Improving Environmental Performance Assessment

A Comparative Analysis of Weighting Methods Used to Evaluate Chemical Release Inventories

Michael W. Toffel and Julian D. Marshall

Keywords

emissions
environmental health
environmental management
indicators
metrics
toxic release inventory (TRI)

Summary

Managers, management scholars, regulators, nonprofit organizations, and the media are increasingly using emissions inventory data to measure organizations' environmental performance. Whereas some analysts use total mass emitted, others have applied one or more of the growing number of toxicity-weighting databases aimed at predicting the environmental and health impacts of emissions. Little research is available to guide analysts in selecting among these databases. This article compares 13 methods in terms of their sophistication, complexity, and comprehensiveness. Seven of these methods are then evaluated as to their usefulness in weighting emissions data from the U.S. Environmental Protection Agency's (U.S. EPA's) toxic release inventory, and three pair-wise comparisons are conducted. We recommend the U.S. EPA's Risk Screening Environmental Indicators for estimating impacts to human health. We recommend the Tool for the Reduction and Assessment of Chemical Impacts for estimating impacts to human health and the environment.

Address correspondence to:
Michael W. Toffel
Haas School of Business
University of California
545 Student Services Building #1900
Berkeley, CA 94720 USA
⟨toffel@haas.berkeley.edu⟩

Volume 8, Number 1–2

Introduction

A growing body of research in the management literature investigates the motivations and implications of organizations' environmental management practices and strategies. For example, researchers have recently begun examining the diffusion of voluntary environmental standards such as the International Organization for Standardization (ISO) ISO 14001 standard and the chemical industry's Responsible Care program. Many of these researchers create facility-level measures of environmental performance based on pollutant release and transfer registries (PRTRs) such as the U.S. Environmental Protection Agency's (U.S. EPA's) toxic release inventory (TRI) program. The TRI program is one of many PRTRs that have emerged around the world. Australia, Canada, Korea, the Slovak Republic, and the European Union nations operate PRTRs and publicly disseminate collected data at the facility level (OECD 2001a, 2001b; European Environmental Agency 2003). Facility-level performance metrics are often used to compare the facility's performance over time or relative to other facilities. Developing a meaningful performance metric from TRI data remains problematic, however, for reasons discussed below. The purpose of this article is to assess various weighting methods that have been or could be used to aggregate emissions inventory data and to recommend which schemes are most appropriate for use with TRI data to develop a facility-level environmental performance indicator.

TRI Overview

In the TRI program, the U.S. EPA requires facilities to report releases and transfers of specific chemicals if the facility (1) is primarily engaged in manufacturing, mining, electric utilities, hazardous waste treatment, or chemical distribution; (2) has ten or more full-time employees; and (3) manufactures, imports, processes, or otherwise uses any of the listed toxic chemicals in amounts greater than their threshold quantities (U.S. EPA 2002g). Currently, the TRI program requires companies to report emissions of 609 substances (579 chemicals and 30 chemical categories) to air, surface water, land, and underground injection when their amounts exceed a minimum reporting threshold (U.S. EPA 2002h, 2003c). TRI also requires that companies report off-site transfers (e.g., to waste handlers or waste processors). Compared to other environmental performance information, TRI—like many PRTRs—offers researchers the unique combination of consistently reported facility-level data that are required by regulations, publicly available, and free. TRI applies to a wide array of industries and consists of a panel of thousands of facilities reporting annual data since 1987.

Yet creating an environmental performance metric from TRI data remains problematic for several reasons:

- Data accuracy is uncertain because neither the U.S. EPA nor state environmental protection agencies routinely validate TRI data.[1] One study of 60 facilities in three industries found errors of up to 40% in reported TRI emissions (U.S. EPA 1998a).
- Changes in U.S. EPA instructions contributed to "a significant portion of the reported reductions" in TRI's early years (U.S. GAO 1994, 3).
- Because some TRI data are estimated rather than measured, apparent cross-sectional and longitudinal variations in releases can result from different estimation methods (U.S. EPA 2002k; U.S. GAO 1994).

The accuracy of TRI data remains an open issue, and we mention it only to alert researchers contemplating using this database. The challenges in estimating emissions and characterizing their uncertainty have been discussed by others (Frey and Small 2003). Instead, we focus on a second issue regarding the use of TRI data: aggregation techniques.

Aggregation

The potential harm caused by a particular amount of a chemical released to the environment depends on a number of factors, including the properties of the chemical and the medium to which it is released. Simply summing annual emissions of all TRI substances released by a facility in a given year is a poor proxy for its aggregate potential harm to human health or the environment because the toxicity of TRI chemicals varies over more than 6 orders of magnitude (Horvath et al. 1995). In summary, “mass is a crude proxy for environmental effect” (Lifset 2001, 1).

