Risk Analysis, Vol. 22, No. 5, 2002

Tools for Comparative Analysis of Alternatives: Competing or Complementary Perspectives?

Patrick Hofstetter,[1]* Jane C. Bare,[2] James K. Hammitt,[3] Patricia A. Murphy,[4] and Glenn E. Rice[4]

A third generation of environmental policy making and risk management will increasingly impose environmental measures, which may give rise to analyzing countervailing risks. Therefore, a comprehensive analysis of all risks associated with the decision alternatives will aid decision-makers in prioritizing alternatives that effectively reduce both target and countervailing risks. Starting with the metaphor of the ripples caused by a stone that is thrown into a pond, we identify 10 types of ripples that symbolize, in our case, risks that deserve closer examination: direct, upstream, downstream, accidental risks, occupational risks, risks due to offsetting behavior, change in disposable income, macro-economic changes, depletion of natural resources, and risks to the manmade environment. Tools to analyze these risks were developed independently and recently have been applied to overlapping fields of application. This suggests that either the tools should be linked in a unified framework for comparative analysis or that the appropriate field of application for single tools should be better understood. The goals of this article are to create a better foundation for the understanding of the nature and coverage of available tools and to identify the remaining gaps. None of the tools is designed to deal with all 10 types of risk. Provided data suggest that, of the 10 types of identified risks, those associated with changes in disposable income may be particularly significant when decision alternatives differ with respect to their effects on disposable income. Finally, the present analysis was limited to analytical questions and did not capture the important role of the decision-making process itself.

KEY WORDS: Comparative risk assessment; life cycle assessment; risk tradeoff analysis; health-health analysis; benefit-cost analysis; cost-effectiveness analysis

1. INTRODUCTION

The nature and our understanding of environmental problems and solutions underwent major transitions in the recent history of environmental regulation and management. Over the previous three centuries, there has been an increased understanding of the multicausal nature of human disease and the recognition of important groups of causative agents. This understanding has resulted in public health interventions that have significantly improved public health, as evidenced by dramatic reductions in morbidity and increased life expectancy in most parts of the world. A prime example of this early phase of environmental problem recognition and successful intervention is the implementation of filtration and chemical disinfection of public drinking water in the larger cities of Europe and the United States in the late

1 ORISE Research Fellow at U.S. EPA, National Risk Management Research Laboratory, Visiting Scientist at Harvard School of Public Health.
2 U.S. EPA, National Risk Management Research Laboratory (NRMRL), Cincinnati, OH.
3 Center for Risk Analysis, Harvard School of Public Health, Boston, MA.
4 U.S. EPA, National Center for Environmental Assessment (NCEA), Cincinnati, OH.
*Address correspondence to Patrick Hofstetter, Häldelistrasse 7, CH-8173 Neerach, Switzerland; patrick_hofstetter@yahoo.com.

0272-4332/02/0800-0833$22.00/1

19th and early 20th centuries.[1] We recognize three generations of contemporary environmental regulation and management. First, major problems of air and water pollution were dealt with using "end-of-pipe" technologies between the late 1960s and mid 1980s. Second, transboundary pollution effects such as Waldsterben (sick and dying forests), large environmental and human health consequences due to major accidents in Bhopal, Schweizerhalle, and Chernobyl, and extraordinary increases in gross domestic product (GDP) in OECD countries stimulated the generation of more stringent environmental regulations introduced in the late 1980s and early 1990s. Third, increasingly sophisticated methods allow us to detect lower and lower chemical concentrations and better understand causal links to potentially harmful effects, leading to the recognition that countervailing risks[5] may have been inadvertently created by certain risk management actions. As our understanding has expanded, we currently recognize the necessity of a comprehensive evaluation for all aspects of proposed risk management actions. To cut the compliance costs, end-of-pipe technologies were combined with technologies that reduce pollution at the source, better known as pollution prevention[2] and integrated waste management.[3] Environmental management instruments and standards[4] and a whole family of economic instruments have been implemented.[5,6] This third generation of environmental policy aims to reduce both the costs of compliance and the net risks. The complete assessment of the full range of hazards targeted and implied by regulations or management actions requires information and tools that will accommodate some form of comparative analysis of all aspects of risk management technology.

Analytic and quantitative tools are needed and we address them here to show the cause-effect relationships that have become obscured. In early human history, small-scale sufficiency economies and councils of sages facilitated an understanding of causal relationships between human activities and (adverse) effects. Most processes were perceivable and slow innovation cycles usually prevented catastrophes based on delayed effects.[6] Today's economy is based on global work division with several innovation cycles occurring in the same decade. In such a world, cause-effect relationships are hardly perceivable and effects may prove fatal when their cause (and originator) has already been followed by the next technological generation.

A large number of different instruments and tools to compare decision alternatives have been developed and applied in different decision contexts relevant for third-generation environmental decision making. Recent reviews of environmental policy and regulatory tools[7–11] and tools for environmental design and management[12–14] provide overviews of a large set of tools and attempt to characterize them with regard to specified interests.

Here, we start by asking which types of risks need to be analyzed when management options are compared regarding their environmental impacts and we illustrate these risks by introducing two examples of contemporary decision questions (Section 2). Based on this outline, we introduce a limited set of tools and techniques for comparative analysis and analyze their coverage of the identified risks (Section 3). In an overview, we will then show how they differ from each other when applied in their initial setting (Section 4).

More recently, these tools have been used in new contexts or used in attempts to cover aspects that have initially not been considered, for example, the comparative risk analysis of alternatives can be used to inform regulatory development, consumers,[15] and municipalities.[16] Life cycle assessments (LCA) are not only used to inform designers and consumers on design and purchase alternatives, but should also steer integrated product policy[17] or, according to the president of the German EPA, be "considered where fundamental decisions are on the way or shifts with long-term impacts on environmental protection."[18] Although this extension of LCA is largely supported by an international standard,[4] we question here whether the analytical framework in its present form is suitable for these additional tasks.

This formerly independent development of similar tools and the more recent overlap in terms of their application suggests either that the tools should be linked in a unified framework for comparative analysis or that the appropriate field of application for

5 The terms "risk," "impact," "effect," and "hazard" will be used interchangeably in this article, given that the covered disciplines currently use them to reflect different but related meanings and in reference to both probabilistic and deterministic events.

6 However, nonhuman-induced meteorological changes that, e.g., caused the disappearance of the early flourishing Mid-Eastern farming communities, overexploitation of natural resources as practiced by, e.g., the Pueblo Bonito in New Mexico or the population of Easter Islands, and effects that were difficult to perceive, like the lead problems of the Romans, are exceptions that led to major catastrophes.

single tools should be better understood.[19] Section 5 provides a set of useful questions that help identify appropriate tools that can provide the information needed in decision support. This article contrasts the need for information when actions to reduce environmental risks require evaluation and the coverage of this need by available tools. The ultimate goal is to guide the selection of tools within the present decision support and to indicate gaps that identify specific needs for further research and refinement of tools.

2. WHICH INFORMATION ON WHICH RISKS IS NEEDED?

2.1. Two Illustrative Examples

To make the discussion more concrete, we will use two illustrative examples. The prevention of health effects due to the West Nile virus (WNV) and due to methyl tertiary butyl ether (MTBE) in drinking water are two typical examples of environmental decision making where comprehensive information on management alternatives is needed.

2.1.1. Managing the West Nile Virus (WNV)

WNV is an emerging pathogen in the western hemisphere, first being reported in 1999. WNV infection typically results in an onset of mild symptoms, including fever, headache, and malaise; symptoms associated with a more severe WNV infection may include a high fever, disorientation, muscle weakness, and, rarely, death. WNV primarily appears to be transmitted through a bird-mosquito cycle. Infected mosquitoes transmit WNV to humans during blood feeding. The emergence of the pathogen has increased WNV surveillance efforts in birds and in adult and larval mosquito populations. The surveillance efforts are designed to evaluate the distribution and spread of WNV in the western hemisphere. These efforts have led to evaluations of the risk posed to public health as well as the consideration of the following risk management interventions (this list is intended for illustration only and is not exhaustive).

1. Education concerning adult mosquito feeding habits and capacity to transmit disease. The goal of these types of interventions is to reduce human exposures to the vector by influencing human behaviors (e.g., avoiding outdoor evening activities in WNV infested areas, installation of home window and door screens).
2. Reduction of larval habitats. The goal of these types of interventions is to decrease mosquito populations by reducing the quantity of standing water bodies where mosquito larvae develop, (e.g., flushing storm drains, removing tire piles, eliminating containers holding stagnant water such as abandoned pools and rain gutters).
3. Control larval mosquito populations. The goal of these types of interventions is to decrease mosquito populations in the larval stage through introduction of larvicides or insectivorous fish.
4. Control adult mosquito populations following surveillance targeting mosquito species distribution. The goal of these interventions is to reduce the adult WNV-infected mosquito populations directly by spraying organophosphates, pyrethrins, and pyrethroid-based insecticides.

As we will see later, each of these interventions potentially reduces the target risk of WNV infections of humans but has countervailing risks that arise from the intervention itself or by averting human behaviors.

2.1.2. Fuel Additives—Managing the Phase-Out of Methyl Tertiary Butyl Ether (MTBE)

Gasoline additives improve performance and reduce engine emissions. Lead was initially added to reduce engine knocking, but later was found to be a neurotoxicant. Based on this, the use of lead in gasoline in the United States was phased out starting in 1973.[20] It was also known at that time that gasoline with higher oxygen content would lead to a more complete combustion and reduce volatile organic compounds, carbon monoxide, NO_x, and air toxics such as benzene and 1,3-butadiene. Therefore, risk managers had to contend with two target risks: neurotoxicity of lead and air quality impact of using gasoline with lower oxygen content. Countervailing risks considered included decreased fuel efficiency (with its effects on resource use and increased pollution) and an increase in fuel prices.

MTBE was chosen as one of the additives because it dealt effectively with the two target risks and was believed to have acceptable countervailing risks. After a preliminary evaluation of its environmental fate, MTBE was approved for use in gasoline in the United States in 1979. The fraction of gasoline that contained MTBE increased from 8% in 1984 to 22% in

1988[21] and then continued to increase even more after the 1990 Clean Air Act Amendments, which required the use of reformulated gasoline (RFG) containing 2% oxygen by weight in certain regions experiencing high tropospheric ozone concentrations. By 1999, over 85% of the RFG contained MTBE—a highly mobile and persistent chemical in water. Through various leaks in underground storage tanks and other spills, by 1999 approximately 5–10% of the drinking water supplies in high oxygenate areas were recording detectable levels of MTBE. Concerns were raised primarily about the taste and odors related to MTBE contamination, although many began to question the possible health effects associated with exposure to MTBE in drinking water, and thus the wisdom of utilizing MTBE as a fuel additive. Therefore, the new target risks are the public health concerns and potential environmental impacts due to MTBE.

