TRACI

The Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts

Jane C. Bare, Gregory A. Norris, David W. Pennington, and Thomas McKone

Keywords

environmental impact modeling
life-cycle assessment (LCA)
life-cycle impact assessment (LCIA)
midpoint
site-dependent impact assessment
software

e-supplement available on the *JIE* Web site

Summary

The tool for the reduction and assessment of chemical and other environmental impacts (TRACI) is described along with its history, the research and methodologies it incorporates, and the insights it provides within individual impact categories.

TRACI, a stand-alone computer program developed by the U.S. Environmental Protection Agency, facilitates the characterization of environmental stressors that have potential effects, including ozone depletion, global warming, acidification, eutrophication, tropospheric ozone (smog) formation, ecotoxicity, human health criteria–related effects, human health cancer effects, human health noncancer effects, fossil fuel depletion, and land-use effects. TRACI was originally designed for use with life-cycle assessment (LCA), but it is expected to find wider application in the future.

To develop TRACI, impact categories were selected, available methodologies were reviewed, and categories were prioritized for further research. Impact categories were characterized at the midpoint level for reasons including a higher level of societal consensus concerning the certainties of modeling at this point in the cause-effect chain. Research in the impact categories of acidification, smog formation, eutrophication, land use, human cancer, human noncancer, and human criteria pollutants was conducted to construct methodologies for representing potential effects in the United States. Probabilistic analyses allowed the determination of an appropriate level of sophistication and spatial resolution necessary for impact modeling for each category, yet the tool was designed to accommodate current variation in practice (e.g., site-specific information is often not available). The methodologies underlying TRACI reflect state-of-the-art developments and best-available practice for life-cycle impact assessment (LCIA) in the United States and are the focus of this article. TRACI's use and the impact of regionalization are illustrated with the example of concrete production in the northeastern United States.

Address correspondence to:
Jane C. Bare
U.S. EPA (MS-466)
26 W. MLK Dr.
Cincinnati, OH 45268
bare.jane@epa.gov

Volume 6, Number 3–4

Introduction

Life-cycle assessment (LCA), a method for the systematic environmental assessment of facility, process, service, and product choices from raw material extraction through manufacturing and use to end-of-life management, has grown considerably in prominence and sophistication over the last decade. As interest in LCA has increased, there has been increasing discussion of the sophistication, accuracy, and complexity in life-cycle impact assessment (LCIA), including the consideration of temporal and spatial dimensions of potential impacts (ISO 14042) (Udo de Haes et al. 2002; Bare et al. 1999, 2000; Udo de Haes et al. 1999a, 1999b; Owens 1997). An ideal LCA without budget or time constraints would use the highest quality data to incorporate all disaggregated impact categories, all stressors, and all life-cycle stages. Unfortunately, completing comprehensive assessments for all potential effects at a high level of simulation sophistication and disaggregation would require impossibly large amounts of time, data, knowledge, and resources. It therefore follows that every study must be limited in some aspect of sophistication and/or comprehensiveness.

LCIA methods must be usable with existing and foreseeable life-cycle inventory (LCI) data. LCIs contain information about the materials extracted from and released to the environment over entire product life cycles, including manufacturing supply chains, the use phase, and end-of-life processes. These inventories often lack spatial information. LCIA methods that can accommodate the absence of data from some of the many hundreds or even thousands of sites contributing to the LCI totals are required to generate results that are as meaningful as possible. At the same time, where the use of additional source information could significantly reduce the uncertainty of LCIA results, the LCIA methods should be able to make use of (and thus encourage future tracking of) such information.

Researchers have been discussing the most appropriate number and representation of impact categories, as well as the best available methodologies, in various public and literature forums (Udo de Haes et al. 1999; Bare et al. 1999, 2000). Hofstetter (1999) pointed out that many of the early impact categories appeared to emerge from existing environmental regulations, perhaps relying on the assumption that the most significant environmental problems would have a corresponding regulation. Later methodologies included stressors and impact categories that were not included within environmental regulations but were assumed to be of interest to society. At this point in time, there is no single worldwide consensus on either the list of impact categories for inclusion or the associated methodologies for use in LCIA; instead, many methodologies for LCIA exist. The evaluation of the appropriate level of method sophistication and comprehensiveness in individual studies depends upon the application, goal, and scope of the study and is ultimately the responsibility of the person or organization sponsoring the study and those that conduct it.

TRACI Development

In 1995, while conducting several LCA case studies, the U.S. Environmental Protection Agency (U.S. EPA) sought to find the best impact assessment tool for LCIA, pollution prevention, and sustainability metrics for the United States. A literature survey was conducted to ascertain the applicability, sophistication, and comprehensiveness of all existing methodologies. One of the most popular methodologies being used in the United States was developed by the Dutch for stressors released in Europe (Heijungs et al. 1992). When the development of a software tool began, nearly all U.S. practitioners used these chemical potencies when conducting LCIAs for U.S. conditions simply because similar simulations had not been conducted within the United States. Because it was apparent that no tool existed that would allow the sophistication, comprehensiveness, and applicability to the United States that was desired, the U.S. EPA decided to begin development of software to conduct impact assessment with the best applicable methodologies within each category. The result was the tool for the reduction and assessment of chemical and other environmental impacts (TRACI).

Because the U.S. EPA decided to make TRACI widely available,[1] it was important that it be simple and small enough to run on a personal computer (PC). This provided some con-

straints because advanced features such as geographical information system spatial linking and the inclusion of uncertainty modeling, such as Monte Carlo analysis for propagation of errors, could have exceeded the memory of many PCs and may have significantly complicated the use of TRACI. Although future versions of TRACI may allow for the inclusion of such features, this version is designed for simplicity and therefore does not facilitate a quantification of propagated uncertainty.

The first step in developing this tool was to select the impact categories for analysis and methodology development. An updated literature search revealed several researchers that have discussed this issue, including Heijungs and colleagues (1992), Udo de Haes (1996), and Guinée and colleagues (1996). Impact categories are generally of two types: (1) the depletion categories, which include abiotic resource depletion, biotic resource depletion, land use, and water use, and (2) the pollution categories, which include ozone depletion, global warming, human toxicology, eco-toxicology, smog formation, acidification, eutrophication, odor, noise, radiation, and waste heat. The applicability of many of these impact categories in such a broad-reaching tool as LCA has been discussed widely (e.g., applying generic methodologies to unknown locations for impact categories such as odor and noise can be difficult). Indeed, it was recognized by the U.S. EPA that the selection of these impact categories is a normative decision depending on what is valued; it is also apparent that these impact categories can be taken to further points along the cause-effect chain or can be subdivided or aggregated. Additional efforts to develop a global listing comprising all impact categories and acceptable methodologies continue (UNEP-SETAC 2000). In the absence of such a global consensus, the selection of the impact categories is left as one part of the goal and scope of each individual case study or is left to the discretion of the tool designer.

For the development of TRACI, each of the above impact categories was considered and its current state of development and perceived societal value were assessed. The traditional pollution categories of ozone depletion, global warming, human toxicology, eco-toxicology, smog formation, acidification, and eutrophication were included within TRACI because various programs and regulations within the U.S. EPA recognize the value of minimizing effects from these categories. The category of human health was further subdivided into cancer, noncancer, and criteria pollutants[2] (with an initial focus on particulates) to better reflect the focus of U.S. EPA regulations and to allow methodology development consistent with U.S. regulations, handbooks, and guidelines (e.g., the *EPA Risk Assessment Guidance for Superfund*, U.S. EPA 1989b and the *Exposure Factors Handbook*, U.S. EPA 1989a). Smog-formation effects were kept independent and not further aggregated with other human health impacts because environmental effects related to smog formation would have become masked and/or lost in the process of aggregation. Criteria pollutants were maintained as a separate human health impact category, allowing a modeling approach that can take advantage of the extensive epidemiological data associated with these well-studied impacts.

The categories of odor, noise, radiation, waste heat, and accidents are outside of the U.S. EPA's purview and are usually not included within case studies in the United States for various reasons, including, perhaps, because the perceived threat from these categories is often considered minimal, local, or difficult to predict. The resource depletion categories are recognized as being of significance in the United States, especially for fossil fuel, land, and water use. Therefore, the categories selected at this time include the following:

- Ozone depletion
- Global warming
- Smog formation
- Acidification
- Eutrophication
- Human health cancer
- Human health noncancer
- Human health criteria pollutants
- Eco-toxicity
- Fossil fuel depletion
- Land use
- Water use

It should be noted, however, that the impact categories selected for inclusion within TRACI are considered a minimal set that may be expanded in future versions.

The TRACI Framework

TRACI is a stand-alone application that can run on computers with Windows 95, Windows 98, or Windows NT. Because TRACI uses a runtime version of Microsoft Access, it runs on computers whether or not they include a version of Access.

As shown in figure 1, the TRACI software utility allows the storage of inventory data, classification of stressors, and characterization for the listed impact categories. Inventory data are stored as either inputs or outputs on specific process levels within the various life-cycle stages. Studies can be conducted on products or processes and compared to alternatives.

TRACI's Modular Design

TRACI is a modular set of LCIA methods intended to provide the most up-to-date possible treatment of impact categories for the North American context. The methods have been made operational within a public-domain software tool available from the U.S. EPA. The software reads inventory data and applies the TRACI methods to provide results. These methods can also be used within other LCA tools, such as LCA modeling and decision support software. TRACI's modular design allows the compilation of the most sophisticated impact assessment methodologies that can be utilized in software developed for PCs. Where sophisticated and applicable methodologies did not exist, simulations were conducted to determine the most appropriate characterization factors for the various conditions within the United States. As the research, modeling, and databases for LCIA methods continue to improve, each module of TRACI can be improved and updated. A framework and software utility for quantitative uncertainty analysis in conjunction with the TRACI methodologies is under development. Future research is expected to advance methodologies for resource-related impact categories.

A framework incorporating normalization and valuation processes allows decision makers to determine their values for one situation and to maintain the use of these values for consistency in other environmental situations requiring similar trade-offs between impact categories. At the time of writing, the current best practice for normalization and valuation is still very much under debate (Hofstetter 1998; Finnveden et al. 1997; Volkwein and Klopffer 1996; Volkwein et al. 1996; Grahl and Schmincke 1996). Examples exist demonstrating the use of normalization in European situations with European normalization databases, whereas a comparable U.S. normalization database is not yet available. The valuation process was (and is) also in need of advancement. Participants in an international workshop on midpoints versus endpoints held in Brighton, England, in May 2000 (Bare et al. 2000) could not point out any examples of successful, unbiased panel procedures that could guide future valuation processes. Because of these issues, and because of possible misinterpretation and misuse, it was determined that the state of the art for the normalization and valuation processes did not yet support inclusion in TRACI.

TRACI Impact Assessment Methodology Development

A workshop was held in Brussels, Belgium, in 1998 to discuss, among experts worldwide, the degree of sophistication in LCIA in the conduct of case studies, research, and tool development, as well as in the development of ISO 14042 (Bare et al. 1999). Most researchers agreed that the most sophisticated methodologies should be used, as long as they did not require significant additional work in compiling the inventory and did not require significant additional modeling assumptions that were not already incorporated into regulations or societal standards. They recognized that this level of sophistication would vary across impact categories, with some of the categories (e.g., human toxicity) having had years of research and standardization, whereas other categories (e.g., land use) were still very much in their infancy, even in determining what is to be valued. The U.S. EPA decided to incorporate the most sophisticated methodology for each impact category in TRACI, minimizing new modeling assumptions. A modular approach was chosen to facilitate updates as new impact assessment research becomes available.

As an example, prior to the development of TRACI, many researchers used simple measures of toxicity (Heijungs et al. 1992) or scoring pro-

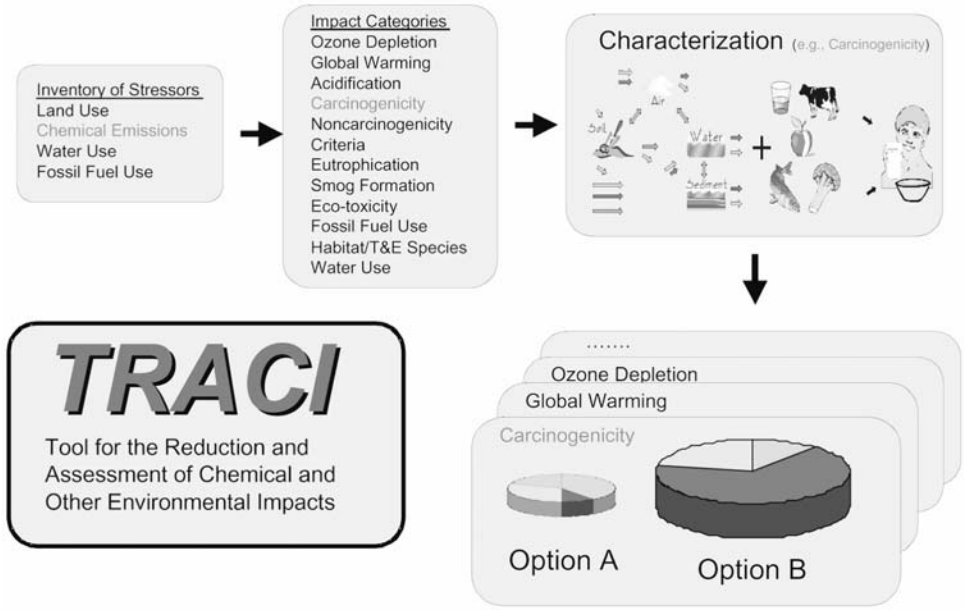


Figure 1 TRACI's framework, using carcinogenicity due to chemical emissions as an example.

cedures based on persistence, bioaccumulation, and toxicity (Swanson and Socha 1997) to provide indicators for human toxicity. These methods yielded proxy indicators that did not fully quantify the potential for effects, but simply provided a measure of a related parameter. By incorporating a sophisticated multimedia model followed by human exposure modeling, the current methodology within TRACI provides a more sophisticated output that is related to the potential impacts being quantified.