Unfortunately, this raw summing technique remains a common method among mass media outlets, including the Associated Press, the *Wall Street Journal,* and the *San Francisco Chronicle* (Associated Press Newswires 2002; Kay 2002; Noah 1996; Shields 1999). This technique has also been widely employed in the management literature (e.g., Cohen et al. 1997; Dooley and Fryxell 1999; Eskeland and Harrison 1997; Feldman et al. 1997; Khanna and Anton 2002; Konar and Cohen 2001) and even in leading scientific journals (e.g., Rubin 1999). It continues to be used by several prominent nonprofit organizations, including the Investor Responsibility Research Center, Bridges to Sustainability, and Environmental Defence Canada (Environmental Defence Canada 2002; IRRC 2002), and in government publications, including the U.S. Census Bureau’s *Statistical Abstract of the United States,* the California Environmental Protection Agency’s *Strategic Vision,* and the North American Commission for Cooperation’s annual *Taking Stock* reports (Cal/EPA 2000; CECS 2002; U.S. Census Bureau 2001). As one example, Lucent Technology’s EcoPro, which has been used to evaluate desktop computers’ environmental performance (Caudill et al. 2000), is sufficiently sophisticated to combine environmental impact, resource productivity, and eco-efficiency into one metric to assess performance of products and facilities; however, it uses an unweighted sum of hazardous pollutants (Mosovsky et al. 2000). Finally, Kleijn (2001) pointed out that two major reports by the World Resources Institute (Adriaanse et al. 1997; Matthews et al. 2000) calculated bulk-mass flow analyses by aggregating all materials flows without any weighting scheme. Yet as Kleijn (2001, 8) notes, “bulk-[mass flow analysis] indicators are . . . not very good indicators for environmental pressure, as they ignore differences in the environmental impacts of the materials being accounted.”

The Importance of Weighting Schemes

More rigorous approaches weight toxic emissions in terms of relative harm (e.g., based on their toxicity to humans, relative to a reference chemical) before aggregating them. TRI emissions data should be weighted before comparing the environmental performance between firms or over time (Horvath et al. 1995). Some weighting techniques go further by incorporating the medium of release (e.g., air, water), modeling chemical transport and fate, and assessing exposures. The most sophisticated methods estimate potential population exposure based on the physical and demographic characteristics proximate to the release. No single method has been established as the standard because different approaches trade off particular benefits and drawbacks. Often, implementation of more complex weighting methods is difficult or infeasible due to data and time constraints.

Management scholars often use environmental inventory data to develop firm- or facility-specific measures of environmental performance. The decision of which weighting scheme to use is important because the various schemes differ in their objectives, comprehensiveness, and weighting values. The choice of scheme can lead to different conclusions regarding which substances are most damaging to human and ecosystem health (Pennington and Yue 2000). This choice can affect the results of environmental justice investigations that explore the correlation between socioeconomic status and environmental health burden (Cutter et al. 2002).

Selecting an appropriate weighing scheme is also important in other arenas, such as creating and interpreting life-cycle assessments (LCAs) and implementing design for environment (DfE) objectives. Within the LCA methodology, the weighting schemes discussed in this article are relevant in the characterization component of life-cycle impact assessment (LCIA). LCIA can be viewed as a five-step process: (1) selection of impact categories, (2) classification of resources and releases, (3) characterization of resources and releases to estimate the potential resulting human health and environmental impacts, (4) normalization, and (5) weighting[2] (Guinée 2002; ISO 1997, 2000; Seppälä et al. 2001). In LCA terminology, this article focuses on characterization models, defined as mathematical models of "the impact of environmental interventions with respect to a particular impact category" (Guinée 2002, 92), for two impact categories: human health and ecosystem quality. LCIA uses "measures of hazard to compare the relative importance of pollutants within a defined impact category" (McKone and Hertwich 2001, 106).

In this article, we compare several weighting methods and assess their value for use with TRI data. Previous work has highlighted general issues associated with weighting schemes. Hertwich and colleagues (2002) described three levels of sophistication in models that assess the fate and exposure of toxic chemicals and compare multimedia to single-medium models. Various frameworks have been proposed to distinguish among available methodologies to weight toxic chemicals (e.g., Bengtsson and Steen 2000; Hertwich et al. 1998). Bengtsson and Steen (2000) categorized weighting methods as "distance to target," where weights are higher for substances as they approach critical health or environmental levels, or "damage models" that address impacts on ecosystems, human health, and nonrenewable resources. Hertwich and colleagues (1998) categorized weighting methods based on the extent to which they incorporate toxicity and three exposure factors: persistence, fate, and exposure pathways. Krewitt and colleagues (2002) described advantages and drawbacks of several potency- and severity-based methods to characterize human health impacts resulting from exposure to toxic chemical releases and evaluated their suitability for LCIA. Others have evaluated the appropriateness of various weighting schemes to environmental management (Steen 1999), product design (Yarwood and Eagan 2001), and LCA (Hauschild and Pennington 2002a).

