargement of the EU for the Environmental Dimensions of Eastern Enlargement, in Particular for Poland, Nota di Lovoro, 59.2001: FEEM, Milan.
Kuznets, S. 1995. Economic growth and income inequality.
Markandya, A. and A. Golub. 2002. On environmental regrets. Draft.
Markandya, A. Expenditure Policies in the Context of EU Accession. Dealing with the Challenges of Accession: The Environmental Dimension. Prepared for World Bank/IMF Conference on "EU Accession - Developing Fiscal Policy Frameworks for Sustainable Growth," September 2002.
Markandya, A., and F. Patrick. 2001. Promoting Private Sector Participation in the Financing of Sustainable Development. In *Financing Sustainable Development: Testing Policy Approaches,* J. Holst, ed. United Nations: New York.
OECD. 1998. Environmental Performance Review of the Russian Federation.
Ostro, B. 1994. Estimating the Health Effects of Air Pollutants. A Method with an Application to Jakarta. Policy Research Working Paper No. 1301.
State Committee of Russian Federation on Statistics, Statistical bulletin #6 (80), 2001.
Strukova, E., V. Sidorenko, S. Bobylev, A. Golub, G. Safonov, & S. Avaliani. 2001. Macroeconomic Assessment of Environment-Related Human Health Damage Cost for Russia. World Bank working paper.
United States Environmental Protection Agency. 1997. The Benefits and Costs of the Clean Air Act, 1970 to 1990. Washington, D. C.
United States Environmental Protection Agency. 1999. The Benefits and Costs of the Clean Air Act, 1990 to 2010. Washington, D. C.
Using Valuation of Environmental Impacts in Decision-Making. Paper for the NIS Ministerial Conference in Almaty; World Bank, 2001.
Yandle, B. et al. May 2002. The Environmental Kuznets Curve.

Notes

[1] Hughes, G. et al. 1997, "Can the Environment Wait?" World Bank.

[2] Below are two U.S. examples of the Clean Air Act using a retrospective and future benefit-cost analysis. In addition, we present an example from EU accession countries, which demonstrates that some countries face much higher costs to reach EU environmental standards because they missed the time to start water protection and abate water discharges at less cost.

[3] The most obvious example of the consumer's lack of choice is a restricted real estate market. In Soviet times, even if a consumer had information about polluted areas within the city, he usually could not move to another apartment to exercise his desire for a cleaner environment. Market economies not only allow choice to be exercised, but also provide information about pollution.

[4] Presented based on United States Environmental Protection Agency. 1997. The Benefits and Costs of the Clean Air Act, 1970 to 1990. Washington, D. C.; United States Environmental Protection Agency. 1999. The Benefits and Costs of the Clean Air Act, 1990 to 2010. Washington, D.C.; and materials of Environmental Defense.

[5] For present value of costs and benefits.

[6] The table has been constructed by annualizing the benefits in Table 12 at a real discount rate of 4% over 15 years. This figure has been divided by the sum of annualized 20 year investment estimates, and the annual operating costs. Although the data is from an EC study, the analysis of net benefits done here has not, to our knowledge, been attempted before.

[7] The calculations were carried out using a real discount rate of 4%. This is somewhat lower than the rate normally used to discount public sector projects by the Commission. A recent Commission document on Benefit-Cost Analysis proposes a real rate of 5% for projects involving structural funds (EC 1997). The reason for this small difference is not clear but it does not significantly change the results shown in Table 13. The countries themselves, however, may prefer a higher rate, and select those investments that yield a higher 'up front' benefit.

e

environmental defense

finding the ways that work

National Headquarters
257 Park Avenue South
New York, NY 10010
212-505-2100

1875 Connecticut Avenue, NW
Washington, DC 20009
202-387-3500

5655 College Avenue
Oakland, CA 94618
510-658-8008

2334 North Broadway
Boulder, CO 80304
303-440-4901

2500 Blue Ridge Road
Raleigh, NC 27607
919-881-2601

44 East Avenue
Austin, TX 78701
512-478-5161

18 Tremont Street
Suite 850
Boston, MA 02108
617-723-5111

Project Office
3250 Wilshire Boulevard
Suite 1400
Los Angeles, CA 90010
213-386-5501