In addition to model sophistication, factors involved in methodology development included portability and ease of use, consistency with existing U.S. EPA regulatory guidance, and last, but importantly, minimization of assumptions and value choices. Significant emphasis was given to applicability to the United States.

Applicability to the United States is important for many of the categories that have location-specific input parameters. In those categories (e.g., human health cancer and noncancer) where sophisticated models allowed it, the location-related input parameters (e.g., meteorology and geology) were varied according to a sensitivity analysis structure. Within the human health cancer and noncancer categories, research suggested that uncertainties not related to location—for example, uncertainties in toxicity or cancer potency—exceeded regional sensitivities for the impact factors by several orders of magnitude. For this reason, U.S. average impact factors were developed for the human cancer and noncancer categories. For categories such as acidification and smog formation, detailed U.S. empirical models, such as those developed by the U.S. National Acid Precipitation Assessment Program (Shannon 1992) and the California Air Resources Board (Carter 2000), allowed the inclusion of more sophisticated location-specific approaches and characterization factors. In all impact categories, values are available for the United States where location (e.g., state or county level) information about the inventory data is otherwise lacking.

Consistency with previous modeling assumptions (especially those of the U.S. EPA) was important for every category. The human health cancer and noncancer categories were heavily based on the assumptions made for the U.S. EPA's *Risk Assessment Guidance for Superfund* (U.S. EPA 1989b). The U.S. EPA's *Exposure Factors Handbook* (U.S. EPA 1989a) was utilized to make decisions related to the various input parameters for both of these categories as well. Another example of consistency with U.S. EPA

modeling assumptions includes the use of the 100-year time frame reference for global warming potentials.

When there was no U.S. EPA precedent, assumptions and value choices were minimized in some cases by the use of midpoints. Although other modelers have extrapolated to endpoints in published LCIA methodologies (e.g., Goedkoop and Spriensma 1999), the U.S. EPA decided that TRACI should not conduct these calculations because several assumptions and value choices are necessary to extrapolate to the endpoints. Other advantages to the use of midpoints (such as comprehensiveness) are discussed in the next section.

TRACI's Midpoint Choices

Many of the impact assessment methodologies within TRACI are based on "midpoint" characterization approaches (Bare et al. 2000). The impact assessment models reflect the relative potency of the stressors at a common midpoint within the cause-effect chain; see the example in figure 2 for ozone depletion. This diagram shows that characterization could take place at the level of midpoints, such as ozone depletion potential, or endpoints, such as skin cancer, crop damage, immune-system suppression, damage to materials such as plastics, marine-life damage, and cataracts. Analysis at a midpoint minimizes the amount of forecasting and effect modeling incorporated into the LCIA, thereby reducing the complexity of the modeling and often simplifying communication. Another factor supporting the use of midpoint modeling is the incompleteness of model coverage for endpoint estimation. For example, as shown in figure 2, models and data may exist to allow a prediction of potential endpoint effects such as skin cancer and cataracts, but the inclusion of effects such as crop damage, immune-system suppression, damage to materials, and marine-life damage is less well supported. These associated endpoints and their expected effects remain important but are not often captured in certain endpoint analyses.

Within this article, under the individual discussions of impact categories, simple cause-effect chains are drawn showing the point at which each impact category is characterized. The indicators and associated level of site specificity adopted in the current version of TRACI are summarized in table 1.

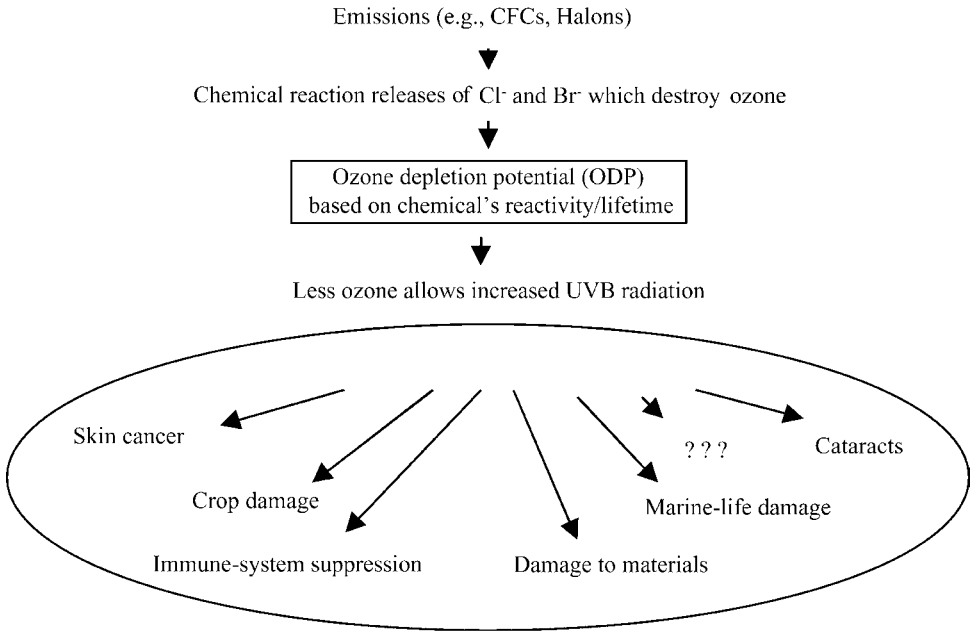


Figure 2 Ozone depletion midpoint/endpoint modeling. The rectangle indicates the midpoint and the oval indicates the endpoints.

Table 1 Cause-effect chain selection

Impact category	*Midpoint level selected*	*Level of site specificity selected*	*Possible endpoints*
Ozone depletion	Potential to destroy ozone based on chemical's reactivity and lifetime	Global	Skin cancer, cataracts, material damage, immune-system suppression, crop damage, other plant and animal effects
Global warming	Potential global warming based on chemical's radiative forcing and lifetime	Global	Malaria, coastal area damage, agricultural effects, forest damage, plant and animal effects
Acidification	Potential to cause wet or dry acid deposition	U.S., east or west of the Mississippi River, U.S. census regions, states	Plant, animal, and ecosystem effects, damage to buildings
Eutrophication	Potential to cause eutrophication	U.S., east or west of the Mississippi River, U.S. census regions, states	Plant, animal and ecosystem effects, odors and recreational effects, human health impacts
Photochemical smog	Potential to cause photochemical smog	U.S., east or west of the Mississippi River, U.S. census regions, state	Human mortality, asthma effects, plant effects
Ecotoxicity	Potential of a chemical released into an evaluative environment to cause ecological harm	U.S	Plant, animal, and ecosystem effects
Human health: criteria air pollutants	Exposure to elevated particulate matter less than 2.5μm	U.S., east or west of the Mississippi River, U.S. census regions, states	Disability-adjusted life-years (DALYs), toxicological human health effects
Human health: cancer	Potential of a chemical released into an evaluative environment to cause human cancer effects	U.S.	Variety of specific human cancer effects
Human health: noncancer	Potential of a chemical released into an evaluative environment to cause human noncancer effects	U.S.	Variety of specific human toxicological noncancer effects
Fossil fuel	Potential to lead to the reduction of the availability of low cost/energy fossil fuel supplies	Global	Fossil fuel shortages leading to use of other energy sources, which may lead to other environmental or economic effects
Land use	Proxy indicator expressing potential damage to threatened and endangered species	U.S., east or west of the Mississippi River, U.S. census regions, state, county	Effects on threatened and endangered species (as defined by proxy indicator)
Water use	Not characterized at this time		Water shortages leading to agricultural, human, plant, and animal effects

Individual Impact Assessment Methodologies

Stratospheric Ozone Depletion

Stratospheric ozone depletion is the reduction of the protective ozone within the stratosphere caused by emissions of ozone-depleting substances. Recent anthropogenic emissions of chlorofluorocarbons (CFCs), halons, and other ozone-depleting substances are believed to be causing an acceleration of destructive chemical reactions, resulting in lower ozone levels and ozone "holes" in certain locations. These reductions in the level of ozone in the stratosphere lead to increasing ultraviolet-B (UVB) radiation reaching the earth. As shown in figure 2, increasing UVB radiation can cause additional cases of skin cancer and cataracts. UVB radiation can also have deleterious effects on crops, materials, and marine life.

International consensus exists on the use of ozone depletion potentials, a metric proposed by the World Meteorological Organization for calculating the relative importance of CFCs, hydrochlorofluorocarbons (HFCs), and halons expected to contribute significantly to the breakdown of the ozone layer. The ozone depletion potentials have been published in the *Handbook for the International Treaties for the Protection of the Ozone Layer* (UNEP-SETAC 2000), where chemicals are characterized relative to CFC-11. The final sum, known as the ozone depletion index, indicates the potential contribution to ozone depletion:

$$\text{Ozone Depletion Index} = \Sigma_i\, e_i \times \text{ODP}_i \quad (1)$$

where e_i is the emission (in kilograms) of substance i and ODP_i is the ozone depletion potential of substance i.

Climate Change

The impact category of global climate change refers to the potential change in the earth's climate caused by the buildup of chemicals (i.e., "greenhouse gases") that trap heat from the reflected sunlight that would have otherwise passed out of the earth's atmosphere (see figure 3 for the global warming chain of potential impacts). Since preindustrial times, atmospheric concentrations of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) have climbed by over 30%, 145%, and 15%, respectively. Although "sinks" exist for greenhouse gases (e.g., oceans and land vegetation absorb carbon dioxide), the rate of emissions in the industrial age has been exceeding the rate of absorption. Simulations are currently being conducted by global warming researchers to try to quantify the potential endpoint effects of these exceedences, including increased droughts, floods, loss of polar ice caps, sea-level rise, soil moisture loss, forest loss, change in wind and ocean patterns, changes in agricultural production, decreased biodiversity, and increasing occurrences of extreme weather events.

TRACI uses global warming potentials, a midpoint metric proposed by the International Panel on Climate Change (IPCC), for the calculation of the potency of greenhouse gases relative to CO_2 (IPCC 1996). The 100-year time horizons recommended by the IPCC and used by the United States for policy making and reporting (U.S. EPA 2001) are adopted within TRACI. The final sum, known as the global warming index, indicates the potential contribution to global warming:

$$\text{Global Warming Index} = \Sigma_i\, e_i \times \text{GWP}_i \quad (2)$$

where e_i is the emission (in kilograms) of substance i and GWP_i is the global warming potential of substance i.

Acidification

Acidification comprises processes that increase the acidity (hydrogen ion concentration, $[H^+]$) of water and soil systems. Acid rain generally reduces the alkalinity of lakes. Changes in the alkalinity of lakes, related to their acid-neutralizing capacity, are used as a diagnostic for freshwater systems analogous to the use of H^+ budgets in terrestrial watersheds (Schlesinger 1997). Acid deposition also has deleterious (corrosive) effects on buildings, monuments, and historical artifacts.

The stressor-effects diagram for acidification has three stages (figure 4). Emissions lead to deposition (via a complex set of atmospheric trans-

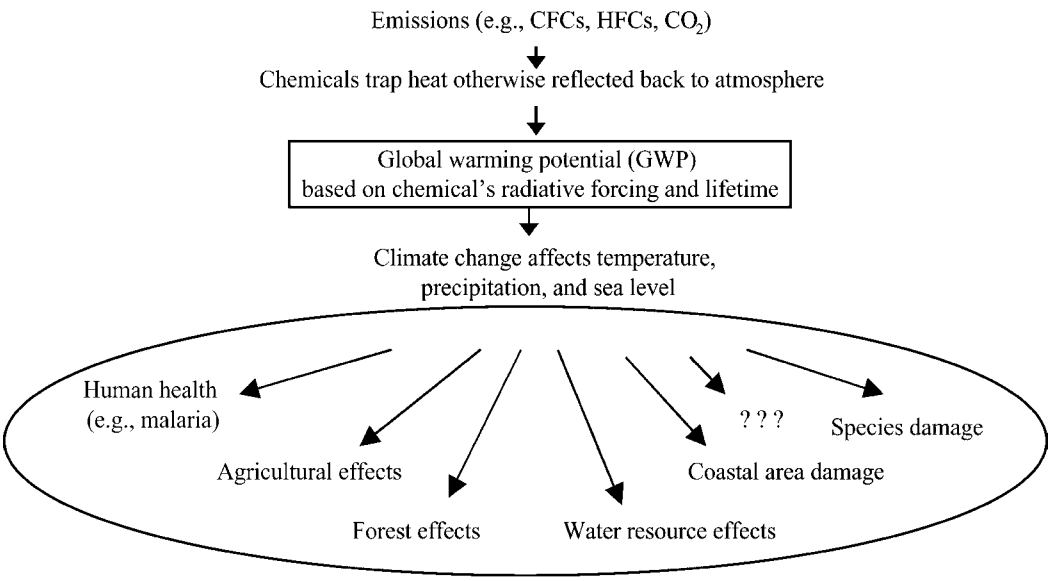


Figure 3 Global warming chain of potential impacts. The rectangle indicates the midpoint and the oval indicates the endpoints.

port and chemistry processes), which in turn can lead to a variety of site-dependent ecosystem impacts—damage to plant and animal populations (via a complex set of chemical and ecological processes). Deposition occurs through three routes: wet (rain, snow, sleet, etc.), dry (direct deposition of particles and gases onto leaves, soil, surface water, etc.), and cloud water deposition (from cloud and fog droplets onto leaves, soil, etc.).

As described in a companion article (Norris 2002), the acidification model in TRACI makes use of the results of an empirically calibrated atmospheric chemistry and transport model to estimate total North American terrestrial deposition of expected H^+ equivalents due to atmospheric emissions of NO_x and SO_2, as a function of the emissions location.