The Blue Ribbon Panel recommended:[22] (1) begin the reduction of MTBE use in RFG, (2) continue the testing and replacement of older leaking underground storage tanks, and (3) continue protection, treatment, and remediation of water supplies. The Panel recognized ethanol as the most likely replacement for MTBE.[22] Ethanol has been praised in the past because it can be produced from renewable biomass (grain crops), may lower impacts on climate change,[21] and is expected to have many other positive effects, including increased employment and an improved balance of international trade.[22]

2.2. The Ripple Metaphor

Fig. 1 introduces the metaphor of the ripples caused by throwing a stone in a pond, suggested by Graham and Wiener.[23] Throwing a stone stands for making a decision and the ripples represent the different consequences this decision may have in society, the economy, and the environment. Here, we focus on ripples that pose risks to human health and ecosystems where the height of the ripples corresponds with the size of the risks. First, the reader recognizes that the number of ripples is large and increases the longer the photographer waits for the shot. Second, the photograph illustrates that the inner ripples are not necessarily the largest. Third, the ripples have hills and "valleys." And, finally, the light may delude our perception. We propose the following interpretation of this metaphor.

- Decisions tend to alter many risks and actions tend to change the whole system. For instance, the use of larvicides may affect organisms

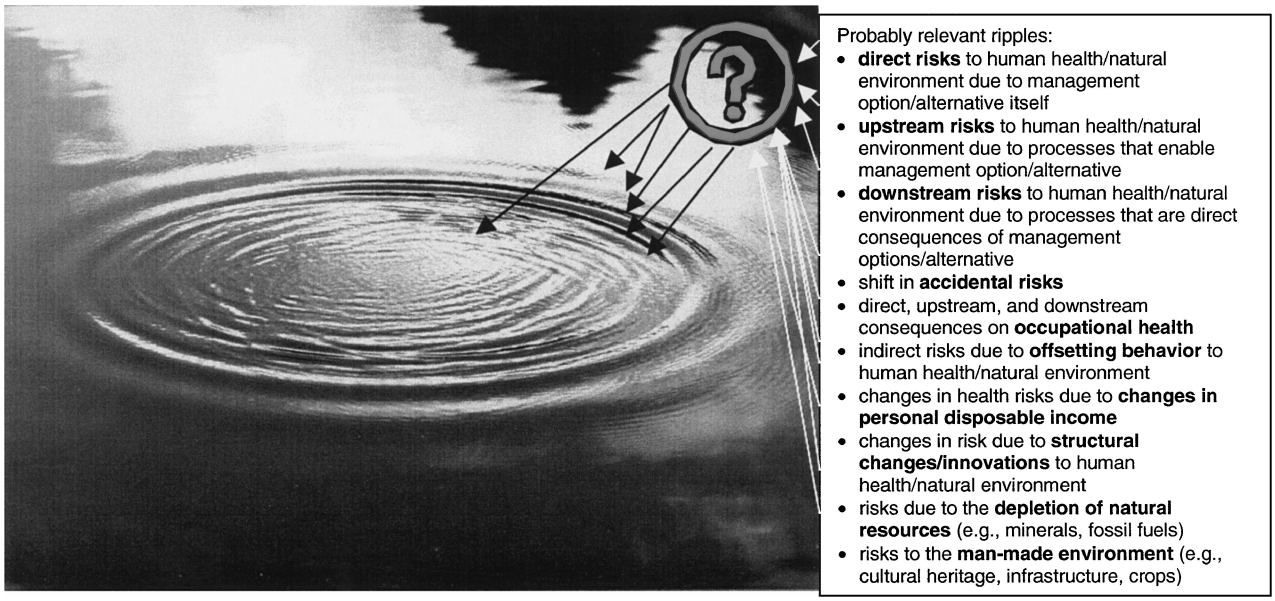

Fig. 1. Making a decision is symbolized by throwing a stone in a pond (analogy suggested by Graham and Wiener[23]). The water level symbolizes the level of risk and will immediately be lowered at the place the stone lands, i.e., the target risk is reduced. The ripples symbolize the countervailing and secondary risks due to the management action or suggested alternatives.

other than mosquitoes and new resistant mosquito strains may develop. If, for example, we wait long enough after tough environmental regulations have been introduced, induced innovations and market leadership may become a major ripple.[24][7] This may become true with respect to WNV control technologies and also with respect to "new" gasoline additives, more efficient ways to produce ethanol, or new engine concepts that do not require additives at all.

- The direct effects are not necessarily the largest; expensive measures to reduce health effects may induce even larger indirect health effects.[29] For example, if residents of a city are not allowed to use a local park during dusk to avoid mosquito bites or because insecticides were sprayed and instead drive to more distant recreational areas, the health risks due to traffic accidents and pollution may be larger than the reduction of the targeted risk.
- Next, to the reduction of the target risk, there are not only countervailing risks but also synergy effects (the "valleys"), for example, if the replacement of MTBE causes increases in gasoline prices, then the driven mileage goes down, which may reduce health risks due to accidents and pollution; alternatively, closing parks during dusk may reduce the crime rate during these hours.
- New regulations or management actions may induce economic changes that are largely influenced by human behavior. For a realistic prediction of these changes, we would need general equilibrium models that are at least able to predict changes in behavior due to changes in prices. In principle we would also like an economic model that considers changes in the level of information people have and use. In the case of increases in gasoline prices, we would need to predict whether consumers just absorb the increase by making driving a higher priority in their spending or if they demand more fuel-efficient cars.

In Fig. 1 we name some of the ripples that may be relevant depending on the type of decision at hand. Additional types of risks could be added and Section 4 provides some alternative labels. The direct risk of the larvicide spraying (WNV) is the potential health effects of the larvicide on park visitors. Upstream risks would be impacts from agricultural crop production and ethanol synthesis, while downstream risks would include changes in the emission behavior of engines (MTBE). The leakage from underground storage tanks may be considered an accidental risk; other new accidental risks may occur when large amounts of ethanol need to be transported from the midwest to the coasts (MTBE). Occupational health risks are incurred when larvicides are sprayed (WNV), underground storage tanks retrofitted, or petrochemical workplaces substituted by more risky farm jobs (MTBE). Offsetting behavior[8] would occur if, for instance, former park visitors substituted these visits with vacations in tropical areas (WNV) or if the use of ethanol prevented owners from properly maintaining their storage tanks (MTBE). The substitution of MTBE is anticipated to increase fuel prices. This increase will reduce the disposable income of drivers and may lower their spending for other safety and health measures. The need to replace MTBE may trigger the development of completely new additives that dramatically reduce engine emissions or may revolutionize farming of crops for ethanol production. Such innovations will lead to structural changes (change in sector structure and contributions to the economy) that cause long-term changes in risk patterns. If ethanol can be produced without using more fossil fuels than it replaces, the risks of fossil fuel depletion can be reduced (MTBE). The limitation of public access to a local park can be considered as a devaluation of a manmade environment (WNV).

7 The suggestion that environmental regulations create innovation that offsets the cost of regulation is known as the Porter Hypothesis.[24,25] Although regulation may induce firms to innovate in ways to reduce the burden of regulation, economists are generally skeptical of the claim that regulation can lead to net gains in efficiency, as firms would be expected to search out these innovations even in the absence of regulation. Empirically, there is little compelling evidence and the issue is difficult to evaluate as it requires comparing the innovations firms make in response to regulation with innovations they would have made without the regulations.[26–28]

8 The risk homeostasis theory[30] says that individuals have a target level of risk and that if management actions only reduce the risk but do not alter this target level, the gain in risk reduction is likely to be offset by offsetting behavior or "rebound" effects. This means that, e.g., risk reductions in the workplace may be compensated by risky hobbies, or laws to use seatbelts may be met by driving faster, etc.[31,32] Wilde[30] claims that in the case of traffic safety, the perceived costs and benefits of risky and cautious behavior determine this target level. Therefore, risk reduction is successful if these perceived costs and benefits are altered as well.

3. DESCRIPTION OF TOOLS FOR COMPARATIVE ANALYSIS

From a large number of available environmental assessment tools we selected a subset of tools that have been used in comparative analysis and/or are designed to quantitatively evaluate some of the risks identified in Fig. 1. We also concentrate on decision-support tools rather than monitoring tools. These criteria justify excluding less comparative tools such as environmental impact assessment, risk assessment, green accounting or eco-audit, and substance and material flow analysis. This section introduces the tools and concludes with an overview table that reports the coverage of different types of risks.

Life cycle assessment (LCA) is a method for the comprehensive environmental assessment of products and services.[9] The comprehensiveness has two dimensions. First, a large variety of environmental impacts that affect resource stocks, ecosystems, and human health are included and second, impacts along the full life cycle of a product—from cradle to grave—are considered and allocated to the chosen product function.

Fig. 2 illustrates a life cycle for a pair of skis that fulfills certain minimum requirements, for example, a technical lifetime of five years, the performance required for recreational use, and delivery within two days for an anxious skier. The production/purchase of an additional pair of skis will cause additional resource extraction of, for example, iron ore and crude oil; will involve many raw material acquisition and fabrication steps; will require banking and insurance services to the involved industries; and will increase transportation of goods and energy production. In turn, each of these businesses will stimulate additional economic activities. To advertise their product, ski producers will sponsor ski racers who will travel around the world several times during the winter race season. If additional purchasing supports the marketing strategy, the sponsorship will be maintained or even increased in future seasons. Therefore, a small share of the marketing efforts results from individual purchase of skis. After the skis lose their tension or become obsolete by successful advertisement for the next generation of skis, they will take up (heated) storage space for a few more years and finally be disposed

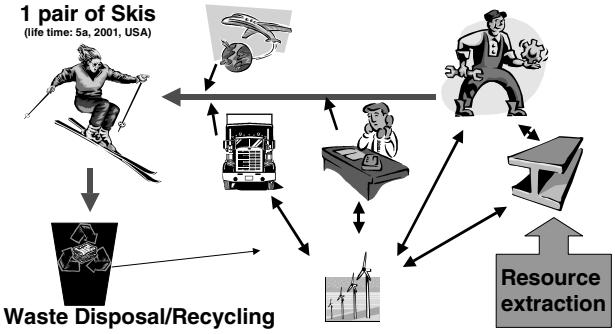

Fig. 2. Simplified illustration of a web of processes that are related to the service that a pair of skis can provide during five years. The cycle from resource extraction through production, use, and disposal is usually referred to as the life cycle.

of or recycled. As the double arrows indicate, this life cycle is not a linear chain of processes but a connected web of economic relations.