The literature on this topic contains only a few in-depth comparisons of a few methods. For example, scores from the U.S. EPA's waste minimization prioritization tool (WMPT), which reflect chemicals' persistence, bioaccumulation, and toxicity, have been compared to toxic equivalency potentials, which incorporate chemical fate, multipathway exposure, and toxicity (Pennington and Bare 2001). The study reported that although there were significant structural differences between the methods and the values for specific chemicals differed substantially, overall results from implementing the two methods were strongly correlated. They concluded that results of both methods generally agree. This finding is consistent with another study, wherein four weighting schemes (EcoIndicator95, Eco-Indicator99, EcoPro, and Ecological Footprint) were used to evaluate the relative environmental impact of three products (Luo et al. 2001). In each of the three cases evaluated, the four schemes yielded markedly different values, yet all four agreed on which product had less impact. Others have found little or no correlation between various schemes (Hofstetter et al. 2000), suggesting that different schemes can lead to widely varying results. This observation emphasizes the importance of selecting a weighting scheme that is appropriate to the particular analysis being conducted. A study of six schemes concluded that each scheme (1) had the potential to yield incorrect evaluations and (2) required data that an analyst might not be able to acquire at the necessary level of accuracy (Hertwich et al. 1997). By applying four increasingly sophisticated risk assessment methods to TRI data, Zhang and colleagues (2001) showed that site-specific exposures and chemical-specific fate estimations can be important for pollution prevention decision making.

This article extends prior research by examining 13 weighting methods, many of which have been developed over the past decade. We highlight sources of the data underlying each model to aid researchers. We conclude by recommending methods to aggregate TRI data. Although our results may be useful in a number of settings, our target application is the use of TRI emissions data to generate environmental performance indicators for facilities located in the United States. We assume the analyst does not intend to conduct environmental fate and transport modeling.

Methods

Selection of Weighting Methods

Literally hundreds of ways exist to quantify the potential harm caused by releasing chemicals into the environment.[3] Because this article focuses on toxic impacts, we have not evaluated weighting methods strictly for issues such as climate change or stratospheric ozone depletion. The motivation for this article is to evaluate weighting schemes for use in future research, especially related to TRI emissions data. Thus, we only evaluate databases for which the data and documentation are readily available.

Based on these criteria, we compiled a list of 13 weighting schemes (see table 1) that have been or could be used to weight TRI and other toxic release data. This list came from a literature review of leading journals in the fields of environmental science (e.g., *Environmental Science and Technology, Environmental Toxicology and Chemistry, Environmental Progress*), industrial ecology (e.g., *Journal of Industrial Ecology, International Journal of Life-Cycle Assessment*), and business management (e.g., *Academy of Management Journal, Production and Operations Management*).

We have omitted several schemes that did not meet our selection criteria. Four examples are, first, the Pratt Index (also known as the "hazardous chemical pollutant index") (Pratt et al. 1993), which has been used recently in environmental justice analyses in Minnesota (Sheppard et al. 1999) and South Carolina (Cutter et al. 2002). We exclude it from our analysis because the scheme is not readily available and because the developer, the Minnesota Pollution Control Agency, is no longer using this method owing to lack of funding to update the data (Pratt 2002). Second, although we found the environmental burden approach (Clift and Wright 2000; ICI 1997; Wright et al. 1997) to be compelling, we omitted it because it contains values for only the small number of TRI chemicals emitted by the ICI Group. We excluded the WMPT, which was developed by U.S. EPA to prioritize hazardous chemicals and focus waste minimization program initiatives. The WMPT's values were never finalized, it is no longer supported by the U.S. EPA (U.S. EPA 2002l), and its current set of waste management priority chemicals is based on a method that is not publicly available. Finally, although we are impressed with the EPA's persistent bioaccumulative and toxic (PBT) profiler (U.S. EPA 2002k), it is too narrowly focused for the objectives of this article. The PBT profiler is a "screening tool . . . for chemicals without experimental data" (Environmental Science Center 2003), and it aims to identify persistent and bioaccumulative toxic compounds. Consistent with this scope, its toxicity values are for chronic aquatic ecosystem toxicity.