The resulting acidification characterization factors are expressed in H^+ mole equivalent deposition per kilogram of emission. Characterization factors take account of expected differences in total deposition as a result of the pollutant release location. Factors for acidification are available for each U.S. state. In many LCIA applications, the location of the emission source is known with less precision than the state level. Therefore, additional characterization factors were developed for each of four U.S. regions, for two larger regional divisions (either east or west of the Mississippi River), and for the United States as a whole. For each of these larger regions, the composite factor was created using an annual emissions-weighted average of its constituent states.

As reported by Norris (2002), regional characterization factors range from roughly 20% of the U.S. average to 160% of the U.S. average. Also, the deviation of the region-specific factors from the U.S. average is not identical between SO_2 and NO_x; that is, the ratio of region specific to U.S. average characterization factors for a given region varies somewhat between SO_2 and NO_x. Although the majority of acidic deposition in North America stems from emissions of NO_x (NO and NO_2) and SO_2 (including SO_2 as SO_x), significant amounts are also due to emissions of other substances. TRACI adopts U.S. average characterization factors for these trace emissions, based on their H^+ formation potentials per kilogram emitted in relation to SO_2.

The benefit of the new TRACI method for characterization of acidifying emissions, relative to a prior nonregionalized method such as that of Heijungs and colleagues (1992), is the increased ability of LCIA results to take into account location-based differences in expected impact. This benefit arises because the TRACI

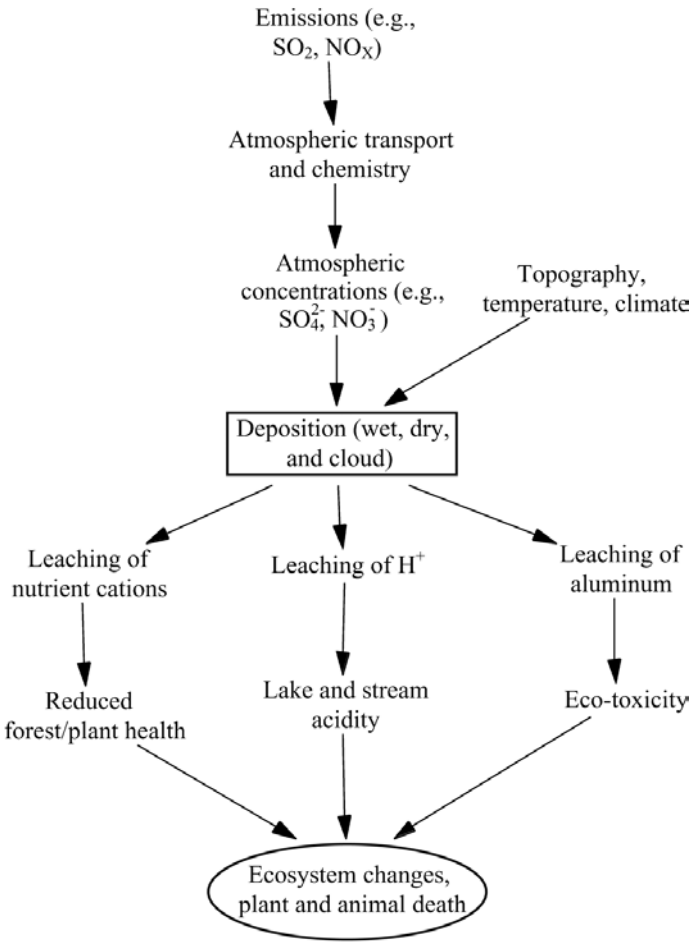


Figure 4 Acidification chain of potential impacts. The rectangle indicates the midpoint and the oval indicates the endpoints.

acidification factors pertain to a focused midpoint within the impact chain (total terrestrial deposition) for which there is considerable, well understood, and quantifiable variability among source regions.

In at least two ways, the regional variability in deposition potential can impact the acidification potential. In the event that the alternatives being studied in the LCA have their processes (and thus their emissions) clustered in different regions, the overall deposition potentials for both SO_2 and NO_x can vary by as much as a factor of 5 or more (Norris 2002). Another possibility is that the alternatives have their processes predominantly clustered in the same regions. If this is the case, then the relative deposition potentials of a kilogram of NO_x versus SO_2 emissions can vary by nearly a factor of 2 from one region to another. In this instance, using the region-appropriate characterization factors may be important to the overall study outcome.

The modeling stops at the midpoint in the cause-effect chain (deposition) because in the United States there is no regional database of receiving environment sensitivities (as is available in Europe[3]). Thus, the source-region-based variability in total terrestrial deposition has been captured, but not the receiving-region-based variability in sensitivity or ultimate damage. Future advances of the TRACI acidification method may address regionalized transport and deposition of ammonia emissions and investigate the potential of accounting for re-

gional differentiation of receiving environment sensitivities.

Eutrophication

Eutrophication is the fertilization of surface waters by nutrients that were previously scarce. When a previously scarce (limiting) nutrient is added, it leads to the proliferation of aquatic photosynthetic plant life. This may lead to a chain of further consequences, including foul odor or taste, death or poisoning of fish or shellfish reduced biodiversity, or production of chemical compounds toxic to humans, marine mammals, or livestock (figure 5). The limiting-nutrient issue is key to characterization analysis of phosphorus (P) and nitrogen (N) releases within LCIA. If equal quantities of N and P are released to a freshwater system that is strictly P limited, then the characterization factors for these two nutrients should account for this fact (e.g., the characterization factor for N should approach zero in this instance).

> The most common impairment of surface waters in the U.S. is eutrophication caused by excessive inputs of P and N. Impaired waters are defined as those that are not suitable for designated uses such as drinking, irrigation, by industry, recreation, or fishing. Eutrophication is responsible for about half of the impaired lake area, 60% of the impaired river reaches in the U.S., and is also the most widespread pollution problem of U.S. estuaries. (Carpenter et al. 1998, 560)

Prior to utilization of TRACI, it is important to determine the actual nutrient emissions that are transported into water. As an example, fertilizers are applied to provide nutrition to the vegetation that covers the soil; therefore, only the runoff of fertilizer makes it into the waterways. The overapplication rate is highly variable and may depend on soil type, vegetation, topography, and even the timing of the application relative to weather events.

The TRACI characterization factors for eutrophication are the product of a nutrient factor and a transport factor. The nutrient factor captures the relative strength of influence on algae growth in the photic zone of aquatic ecosystems of 1 kg of N versus 1 kg of P, when each is the limiting nutrient. The location or context-based transport factors vary between 1 and 0 and take into account the probability that the release arrives in an aquatic environment (either initially or via air or water transport) in which it is a limiting nutrient (figure 6). The TRACI characterization method for eutrophication is described in more detail in the companion article (Norris 2002).

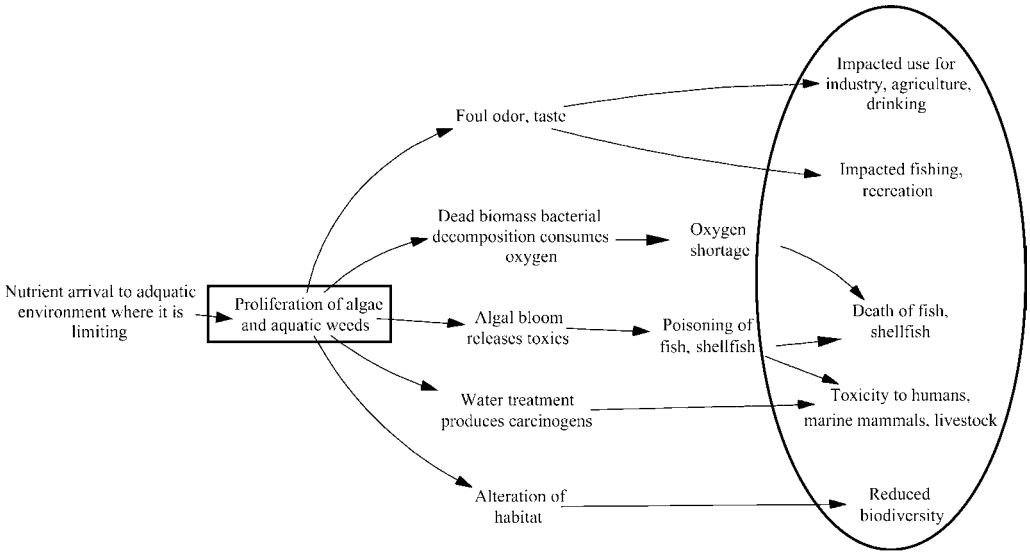


Figure 5 Chain of potential impacts of eutrophication. The rectangle indicates the midpoint and the oval indicates the endpoints.

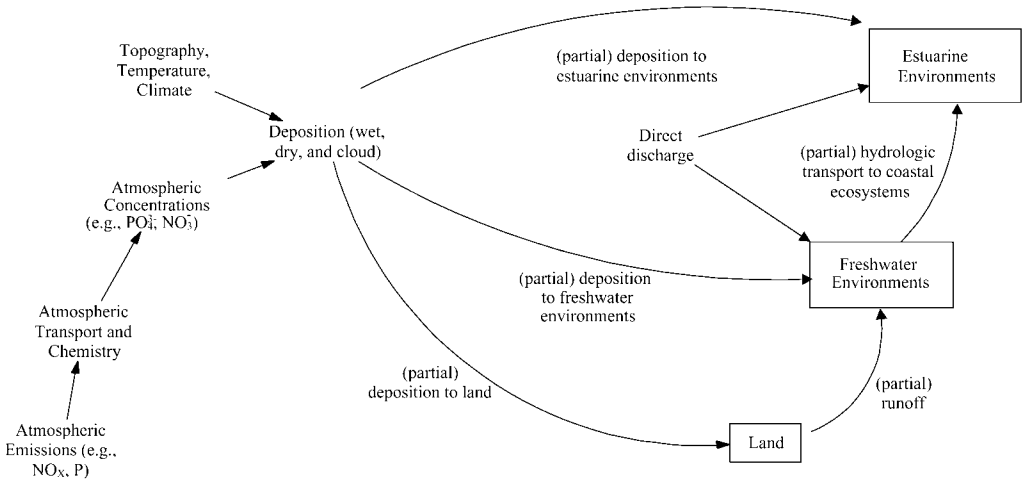


Figure 6 Pathways for nutrient transport to fresh and coastal waters. The rectangles indicate the receiving environment.

The characterization factors estimate the eutrophication potential of a release of chemicals containing N or P to air or water, per kilogram of chemical released, relative to 1 kg N discharged directly to surface freshwater. The regional variability in the resulting eutrophication factors shows that the location of the source influences not only the relative strength of influence for a unit emission of a given pollutant, but it also influences the relative strength of influence among pollutants. The benefit of the new TRACI method for characterizing eutrophying emissions, relative to a prior nonregionalized method such as that of Heijungs and colleagues (1992), is the increased ability of LCIA results to take into account the expected influence of location on both atmospheric and hydrologic nutrient transport and thus the expected influence of release location upon expected nutrient impact. The combined influence of atmospheric transport and deposition along with hydrologic transport can lead to total transport factors differing by a factor of 100 or more (Norris 2002).

As with both acidification and photochemical oxidant formation (below), TRACI provides characterization factors for nine different groups of U.S. states, known as census regions,[4] for eastern and western regions and for the United States as a whole, for use when the location of the release is not more precisely known. For each of these larger regions, the composite factor was created using an average of those for its constituent states.

Photochemical Oxidant Formation

Ozone (O_3) is a reactive oxidant gas produced naturally in trace amounts in the earth's atmosphere. Ozone in the troposphere leads to detrimental impacts on human health and ecosystems (figure 7). The characterization point associated with photochemical oxidant formation is the formation of ozone molecules in the troposphere. Rates of ozone formation in the troposphere are governed by complex chemical reactions, which are influenced by ambient concentrations of nitrogen oxides (NO_x) and volatile organic compounds (VOCs), as well as the particular mix of VOCs, temperature, sunlight, and convective flows. In addition, recent research in the Southern Oxidants Study (e.g., Chameides and Cowling 1995) indicates that carbon monoxide (CO) and methane (CH_4) can also play a role in ozone formation.

More than 100 different types of VOCs are emitted to the atmosphere, and they can differ by more than an order of magnitude in terms of their estimated influence on photochemical oxidant formation (e.g., Carter 1994). Further complicating the issue is the fact that in most regions

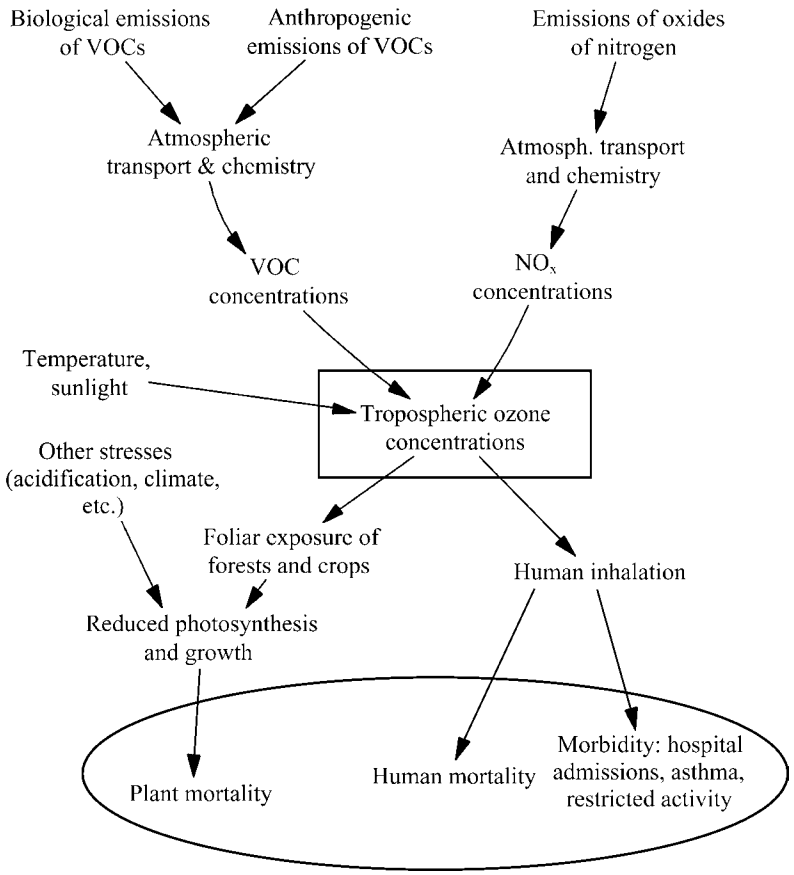


Figure 7 Cause-effect linkages for tropospheric ozone. The rectangle indicates the midpoint and the oval indicates the endpoints.

of the United States, ambient VOC concentrations are due largely to biological sources (trees). For example, in urban and suburban regions of the United States at midday, biogenic VOCs can account for a significant fraction (10% to 40%) of the total ambient VOC reactivity (NRC 1991). In rural areas of the eastern United States, biogenic VOCs contribute more than 90% of the total ambient VOC reactivity in near-surface air.