LCA has its methodological roots in energy analysis[33] and was applied in the late 1960s and early 1970s to questions of energy-supply systems and product-packaging systems.[34,35] The LCA methodology was further developed in the 1990s under the auspice of SETAC[14,36,37] and standardized by ISO.[4] The ISO framework distinguishes four phases that are iteratively applied: goal and scope definition, life cycle inventory analysis (LCI), life cycle impact assessment (LCIA), and interpretation. Major methodological developments have been made: in LCI by developing operational methods to divide the product system from the remaining economy and by linking economic input-output tables, environmental statistics, and conventional process-chain analysis; and in LCIA by linking indicators for relevant environmental effects with information of emissions and resource use, combining and adopting the wealth of models and data available from environmental chemistry, toxicology, epidemiology, medicine, etc.[38–40]

The comprehensive scope of this tool, including hundreds of environmental stressors and thousands of processes that are located worldwide and take place in the past, present, or future, led to a number of simplifying assumptions implemented in most LCA approaches: ignoring the temporal and spatial scale, the assumption of *ceteris paribus* (that all other processes that are not considered part of the product system remain constant), the linearizing of emission-dose-response relationships, or ignoring disparate subpopulation effects. Another simplification is that products (or a set of products) that provide the same service

[9] In accordance with ISO,[4] we will use "product" as the term for the subject of analysis, even though all kind of services are included.

(called a functional unit) are compared. Therefore, no product benefit analysis is needed.

ISO[4] lists the following applications for LCA: product development and improvement, marketing and product information to consumers, strategic planning, and public policy making. LCA also assists in environmental management systems, performance evaluation, and labeling.

This additional use for strategic planning and public policy making and the aim to improve the realism of the models recently led researchers to develop site-dependent,[41–49] nonlinear,[50] or more risk-based approaches.[51–54]

Programmatic comparative risk analysis (PCRA) refers to the application of comparative risk assessment to set priorities for research and risk management actions. PCRA describes a process of creating a risk-based, scientific ranking of relatively broad-based environmental problem areas, for example, indoor radon, drinking water contaminants, and criteria air pollutants. The outcome of such an exercise is a relative ranking of the multiple environmental problem areas that may then serve as a template for creating applied research programs and for prioritizing resources. The question usually addressed at this level is: "Are we as a nation (region, state, etc.) addressing/spending our resources on the most important environmental problems?" This type of PCRA has been used as an analytical policy tool by public agencies at all levels of government to inform a variety of environmental priority-setting efforts.[55–62]

PCRA includes three steps of analysis.[63] Identifying a list of environmental problems to be analyzed and compared is the first step. A second step identifies the specific risk domains or arenas that are to be included in the evaluation. The risk domains usually considered in a PCRA have included human health, ecosystems, and welfare or quality of life, with human health appearing to be a common element in most of the currently published PCRAs.[62] Determining and quantifying the criteria to measure risks in the mentioned domains is also part of the second step. In the third step, the environmental problems are ranked to establish an ordered or categorized list. Available studies vary between pure expert assessments,[55] largely participatory processes, and stakeholder assessments.[57]

Analytic challenges include ensuring that the chosen environmental problems are not overlapping or double-counted by listing problems on the source, substance, and damage level (e.g., use of cars, VOC emissions, summer smog, and asthma). In the application of the results two major problems have been realized. First, risk rankings usually refer to the *status quo*, that is, the residual risks that may be useful information in prioritizing research expenditures. However, risk management expenditures need to be allocated to risks where reductions can be made at the lowest cost.[59,62–64][10] Second, resources are spent on research programs and risk management actions and not on environmental problems. Therefore, results of PCRA need a careful translation to be useful in supporting actual decision making. We understand PCRA here as one element of information needed in comparative decision support.

The analysis and comparison of risks from two or more risk management alternatives is called *comparative risk analysis of alternatives* (CRAoA). The term "comparative risk" was probably introduced in the 1970s and methods for CRAoA have been developed in the fields of energy system analysis,[65] hazardous-waste sites,[66] and technology assessment.[67] Although the developments in the field of energy supply comparisons concentrated on human health impacts, other early methods considered both human health and environmental impacts. More recently, CRAoA has been used to assess human health tradeoffs of eating healthy but contaminated fish,[15,68] to optimize the treatment of drinking water by minimizing risks due to both bacterial infections and chemical byproducts,[16,69] and to regulate sulfur in fuel.[70] Comparative ecological risk assessments are not routinely performed. Combined health and ecological risk approaches are rare[71,72] but advances in the development of metrics to compare ecological impacts may increase the number and utility of comparative ecological assessments.[73,74]

Risk tradeoff analysis (RTA), sometimes called risk-risk tradeoff analysis or risk-risk analysis, was introduced in 1995 for the purpose of analyzing countervailing risks that occur due to the management of target risks.[23] Based largely on earlier work of Lave[75] and Viscusi,[76] the concept draws from previous experiences with medical treatment method analyses and their countervailing risks. Within RTA, the target risk to be regulated is specified, and policy options that reduce target risks are developed. Then the most likely consequences of a regulation/management option in

[10] This does not imply that distributional requirements are not considered.

		Compared to the target risk, the countervailing risk is	
		SAME TYPE OF RISK	DIFFERENT TYPE OF RISK
Compared to the target risk, the countervailing risk affects	SAME POPULATION	Risk offset	Risk substitution
	DIFFERENT POPULATION	Risk transfer	Risk transformation

Fig. 3. Terminology and characterization used in risk tradeoff analysis.[23]

terms of product substitution or market behavior are identified and their countervailing risks are assessed. Fig. 3 proposes a topology on how these countervailing risks are named and ordered. Changes in the type of risks and risk carriers are both made transparent by following this structure. The underlying reasons for presenting countervailing risks in this manner are the findings of risk perception[77] and experiences with PCRA listed above, that is, it does matter who will be at risk and by what types of risk.

Recent applications studied risk tradeoffs in the remediation of hazardous-waste sites,[78] potential countervailing risks due to changes in building codes,[79] and risk tradeoffs of banning organophosphate and carbamate pesticides.[80] Although RTA is in principle considering risks to ecosystems, human health is the primary focus of these examples. Number of fatalities and quality adjusted life years (QALYs) serve as the primary endpoint indicators.[81,82] Since the quantitative modeling of potential damages to human health due to decreased biodiversity or changing climate is difficult, such impacts are usually not quantified, but may be addressed qualitatively or descriptively.

The largest share of the risks quantified by Hammitt *et al.*[79] and Gray and Hammitt[80] were fatalities due to income effects, that is, potential lives shortened when regulations lower the income of a group in the population, which results in a statistical reduction in purchase of health and safety (see health-health analysis below). This result is surprising because the change in consumption patterns due to a regulation is expected to occur as a very distant ripple.

Although Graham and Wiener[23] provided a framework for RTA, additional developments are needed to make this approach a more useful analytical tool. Little guidance is given on which ripples should be analyzed and how. Conceptually, we see no fundamental differences between RTA and CRAoA. Therefore, we will discuss them together in the remainder of this article.

Health-health analysis (HHA) (also known as wealth-health analysis), attempts to capture the complex relationship between induced changes in personal disposable income that result from regulation or policy and their public health consequences. Wildavsky[29] suggested that the fact that poorer people live at higher risk should be considered when regulations reduce the public's disposable income to ensure that these risks do not countervail the regulations' intent to reduce target risks. This occurs because less disposable income means less expenditure and, statistically speaking, fewer investments in safety features and healthy lifestyles. This includes car maintenance and replacement, adverse effects of overcrowding in poor housing, access to nutritious food, and the availability of timely medication and health care. However, it was not until Keeney[83] quantified the amount of economic expenditures for regulations that induces one additional statistical fatality that health-health analysis entered the policy arena. Keeney's estimates were first used in 1991 to oppose a regulation that protected workers from accidental startups of hazardous machinery suggested by the U.S. Occupational Safety and Health Administration (OSHA).[84] After this, applications to evaluate air-quality standards at workplaces, cadmium standards, food labeling,[84] CO_2-reduction policies,[85] building codes,[79] the 1990 U.S. Clean Air Act,[86] and a ban on specific pesticides[80] followed.

Two methods to estimate the relationship between change in disposable income and fatality risks have been applied in the literature: direct empirical estimates, which use cross-sectional and longitudinal data,[83,84,87–91] and indirect estimates, which are based on the theoretical relationship between willingness to pay to reduce risk and the average fraction of income spent on risk reduction.[92] These calculations suggest that a reduction of between $5–40 million in disposable income within the U.S. population induces the shortening of one additional life. Because low-income groups live at higher baseline risk and use a larger share of their income to improve health and safety, the lower range of these estimates applies to low-income groups and the higher end to high-income groups.

From an application point of view, it is important to acknowledge that the decrease of disposable income of consumers due to increased costs of, for example, food due to a ban of certain pesticides,[80]

causes a second-order income effect. The additional expenditures by consumers may increase the income of workers in the sectors with higher costs, although if the cost increase results from factors such as technology restrictions that reduce worker productivity, the higher cost may not increase worker incomes. In addition, the higher costs of some products will reduce demand for other products, potentially decreasing income in those sectors. Because all these income changes affect different populations with different income levels, very data-intensive general equilibrium models need to be run to calculate net effects.

Reversed causalities (because people are sicker they earn less) or omission of additional independent variables, such as education, that influence both income and health may confound these results.[88,90,93] Portney and Stavins[94] suggest that health impacts due to regulation costs will not be significant in most applications[11] and that benefit-cost analysis (BCA) (see below) should remain the first choice to analyze policy implications.[97] Portney and Stavins also highlight the importance of the other 70–80% of income that is (on average) not spent to reduce health risks. Therefore, they suggest using HHA only if net health benefits should be analyzed or when the use of BCA is prohibited. Wilkinson[98] provides evidence from many different disciplines that inequality in income and social capital distribution are the key factors for population health and not absolute income as suggested above. As pointed out by Wagstaff and van Doorslaer,[99] it is very difficult to find statistical evidence from regression analysis for hypotheses other than the absolute income hypotheses. However, this is also true for the rejection of the income inequality hypothesis. Further empirical research is needed to reject one or the other hypothesis and quantify the magnitude of this ripple effect.