Below, we describe a framework others have proposed to compare schemes' complexity and realism. Subsequently, we describe each weighting scheme.

Model Complexity and Realism

Several researchers have used a four-tiered hierarchy of increasing complexity and realism in modeling the true potential environmental and human health impacts of chemical releases (e.g., Hertwich et al. 1998; Jia et al. 1996). Each increasing level adds complexity to the more basic models.

Level 1: Toxicity. These schemes account for the toxicity and mass of emissions. Toxicity values typically quantify three types of risk from chemical exposure: chronic exposure that may cause cancer; chronic exposure that may cause noncancerous health impacts, such as reproductive or neurological disorders; and acute exposure that may cause noncancerous impacts, such as kidney damage. Both increased emissions and increased toxicity increases the potential harm caused by a chemical.

Level 2: Persistence. The persistence of a chemical in the environment is typically incorporated in these schemes by tracking the characteristic time for removal and degradation of a compound. More persistent chemicals, which take longer to degrade or otherwise be removed from ecosystems, have a greater potential to cause harm.

Level 3: Concentrations. These schemes predict chemical concentrations in various media (e.g., air, water, soil) and model transfers of chemicals between media. For example, these models would estimate the transport of a chemical released to air into surface water and soil.

Level 4: Intake. These methods account for exposures to environmental media. For example, intake of an airborne chemical can occur directly via inhalation and, if the chemical deposits to surface water, it could be ingested via drinking water or by eating fish. The most advanced schemes incorporate site-specific climate and geography into fate and transport models, and demographic surveys into population exposure calculations.

Some authors identify level "zero" as evaluating mass emissions without a weighting scheme (Fava et al. 1992; Pennington and Yue 2000). The above levels of sophistication can be applied to schemes that use generic environmental conditions, site-specific data, or both (Pennington and Yue 2000). When employing a weighting method, analysts typically provide the mass emission rate of various chemicals released to each medium (e.g., from TRI data) and the weighting method database provides weightings for each chemical and sometimes for each release medium.

In addition to the four-level framework provided above, there are several other dimensions of complexity and realism that can be included in weighting methods. For example, some methods only characterize toxicity in terms of chronic cancer potency, whereas others also include chronic noncancer potency and acute potency. Some methods characterize human toxicity, whereas others characterize ecosystem toxicity. Several schemes make a simplification by characterizing chemical fate, transport, and toxicity only in terms of inhalation of pollutants released to air. This simplification ignores the reality that chemicals are released to air, water, underground injection wells, and soil; that emissions can migrate between media; and that pollution can be taken into the body via multiple pathways, including ingestion of food and water and dermal absorption of pollutants in soil and water.

Description of Schemes Evaluated

Table 1 lists the developer of each of the 13 schemes, a few examples of how each has been applied, and where each database can be obtained. Each scheme is described below.

Human Toxicity Potential

The Human Toxicity Potential (HTP) was developed to rank toxic emissions, such as from TRI and LCA data (Hertwich et al. 2001; Hertwich et al. 1998; McKone and Hertwich 2001). HTP values are based on the CalTOX model, which uses multimedia fate and transport to estimate intake via inhalation, ingestion, and dermal absorption (DTSC 1993a, 1993b; McKone et al. 2002; McKone and Enoch 2002). HTPs incorporate the different toxicities implicit in these different exposure routes. Up to four values are provided for each substance to reflect its relative cancer and noncancer health impact from releases to air and water. CalTOX is a sophisticated and comprehensive multimedia fate and transport model, and HTPs have been used to weight TRI emissions (e.g., Hertwich et al. 2001).

Indiana Relative Chemical Hazard Score

The Indiana Relative Chemical Hazard Score (IRCHS) scheme has values for many more chemicals than the HTP, but it is an imprecise measure of potential damage. It generates a worker exposure hazard score and an environmental hazard value score based on various categorizations of a chemical, such as the number of regulatory lists it has been placed on (CMTI 2001).[4] For example, the "air" component of the IRCHS environmental hazard value is the sum of the points assigned if the chemical is a criteria pollutant (20 points), a hazardous air pollutant (40 points), a high-risk pollutant (20 points), and/or an extremely hazardous substance (20 points). Three weighting schemes within IRCHS represent worker hazard, environmental hazard, and a combined (worker plus environment) hazard.