Conventional smog characterization factors for LCIA have been based on European modeling of the relative reactivities among VOCs and have neglected NO_x entirely. This neglect of NO_x is a highly significant omission. Throughout the past decade, numerous U.S. studies have found spatial and temporal observations of near-surface ozone concentrations to be strongly correlated with ambient NO_x concentrations and more weakly correlated with anthropogenic VOC emissions (see, for example, NRC 1991; Cardelino and Chameides 1995). Another omission in all existing smog characterization factors has been the potential influence of emission location.

The approach to smog characterization analysis for VOCs and NO_x in TRACI incorporates the following components: (1) relative influence of individual VOCs on smog formation, (2) relative influence of NO_x concentrations versus average VOC mixture on smog formation, (3) impact of emissions (by release location) upon concentration by state, and (4) methods for aggregation of effects among receiving states by area.

To characterize the relative influence on O_3 formation among the individual VOCs, Carter's latest maximum incremental reactivity calculations are used (Carter 2000). These reflect the

estimated relative influence on conditions under which NO_x availability is moderately high and VOCs have their strongest influence upon O_3 formation. For the relative influence of NO_x emissions in comparison to the base reactive organic gas mixture, a midrange factor of 2 is used, which is in agreement with empirical studies on regional impacts for the eastern United States (e.g., Cardelino and Chameides 1995) and is at the middle of a range of model-based studies (Rabl and Eyre 1997; Seppälä 1997).

The influence of NO_x emissions on regional ambient levels has been modeled using source/receptor matrices that relate the quantity of seasonal NO_x emissions in a given source region to changes in ambient NO_x concentrations in each receiving region across North America. These source/receptor matrices were obtained from simulations of the Advanced Statistical Trajectory Regional Air Pollution model (Shannon 1991, 1992, 1996). Source and receptor regions are the contiguous U.S. states and Washington, D.C., plus the ten Canadian provinces and northern Mexico. Recent empirical research (e.g., St. John et al. 1998; Kasibhatla et al. 1998) shows that average O_3 concentrations exhibit strong and stable correlations with regional ambient NO_x concentrations.

The assumption was made that VOC emission impacts on regional O_3 concentrations have the same spatial distribution as the ambient NO_x concentration impacts (i.e., similar regional transport for VOCs and NO_x). Finally, the outcome of the source/transport modeling is proportional to estimated O_3 concentration impacts (grams per square meter) per state, given an assumed linear relationship between the changes in NO_x and O_3 concentrations (with VOC concentrations converted to NO_x equivalents).

Human Health Criteria Pollutants

Ambient concentrations of particulate matter (PM) are strongly associated with changes in background rates of chronic and acute respiratory symptoms, as well as mortality rates. Ambient particulate concentrations are elevated by emissions of primary particulates, measured variously as total suspended particulates, PM less than 10 μm in diameter (PM_{10}), PM less than 2.5 μm in diameter ($PM_{2.5}$), and by emissions of SO_2 and NO_x, which lead to the formation of the so-called secondary particulates sulfate and nitrate.

A three-stage method for LCIA of the human health impacts of these emissions has been developed for use in TRACI. The first stage of the model uses the output of atmospheric transport models to estimate the expected change in exposure to $PM_{2.5}$ due to emissions; this modeling incorporates atmospheric reactions and transport, as well as regional variability in population densities. The second stage in the model relies on epidemiological studies to provide concentration-response (C-R) functions that are used to relate changes in exposure to changes in mortality rates and a variety of morbidity effects. The third stage in the model translates the different mortality and morbidity effects into a single summary measure of disability-adjusted life-years (DALYs). Uncertainties at each stage of the model are estimated and modeled quantitatively. The second and third optional stages of the model provide a means to increase the relevance to final decision making.

This methodology has been adopted for this particular impact category as a way of aggregating the different human health effects that are correlated with exposures to ambient particulates. The epidemiological data that are used enable estimation of the incidence of a variety of human health endpoints that differ in severity. This contrasts with the human health cancer and noncancer effects categories that include many chemicals that may have multiple effects of different severities at various levels—many of which may be unknown as a result of inadequate toxicity testing data. Also, the potential effects of criteria pollutants are well matched with the DALY methodology; that is, DALYs exist for respiratory effects and mortality effects, which contrasts with some of the human health effects for the other categories. For example, DALYs cannot currently be used to quantify the potential value of certain reproductive or teratogenic effects.

To determine the relationship between emissions and exposure by setting (model stage 1), the approach described by Nishioka and colleagues (2000) is used and the state-level "intake

fractions" are derived in the study. The intake fraction is the fraction of a pollutant emitted that is actually inhaled; in fact, it represents more precisely the probability that an emitted molecule of emission will be inhaled. Nishioka and colleagues (2002) used the regression models reported by Levy and colleagues (2000) to estimate state-level point and area intake fractions. The final predictive equations for the intake fraction of primary $PM_{2.5}$, secondary sulfates, and secondary nitrates were able to predict these intake fractions quite well (R^2 between 0.5 and 0.9) based on a limited number of simple parameters (e.g., total population within 500 km, annual average temperature at the source). Levy and colleagues (2000) selected these parameters both for their predictive power and because they could be easily collected for point and area sources.

To determine the health benefits associated with the estimated reductions in $PM_{2.5}$ exposure (model stage 2), a survey of the relevant epidemiological literature was used. Premature mortality is the principal focus, because it has contributed a large portion of the total impacts (after valuation) in past studies (U.S. EPA 1999). In addition, to help communicate the range of health effects, C-R functions are applied for chronic bronchitis, cardiovascular hospital admissions, and restricted activity days. A number of studies report associations between various morbidity categories and PM. Morbidity categories with a nonnegligible contribution to total impacts were selected (e.g., based on U.S. EPA 1999); studies were included for which reasonable scientific and epidemiological evidence were available. For this analysis, the study used the considerable evidence for a linear C-R function throughout the range of ambient concentrations in the United States, with any potential effects threshold being below the lowest ambient concentration. The evidence of the linear relationship between particulate air pollution at the current level and daily mortality without a threshold is supported by Daniels (2000) in a study in which daily time-series data for the 20 largest U.S. cities for 1987–1994 was analyzed using a spline model. This also greatly simplifies the impact assessment; because ambient $PM_{2.5}$ concentrations vary across the United States, any nonlinearities would imply that different slopes should be applied in different locations. For simplicity, background disease prevalence and mortality rates are assumed constant across the United States. The potential effects of regional variation in background rates are explored separately (Nishioka et al. 2002).

Stage 3 of the model entails aggregation of the mortality and morbidity outcomes. When an individual dies of exposure to PM, the number of years lost is calculated based on age- and gender-specific life expectancy tables. Estimates of the total life-years lost by a population exposed to PM depend on several factors, including the age distribution and the size of the exposed population, the magnitude of the PM change, the relative risk assumed to be associated with the change in PM, and the length of exposure (U.S. EPA 1999).

The "burden of disease" measure developed by Murray and Lopez (1996) can be used to estimate the health loss associated with air pollution. Several hundred severity weights are reported in their study (Murray and Lopez 1996). Also, disability weights have been derived for the Netherlands, addressing 56 diagnostic groups separating more than 100 different disease stages (Stouthard et al. 2000). Environmental disease-related disability weights have been provided by De Hollander and colleagues (1999) based on the work of Stouthard and colleagues (2000). De Hollander and colleagues (1999) used both the Global Burden of Disease project and the Dutch Burden of Disease project to attribute weight to environmental health impacts.

The DALYs measure combined years of life lost and years lived with disability that are standardized by means of severity weights. Thus, the annual number of DALYs lost can be calculated as follows:

$$\text{DALY} = N \times D \times S \qquad (3)$$

where N is the number of cases, D is the average duration of the response, including loss of life expectancy as a consequence of premature mortality, and S is the discount weights to the unfavorable health conditions.

For premature mortality due to long-term exposure to particulate air pollution, De Hollander and colleagues (1999) reported a severity score[5] of 1 and the years of life lost to be 10.9 yr per

case. Thus, the DALY value of 10.9 per mortality case is used.

Human Cancer and Noncancer Effects

Reliable ranking and relative comparisons of a large number of chemicals in terms of their potential to cause toxicological impacts are more important than trying to characterize absolute risk. Indeed, a number of researchers believe that there is not currently sufficient data to make reliable estimates of low-dose risk on an absolute basis, and a large number of methodologies have been proposed in the literature to help compare chemical emissions in this context (Pennington and Bare 2002; Pennington and Yue 2000; Swanson and Socha 1997). As local and worker health impacts are controlled largely through site-specific initiatives, comparisons are performed in LCA in the context of long-term exposures at a macro scale (regional and global). Such approaches require considering the long-term fate of a chemical, human exposure, and the effects on human health.

Quantitative characterization factors are usually predicted in LCIA in the context of toxicological impacts using integrated multimedia fate, exposure, and effect models (e.g., Guinée et al. 1996; Hertwich et al. 2001; Huijbregts 1999a). Generic models are usually adopted, particularly in the absence of site-specific emissions inventory and chemical property data. The relative toxicological concern of an emission in the context of human health is currently calculated in TRACI based on human toxicity potentials (HTPs) (Hertwich et al. 2001). These best-estimate HTPs (usually between the mean and the median of the HTP distributions given parameter uncertainty and variability) were derived using a closed-system, steady-state version of CalTOX (Version 2.2) (McKone 1993), a multimedia fate and multiple-exposure pathway model with fixed generic parameters for the United States. The closed system is used to provide a scenario in which a chemical can only be removed from the environment, or modeled unit world, by transformation and not by relocation to other regions.

The modified model used to calculate the HTPs for LCIA, CalTOX (McKone 1993), consists of a regional multimedia fate model coupled with human exposure correlations for 23 pathways (U.S. EPA 1989a, 1992a; CalEPA 1996; Yang and Nelson 1986) to estimate exposure doses. These doses are then compared with effect potency data, as illustrated in figure 8. Exposure via dermal contact with soils, ingestion (of water and contaminated food), and inhalation are taken into account.

Using CalTOX, the dose per unit release rate (in kilograms per hour) of a chemical is estimated. As in risk assessment, each dose/unit release rate is divided by an acceptable daily intake for noncarcinogens or multiplied by a carcinogenic risk potency factor (q_1^*) to help derive the HTPs. The acceptable daily intakes used in the HTP calculations are currently based on reference doses for ingestion and reference concentrations for inhalation. Separate HTPs are calculated for each exposure pathway, release medium (currently emissions to air and to water), and type of effect (carcinogenic or noncarcinogenic). Aggregate values of the calculated noncancer and cancer HTPs are reported for each release medium. The HTPs for each chemical are then compared to baseline values using benzene for carcinogens and toluene for noncarcinogens, hence chemical emissions are comparable in terms of toxicological equivalencies.

Using CalTOX, a variation analysis was conducted to determine the effect of the various levels of uncertainty in chemical input parameters relative to the variability introduced in exposure factors and landscape parameters.

The largest sources of uncertainty in the model were the result of the chemical input parameters, especially half-life data. For some chemicals, exposure factor uncertainty was significant, but landscape parameters were considered to be of minor significance in general (Hertwich et al. 1999).