Benefit-cost analysis (BCA) attempts to measure both benefits and costs connected to all consequences of decision alternatives. Usually, monetary values are used as common metrics for both costs and benefits. This standard method of economic decision making has been used for a long time in environmental decision making. Crucial elements are the identification and subsequent valuation of environmental and human health endpoints and the assessment of intergenerational health effects. Studies that quantify such external costs use—similar to LCA, CRAoA, and RTA—impact-pathway analysis but convert to monetary values rather than using physical indicators.[100–103] A large variety of methods to measure stated or revealed preferences have been developed and are in use and debates on the most appropriate approaches are ongoing.[104,105]

Pearce *et al.*[10] distinguish three types of BCA: efficiency-oriented BCA, distributionally-weighted BCA, and hybrid approaches. Efficiency-oriented BCA refers to the fact that BCA usually assumes that monetary units that go to or come from different individuals can be added and net benefits should be maximized on the monetary level, that is, little attention is paid to questions of distributional equity. Distributionally-weighted BCA attempts to also reflect distributional goals and was prescribed in early BCA guidelines that applied to developing countries.[106,107] Hybrid approaches monetize as many aspects as feasible and reasonable and include nonmonetary indicators for other attributes that are considered relevant to the decision. Multicriteria analysis techniques may then be used for further aggregation.

Cost-effectiveness analysis (CEA) is similar in scope to BCA but measures nonmonetary consequences in physical indicators. Such analyses are widely used in the medical and public health fields where health costs are compared with health improvements.[108–110] For the measurement of health improvements, many methods have been developed and

11 A theoretical argument would suggest the opposite. U.S. EPA[95] justifies regulation if the benefits calculated with a mean value of statistical life of $4.8 millions exceed the costs. If these regulation costs would be incurred by low-income groups, the income effect could induce fatality risks of the same order of magnitude as the reduction in the target risk that was intended by the regulation, meaning that if regulations are designed to achieve zero net benefits at the margin (maximizing total net benefits from a BCA point of view), then such a regulation does not reduce health risks at the margin, i.e., the assumed health benefits do not materialize. Therefore, to pass the health-health test is only a *necessary* rather than a *sufficient* criterion[84] but also to pass the BCA test is not *sufficient* to allow for net marginal health benefits. In practice, the marginal costs of regulations may be far below or above the mentioned threshold. Tengs *et al.*[96] looked at 587 life-saving interventions, some of which have been implemented, and found average life-saving costs from less than zero to $99 billion per life-year saved. About 40% of all interventions and 80% of interventions to limit environmental impacts (n = 124) cause average costs above $100,000 per life-year saved. These interventions are likely to have marginal costs above $4.8 million per shortened life and may induce at the margin more health effects than they prevent. Although the compilation of Tengs *et al.*[96] is impacted by publication and selection bias, there is evidence that a relevant number of management actions—when we evaluate only human health—may increase rather than decrease health effects.[91]

many take the general form of health-adjusted life years.[82,111] CEA may be useful in ranking risk management options and identifying the option with the least cost to achieve a predetermined level of effectiveness. Graham *et al.*[112] argued that standardized methods are needed for these analyses to make valid and meaningful comparisons. However, CEA alone will not determine whether a risk reduction is "worth" its cost.

None of the described evaluation tools covers all types of risk mentioned in Fig. 1 (see Table I). LCA has probably the widest coverage because it systematically evaluates upstream and downstream risks and considers resource depletion and risks to manmade environments and ecosystems. BCA has a similar coverage but it is limited by the inclusion of endpoints that can be monetized.

4. MAJOR STEPS IN UNDERSTANDING THE TOOLBOX FOR COMPARATIVE ANALYSIS

Table I evaluated the tools for comparative analysis with respect to their coverage of different types of risks. This section characterizes the tools according to their decision-making principle, assessment scales and dimensions, way to deal with distributional questions, the levels at which decisions are supported, and their assumptions on the systems' behavior. More detailed characterization frames applied to a broader range of tools are provided by Wrisberg *et al.*[13] and Pearce *et al.*;[10] attempts toward axiomatic explanations can be found in Heijungs;[117] and applying different tools to the same case study was developed in Bouman *et al.*[118] Section 5 will indicate how the knowledge of these characteristics can be used in tool selection and decision support. Since health-health analysis covers only one type of risk, the income effect, we assume that it would be combined with any other tool. Therefore, we drop this tool in the discussion below.

4.1. Decision-Making Principle and Distributional Aspects

Table II proposes that each alternative usually has "costs" and "benefits" that may be quantified differently and used in different ways to support decisions. The last column indicates the degree to which distributional issues are explicitly addressed. The term

Type of Risks	LCA	PCRA	CRAoA/RTA	HHA	BCA	CEA
Direct risks	×	(4)	×	(6)	(7)	(9)
Upstream risks	×	(4)	(5)	—	(7)	(9)
Downstream risks	×	(4)	(5)	—	(7)	(9)
Accidental risks	(1)	(4)	×	—	(7)	(9)
Occupational health risks	(2)	(4)	(10)	—	(8)	—
Indirect risks due to offsetting behavior	—	—	sometimes	—	(8)	—
Risks due to changes in personal disposable income	—	—	sometimes	×	—	—
Changes in risk due to structural changes/innovations	—	—	sometimes	—	(8)	—
Risks due to the depletion of natural resources	×	(4)	—	—	(7)	—
Risks to the manmade environment	(3)	(4)	—	—	(7)	—

Table I. Types of Risks Generally Considered in Comparative Analyses

(1) Includes only accidents that occur frequently enough that their emissions are included in yearly statistical compilations.
(2) Only included in the north of Europe.[113–115]
(3) Suggested for inclusion by Udo de Haes *et al.*[116]
(4) In principle included but usually related to total residual risk in a specified region.
(5) Sometimes included in a nonsystematic way and with human health focus.
(6) Although not part of HHA, it is assumed that the reduction in target risks to human health is known.
(7) Included if expected damages (quantified with other tools) can be monetized.
(8) In principle possible; in practice rarely done.
(9) Scope usually limited to human health impacts and quantification relies on other tools.
(10) Usually not considered, but see Gray and Hammitt[80] for the case of pesticide regulation and Viscusi and Zeckhauser[97] for an approach that relies on input/output tables.

Table II. Characteristics of Tools for Comparative Analysis

	Units of "Costs"	Units of "Benefits"	Decision-Making Principle	Distributional Questions
Life Cycle Assessment (LCA)	Physical impacts/damages or "ecopoints"	Equal functional unit	Lowest impacts	Not considered
Risk Tradeoff Analysis (RTA)/ Comparative Risk Assessment of Alternatives (CRAoA)	Direct health outcomes or health metrics	Differences expressed as "costs" (risks)	Lowest risks	Considered
Programmatic Comparative Risk Assessment (PCRA)	(Ordinal) ranking	Not applicable	Worst thing first	Sometimes considered
Cost Effectiveness Analysis (CEA)	Monetary	Direct health outcomes or health metrics	Biggest bang for the buck	Mostly not considered
Benefit Cost Analysis (BCA)	Monetary	Monetary	Maximize net benefits	Mostly not considered

"costs" means all types of adverse effects, such as damages, risks, or monetary costs. Under "benefits," we consider nonmonetary and monetary use values and positive effects, such as risk reduction. The "decision-making principle" reflects the main application originally intended when the tool was developed. However, the tools could also be used in different ways, for example, LCA could be used to ban the environmentally worst product providing the same function as other products, or BCA could be used to rank options according to either costs or benefits.

All tools deal with the distributional issues at some level but only a few address them explicitly. LCA assumes that each human being in the past, today, and in the future has the same utility and that utility is maximized when their sum is maximized. RTA, on the other hand, suggests that the utility may vary as a joint function of the specific group that is affected and the type of risk of concern (Fig. 3). To what extent the tools need to explicitly consider distributional questions will be discussed at the end of this section.

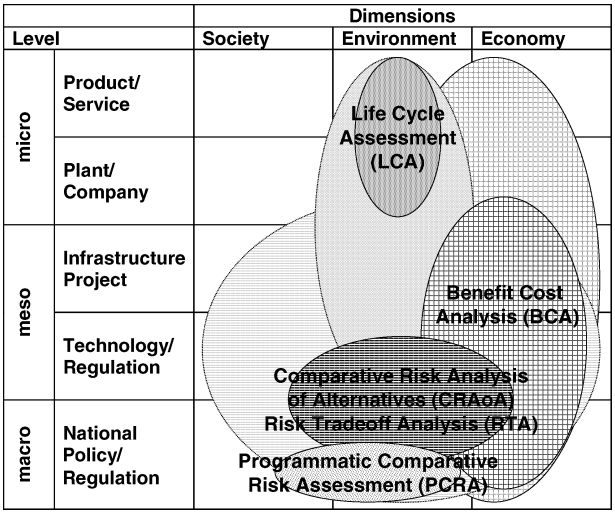

Fig. 4. Positioning of some tools for comparative analysis in the two dimensions "level of analysis" (from micro to macro) and "dimensions of analysis" (from society to economy) (adapted from Hofstetter [120]). Black textured areas indicate the initial scope of the tools and the gray textured areas indicate more recent or potential extensions.

4.2. Analysis Level and Assessment Dimensions

LCA was initially developed to compare and improve products and services and has been applied on a plant level. This we refer to as the micro level (see Fig. 4). On the other end of the scale,[12] PCRA has been applied to inform regulation from a macro perspective. CRAoA and RTA capture both the regulatory and technology assessment level and are used to inform national policy. BCA traditionally has had a wider field of application, being used to assess large infrastructure projects and new technologies and is sometimes requested if new regulations have large anticipated cost consequences. It is important to realize that although an integrated product policy is about products, the level of analysis is on a macro level because this policy does not only attempt to stimulate green product design in green leader companies or influence the behavior of single consumers, but also to initiate major shifts in the way products are designed and marketed. Micro-scale changes will effect incremental changes to the economy and the environment and macro-scale changes may turn markets toward a desired direction with known step-size and speed.

12 There is no international or global scale in Fig. 4. Since BCA, CEA, and "Integrated Assessment" have been applied on a global level in the course of decision support for climate change policy,[119] this level may be added.