Risk-Screening Environmental Indicators

The U.S. EPA's Risk-Screening Environmental Indicators (RSEI) model incorporates the following: year-1988 and later TRI emissions for all U.S. facilities; toxicity for 425 chemicals and chemical categories; fate, transport, and exposure modeling; and size of the exposed population (Bouwes and Hassur 1997; U.S. EPA 2002j, 2003b).[5] RSEI incorporates a combination of site-specific factors and industry-specific generic factors (e.g., air modeling uses the facility's stack height, if known, or else the mean of known stack heights for the facility's three-digit standard industry code) to calculate comparative, risk-related scores. The U.S. EPA has developed one metric based on RSEI so far, the chronic human health indicator (CHHI). RSEI's hazard-based perspective (Pounds × Toxicity Weight) incorporates releases to air, water, land, and underground injection.

RSEI's risk-based perspective calculates certain components of multimedia fate, transport, and exposure pathways. For example, it includes ingestion of fish from recreational and subsistence fishing but not ingestion of agricultural products. RSEI includes evaporation of volatile compounds from publicly owned water treatment facilities but not from industrial wastewater, and it does not include deposition of air emissions. RSEI provides a surrogate dose (via the inhalation and ingestion pathways) for releases to air and water, but it does not model exposure to releases to land or resulting contaminated groundwater. The U.S. EPA is currently developing its risk-based modeling to include releases to land (Hassur 2003). RSEI has been used recently to assess the relative health risk to Oakland residents resulting from manufacturing facilities reporting TRI data (Costa et al. 2002). The model is unique in its inclusion of site-specific exposure and population characteristics (e.g., age and gender) by incorporating U.S. census block-level data. This scheme generates a unique weighting value for each combination of facility, year, chemical, release or transfer pathway, and exposure pathway.

EcoIndicator99

EcoIndicator99 is a compilation of several frameworks that track various environmental and human health impacts (e.g., impacts of ozone depletion on human health). EcoIndicator99 employs the European Union system for the evaluation of substances to model the fate of substances with carcinogenic or ecotoxicological effects (Goedkoop and Spriensma 2000). EcoIndicator99 builds on EcoIndicator95, an earlier model developed by the same researchers, in three ways (Goedkoop 1998). First, disability-adjusted life years (see Murray and Lopez 1996) are adopted as a common unit for quantifying diverse impacts to human health. Unlike most schemes, this provides an absolute rather than a relative measure of impact. Second, environmental damage is extended to include resource depletion due to mineral and fossil fuel consumption and damage to ecosystem quality due to emissions and land-use changes. Third, the methods and the input data are updated to incorporate recent research. EcoIndicator99 explicitly recognizes the need to incorporate user ethical values into weighting various environmental impacts, and it presents several versions. In the quantitative analysis below, we employ the default version of the model, which attempts to use weighting values based on scientific consensus.[6] For EcoIndicator99, we examine three types of indicators: respiratory effects on humans from releases to air, carcinogenic effects on humans from releases to air or water, and ecosystem quality damage due to releases to air or water. Values specific to releases to air, water, and soil are available for the latter two indicators, whereas only values pertaining to air are available for the former indicator.

Environmental Design of Industrial Products

Begun in 1991, the Environmental Design of Industrial Products (EDIP) project seeks to develop methods to consider environmental aspects in product design. The resulting LCA weighting scheme includes an impact assessment stage that incorporates seven impact assessment categories: global warming, stratospheric ozone depletion, photochemical ozone formation, acidification, nutrient enrichment, eco-toxicity, and human toxicity via environmental exposure (Wenzel et al. 1997). We focus on the latter two categories. EDIP provides values that reflect the relative eco-toxicity in water and soil resulting from the release of 71 substances to air, water, and soil, as well as their relative toxicity to microorganisms in sewage treatment plants from

Table 1 Schemes evaluated

Scheme name and location of database	*Developer*	*Sample applications as a weighting scheme*
Human toxicity potential (HTP), ⟨http://design.ntnu.no/ansatte/hertwich/HTP_ETC.html⟩ and ⟨http://eetd.lbl.gov/ied/era/⟩	University of California, Berkeley and the Lawrence Berkeley National Laboratory Environmental Energy Technologies Division	Environmental Defense's internet-based tool (⟨www.scorecard.org⟩) (Environmental Defense 2001) and U.S. Environmental Protection Agency's (U.S. EPA's) tool for the reduction and assessment of chemical and other environmental impacts (Bare et al. 2002)
Indiana Relative Chemical Hazard Score (IRCHS), ⟨www.ecn.purdue.edu/CMTI/⟩	Purdue University's Indiana Clean Manufacturing Technology and Safe Materials Institute	Indiana's Pollution Prevention and Toxic Release Inventory annual report (Davis et al. 2000), the National Organic Standards Board materials database (NOSB 2000), and Environmental Defense's Internet-based tool (⟨www.scorecard.org⟩) (Environmental Defense 2001)
Risk-Screening Environmental Indicators (RSEI), ⟨www.epa.gov/oppt/rsei/⟩	U.S. EPA Office of Pollution Prevention and Toxics	Maine's Department of Environmental Protection uses RSEI in its toxic use reduction program to prioritize site visits and assistance efforts (Dyer 2003); the chronic human health indicator has been used to assess the relative toxicity of facilities' chemical releases reported to Canada's National Pollutant Release Inventory (Antweiler and Harrison 2003; Harrison and Antweiler 2003)
Ecoindicator99, ⟨www.pre.nl/eco-indicator99/⟩	Pré Consultants	An earlier version, Ecoindicator95, has been used in research on design for environment (Lee et al. 2001; Vogtlander et al. 2002) and life-cycle assessment (Brentrup et al. 2001)