Eco-toxicity

For ecological impacts, as well as for human health, LCIA currently uses measures of hazard to compare the relative importance of pollutants

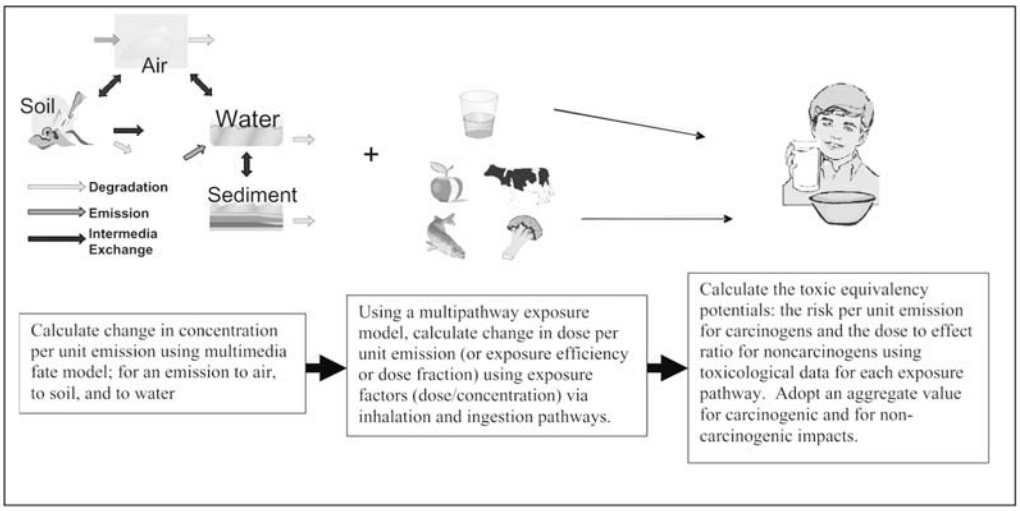


Figure 8 Human toxicity potential. Not all pathways are presented.

within a defined impact category (Guinée and Heijungs 1993; Hertwich et al. 1997; Hofstetter 1998; Udo de Haes et al. 1999; Hertwich at al. 1998). HTPs provide simplified representations of actual processes based on primary attributes for human health. These attributes are developed using measured and/or estimated data in models that focus on factors judged to be crucial (Hertwich et al. 1997, 1998, 2001). Similarly, the ecological toxicity potential (ETP) has been developed as a quantitative measure that expresses the potential ecological harm of a unit quantity of chemical released into an evaluative environment. The goal of the ETP is to establish for LCIA a rank measure of potential ecosystem harm for a large set of toxic industrial and agricultural chemicals.

Impacts of chemical emissions on plant and animal species can have multiple components (Suter 1993; U.S. EPA 1992b, 1998). The ETP is designed to capture the direct impacts of chemical emissions from industrial systems on the health of plant and animal species. The SETAC Europe working group on impact assessments (Hauschild and Pennington 2002) proposed three factors to characterize ecological effects in LCIA: (1) an emission factor to account for mass loading, (2) a source-to-concentration factor to account for transport and transformation, and (3) a toxicity factor to account for harmful effects. In the context of this framework, the goal of the ETP is to provide a system for quantifying the combination of source-to-concentration and toxicity. Overall ecological effects for a functional unit are obtained from the sum of emissions-weighted ETPs.

The ETP includes two components: a generic concentration-to-source ratio (CSR) for pollutant emissions and an impact-to-concentration ratio (ICR). As such, the ETP has been developed as a toxicity-weighted CSR that relates a chemical to be evaluated against one that is relatively well characterized.

$$\mathrm{ETP}_i^{nm} = \frac{[\mathrm{CSR}_i^{nm} \times \mathrm{ICR}_i^{m}]}{[\mathrm{CSR}_x^{nm} \times \mathrm{ICR}_x^{m}]} \qquad (4)$$

where

$$\mathrm{CSR}_i^{nm} = \frac{\mathrm{PEC}_i^{m}}{S_i^{n}},\ \mathrm{ICR}_i^{m} = \frac{\mathrm{FA}^*}{C_i^{m*}} \qquad (5)$$

ETP_i^{nm} is the ecotoxicity potential expressing the impact of the release of chemical i to compartment n in terms of equivalent quantities of chemical x released to that same compartment; CSR_i^{nm} is the concentration-to-source ratio, which is the concentration (in moles per cubic meter) of chemical i in environmental medium m (soil, surface water) based on a unit release (in moles per day) to compartment n (air or surface water);

ICR_i^m is the impact-to-concentration ratio for chemical i in environmental medium m, that is the measure of potential impact (i.e., species affected) associated with a marginal increase of concentration (in moles per cubic meter) in medium m (soil, water); PEC_i^m is the predicted environmental concentration (in moles per cubic meter) of chemical i in environmental medium m from a continuous release S_i^n (in moles per day) to compartment n; FA* is a standardized measure of harm, such as the fraction of species adversely affected, that is used as a consistent measure of potential harm among a large set of chemical substances; and C_i^{m*} is a benchmark concentration (in moles per cubic meter) for chemical i and is the concentration on the exposure-response curve at which there is a point of departure where the likelihood of effect and not the severity of effect increases with increasing exposure.

Emissions to air and surface water (sw) are considered separately ($ETP_i^{air,m}$ and $ETP_i^{sw,m}$). In addition, for each emission type (air and sw), an ETP is calculated based on potential terrestrial ecosystem impacts and potential aquatic ecosystems impact so that there are four ETP values, $ETP_i^{air,soil}$, $ETP_i^{air,sw}$, $ETP_i^{sw,soil}$, and $ETP_i^{sw,sw}$. This is reduced to a set of two ETP values based on a simple combination of the water ETP and soil ETP impact for a release to either air or water. As a result, the overall ETP for each release scenario is calculated as follows:

$$ETP_i^{air}\ (\text{overall}) = 0.5 \times ETP_i^{air,soil} + 0.5 \times ETP_i^{air,sw} \quad (6)$$

$$ETP_i^{sw}\ (\text{overall}) = 0.5 \times ETP_i^{sw,soil} + 0.5 \times ETP_i^{sw,sw} \quad (7)$$

Calculating the ETP

The CSR expresses the concentration (in moles per cubic meter) of chemical i in environmental medium m (soil or surface water) based on a continuous unit release (in moles per day) to compartment n (air or surface water). For the current set of ETP estimates, the same modified version of CalTOX and the chemical and landscape data sets employed by Hertwich and colleagues (2001) for HTP calculations are used. The soil and surface water CSRs are obtained from the steady-state solution of the CalTOX multimedia mass-balance model with continuous air and surface water emissions.

The ICR is based on the ecosystem predicted no-effects concentration (PNEC), which is the measure of ecosystem impact that has been most frequently used in life-cycle impact and for comparative risk assessment (see, for example, Guinée and Heijungs 1993; Guinée et al. 1996; Hauschild et al. 1998; Huijbregts 1999a; Huijbregts et al. 2000; Jolliet and Crettaz 1997; Wenzel et al. 1997). For ecosystem health, the PNEC is defined as the environmental concentration expected to cause no effects, acute or chronic, on the structure or function of ecosystems (European Commission 1996; U.S. EPA 2000).

As has been done by others, notably Huijbregts and colleagues (2000), the potential fraction of species affected is selected as an appropriate measure of harm for ecosystems to estimate PNECs. The potential fraction of species affected is obtained from the species sensitivity distribution of no observable effect concentration (SSDNOEC) values (Klepper et al. 1998; U.S. EPA 2000). Typically, one adopts the lower bound, such as the fifth percentile, of the SSDNOEC to obtain a PNEC (Klepper et al. 1998; Suter 1998; Huijbregts 1999b; Huijbregts et al. 2000; van Beelen et al. 2001).

The current ETP set includes 161 chemicals. For emissions to air and surface water, an ETP is calculated based on potential terrestrial ecosystem impacts (related to soil concentration) and potential aquatic ecosystems impact. 2,4-Dichlorophenoxyacetic acid (2,4-D, CAS ID 94-75-7) is used as a reference substance.

Resource Depletion

Land, fossil fuel, and water use are included as resource depletion categories. Other resource depletion issues (e.g., minerals) could be included, but these three are currently more often included in LCA studies than others.

The methodologies that support the resource depletion categories have the least consensus. To date, much more focus has been on chemical emissions and the assessment of the potential effects from these categories. In the resource de-

pletion categories, there is still no consensus on the "value" of the resource (e.g., what factors are involved in valuing land). The methodologies provided for these categories should not be considered as robust, given current insights, but are the most likely to change with additional research.

Land Use

How to best characterize and aggregate individual land-use modifications is a topic of widespread controversy and research. Recent publications discuss a variety of techniques (Heijungs et al. 1997; Lindeijer et al. 1998; Swan 1998; Goedkoop and Spriensma 1999). Among the various existing techniques are economic appraisals, species area relationships, and land-conversion and land-occupation assessments. Some of these techniques require knowledge of the land's historical condition, as well as its altered condition and expected time for restoration to its original state. Most land-use valuation techniques recognize that the locality of the land is important because it often affects the economic value of the land, but also because it is integrally linked to the climate, topography, and resident plant and animal species, making it a unique location in many respects.

TRACI recognizes that location has a very large role in determining the environmental importance of land. Rather than trying to characterize various different land uses and inhabitant species, TRACI uses the density of threatened and endangered (T&E) species in a specific area (e.g., county) as a proxy for environmental importance. This allows users to characterize land that is to be developed or modified at a specific location and then to tie into a T&E species database to calculate the potential T&E displacement of the proposed altered land. The T&E species proxy approach operates under the assumption that land that has a higher number of T&E species is inherently more valuable in so far as species might completely disappear in the United States if additional land is developed. Land within a county may be expected to be similar in climate, soil type, typography, and many other aspects that may be valuable to these T&E species. The T&E database is provided within TRACI, so that the user does not need to know the T&E species count within the relevant county. In the absence of knowledge of the specific county, a more generalized T&E density value (e.g., state T&E density) may be used; however, the user should be cautioned that there may be significant differences in T&E density value within a particular state, and therefore the value of the model is significantly reduced when the state- or national-level T&E density value is used.

All land modifications are not created equal, so some decisions need to be made about which land modifications are considered significant enough to result in potential habitat effects. Minor land modifications (e.g., land that is simply being converted from one agricultural use to another agricultural use) should not be counted, but significant alterations (e.g., land that is being deforested and converted into a shopping mall) should be counted. Significance can be measured based upon (1) previous use (history) versus proposed use, (2) extent of modifications, (3) occupation time, and (4) recovery time. Although it is recognized that this guidance is currently vague, it is anticipated that this impact category will continue to receive much research attention. The equation for land use is as follows:

$$\text{Land Use Index} = \Sigma_i A_i \times (T\&E_i)/CA_i \quad (8)$$

where A_i is the human activity area per functional unit of product, $T\&E_i$ is the T&E species count for the county, and CA_i is the area of the county.

Fossil Fuel Use

Several ways of analyzing fossil fuel and energy consumption exist. Many of these techniques acknowledge a preference for renewable energy sources as opposed to nonrenewable energy sources. Although a useful measure, total nonrenewable energy consumption does not fully address potential depletion issues associated with these flows. For example, solid and liquid fuels are not perfect substitutes (i.e., solid fuels are not currently practical in personal transportation applications). For this reason, depletion of petroleum has different implications than depletion of coal, and so forth.

An existing technique described in Eco-Indicator'99 was selected for incorporation into

TRACI (Goedkoop and Spriensma 1999). This technique takes into account the fact that continued extraction and production of fossil fuels tends to consume the most economically recoverable reserves first, so that (assuming fixed technology) continued extraction will become more energy intensive in the future. This is especially true once economically recoverable reserves of conventional petroleum and natural gas are consumed, leading to the need to use nonconventional sources, such as oil shale. For each present fuel, experts generated scenarios for replacement fuels at a point in the future when total cumulative consumption equals 5 times the present cumulative consumption. The current energy intensity (energy per unit of fuel delivered) for these future fuel extraction and production scenarios was specified. The increase in unit energy requirements per unit of consumption for each fuel provides an estimate of the incremental energy input "cost" per unit of consumption. These factors then provide a basis for weighting the consumption of different fossil fuel energy resources.

$$\text{Fossil Fuel Index} = \Sigma_i N_i \times F_i \qquad (9)$$

where N_i is the increase in energy input requirements per unit of consumption of fuel i and F_i is the consumption of fuel i per unit of product.

Water Use

Water use has generally been tracked in simple mass or volume terms in LCI, without subsequent characterization analysis that would weight different usage flows to take into account important differences among source types and usage locations (Owens 2001). Rather than trying to capture the addition of water pollutants into the environment, this impact category is structured to capture the significant use of water in areas of low availability. As it is a relatively new consideration in LCIA, an impact assessment methodology for water use is not incorporated within TRACI.

Application Example

A simplified case study was conducted for various reasons: (1) to demonstrate the use of TRACI, (2) to determine the effects of site-specific LCIA on deterministic results, and (3) to analyze the uncertainty reductions by the use of site-specific LCIA.

The case study is based on LCI models for concrete manufacturing developed for the Athena Sustainable Materials Institute by George Venta (ASMI 1993, 2001). Because the purpose of the case study was to demonstrate and illustrate the three conditions above and not to support or inform a specific decision, the case study was simplified to include only the LCI data for the criteria pollutants and CO_2.

The process tree for the manufacture of cement is shown in figure 9. In this example, assumptions are made about process locations based on concrete manufacturing taking place in Massachusetts in the northeastern United States. Process location assumptions are shown following the process name in figure 9. The quarrying of coarse and fine aggregates takes place in Massachusetts, as does fuel combustion in the final concrete manufacturing stage. Electricity input to the concrete plant is generated somewhere in the northeast United States, but cannot be regionally resolved more precisely than that, given the extent of regional power wheeling. The manufacture and transport of "secondary cementatious materials" into concrete manufacturing could occur anywhere in the United States, as this is not a massive material input.

As processes further up the supply chain (which are processes appearing toward the bottom of figure 9) are considered, their location becomes less precisely known. Thus, production of cement, clinker, gypsum, and raw meal (the fine powder that results from dry grinding of these raw materials) takes place in New England, with electricity generation and most inputs to raw meal production coming from the northeastern United States. Iron ore production is assumed to come from an unspecified location in the United States.