Although LCA exclusively includes effects on the environment, PCRA includes "quality of life" aspects that reflect societal effects and RTA makes distributional questions transparent and may consider income effects. BCA attempts to include all important consequences to the extent they can be monetized. Of course, all tools are used to improve decision making in favor of societal goals and LCA and CRAoA/RTA do analyze the economic system. Fig. 4 attempts to indicate the dimensions in which changes are finally assessed.

Based on the gray areas in Fig. 4, we indicate present or intended extensions compared to the initial design of the tools. As mentioned earlier, today LCA is seen as a tool that can also support public policy and strategic long-term decisions. BCA has also been applied on a company or product level and attempts to capture additional environmental effects by improving methods to value nonmarket goods. CRAoA/RTA can be used not only to evaluate countervailing risks of planned regulations, but also for technology assessment (drinking water treatment), informing consumers (fish consumption), or optimizing plants (optimize the level of disinfectants in drinking water treatment). Further, although this is not indicated in the figure, elements of PCRA will be helpful within the tradeoff steps of LCA and CRAoA/RTA.

4.3. Indirect and Induced Risks

The short description of different risks (ripples) following Fig. 1 separated different safeguard subjects (human health, ecosystem health, natural resources, manmade environment) and the characteristic causing the risk (accidental, nonaccidental) and distinguished the directness of the link between action and risk. Here we differentiate among direct, indirect, and induced relations between action and risk because the analysis of each of these relations makes use of different methods and data. Direct risks of actions include those risks that have a perceivable cause-effect relationship for the user or beneficiary of a certain action. In the example of fuel additives, this corresponds with the (reduced) air emissions of cars. However, for a risk manager in a locality with extensive groundwater pollution by MTBE, the focus may change and the water pollution by MTBE may become both the target and direct risk.

The differentiation between indirect and induced effects is based on the assumptions of what is assumed to stay the same and what changes, that is, the *ceteris paribus* assumptions. By "indirect effect" we mean those causal links that are inherent to or follow necessarily from a management action because of physical laws or established practice, for example, the use of alternative pesticides will have a certain toxic potential and their production will require certain amounts of resources and processes to manufacture. The manufacturer will also need insurance and banking services, which are provided by the present economy. However, indirect consequences of a ban of organophosphate insecticides would not include, for example, a decrease in toy demand because of reduced disposable income of consumers; a shift of agriculture into developing countries due to tougher environmental regulations in industrialized countries; or new innovations in the agriculture/agro-chemical industry. These consequences do not necessarily need to happen but typically do.

Energy economics is one applied field of economics that has some tradition in looking closely into these induced effects. The idea is to identify to what extent energy efficiency improvements and energy policies suffer from rebound effects that cause backfire or take-back effects in the economy, that is, to what extent energy use can really be reduced or whether overall increases have to be expected after considering all induced effects. Greening *et al.*[121] and Binswanger[122] suggest the following terminology.

- *Direct rebound effect* (substitution effect, pure price effect): Greater efficiency may lead to a lower price of the service (or product or technology) and induce an increased use of this cheaper service.
- *Indirect rebound effect* (income effect, secondary effect): If prices of other commodities and income are constant, the reduction of costs due to a particular efficiency increase for a service implies that consumers have more money to spend on other goods.
- *General equilibrium effect* (economywide effects): The direct and indirect rebound effect lead to changed prices and consumption throughout the economy, which may increase or decrease production in distant sectors and changes in production functions.
- *Transformational effect*: This type of effect includes changes in consumer preferences, alteration of social institutions, and the rearrangement of the organization of production.

This terminology—although foreign to the field of risk assessment—may alternatively be used to label the ripples in Fig. 1. Offsetting behavior would then

fall within the direct rebound effect, effects due to the change in disposable income refers to indirect rebound effects, and structural changes/innovations would fall within general equilibrium effects. How do some of the discussed tools compare on the type of modeled consequences? RTA/CRAoA may consider risks by income effect but usually do not include other macro-economic effects such as increased import of cheap fruits, new developments of cheaper, more effective, and less toxic pesticides, or consumer willingness to buy more expensive but less polluted fruits. LCA is more thorough in analyzing indirect effects along the upstream and downstream of a management option, that is, considering the full life-cycle impacts. However, LCA does not—in its initial setup[38]—take into account induced effects. BCA, on the other hand, in principle, covers induced effects, especially if general equilibrium models are used to assess the market costs.[13]

4.4. Voluntary and Micro-Scale Changes

Should LCA model induced effects? Fig. 4 shows that LCA was initially designed for micro-level applications that cause changes on a micro level, where the induction of macro-economic effects is, although unlikely, certainly small. Assume the use of LCA to support consumers in their purchasing decisions. These are voluntary decisions and many other considerations in addition to LCA results influence the actual purchasing decision. In this situation, the decision-maker is the consumer with full understanding and control (but not necessarily consideration) of the implications an increased product price will have on her remaining possibilities to purchase health and safety. Although the purchase of a more expensive "eco-product" may affect the health risks due to all other purchases, we can predict neither the size nor the sign of such a change for this single consumer. Because we are able to estimate only statistically induced changes in fatality risk by higher product prices, these may not apply in this case, as the consumer may choose to finance her extra expenditure on the eco-product by reducing consumption of some other good that does not affect health, or even by reducing expenditures on harmful goods, further reducing her risk. A similar argument can be made in the case where the design department of a green company uses LCA to improve its products. This company's products do not compel consumers to spend less on health and safety, although this may be the consequence for some customers. However, if LCA is used to support (inter)national policy and regulations and would result in a general price increase of, for example, gasoline containing ethanol produced from crops, then consumers' choices become limited and a statistically induced increase in fatality risk is likely. Consequently, LCA must be supplemented with analyses that allow the inclusion of induced effects in these cases.

4.5. Distribution of Risks

A similar argument applies to the question of whether LCA should consider not only the aggregate population risk but also its distribution. LCA applied on a micro level in voluntary decision making will change the distribution of risk but this change will be small because potentially all sources of risks are minimized if we consider all LCA applications we can think of, and its direction and size is hardly predictable[14] because of the voluntary nature of the supported decisions. If the distribution could be predicted, it could turn out that its consideration would be inefficient because of its uncertain total size and direction. Compliance with environmental regulations may be a much more efficient means of minimizing individual risks. However, the application of LCA in technology assessment and regulation may indeed cause changes in the distribution of risks that contradict societal goals. If the correction of such a shift would imply inefficient countermeasures from the regulator, it may be necessary to include distributional considerations in macro LCA. This last statement holds as well for the other tools that have been designed for application on the macro level. Nevertheless, the regulator may have much more efficient instruments to correct distributional inequities than using the indirect impacts of environmental decision making.[8] If management actions are designed to avoid distributional changes in risk, they should also estimate the increase in costs due to these measures or the sacrificed benefits on a population level.

13 In practice, general equilibrium models are rarely applied. Exceptions include Hazilla and Kopp[123] and Jorgenson and Wilcoxen.[124]

14 Exceptions occur. The same amount of emissions released at rural production sites or sites with low downwind populations (like coastal locations) may cause the same risk to exposed people but less population risk. Therefore, minimizing population risk may cause a shift of the production to such sites.

4.6. Increasing Overlap

Table II and the black textured areas in Fig. 4 indicate that the tools for comparative analysis have distinct characteristics and fields of applications. However, these differences are vanishing due to broader fields of application for single tools. We see two reasons for this increasing overlap and mixture.

1. Decisionmakers and analysts lost the broader picture on available tools and are biased toward those that they have used before or that were developed within their own discipline.
2. Decisionmakers and analysts do recognize special features of some tools as being important and useful and want to incorporate them into a tailored decision-support system.

The first reason is understandable but should be avoided by specific guides to environmental toolboxes,[10,12,13] cross-disciplinary teaching of environmental decision-support tools at universities and in seminars for further education and, we hope, by articles such as this. The second reason is a challenge for tool developers.

5. QUESTIONS TO BE ASKED WHEN TOOLS FOR COMPARATIVE ANALYSIS ARE SELECTED AND TAILORED

This article focused on characteristics that distinguish the available tools for comparative assessment. Out of this analysis, we suggest five questions that decisionmakers or their analysts should answer when they select tools for providing needed information.

1. *Assessment dimensions*: Is the focus on environmental impacts or will adverse societal and economic effects be assessed as well? Is there a need to transform all information into monetary or other common units? (See Table II; Fig. 4.)
2. *Object/decision level*: Who are the decisionmakers, who is affected by the decision, and what type of change is expected (micro or macro)? (See Fig. 4.)
3. *Decision principle*: Is it about prioritizing a list of measures, selecting the one least damaging alternative, or implementing any measure with net benefit? (See Table II.)
4. *Distribution*: Shall aggregate population risk alone be minimized or do distributional aspects need to be considered? How much reduction in population risk shall be sacrificed for how much equity in distribution? (See Table II.)
5. *Type of risks*: Is it possible that the alternatives to be compared differ with respect to their direct, upstream, downstream, accidental and occupational risks, risks due to offsetting behavior, change in disposable income, macro-economic changes, depletion of natural resources, or risks to the manmade environment? (See Table I.)

The references in the parentheses refer to illustrations that allow one to narrow down the choices for the most suitable tools based on the given answers. However, while Questions 1 to 4 may be valuable in narrowing down the tools to be used, Question 5 is most probably very difficult to answer and needs either rich experience in a specific field of application or the initial consideration of all types of risks. Although all the listed types of risks prove decisive in some applications, it would be most valuable to have efficient screening tools that support the tailoring of the analysis at hand.

An attempt to coarsely estimate the importance of the ripples in Fig. 1 finds that induced health effects of management actions that reduce the disposable income are a major ripple that needs more attention in the future.[15] This was already argued in the description of the health-health analysis (page 839, footnote 11) and corroborated by Hammitt *et al.*[79] and Gray and Hammitt.[80] Other behavioral and macroeconomic changes, including the possibility for induced technological innovations, may be important but often beyond the realm of prediction.

Let us return to the two cases described in Section 2.1. Which tools may be best suited to shed light on potential countervailing risks? Applying the five questions reveals:

1. In both cases, concerns about human health effects have put these cases on the agenda of risk managers. However, at least in the case of the fuel additives, we know that economic consequences (increase in fuel prices) and societal questions (agricultural policy) are factors in the management decision. Therefore, the management of fuel additives cannot solely rely on environmental tools, but transformation of all information into monetary units is not necessary.