Environmental Design of Industrial Products (EDIP), (Hauschild et al. 1998a, 1998b; Wenzel et al. 1997)	The Danish Environmental Protection Agency, the Technical University of Denmark, Confederation of Danish Industries, and five Danish companies	Used to develop a decision model for material management of end-of-life products (Yu et al. 2000) and normalization references and weighting factors for three regions in China (Yang and Nielsen 2001)
Tool for the Reduction and Assessment of Chemical Impacts (TRACI), ⟨www.epa.gov/ORD/NRMRL/std/sab/download2.htm⟩	U.S. EPA Office of Research and Development	Released in 2002; no applications outside U.S. EPA were identified for this tool
Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) Reportable Quantities (RQs),[1] ⟨www.epa.gov/ceppo/ds-epds.htm⟩	U.S. EPA Office of Emergency and Remedial Response and U.S. EPA Office of Solid Waste	Several articles in the management literature that analyze the environmental performance of companies use the inverse of RQ (Harrison 2002; King and Lenox 2000, 2001a, 2001b; King and Shaver 2001; Lenox 2001; Lenox and Nash 2003; Russo 2002)
Threshold Limit Value–Time Weighted Average (TLV-TWA),[2] ⟨www.worldbank.org/nipr/data/toxint/index.htm⟩	American Conference of Governmental Industrial Hygienists (ACGIH)	Used to weight toxic release inventory (TRI) emissions in an economic input-output LCA tool (Horvath et al. 1997), by the World Bank for its toxic intensities database (World Bank 2002), and by the hazards research lab at the University of South Carolina (Hazards Research Lab 1997)
Minimal risk levels (MRLs) ⟨www.atsdr.cdc.gov/mrls.html⟩	Agency for Toxic Substances and Disease Registry	—

Table continued on next page

Table 1 *(continued)*

Short-Term Exposure Limit (STEL) ⟨toxnet.nlm.nih.gov⟩	U.S. Occupational Safety and Health Administration (OSHA), U.S. National Institute for Occupational Safety and Health (NIOSH), and ACGIH	—
Permissible Exposure Limit (PEL),[3] ⟨www.osha-slc.gov/SLTC/pel/⟩	U.S. OSHA	Used to relate carcinogenic and noncarcinogenic pollutants (Tellus Institute 1992)
		Used to weight emissions within Brazil's municipalities (Dasgupta et al. 1998)
Reference Exposure Levels (RELs), Acute RELs for Airborne Toxicants, Chronic RELs for Airborne Toxicants, ⟨www.cdc.gov/niosh⟩ and ⟨www.oehha.org/air.html⟩	U.S. NIOSH and California Office of Environmental Health Hazard Assessment (OEHHA)	Used to weight emissions within Brazil's municipalities (Dasgupta et al. 1998) and widely used in environmental health risk assessments in the United States
Cancer Unit Risk Potency Factors, ⟨www.epa.gov/iris/index.html⟩ and ⟨www.oehha.ca.gov⟩	California OEHHA and U.S. EPA Integrated Risk Information System (IRIS)	Used to weight TRI emissions (Hamilton 1999) and widely used in environmental health risk assessments in the United States

[1] RQ values are also available in the U.S. Code of Federal Regulations (U.S. CFR 2003a and U.S. CFR 2003b). In addition, RQs for extremely hazardous substances are available from the U.S. EPA (U.S. EPA 1998b).

[2] TLV values are also available from the International Labor Organization (ILO 1991). TLVs for hazardous air pollutants are available from the U.S. EPA (U.S. EPA 2002a).