In this example, the LCIA results with TRACI change when process (and thus emission) location information is used, as compared with an alternative scenario that occurs anywhere in the United States. Because TRACI has developed regionalized factors for the impact categories of acidification, eutrophication, and smog, these impact categories are the focus of

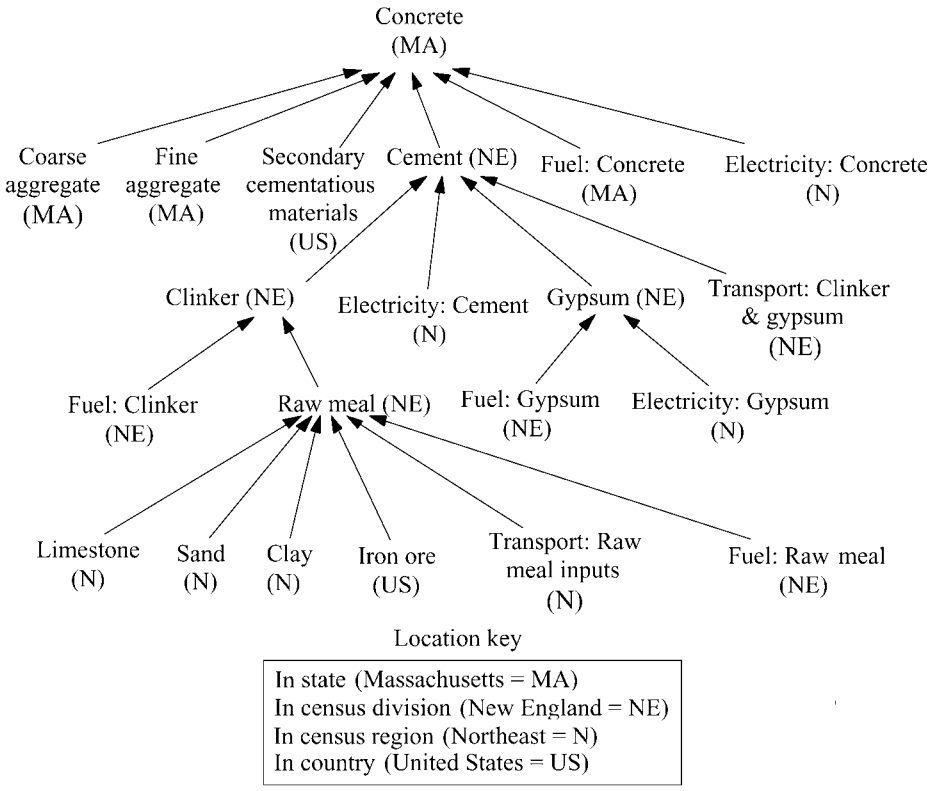


Figure 9 Process tree with location indications for concrete manufacturing.

this case study. The actual LCI data contain 107 different inventory flows, for releases to air, water, and land, and are available on the journal's Web site (http://mitpress.mit.edu/JIE). LCIA data processing and results presentation are simplified for this example by focusing strictly on the releases of five pollutants to air, most of which are relevant to the three impact categories addressed here. These condensed LCI data are presented in table 2. The bulk of the emissions of each pollutant tends to come from half a dozen or so of the downstream processes, including aggregates quarrying and transport, clinker production, fuel combustion at the concrete plant, secondary cementatious materials manufacture and transport, and electricity inputs to cement manufacturing.

TRACI contains U.S. average LCIA factors for all of the impact categories for use in applications when the process location is not known. By using these regionalized characterization factors in place of U.S. average factors, the impacts on deterministic results as well as uncertainty reductions were analyzed and determined to be significant.

The uncertainty inherent in national and macroregional characterization factors relative to state-specific characterization factors is quantified as follows. For each process in figure 9 and table 1, if location information is not specified, then the process could in principle be located anywhere in the United States (assuming an entirely domestic supply chain, which is reasonable for the processes in this example). To reflect this indeterminacy, the characterization factor for each process is expressed as a discrete probability distribution, whose possible values are the state-level characterization factors, each with given nonzero probability. Two plausible sets of probabilities are tested: one where each release location (e.g., each U.S. state) is considered equally likely, and another where the likelihood of each release location being the source of a process release is given by the share of U.S. emis-

Table 2 Condensed life-cycle inventory data for the case example in figure 9

Unit process	CO_2	NO_x	*PM*	SO_x	*VOCs*
Concrete	0	0	120	0	0
Coarse aggregate	3781	52	53.47	9.82	12.1
Fine aggregate	5001	59.7	54.6	14.9	14.6
SCM	717	5.88	0.889	1.97	2.29
Cement	0	0	0	0	0
Electricity: concrete	355	1	0	2	0
Fuel: concrete	5401	66.2	4.57	41.15	19.8
Clinker	843	2	0.4	0.1	0.001
Electricity: cement	5	0.02	0.01	0.03	0.001
Gypsum	0	0	0	0	0
Transport: clinker	10.92	0.10	0.01	0.03	0.04
Fuel: clinker	7.92	0.05	0.24	0.43	0.14
Raw meal	0	0	0	0	0
Fuel: gypsum	7.72	0.06	0.003	0.09	0.03
Electricity: gypsum	3.76	0.01	0.00	0.02	0.0005
Limestone	1.94	0.04	0.47	0.0043	0.01
Sand	0.03	0.0005	0.01	0.0001	0.0001
Clay	0.08	0.0015	0.02	0.0002	0.0003
Iron ore	0.05	0.0010	0.01	0.00	0.0002
Transport: raw meal inputs	4.56	0.05	0.004	0.01	0.02
Fuel: raw meal	5.54	0.02	0.001	0.08	0.02
Electricity: raw meal	3.80	0.01	0.005	0.02	0.0005

sions for the pollutant that is emitted within each U.S. state. Thus, the probabilistic characterization factors (CFs) for each region in the example, for a given pollutant and impact category, are given by the following:

$$\mathrm{CF}_{\mathrm{USA}} = \sum_{i \in \mathrm{USA}} v_i p_i \tag{10a}$$

$$\mathrm{CF}_{\mathrm{NorthEast}} = \sum_{j \in \mathrm{NorthEast}} v_j p_j \tag{10b}$$

$$\mathrm{CF}_{\mathrm{NewEngland}} = \sum_{k \in \mathrm{NewEngland}} v_k p_k \tag{10c}$$

$$\mathrm{CF}_{\mathrm{Massachusetts}} = v_{\mathrm{Massachusetts}} \tag{10d}$$

where v_k indicates the characterization factor value for state k and p_k is the probability of the emission occurring in that state.

Characterization results for two cases are compared. In the first case, the locations of the processes are known (as summarized in figure 9), in which case the region-appropriate characterization factors as given by equations (10a–d) are applied. In the second case, the locations of the processes are known only to be somewhere in the United States, so that the U.S. characterization factor given by equation (10a) is used for each process.

Figure 10 summarizes the expected results for the regionalized versus nonregionalized (U.S.) approaches to LCIA for the case study LCI data for the impact categories of acidification, eutrophication, and smog. The results are normalized relative to the U.S. results. The emissions-weighted probability assumption is applied to the states in each multistate region (U.S., Northeast, and New England). Using this equal-probability assumption introduces less than a 5% change in the expected results for this case study. The results in figure 10 clearly show that the expected result for each impact category is significantly influenced by the process location information together with the regionalized TRACI characterization factors, cutting the total characterization value by more than half for two of the three impact categories.

The results of regionalization are even more striking with respect to their uncertainty reduction. Figure 11a is a comparison of the probability density functions for acidification. The cu-

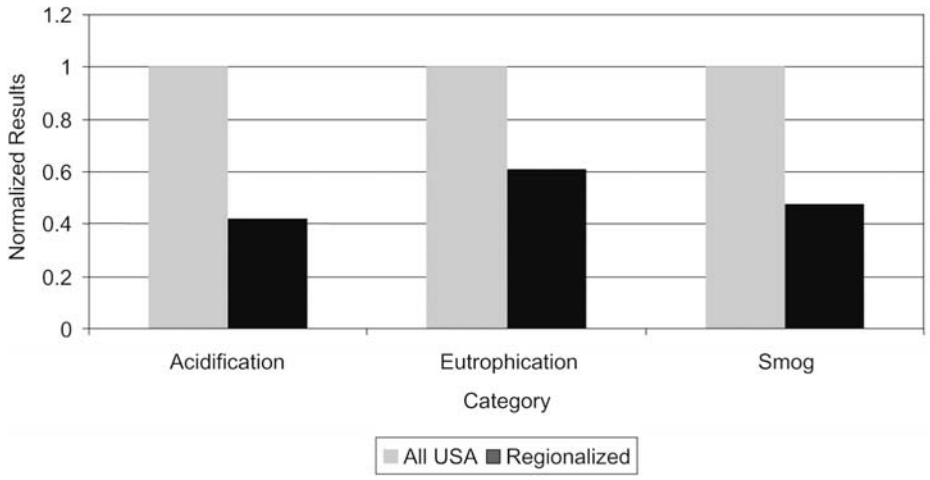


Figure 10 Results for regionalized versus nonregionalized LCIA for the concrete case study.

mulative probability distribution is the integral of the probability density function. Figures 11b, 11c, and 11d compare regionalized and nonregionalized cumulative probability distributions for each of the three impact categories for the emissions-weighted probability assumption. In each case, the uncertainty (as measured by the spread of possible values) is reduced by orders of magnitude. Further, in this case study, the plausible range for the regionalized results falls at the very tail end of the distribution of possible values for the nonregionalized results. This latter effect is primarily because the case study region is the northeastern United States, for which a significant share of the emissions are carried out to sea rather than across populated areas of the North American continent. Still, the differences in uncertainties (magnitudes of plausible ranges of values) would be expected even if the case study region were changed.

Figure 12 demonstrates that the probability basis has very little influence upon the results. Selecting emissions-weighted probabilities of the states within a region has the effect of slightly narrowing the plausible range of values for the total characterized result.

These results demonstrate the powerful influence of TRACI's regionalization for LCIA results for acidification, eutrophication, and smog. Regionalization changes the expected values and helps to reduce uncertainties. Many other sources of uncertainty in LCIA results exist besides those introduced by regional variability in expected impact per unit emission. Ongoing TRACI research is developing a generalized framework for probabilistic uncertainty analysis that integrates the various sources of inventory and impact assessment uncertainty, as well as the appropriate display and communication of the sophistication of probabilistic results. This capability should enable users to assess the potential information (that is, uncertainty reduction) benefits of using more sophisticated LCIA models, such as region-specific characterization factors.

Conclusions

This article presents the research behind the selection and development of impact assessment methodologies within TRACI. Cause-effect chain mechanisms are discussed for each impact category along with the research and methodologies that were constructed to best represent the characterizations for the United States. Whenever possible, existing U.S. EPA guidelines, handbooks, and databases were used to ensure consistency with regulations and policy. Impact categories were often characterized at the midpoint level to minimize assumptions and value choices and to reflect a higher level of societal consensus. Spatial resolution is considered for

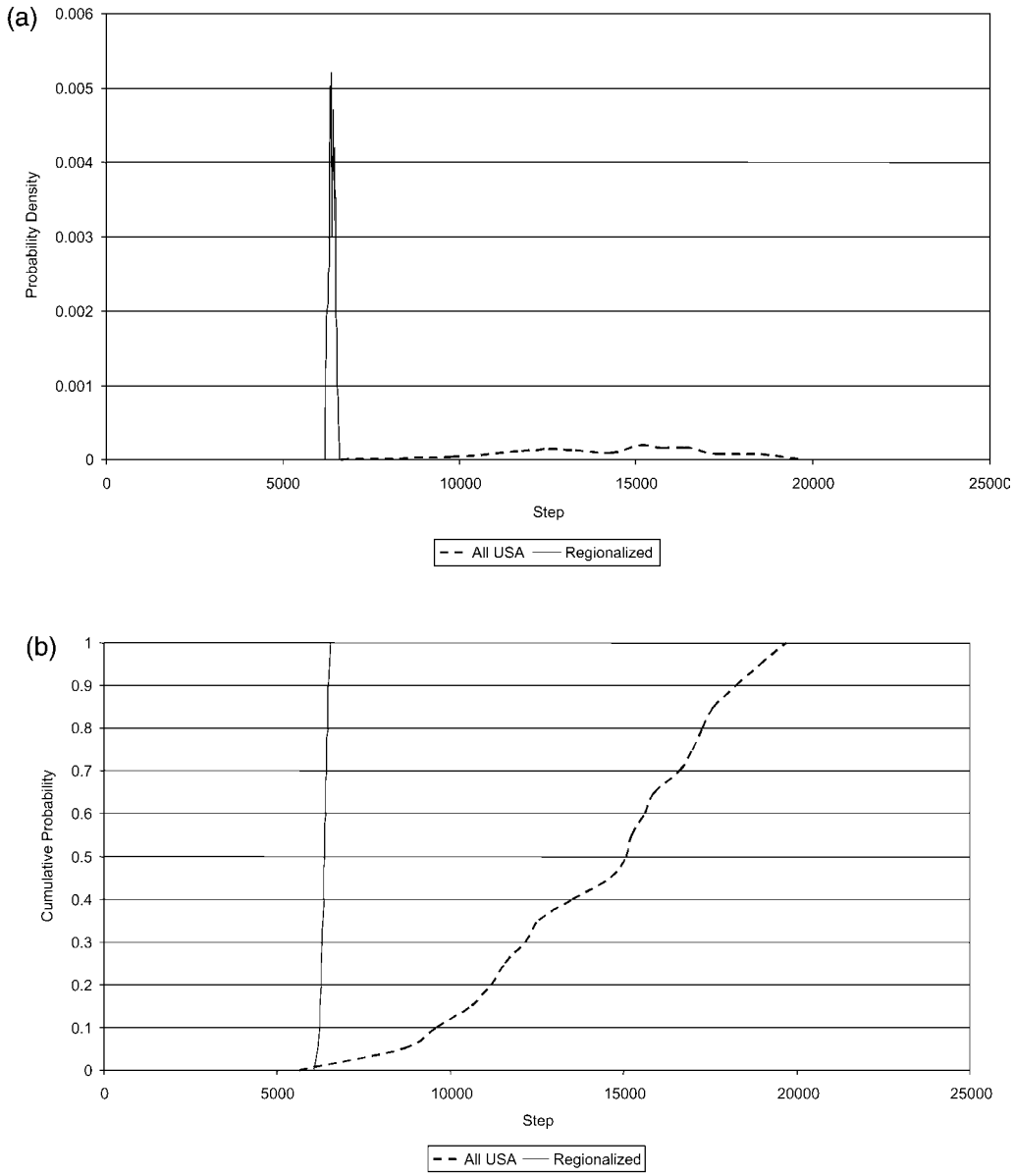
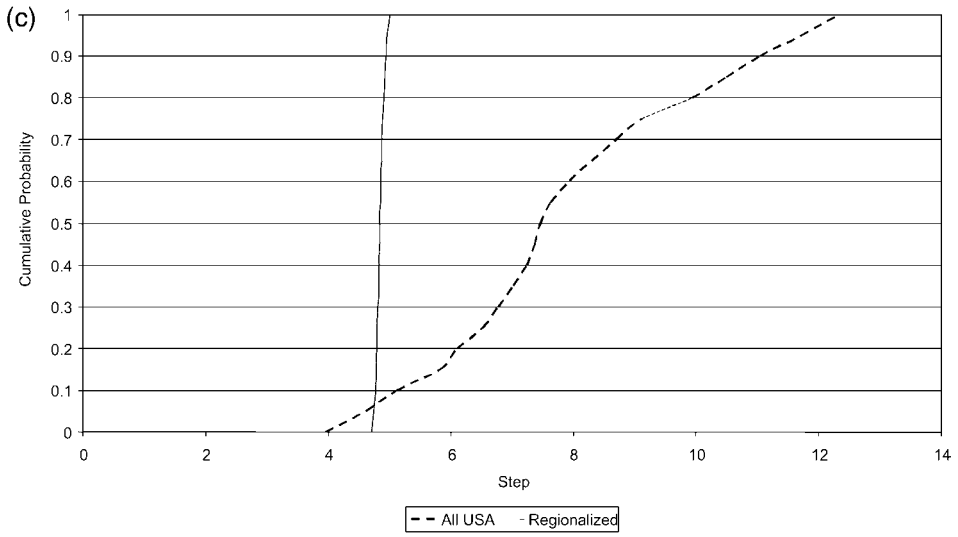