[15] Data available from the corresponding author.

2. In the case of WNV, immediate risk management was deemed necessary on a community level. In this case, decisionmakers are local and state government officials. The affected population is locally identifiable. However, if WNV spreads into larger regions of the western hemisphere, the problem may shift from a meso to a macro level. Both CRAoA and BCA fulfill these requirements, according to Fig. 4. The case of fuel additives is considered a macro-level problem because of the widespread use of MTBE. Decisionmakers are state and federal governments. Gasoline consumers, residents in MTBE-contaminated areas, farmers, and the petrochemical industry are most likely to be affected by the decision. According to Fig. 4, this makes the use of LCA to compare different fuel additives less straightforward, as one would expect.
3. In both cases, a number of nonexclusive management actions have been suggested. Therefore, we are not just interested in the best management actions, but in those that provide net benefits (BCA).
4. It is difficult to predict the distributional consequences without analysis of the countervailing risks. However, if fuel prices increase due to a more expensive additive, this may disproportionately affect lower-income groups.
5. None of the 10 types of risk can at this stage be excluded or considered to dominate the analysis. Since each cent/gallon increase in gasoline price increases the costs borne by consumers by $1–1.3 billion /year,[22] 25–260 induced life shortenings per year may be expected from the income effect alone.[16] Analyses in regard to WNV have yet to be explored in this context.

From this first overview, it appears that both BCA and CRAoA/RTA can play an important role in the analysis of these cases. Although upstream and downstream effects need to be analyzed at least in the case of fuel additives, LCA may need to be combined with general equilibrium models and elements of risk assessment to adequately deal with the implied life cycle effects.

[16] This assumes a range of $5–40 million per induced life shortening and assumes that generated income by the new fuel additive will be distributed as equally as the reduced income among the population that suffers due to the reduced expenditures.

6. DISCUSSION AND CONCLUSIONS

We advocate both the application and continued refinement of tools for comparative analysis by highlighting the increasing importance of countervailing risks. We also note a recent increase in the practice of applying the available tools to questions beyond their initial design and the need for better guidance for decisionmakers, analysts, and risk managers. To evaluate the existing tools and provide better guidance in their use, we first introduced two contemporary environmental management problems and provided a more generic typology of potentially important countervailing risks—the ripples in the pond metaphor. When combined with the analysis of the toolbox from different angles, this provides the characteristics that actually distinguish the tools and the need to be transparent for the toolbox user. Indeed, we find that the tools we looked at have distinguishing characteristics and little overlap if applied in their initial setting. However, we also find that current management problems probably have more relevant dimensions than any single tool alone is capable of analyzing. A matrix that illustrates the coverage of the tools in terms of analyzed types of risks (Table I) reveals gaps. Gaps that may be of major importance are behavioral changes and the consideration of innovation that may be captured—if at all—only by using sophisticated general equilibrium models. The fundamental limitation in expanding the analysis of risks upstream and downstream and to indirect and induced impacts is that while the sphere of considered processes is broadened and the numbers of affected individuals, and thus the overall consequences, are increased, the inherent uncertainties tend to increase even more, that is, tertiary effects may outweigh the primary, but we may not even be certain whether they are positive or negative. One has to assume *ceteris paribus* at some point, but at what point this assumption is made may critically affect the results of the assessment. Nevertheless, the overarching relevance of income effects due to reduced disposable income on a population level suggests that a health-health analysis become a common element of all comparative analyses if the disposable income of others than the decisionmaker is altered by the decision at hand.

It is not surprising that none of the available tools provide all the information that the holistic management of today's problems may need. Tools for analysis tend to be developed within disciplines and, consequently, focus on a limited number of aspects using a limited set of available techniques. Also,

an all-encompassing tool would be very expensive and time consuming to apply. However, it is surprising that guides to both selecting the relevant tools and identifying the types of risks that are most important in a specific case have been developed only recently.[10,13,125] Analysts should devote considerable effort and time to considerations of framing, definition, and boundaries of analysis before launching into detailed numerical analysis.

Our analysis is limited in several respects. We elaborated on the family of instruments, tools, and techniques of comparative analysis and concentrated on analytical rather than policy and implementation questions. We acknowledge that such a focus will allow only limited conclusions because we neglect criteria that rely on the interplay with other policy tools[7] and on the possibility of implementing them in regulation and management schemes.[4] Therefore, the discussed tools are not a comprehensive set of environmental decision-support tools and no attention has been paid to their position within a decision-making process (deliberative process, stakeholder involvement) nor to the procedural aspects in implementing them. Neither was the analysis focused on providing the quantitative background that would allow a screening of potentially important types of risks.

There are common needs for further tool developments. In all subfields of environmental sciences we use experiments in laboratories where conditions are controlled but the transferability to the real world limited, experiments in the field where fewer experimental conditions can be controlled but the transferability of observations is increased, and monitoring and observation of the real world to find relations between different factors. All these settings generate information on the relationships between dependent and independent factors, quantify their transfer function, and may or may not provide information on causality. Thus, information is the true bottleneck of all analyses that deal with environmental effects of management actions. Therefore, it is important that tools have interfaces to all types of techniques that increase information on the magnitude and causality of relations and allow weight of evidence considerations. A second major common research need is that all tools imply, one way or another, that we are able to make predictions about the future. Scenario developments and forecasting, possibly based on general equilibrium models, is therefore a common element to most tools and deserves special attention in future research activities.

When former U.S. EPA Administrator C. M. Browner stated recently:[17] "my goal was to protect public health and the environment by ensuring that Americans have both cleaner air and cleaner water—and never one at the expense of the other," she reflected the continued and increasing awareness of the complex and interrelated nature of different environmental exposures and risks. Our analysis reveals that none of the available tools is able to provide in isolation the necessary information to support these kinds of decisions. The combined use of data and observations from different disciplines and their integration in a combination of analytical approaches and tools that is tailored to the question at stake will facilitate identifying and comparing the different risks and benefits of environmental risk management policies.

ACKNOWLEDGMENTS

Patrick Hofstetter was supported, in part, by an appointment to the Postdoctoral Research Program at the National Risk Management Research Laboratory administered by the Oak Ridge Institute for Science and Education (ORISE) through an interagency agreement between the U.S. Department of Energy and the U.S. Environmental Protection Agency. This article may or may not reflect the views of the supporting agencies. We thank the reviewers and John Evans (Harvard School of Public Health, Boston) for useful suggestions.

REFERENCES

1. CDC. (1999). Achievements in public health, 1900–1999: Control of Infectious Diseases. *MMWR*, *48*(29), 621–629.
2. U.S. EPA. (1990). *Pollution Prevention Act of 1990*. EPA/742/L-90/100. Available at http://www.epa.gov/opptintr/library/ppicdist.htm#12.
3. U.S. EPA. (1994). *RCRA waste minimization national plan*. EPA530-D-94-001. Available at http://www.epa.gov.epaoswer/hazwaste/minimixe/docs.thm.
4. ISO. (1997). *Environmental management—Life cycle assessment—Principles and guidelines*. EN ISO 14040. Brussels, Belgium: ISO.
5. Baumol, W. J., & Oates, W. E. (1988). *The theory of environmental policy*. Cambridge, UK: Cambridge University Press.
6. U.S. EPA. (2001). *The United States experience with economic incentives for protecting the environment*. EPA-240-R-01-001. Available at http://www.epa.gov/economics/.
7. Office of Technology Assessment (OTA). (1995). *Environmental policy tools: A user's guide*. OTA-ENV-634. Washington, DC: U.S. Government Printing Office.

17 Statement by Carol M. Browner, U.S. EPA Administrator, in "Findings by EPA's Blue Ribbon MTBE Panel," July 26, 1999. Availabe at http://www.epa.gov/otaq/consumer/fuels/oxypanel/browner/pdf.