[3] PELs are also listed in the U.S. Code of Federal Regulations (U.S. CFR 2003c).

emissions to wastewater treatment plants (Hauschild, Wenzel et al. 1998; Wenzel et al. 1997). These values incorporate each substance's toxicity, biodegradability, and dispersion in the environment. Human toxicity potential values quantify the relative toxicity from exposure via air, surface water, and soil pathways resulting from emissions of 100 substances to air, water, and soil (Hauschild, Olsen et al. 1998; Wenzel et al. 1997). Nine human toxicity potential values are generated for each substance. The Danish Environmental Protection Agency released a beta version of the EDIP personal computer tool in 1998, but a finalized version is not yet available.

Tool for the Reduction and Assessment of Chemical Impacts

According to its developers, the Tool for the Reduction and Assessment of Chemical Impacts (TRACI) is "a software tool that allows the evaluation of the environmental and human health impacts associated with the raw material usage and chemical releases resulting from the processes involved in producing a product. TRACI allows one to examine the potential for impacts for a single life-cycle stage, or the whole life cycle, and to compare the results between products or processes" (U.S. EPA 2002d). TRACI provides several environmental impact categories. Eco-toxicity, eutrophication, human health cancer, and human health noncancer are available for releases to both air and water (Bare et al. 2002; Norris 2002). For human health, TRACI employs the CalTOX HTP values discussed above. For releases to air, TRACI calculates additional environmental impact categories, including ozone depletion, global warming, acidification, and photochemical smog. Fossil fuel depletion, land use, and water intake are also incorporated (U.S. EPA 2002d, 2002e, 2002f, 2002i). We focus on TRACI's values that estimate potential impacts to terrestrial and aquatic ecosystems, which are calculated separately for emissions to air and water.

Comprehensive Environmental Response, Compensation, and Liability Act Reportable Quantities

This scheme was established by the U.S. Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) (commonly known as "Superfund"). The U.S. EPA requires immediate notification to local authorities should a listed substance be released into the environment in an amount beyond its reportable quantity (RQ).[7] The U.S. EPA establishes these values based on a chemical's aquatic toxicity, acute mammalian toxicity (oral, dermal, and inhalation), ignitability, reactivity, chronic toxicity, and potential carcinogenicity, as well as its susceptibility to degradation via biodegradation, hydrolysis, and photolysis (U.S. EPA 2002b). CERCLA RQ values derive from several other regulations, including the Clean Water Act, the Clean Air Act, and the Resource Conservation and Recovery Act.[8] For simplicity, there are only five RQ values (1, 10, 100, 1,000, and 5,000 lb), and they are not media specific. King and Lenox (2000) used the inverse of RQ values as a proxy for the environmental harm of TRI emissions to air. This method has been employed in much of the subsequent management research that uses TRI data (e.g., Harrison 2002; King and Lenox 2001a; King and Shaver 2001; Lenox 2001; Lenox and Nash 2003; Russo 2002).

Toxicity Schemes

The remaining entries in table 1 are toxicity values that do not incorporate fate and transport. These include the following:

- Threshold Limit Value–Time-Weighted Average (TLV-TWA)
- Minimal Risk Levels (MRLs) (ATSDR 2002)
- Short-Term Exposure Limit (STEL)
- Permissible Exposure Limit (PEL) (U.S. OSHA 2001)
- Acute Reference Exposure Levels (acute RELs) for Airborne Toxicants (Alexeeff et al. 1999)
- Chronic Reference Exposure Levels (chronic RELs) for Airborne Toxicants (Alexeeff et al. 2000)
- Cancer Unit Risk Potency Factors (U.S. EPA 2002c)

Three of these seven toxicity values (TLV-TWA, STEL, and PEL) are intended for worker protection. Toxicity schemes have been employed frequently to estimate the relative impact of emissions, as noted in table 1.

Evaluation

Various characteristics are important in evaluating weighting schemes. Hauschild and Pennington (2002b) listed seven criteria for evaluating eco-toxicity indicators: (1) scientific validity of the approach, assumptions, and interpretation; (2) relevance to environmental impact or damage; (3) reproducibility of values; (4) transparency of the method and how it addresses uncertainty; (5) extent to which practitioners can comprehend the method; (6) feasibility of calculating values for all relevant substances; and (7) availability of data required to calculate values.

We evaluate the schemes in table 1 based on two criteria: First, schemes that incorporate the complexities of fate, transport, toxicity, and exposure are more likely to produce values that accurately reflect actual impacts to human health and the environment. This emphasis on realism is consistent with the modeling hierarchy proposed by Hertwich and colleagues (1998) and by Jia and colleagues (1996). Second, the scheme should be comprehensive, including as many TRI chemicals as possible.