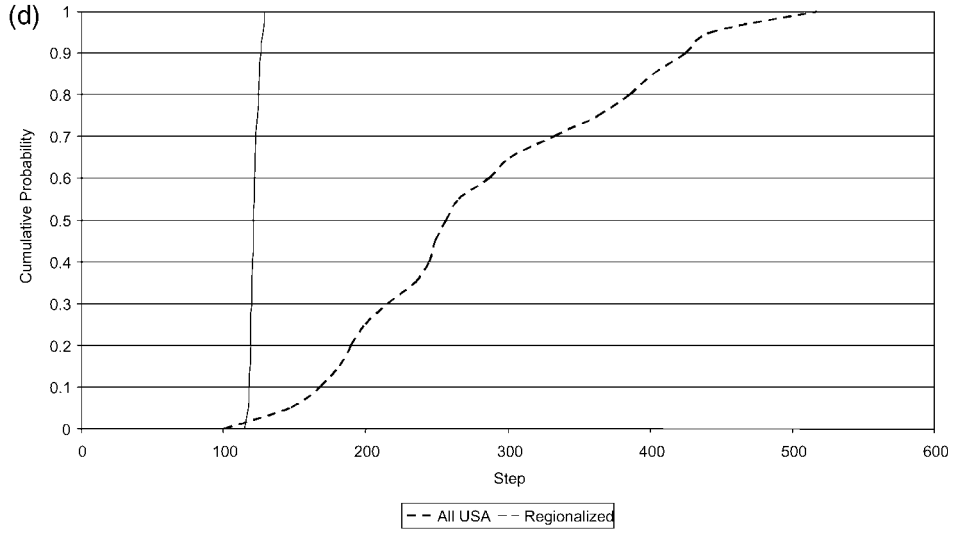
Figure 11 (a) Probability density function for acidification. (b) Cumulative probability density for acidification. (c) Cumulative probability density for eutrophication. (d) Cumulative probability density for smog.

each of the research categories, including the spatial detail of the input parameters, the level of differentiation within various spatial divisions, and the relative uncertainty introduced by lowering the spatial resolution. Seven categories were developed, using U.S. parameters to make them more applicable to the U.S. situation. Four of these categories allow a spatial resolution at the state level or lower. One category allows a county-level resolution.

Within specific impact categories in TRACI, the following new research findings are significant:

- *Human cancer and noncancer effects*. The methodology developed for TRACI is





based on a multimedia fate, multipathway human exposure and toxicological potency approach using CalTOX. Twenty-three exposure pathways were taken into account within the analysis, including inhalation, ingestion of water and various plants and animals, and dermal contact with the soil and water. Toxicity is based on cancer potencies for carcinogens and reference doses or concentrations for noncarcinogens. HTPs were calculated for 330 chemicals, including chemicals representing 80% of the total weight of toxics release inventory releases in 1997. Probabilistic analysis of uncertainty using the proposed model indicates that uncertainty associated with half-life and toxicity represents a large portion of the total uncertainty in calculating HTPs. As landscape information was considered here to be fairly insignificant in contributing to uncertainty, and because an analysis of the U.S. states did not reveal a reordering of HTP values relative to other chemicals, one U.S. value was selected for each chemical to represent releases anywhere within the United States.

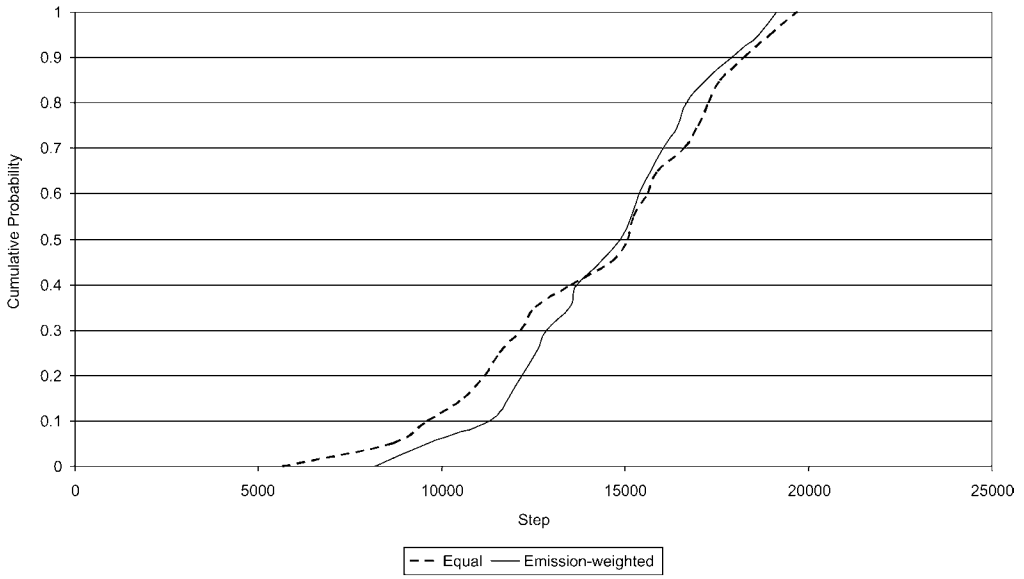


Figure 12 The influence of state probability basis on nonregionalized case study results for acidification.

- *Acidification, eutrophication, and smog.* The methods developed for TRACI for each of these impact categories have taken region-specific differential influence on fate, transport, and expected effects into account, with U.S.-specific modeling. We found that source location can influence acidification impact factors by up to a factor of 4 and eutrophication impact factors by more than a factor of 100. The TRACI methods show, for example, that differentiating source location can be even more important than differentiating the actual chemical species for some of these impact categories. These methods have taken a midpoint rather than an endpoint approach to characterization in each category, stopping at points in the cause-effect impact chain where there is a unique mechanism taking place as a result of each chemical and from which there is consistency of expected impacts among chemicals later in the impact chain.

Finally, a case study of cement manufacture was conducted, and the impacts of site-specific analysis were shown to have significant effects in both the deterministic results and the uncertainty reductions.

Acknowledgments

We gratefully acknowledge the contributions of Edgar Hertwich (currently of the International Institute for Applied Systems Analysis in Laxenburg, Austria) for the development of the human health toxicity potentials (HTPs) and Patrick Hofstetter (U.S. EPA–ORISE) for providing comments. The work on factors for acidification, eutrophication, and smog benefited from expert input at an EPA-sponsored workshop in 1998 and from contributions by colleagues including Jyri Seppälä, José Potting, Mark Goedkoop, Patrick Hofstetter, and Jeroen Guinée.

Disclaimer

Use of TRACI, including but not limited to impact assessment modeling, does not create regulatory or scientific approval by the U.S. EPA on any issues to which it is applied, nor does it release any users from any potential liability, either administratively or judicially, for any damage to human health or the environment. The U.S. EPA does not make any warranty concerning the correctness of the database, any actions taken by third parties as a result of using the model, or the merchantability or fitness for a particular purpose of the model. The U.S. EPA does not endorse any products or services.

Notes

1. TRACI is available on the Internet at no cost at ⟨http://epa.gov/ORD/NRMRL/std/sab/iam_traci.htm⟩.
2. "Criteria pollutants" is a term used in U.S. environmental regulation to describe those regulated air pollutants for which the U.S. EPA uses health-based criteria to set permissible levels. The criteria pollutants are ozone, carbon monoxide, total suspended particulates, sulfur dioxide, lead, and nitrogen oxides.
3. Editor's note: For a discussion of a regional database for receiving environment sensitivities in Europe, see the article by Potting et al., *Journal of Industrial Ecology* 2(2): 63–87.
4. U. S. census regions are available at ⟨http://www.eia.doe.gov/emeu/reps/maps/us_census.html⟩.
5. The severity weight in DALY calculations is 1 for death and 0 for perfect health.

References

ASMI (Athena Sustainable Materials Institute). 1993. *Cement and structural concrete products: Raw materials balances, energy profiles, and environmental unit factor estimates*. Ottawa, Canada: ASMI.

ASMI. 2001. *Cement and structural concrete products. Update: Raw materials balances, energy profiles, and environmental unit factor estimates*. Ottawa, Canada: ASMI.

Bare, J. C., D. W. Pennington, and H. A. Udo de Haes. 1999. Life cycle impact assessment sophistication: International workshop. *International Journal of Life Cycle Assessment* 4: 299–306.

Bare, J. C., P. Hofstetter, D. W. Pennington, and H. A. Udo de Haes. 2000. Life cycle impact assessment midpoints vs. endpoints: The sacrifices and the benefits. *International Journal of Life Cycle Assessment* 5: 319–326.

CalEPA (California Environmental Protection Agency). 1996. *Risk assessment guidelines, Part 4: Technical support document: Exposure assessment and stochastic analysis*. Sacramento, CA: State of California Office of Environmental Health Hazard Assessment.

Cardelino, C. A. and W. L. Chameides. 1995. An observation-based model for analyzing ozone precursor relationships in the urban atmosphere. *Journal of the Air and Waste Management Association* 45: 161–180.

Carpenter, S., N. Caraco, D. Correll, R. Howarth, A. Sharpley, and V. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* 8(3): 559–568.

Carter, W. 1994. Development of ozone reactivity scales for volatile organic compounds. *Journal of the Air and Waste Management Association* 44: 881–899.

Carter, W. 2000. *Updated maximum incremental reactivity scale for regulatory applications*. Sacramento, CA: California Air Resources Board.

Chameides, W. L. and E. B. Cowling, eds. 1995. The state of the southern oxidants study: Policy-relevant findings in ozone pollution research 1988 to 1994. Raleigh, NC: North Carolina State University.

Daniels, M. J., F. Dominici, J. M. Samet, and S. L. Zeger. 2000. Estimating particulate matter-mortality dose-response curves and threshold levels: An analysis of daily time series for the 20 largest U.S. cities. *American Journal of Epidemiology* 152: 407–412.

De Hollander, A. E. M., J. M. Melse, E. Lebret, and P. G. N. Kramers. 1999. An aggregate public health indicator to represent the impact of multiple environmental exposures. *Epidemiology* 10: 606–617.

European Commission. 1996. *Technical guidance documents in support of Directive 93/67/EEC on risk assessment of new notified substances and Regulation (EC) No. 1488/94 on risk assessment of existing substances (Parts I, II, III, and IV)*. EC catalog numbers CR-48-96-001, 002, 003, and 004-EN-C. Luxembourg: Office for Official Publications of the European Community.

Finnveden, G. 1997. Valuation methods within LCA: Where are the values? *International Journal of Life Cycle Assessment* 2: 163–169.

Goedkoop, M. and R. Spriensma. 1999. *The Eco-Indicator 99: A damage orientated method for life cycle impact assessment*. The Hague, The Netherlands: VROM, Ministry of Housing, Spatial Planning, and the Environment.

Grahl, B. and E. Schmincke. 1996. Evaluation and decision-making processes in life cycle assessment. *International Journal of Life Cycle Assessment* 1: 32–35.

Guinée, J. B. and R. Heijungs. 1993. A proposal for the classification of toxic substances within the framework of life cycle assessment of products. *Chemosphere* 26: 1925–1944.

Guinée, J., R. Heijungs, L. van Oers, D. van de Meent, T. Vermeire, and M. Rikken. 1996. *LCA impact assessment of toxic releases: Generic modelling of fate, exposure, and effect for ecosystems and human beings with data for about 100 chemicals*. No. 1996/

21. The Hague, The Netherlands: VROM, Ministry of Housing, Spatial Planning, and the Environment.

Hauschild, M. and D. W. Pennington. 2002. Methods of effect assessment for ecotoxicity. In *Towards best available practice in life cycle impact assessment*, edited by Udo de Haes et al. Brussels: SETAC Europe Press.