8. Hammitt, J. K. (2000). Analytic methods for environmental regulation in the United States: Promises, pitfalls, and politics. *International Journal of Risk Assessment and Management*, *1*(1/2), 105–124.
9. Power, M., & McCarthy, L. S. (1998). A comparative analysis of environmental risk assessment/risk management frameworks. *Environmental Science and Technology*, *May 1*, 224A–231A.
10. Pearce, D., Hett, T., Ozdemiroglu, E., & Howarth, A. (1999). *Review of technical guidance on environmental appraisal*. U.K. Department of the Environment, Transport and the Regions, EFTEC. Available at http://www.environment.detr.gov.uk/rtgea/index.htm.
11. Huppes, G., & Simonis, U. E. (2000). *Environmental policy instruments in a new era*. CML-SSP Working Paper 2000.002. Leiden, the Netherlands: Centre of Environmental Science. Available at http://www.leidenuniv.nl/interfac/cml/ssp/publications/index.html.
12. Lafleur, M., Kortman, J., & van Berkel, R. (1998). *The development of a toolbox for the practical application of industrial ecology in companies*. Available at http://www.ivambv.uva.nl/ivam/produce/about.html.
13. Wrisberg, N., Udo de Haes, H. A., Triebswetter, U., Eder, P., & Clift, R. (Eds.). (2000). *Analytical tools for environmental design and management in a systems perspective*. Leiden, the Netherlands: Centre of Environmental Science.
14. Udo de Haes, H. A., Heijungs, R., Huppes, G., van der Voet, E., & Herrelingh, J.-P. (2000). Full mode and attribution mode in environmental analysis. *Journal of Industrial Ecology*, *4*(1), 45–56.
15. Ponce, R. A., Bartell, S. M., Wong, E. Y., LaFlamme, D., Carrington, C., Lee, R. C., Patrick, D. L., Faustman, E. M., & Bolger, M. (2000). Use of quality-adjusted life year weights with dose-response models for public health decisions: A case study of the risks and benefits of fish consumption. *Risk Analysis*, *20*(4), 529–542.
16. U.S. EPA. (1998). *Comparative risk framework; Methodology and case study*. NCEA-C-0135, SAB External Review Draft. Available at http://www.epa.gov/ncea/frame.htm.
17. VROM. (Ed.). (1994). *Policy document on products and the environment*. The Hague: Ministry of Housing, Spatial Planning and the Environment (VROM).
18. Troge, A. (2000). Life cycle assessment and product-related environmental policy; The position of the Federal Environmental Agency (Berlin, Germany). *International Journal of LCA*, *5*(4), 195–200.
19. American Chemistry Council (ACC). (2000). *Emmitsburg plan for fostering the rational integration of human and environmental considerations into public and private decision-making processes*. Available at www.fivewinds.com.
20. Nadim, F., Zack, P., Hoag, G. E., & Liu, S. (2001). United States experience with gasoline additives. *Energy Policy*, *29*, 1–5.
21. U.S. EPA. (1992). *Alternative fuels research strategy*. EPA 600/AP-92/002. Washington, DC: Office of Research and Development.
22. U.S. EPA. (1999). *Achieving clean air and clean water. The report of the Blue Ribbon Panel on Oxygenates in Gasoline. September 15, 1999*. EPA420-R-99-021. Washington, DC: U.S. EPA.
23. Graham, J. D., & Wiener, J. B. (1995). *Risk versus risk: Tradeoffs in protecting health and the environment*. Cambridge/London: Harvard University Press.
24. Porter, M. E. (1991). America's green strategy. *Scientific American*, *April*, 168.
25. Porter, M. E., & van der Linde, C. (1995). Toward a new conception of the environment-competitiveness relationship. *Journal of Economic Perspectives*, *9*, 97–118.
26. Jaffe, A. B., Peterson, S. R., Portney, P. R., & Stavins, R. N. (1995). Environmental regulation and the competitiveness of U.S. manufacturing: What does the evidence tell us? *Journal of Economic Literature*, *33*, 132–163.
27. Palmer, K., Oates, W. E., & Portney, P. R. (1995). Tightening environmental standards: The benefit-cost or the no-cost paradigm? *Journal of Economic Perspectives*, *9*, 119–132.
28. Smith, V. K., & Walsh, R. (2000). Do painless environmental policies exist? *Journal of Risk and Uncertainty*, *21*(1), 73–94.
29. Wildavsky, A. (1980). Richer is safer. *Public Interest*, *60*, 23–39.
30. Wilde, G. J. S. (1982). The theory of risk homeostasis: Implications for safety and health. *Risk Analysis*, *4*, 209–225, 227–258.
31. Peltzman, S. (1975). The effects of automobile safety regulation. *Journal of Political Economy*, *83*, 677.
32. Adams, J. (1995). *Risk*. London: UCL Press.
33. International Federation of Institutes for Advanced Study (IFIAS). (1974). *Energy analysis workshop on methodology and conventions*. Sweden: Guldsmedshyttan.
34. Hunt, R. G. *et al*. (1974). *Resource and environmental profile analysis of nine beverage container alternatives*. Washington, DC: Midwest Research Institute for U.S. Environmental Protection Agency.
35. Basler & Hofmann Ingenieure und Planer. (1974). *Studie Umwelt und Volkswirtschaft, Vergleich der Umweltbelastung von Behältern aus PVC, Glas, Blech und Karton*. Bern: Eidgenössisches Amt für Umweltschutz.
36. SETAC. (1993). Guidelines for life-cycle assessment: A "code of practice." Workshop Sesimbra, March 3–March 4, 1993. Brussels, Belgium.
37. Udo de Haes, H. A. (Ed.). (1996). *Towards a methodology for life cycle impact assessment*. Brussels, Belgium: SETAC-Europe Working Group on Impact Assessment.
38. Heijungs, R., Guinée, J. B., Huppes, G., Lankreijer, R. M., Udo de Haes, H. A., Wegener Sleeswijk, A., Ansems, A. M. M., Eggels, P. G., van Diun, R., & de Goede, H. P. (1992). *Environmental life cycle assessment of products; Backgrounds & guide*. Leiden the Netherlands: Centre of Environmental Science (CML).
39. Goedkoop, M., & Spriensma, R. (1999). *The eco-indicator '99: A damage oriented method for life cycle impact assessment*. No. 1999/36A/B. Zoetermeer, the Netherlands: VROM. Available at http://www.pre.nl/eco-indicator99/ei99-reports.htm.
40. Steen, B. (1999). *A systematic approach to environmental priority strategies in product development (EPS)* (Version 2000). CPM Report 1999:4 and 5. Göteborg: Chalmers University.
41. Potting, J., & Blok, K. (1994). Spatial aspects of life-cycle impact assessment. In *Integrating impact assessment into LCA*. Proceedings of the 4th SETAC-Europe Congress, April 11–14, 1994. Brussels, Belgium.
42. Potting, J., & Hauschild, M. (1997). Predicted environmental impact and expected occurrence of actual environmental impact part II: Spatial differentiation in life-cycle assessment via the site-dependent characterisation of environmental impact from emissions. *International Journal of LCA*, *2*(4), 209–216.
43. Potting, J., Schöpp, W., Blok, K., & Hauschild, M. (1998). Site-dependent life-cycle impact assessment of acidification. *Journal of Industrial Ecology*, *2*(2), 63–87.
44. Potting, J. (2000). Spatial differentiation in life cycle impact assessment; A framework, and site-dependent factors to assess acidification and human exposures. Ph.D thesis. Utrecht, the Netherlands: University Utrecht.
45. Giegrich, J. (1994). Modelling spatial information for impact assessment. In P. Hofstetter, S. Schaltegger, & H. A. Udo de Haes (Eds.), *First working document on life cycle impact assessment methodology*. Zürich: ETH.

46. Herrchen, M. (1998). Perspective of the systematic and extended use of temporal and spatial aspects in LCA of long-lived products. *Chemosphere*, *37*(2), 265–270.
47. Spadaro, J. V., & Rabl, A. (1999). Estimates of real damage from air pollution: Site dependence and simple impact indices for LCA. *International Journal of LCA*, *4*(4), 229–243.
48. Krewitt, W., Trukenmüller, A., & Friedrich, R. (1999). Site-dependent LCIA impact indicators for human health from integrated air quality and exposure modeling. Poster at the 9th annual meeting of SETAC-Europe, May 25–29, 1999. Leipzig.
49. Nigge, K.-M. (2000). *Life cycle assessment of natural gas vehicles. Development and application of site-dependent impact indicators*. Berlin: Springer Verlag.
50. Hogan, L. M., Beal, R. T., & Hunt, R. G. (1996). Threshold inventory interpretation methodology: A case study of three juice container systems. *International Journal of LCA*, *1*(3), 159–167.
51. Assies, J. A. (1998). A risk-based approach to life-cycle impact assessment. *Journal of Hazardous Materials*, *61*, 23–29.
52. Hillier, W., Murphy, R. J., Dickinson, D. J., & Bell, J. N. B. (1997). *The risk of life cycle impact assessment for preservative treated timber products*. WEI Document 2601. Oslo: WEI-Congrerss.
53. Owens, J. W. (1997). Life-cycle assessment in relation to risk assessment: An evolving perspective. *Risk Analysis*, *17*(3), 359–365.
54. Moll, H. C., & Schoot Uiterkamp, A. J. M. (1997). Comparative evaluation by lifecycle and risk assessment of agribiological and technological routes of production. *Industrial Crops and Products*, *6*, 333–341.
55. U.S. EPA, Office of Policy, Planning and Evaluation, Office of Policy Analysis. (1987). *Unfinished business: A comparative assessment of environmental problems*. Washington, DC: U.S. EPA.
56. U.S. EPA, Office of Policy, Planning, and Evaluation. (1989). *Comparing risks and setting environmental priorities: Overview of three regional projects*. Washington, DC: U.S. EPA.
57. U.S. EPA, Science Advisory Board. (1990). *Reducing risk: Setting priorities and strategies for environmental protection*. Washington, DC: U.S. EPA.
58. U.S. EPA. (1993). *A guidebook to comparing risks and setting environmental priorities (The road map)*. 230-B-93-003. Washington, DC: U.S. EPA.
59. Finkel, A. M., & Golding, D. (Eds.). (1994). *Worst things first? The debate over risk-based national environmental priorities*. Washington, DC: Resources for the Future.
60. Feldman, D. L., Hanahan, R. A., & Perhac, R. (1999). Environmental priority-setting through comparative risk assessment. *Environmental Management*, *23*(4), 483–493.
61. Jones, K., & Klein, H. (1999). Lessons from 12 years of comparative risk projects. *Annual Review of Public Health*, *20*, 159–172.
62. Morgenstern, R. D., Shih, J.-S., & Sessions, L. (2000). Comparative risk assessment: An international comparison of methodologies and results. *Journal of Hazardous Materials*, *78*, 19–39.
63. Konisky, D. M. (1999). *Comparative risk projects: A methodology for cross-project analysis of human health risk rankings*. Discussion Paper 9946. Washington, DC: Resources for the Future. Available at http://www.rff.org.
64. Graham, J. D., & Hammitt, J. K. (1996). Refining the CRA framework. In J. C. Davies (Ed.), *Comparing environmental risks; Tools for setting governmental priorities* (pp. 93–109). Washington, DC: Resources for the Future.
65. U.S. Atomic Energy Commission, Office of Energy Systems Analysis. (1974). *Comparative risk-cost-benefit study of alternative sources of electrical energy*. WASH-1224. Washington, DC: U.S. Atomic Energy Commission.
66. Harris, R. H., Highland, J. H., *et al*. (1984). Comparative risk assessment: Tools for remedial action planning. *Hazardous Waste*, *1*(1), 19–33.
67. Lawless, E. W., Jones, M. V., & Jones, R. M. (1984). *Comparative risk assessment: Towards an analytical framework*. Final report. Kansas City, MO: Midwest Research Institute; Rockville, MO: Impact Assessment Institute.
68. Toxicology Excellence for Risk Assessment (TERA). (1999). *Comparative dietary risks: Balancing the risks and benefits of fish consumption*. Available at http://www.tera.org/news/.
69. Havelaar, A. H., de Hollander, A. E. M., Teunis, P. F. M., Evers, E. G., Van Kranen, H. J., Versteegh, F. M., van Koten, J. E. M., & Slob, W. (2000). Balancing the risks and benefits of drinking water disinfection: Disability adjusted life-years on the scale. *Environmental Health Perspectives*, *108*(4), 315–321.
70. National Round Table on the Environment and the Economy (NRTEE). (1999). *Reducing sulphur in gasoline and diesel fuel—Case study*. Ottawa, Canada. Available at http://www.nrtee-trnee.ca/eng/programs/health/sulphur-12-99.pdf.
71. Suter II, G. W., Cornaby, B. W., Hadden, C. T., Hull, R. N., Stack, M., & Zafran, F. A. (1995). An approach for balancing health and ecological risks at hazardous waste sites. *Risk Analysis*, *15*(2), 221–231.
72. Cirone, P. A., & Duncan, P. B. (2000). Integrating human health and ecological concerns in risk assessments. *Journal of Hazardous Materials*, *78*, 1–17.
73. Suter II, G. W. (2000). Generic assessment endpoints are needed for ecological risk assessment. *Risk Analysis*, *20*(2), 173–178.
74. Posthuma, L., Suter II, G. W., & Traas, T. P. (Eds.). (Forthcoming). The use of species sensitivity distributions in ecotoxicology.
75. Lave, L. B. (1981). *The strategy of social regulation*. Washington, DC: Brookings Institution.
76. Viscusi, W. K. (1994). Risk-risk analysis. *Journal of Risk and Uncertainty*, *8*, 5–17.
77. Fischhoff, B., Slovic, P., Lichtenstein, S., Read, S., & Combs, B. (1978). How safe is safe enough? A psychometric study of attitudes towards technological risks and benefits. *Policy Sciences*, *9*, 127–152.
78. Cohen, J. T., Beck, B. D., & Rudel, R. (1997). Life years lost at hazardous waste sites: Remediation worker fatalities vs. cancer deaths to nearby residents. *Risk Analysis*, *17*(4), 419–425.
79. Hammitt, J. K., Belsky, E. S., Levy, J. I., & Graham, J. D. (1999). Residential building codes, affordability, and health protection: A risk-tradeoff approach. *Risk Analysis*, *19*(6), 1037–1058.
80. Gray, G. M., & Hammitt, J. K. (2000). Risk/risk trade-offs in pesticide regulation: An exploratory analysis of the public health effects of a ban on organophosphate and carbamate pesticides. *Risk Analysis*, *20*(5), 665–680.
81. Pliskin, J. S., Shepard, D. S., & Weinstein, M. C. (1980). Utility functions for life years and health status. *Operational Research*, *28*(1), 206–224.
82. Hofstetter, P., & Hammitt, J. K. (2002). Selecting human health metrics for environmental decision-support tools. *Risk Analysis*, *22*(5), 963–981.
83. Keeney, R. L. (1990). Mortality risks induced by economic expenditures. *Risk Analysis*, *10*, 147–159.
84. Lutter, R., & Morall III, J. F. (1994). Health-health analysis: A new way to evaluate health and safety regulation. *Journal of Risk and Uncertainty*, *8*, 43–66.
85. Cordato, R. E. (1999). *Global warming, Kyoto, and tradeable emissions permits; The myth of efficient central planning*. Institute for Research on the Economics of Taxation. Available at http://www.campbell.edu/lundy/IRET/Cordato.1.htm.

86. Gellhorn, E. (2000). American Trucking Assn's Inc. et al., cross-petitioners, versus Browner, EPA Administrator, respondent. *USA Today*, *July 21*. Available at http://supreme.usatoday.findlaw.com/supreme_court/briefs/99-1426/99-1426fo6/text.html.
87. Keeney, R. L. (1994). Mortality risks induced by the costs of regulations. *Journal of Risk and Uncertainty*, *8*, 95–110.
88. Keeney, R. L. (1997). Estimating fatalities induced by the economic costs of regulations. *Journal of Risk and Uncertainty*, *14*, 5–23.
89. Chapman, K. S., & Hariharan, G. (1994). Controlling for causality in the link from income to mortality. *Journal of Risk and Uncertainty*, *8*, 85–94.
90. Chapman, K. S., & Hariharan, G. (1996). Do poor people have a stronger relationship between income and mortality than the rich? Implications of panel data for health-health analysis. *Journal of Risk and Uncertainty*, *12*, 51–63.
91. Lutter, R., Morrall, J. F., & Viscusi, W. K. (1999). The cost-per-life-saved cutoff for safety-enhancing regulations. *Economic Inquiry*, *37*(4), 599–608.
92. Viscusi, W. K. (1994). Mortality effects of regulatory costs and policy evaluation criteria. *RAND Journal of Economics*, *25*, 94–109.
93. Smith, J. P. (1999). Healthy bodies and thick wallets: The dual relation between health and economic status. *Journal of Economic Perspectives*, *13*, 145–166.
94. Portney, P. R., & Stavins, R. N. (1994). Regulatory review of environmental policy: The potential role of health-health analysis. *Journal of Risk and Uncertainty*, *8*, 111.
95. U.S. EPA. (1999). Valuation of human health and welfare effects of criteria pollutants. In *The benefits and costs of the Clean Air Act 1990–2010* (Appendix H). EPA Report to Congress, EPA-410-R-99-001. Available at http://www.epa.gov/oar/oaq_caa.html.
96. Tengs, T. O., Adams, M. E., Pliskin, J. S., Safran, D. G., Siegel, J. E., Weinstein, M. C., & Graham, J. D. (1995). Five-hundred life-saving interventions and their cost-effectiveness. *Risk Analysis*, *15*(3), 369–389.
97. Viscusi, W. K., & Zeckhauser, R. J. (1994). The fatality and injury costs of expenditures. *Journal of Risk and Uncertainty*, *8*, 19–41.
98. Wilkinson, R. (1996). *Unhealthy societies; The afflictions of inequality*. London/New York: Routledge.
99. Wagstaff, A., & van Doorslaer, E. (2000). Income inequality and health: What does the literature tell us? *Annual Review of Public Health*, *21*, 543–567.
100. Rowe, R., Lang, C., Chestnut, L., Latimer, D., Rae, D., Bernow, S., & White, D. (1995). *The New York electricity externality study*. Dobbs Ferry, NY: Oceana Publications.
101. ExternE. (1999). *Externalities of energy* (Vols. 7–10). EUR 19083 EN. Luxembourg: European Union.
102. Seethaler, R. (1999). *Health costs due to road traffic-related air pollution; An impact assessment project of Austria, France and Switzerland*. Synthesis report. Prepared for the WHO Ministerial Conference on Environment and Health, London. GVF-Bericht 1/99. Bern, Switzerland: Federal Department of Environment, Transport, Energy and Communications.
103. Rabl, A., & Spadaro, J. V. (2000) Health costs of automobile pollution. *Rev. fr. Allergol*, *40*(1), 55–59.
104. Freeman III, A. M. (1993). *The measurement of environmental and resource values: Theory and methods*. Washington, DC: Resources for the Future.
105. Carson, R. T. (2000). Contingent valuation: A users guide. *Environmental Science and Technology*, *34*, 1413–1418.
106. Dasgupta, P., Marglin, S., & Sen. A. (1972). *Guidelines for project evaluation*. Vienna: UNIDO.
107. Squire, L., & van der Tak, H. (1975) *Economic analysis of projects*. Baltimore, MD: John Hopkins University Press.
108. Gold, M. R., Siegel, J. E., Russel, L. B., & Weinstein, M. C. (1996). *Cost effectiveness in health and medicine*. New York: Oxford University Press.
109. CDC. (1992). A framework for assessing the effectiveness of disease and injury prevention. *MMWR*, *41*(RR-3). Available at http://www.cdc.gov/mmwr/preview/mmwrhtml/00016403.htm.
110. Haddix, A. C., Teutsch, S. M., & Shaffer, P. A. (1996). *Prevention effectiveness: A guide to decision analysis and economic evaluation*. New York: Oxford University Press.
111. Hammitt, J. K. (This issue). QALYs versus WTP. *Risk Analysis*.
112. Graham, J. D., Corso, P. S., Morris, J. M., Segui-Gomez, M., & Weinstein, M. C. (1998). Evaluating the cost-effectiveness of clinical and public health measures. *Annual Review of Public Health*, *19*, 125–152.
113. Andersson, A.-B., & Carlson, H. (1995). The basis for a method to integrate work environment in life cycle assessment. *Journal of Cleaner Production*, *3*(4), 215–220.
114. Potting, J., Møller, B. T., & Jensen, A. A. (1997). Work environment and LCA. In H. A. Udo de Haes & N. Wrisberg (Eds.), *Life cycle assessment: State-of-the-art and research priorities* (LCA Documents Volume 1, pp. 149–180). Bayreuth: Eco-Informa Press.
115. Schmidt, A., Rasmussen, P. B., Andreasen, J., Fløe, T., & Poulsen, K. E. (In press). Integration of the working environment in life cycle assessment (LCA)—A new methodology. Copenhagen: Danish Environmental Protection Agency.
116. Udo de Haes, H. A., Jolliet, O., Finnveden, G., Hauschild, M., Krewitt, W., & Müller-Wenk, R. (Eds.). (1999). Best available practice regarding impact categories and category indicators in life cycle impact assessment. Background document for the second working group on life cycle impact assessment of SETAC-Europe (WIA-2). *International Journal of LCA*, *4*(2), 66–74; *4*(3), 167–174.
117. Heijungs, R. (1997). *Economic drama and the environmental stage: Formal derivation of algorithmic tools for environmental analysis and decision-support from a unified epistemological principle*. Leiden, the Netherlands: Centre of Environmental Science.
118. Bouman, M., Heijungs, R., van der Voet, E., van den Bergh, J. C. J. M., & Huppes, G. (1999). *Materials flows and economic models: An analytical comparison of SFA, LCA and equilibrium models*. CML-SSP Working Paper 99.001. Leiden, the Netherlands: Centre of Environmental Science, Leiden University.
119. Houghton, J. T., Meira Filho, L. G., Callander, B. A., Harris, N., Kattenberg, A., & Maskell, K. (Eds.). (1996). *Climate change 1995; The science of climate change*. Cambridge: Cambridge University Press.
120. Hofstetter, P. (1998). *Perspectives in life cycle impact assessment; A structured approach to combine models of the technosphere, ecosphere, and valuesphere*. Boston, MA: Kluwer Academic Publishers.
121. Greening, L. A., Greene, D. L., & Difiglio, C. (2000). Energy efficiency and consumption—The rebound effect—A survey. *Energy Policy*, *28*, 389–401.
122. Binswanger, M. (2001). Technological progress and sustainable development: What about the rebound effect? *Ecological Economics*, *36*, 119–132.
123. Hazilla, M., & Kopp, R. J. (1990). Social cost of environmental quality regulations: A general equilibrium analysis. *Journal of Political Economy*, *98*, 853–873.
124. Jorgenson, D. W., & Wilcoxen, P. J. (1993) Reducing U.S. carbon dioxide emissions: An econometric general equilibrium assessment. *Resource & Energy Economics*, *15*(1), 7–25.
125. Scholz, R. W., & Tietje, O. (2001). *Embedded case study methods: Integrating quantitative and qualitative knowledge*. Thousand Oaks, CA: Sage.