Hauschild, M., H. Wenzel, A. Damborg, and J. Tørsløv. 1998. Ecotoxocity as a criterion in the environmental assessment of products. In *Environmental Assessment of Products*. Vol. 2, *Scientific background*, edited by M. Hauschild and H. Wenzel. London: Chapman and Hall.

Heijungs, R., J. Guinée, G. Huppes, R. Lankreijer, H. Udo de Haes, A. Wegener Sleeswijk, A. Ansems, P. Eggels, R. van Duin, and H. de Goede. 1992. *Environmental life cycle assessment of products*. Vol. 1, *Guide* and Vol. 2, *Backgrounds*. Leiden, The Netherlands: Centre of Environmental Science, Leiden University.

Heijungs, R., J. Guinée, and G. Huppes. 1997. *Impact categories for natural resources and land use*. CML report 138, Section: Substances and products. Leiden, The Netherlands: Centre of Environmental Science, Leiden University.

Hertwich, E. G., W. S. Pease, and C. P. Koshland. 1997. Evaluating the environmental impact of products and production processes: A comparison of six methods. *The Science of the Total Environment* 196: 13–29.

Hertwich, E. G., W. S. Pease, and T. E. McKone. 1998. Evaluating toxic impact assessment methods: What works best? *Environmental Science and Technology* 32: A138–A144.

Hertwich, E., T. McKone, and W. Pease. 1999. Parameter uncertainty and variability in evaluative fate and exposure models. *Risk Analysis* 19: 1193–1204.

Hertwich, E. G., S. F. Mateles, W. S. Pease, and T. E. McKone. 2001. Human toxicity potentials for life cycle analysis and toxics release inventory risk screening. *Environmental Toxicology and Chemistry* 20: 928–939.

Hofstetter, P. 1998. *Perspectives in life cycle impact assessment: A structured approach to combine models of the technosphere, ecosphere, and valuesphere*. Boston: Kluwer.

Hofstetter, P. 1999. *Top-down arguments for a goal-oriented assessment structure*. Global LCA Village. ⟨www.ecomed.de/journals/lca/village/WIA2_99/TopDown.htm⟩. Accessed October 2002.

Huijbregts, M. A. J. 1999a. *Priority assessment of toxic substances in the frame of LCA: Development and application of the multi-media fate, exposure, and effect model USES-LCA*. Amsterdam: Interfaculty Department of Environmental Science, Faculty of Environmental Sciences, University of Amsterdam. ⟨www.leidenuniv.nl/interfac/cml/lca2⟩. Accessed June 2002.

Huijbregts, M. A. J. 1999b. *Ecotoxicological effect factors for the terrestrial environment in the frame of LCA*. Amsterdam: Interfaculty Department of Environmental Science, Faculty of Environmental Sciences, University of Amsterdam. ⟨www.leidenuniv.nl/interfac/cml/lca2/TerrestrialLCA.pdf⟩. Accessed October 2002.

Huijbregts, M. A. J., U. Thissen, J. B. Guinée, T. Jager, D. van de Meent, A. M. J. Ragas, A. Wegener Sleeswijk, and L. Reijnders. 2000. Priority assessment of toxic substances in life cycle assessment, Part 1: Calculation of toxicity potentials for 181 substances with the nested multi-media fate, exposure, and effects model USES-LCA. *Chemosphere* 41: 541–573.

IPCC (Intergovernmental Panel on Climate Change). 1996. *Climate change 1995: The science of climate change*. Edited by Intergovernmental Panel on Climate Change: J. T. Houghton, L. G. Meira Filho, B. A. Callander, N. Harris, A. Kattenberg, and K. Maskell. Cambridge, UK: Cambridge University Press.

Jolliet, O. and P. Crettaz. 1997. *Critical surface-time 95: A life cycle impact assessment methodology including fate and exposure*. Lausanne, Switzerland: École Polytechnique Fédérale de Lausanne.

Kasibhatala, P., W. L. Chameides, R. D. Saylor, and D. Olerud. 1998. Relationships between regional ozone pollution and emissions of nitrogen oxides in the eastern United States. *Journal of Geophysical Research* 103(D17): 22663–22669.

Klepper, O., J. Bakker, T. P. Traas, and D. van de Meent. 1998. Mapping the potentially affected fraction (PAF) of species as a basis for comparison of ecotoxicological risks between substances and regions. *Journal of Hazardous Materials* 61: 337–344.

Levy, J. I., S. Wolff, and J. S. Evans. 2002. A regression-based approach for estimating primary and secondary particulate matter exposure efficiency. *Risk Analysis*. 22(5): 895–904.

Lindeijer, E., M. Kampen, and P. Fraanje. 1998. *Biodiversity and life support indicators for land use impacts in LCA*. Report no. W-DWW-98–059. Amsterdam: IVAM and IBN/DLO.

McKone, T. E. 1993. *CalTOX: A multimedia total exposure model for hazardous-waste sites*. UCRL-CR-111456PtI-IV. Livermore, CA: Lawrence Livermore National Laboratory.

Murray, C. J. L. and A. D. Lopez, eds. 1996. *The global burden of disease*. The global burden of disease and injury, Vol. 1. Cambridge, MA: Harvard University Press.

Nishioka, Y., J. I. Levy, G. A. Norris, A. Wilson, P. Hofstetter, and J. D. Spengler. 2002. Estimating the public health benefits of increased residential insulation for new housing. *Risk Analysis* 22(5): 1003–1017.

Nishioka, Y., G. A. Norris, and J. Levy. 2003. The TRACI midpoint- and endpoint-based method for life cycle impact assessment of primary and secondary particulates. In preparation.

Norris, G. 2002. Impact characterization in the tool for the reduction and assessment of chemical and other environmental impacts: Methods for acidification, eutrophication, and ozone formation. *Journal of Industrial Ecology* 6(3/4): 83–105.

NRC (National Research Council). 1991. *Rethinking the ozone problem in urban and regional air pollution*. Washington, DC: National Academy Press.

Owens, J. W. 1997. Life-cycle assessment: Constraints on moving from inventory to impact assessment. *Journal of Industrial Ecology* 1(1): 37–49.

Owens, J. W. 2001. Water resources in life-cycle impact assessment: Considerations in choosing category indicators. *Journal of Industrial Ecology* 5(2): 37–54.

Pennington, D. W. and J. C. Bare. 2002. Comparison of human health screening and ranking approaches: The waste minimization prioritization tool (WMPT) and toxic equivalency potentials (TEPs). *Risk Analysis* 21: 897–913.

Pennington, D. W. and P. L. Yue. 2000. Options for comparison of process design alternatives in terms of regional environmental impacts. *Journal of Cleaner Production* 8: 1–9.

Rabl, A. and N. Eyre. 1997. *An estimate of regional and global O_3 damage from precursor NO_x* and VOC emissions. Unpublished manuscript. Paris: Ecole des Mines.

Schlesinger, W. 1997. *Biogeochemistry: An analysis of global change*. Boston: Academic Press.

Seppälä, J. 1997. *Decision analysis as a tool for life cycle impact assessment*. Helsinki: Finnish Environment Institute.

Shannon, J. D. 1991. Modeled sulfur deposition trends since 1900 in North America. In *Air pollution modeling and its application*, Vol. 8, edited by H. van Dop and G. Kallos. New York: Plenum.

Shannon, J. D. 1992. Regional analysis of S emission-deposition trends in North America from 1979 through 1988. In *Air pollution modeling and its application*, Vol. 9, edited by H. van Dop and G. Kallos. New York: Plenum.

Shannon, J. D. 1996. Atmospheric pathways module. In *Tracking and analysis framework (TAF) model documentation and user's guide: An interactive model for integrated assessment of Title IV of the Clean Air Act Amendments*. ANL/DIS/TM-36. Argonne, IL: Argonne National Laboratory.

St. John, J. C., W. Chameides, and R. Saylor. 1998. The role of anthropogenic NO_x and VOC as ozone precursors: A case study from the SOS Nashville/Middle Tennessee ozone study. *Journal of Geophysical Research* 103(D17): 22415–22423.

Stouthard, M. E. A., M.-L. Essink-Bot, and J. G. Bonsel. 2000. Disability weights for diseases: A modified protocol and results for a Western European region. *European Journal of Public Health* 10: 24–30.

Suter, G. W., II. 1993. *Ecological risk assessment*. Boca Raton, FL: Lewis Publishers.

Suter, G. W., II. 1998. Ecotoxicological effects extrapolation models. In *Risk assessment: Logic and measurement*, edited by C. L. Strojan and M. C. Newman. Ann Arbor, MI: Ann Arbor Press.

Swan, G. 1998. *Evaluation of land use in life cycle assessment*. Report 1998:2. Göteborg, Sweden: Chalmers University of Technology, Center for Environmental Assessment of Product and Material Systems.

Swanson, M. B. and A. C. Socha, eds. 1997. Chemical ranking and scoring: Guidelines for relative assessments of chemicals. In *Proceedings of the Pellston Workshop on Chemical Ranking and Scoring*. Pensacola, FL: SETAC Press.

Udo de Haes, H. A., ed. 1996. *Towards a methodology for life cycle impact assessment*. Brussels: SETAC Europe.

Udo de Haes, H. A., G. Finnveden, M. Goedkoop, M. Hauschild, E. G. Hertwich, P. Hofstetter, O. Jolliet, W. Klöpffer, W. Krewitt, E. W. Lindeijer, R. Mueller-Wenk, S. I. Olson, D. W. Pennington, J. Potting, and B. Steen. 2002. *Life cycle impact assessment: Striving towards best practice*. Brussels: SETAC-Europe.

Udo de Haes, H. A., O. Jolliet, G. Finnveden, M. Hauschild, W. Krewitt, and R. Mueller-Wenk. 1999a. Best available practice regarding impact categories and category indicators in life cycle impact assessment, Part 1. *International Journal of Life Cycle Assessment* 4(2): 66–74.

Udo de Haes, H. A., O. Jolliet, G. Finnveden, M. Hauschild, W. Krewitt, and R. Mueller-Wenk. 1999b. Best available practice regarding impact categories and category indicators in life cycle impact assessment, Part 2. *International Journal of Life Cycle Assessment* 4(3): 167–174.

UNEP-SETAC (United Nations Environmental Programme–Society of Environmental Toxicology

and Chemistry). 2000. *Letter of intent: Best available practice of life cycle assessment with generic application dependency*. Signed by the International Council of SETAC, and UNEP, Division Technology, Industry, and Economics. ⟨www.uneptie.org/pc/sustain/lca/letter-of-intent.htm⟩. Accessed October 2002.

U.S. EPA (Environmental Protection Agency). 1989a. *Exposure factors handbook*. EPA/600/8–89/043. Washington, DC: Office of Health and Environmental Assessment.

U.S. EPA. 1989b. *Risk assessment guidance for Superfund*. Vol. 1, Human health evaluation manual (Part A). EPA/540/1-89/002. Washington, DC: Office of Emergency and Remedial Response.

U.S. EPA. 1992a. *Dermal exposure assessment: Principles and applications*. EPA/600/8-91/011B. Washington, DC: Office of Health and Environmental Assessment.

U.S. EPA. 1992b. *Framework for ecological risk assessment*. EPA/630/R-92/001. Washington, DC: Government Printing Office.

U.S. EPA. 1998. *Guidelines for ecological risk assessment*. EPA/630/R-95/002F. Washington, DC: Government Printing Office.

U.S. EPA. 1999. *The benefits and costs of the Clean Air Act: 1990 to 2010*. EPA-410-R99-001. Washington, DC: EPA Office of Air and Radiation.

U.S. EPA. 2000. *Ecological soil screening level guidance*. Washington, DC: Office of Emergency and Remedial Response. ⟨www.epa.gov/superfund/programs/risk/ecorisk/frdraft.pdf⟩. Accessed October 2002.

U.S. EPA. 2001. *National emissions: Global warming potentials*. ⟨http://yosemite.epa.gov/oar/globalwarming.nsf/content/EmissionsNationalGlobalWarmingPotentials.html⟩. Accessed Oct. 21, 2002.

van Beelen, P., A. K. Fleuren-Kemila, and T. Aldenberg. 2001. The relation between extrapolated risk, expressed as potentially affected fraction, and community effects, expressed as pollution-induced community tolerance. *Environmental Toxicology and Chemistry* 20: 1133–1140.

Volkwein, S. and W. Klopffer. 1996. The valuation step within LCA, Part 1: General principles. *International Journal of Life Cycle Assessment* 1(1): 36–39.

Volkwein, S., R. Gihr, and W. Klopffer. 1996. The valuation step within LCA, Part 2: A formalized method of prioritization by expert panels. *International Journal of Life Cycle Assessment* 1(4): 182–192.

Wenzel, H., M. Z. Hauschild, and L. Alting. 1997. *Environmental assessment of products*. Vol. 1, Methodology, tools and case studies in product development. London: Chapman and Hall.

Yang, Y. Y. and C. B. Nelson. 1986. An estimation of daily food usage factors for assessing radionuclide intakes in the US population. *Health Physics* 50: 245–257.

About the Authors

Jane C. Bare is a chemical engineer in the systems analysis branch of the sustainable technology division at the National Risk Management Research Laboratory of the U.S. Environmental Protection Agency, Cincinnati, Ohio, USA. **Gregory A. Norris** is a visiting scientist at the Harvard School of Public Health, adjunct professor at the University of New Hampshire, and president of Sylvatica in North Berwick, Maine, USA. **David W. Pennington** is now a temporary agent with the Soil and Water Unit of the Institute for Environment and Sustainability, a unit of the Directorate General Joint Research Center of the European Union in Ispra, Italy. **Thomas McKone** is senior staff scientist at the Lawrence Berkeley National Laboratory and adjunct professor of public health at the University of California, Berkeley, California, USA.