Model Complexity and Realism

We group the schemes into four categories of increasing complexity and realism. For categories 2 through 4, we identify which pollutant release media each scheme is designed for and which exposure pathways are incorporated. We also note whether the scheme's fate, transport, and exposure modeling incorporate location-specific data. Finally, we identify the number of values each scheme generates for each substance and indicate whether the scheme generates relative (unitless) values or absolute values. Table 2 presents the results of this analysis.

Category 1

Category 1 includes schemes that reflect toxicity to workers and schemes that do not characterize fate and transport. The former are not designed to measure the impact of releases to the environment, whereas the latter are less sophisticated and less realistic than the schemes in the other three categories. Several schemes are in category 1, including IRCHS-worker, TLV-TWA, STEL, PEL, MRLs, acute RELs, Chronic RELs, and Cancer Unit Risk Potency Factors.

Category 2

Category 2 contains two schemes that are based on regulations rather than fate and transport models: RQ and IRCHS. As these are not explicitly based on fate and transport models, these schemes are not recommended as toxicity weights (Hertwich et al. 1998; Zhang et al. 2001), although they may well be useful in other contexts such as measuring relative regulatory scrutiny. RQs are problematic as toxicity weights for several additional reasons. First, being divided into only five discrete values reduces their precision as a measure of relative harm. Second, only one RQ value is available for each chemical regardless of the medium to which it is released (e.g., air, water). We elaborate further on this concern below. Third, it is difficult to determine whether any particular RQ value was established because of the chemical's aquatic toxicity, acute toxicity, chronic toxicity, ignitability, reactivity, or potential carcinogenicity. This is particularly problematic because only one RQ value exists per substance, regardless of release medium. Consequently, applying RQs to substances released to air is likely to result in some of the weights being determined by aquatic toxicity, a poor proxy for human health or eco-toxicity impairment resulting from releases of toxic chemicals to air.

IRCHS values seem well designed to provide an indicator of regulatory scrutiny. This may be useful for prioritizing compliance management, a key component of environmental management systems. We believe this scheme is less appropriate, however, as a weighting factor for relative potential to impact human health or the environment. Perhaps the greatest advantages of IRCHS are that its approach is straightforward and easy to understand and that it has values for a large number of TRI chemicals. Similarly, RQs offer the advantage of being straightforward and offering wide TRI coverage. RQs are useful for ranking chemicals in terms of the severity of a potential accidental release (e.g., an accidental spill).

Category 3

Category 3 includes the two schemes (HTP and RSEI) that estimate human health impacts based on fate and transport modeling. HTP is a significant improvement over schemes based solely on toxicity (e.g., TLVs) or on proxies for toxicity (e.g., IRCHS and inverse RQs). RSEI offers the distinct advantage of incorporating site-specific information, such as the population density near an emission source.[9] Although it incorporates a less detailed multimedia fate and transport model than HTP, this disadvantage can be viewed as a trade-off with the benefit of using site-specific data, such as population density. Unlike most schemes, RSEI software integrates several datasets including emissions, toxicity, geography, and population, thereby offering the potential to reduce significantly the resources and effort needed to pose and answer research questions. RSEI is an excellent tool worthy of greater attention by scholars of environmental management.

Category 4

The final category, category 4, contains the three schemes that estimate impact to human health and to the environment (TRACI, EcoIndicator99, and EDIP). These schemes address stratospheric ozone depletion, global warming, land-use changes, acidification, eutrophication, and photochemical ozone formation. Some also model resource depletion. All three schemes use a multimedia fate and transport model to estimate impacts to human health. In aggregating various environmental impacts, EcoIndicator99 is a flexible platform that explicitly incorporates the user's environmental values, whereas TRACI offers a single value for each chemical. With seven values pertaining to eco-toxicity and nine to HTP, EDIP offers the most values per substance. EDIP also provides a method for researchers to develop reliable values for additional substances. A recent study commissioned by the Danish Environmental Protection Agency, one of the co-creators of EDIP, reported that most of the dozen international LCA experts surveyed found EDIP to be the most advanced, complete, and consistent LCA method available (Sørensen 2002).

Comprehensiveness

The TRI program currently consists of 609 substances, which we refer to as "current TRI substances." The U.S. EPA has added many of these substances over the past few years. Analysts who use time series analysis to evaluate changes of facilities' TRI emissions often use the subset of 312 of these substances that were initially required by the TRI program in 1987 (U.S. EPA 2003c). We refer to this group as the "original TRI substances still in the program." Figure 1 presents the number of TRI substances for which values exist under the various forms of each scheme, along with the proportion of coverage of original and current TRI substances.

Note that IRCHS environmental scores, RSEI CHHI, and RQ are the most comprehensive schemes, as they cover the largest proportion of current and original TRI substances. The problems associated with limited coverage are addressed below in the discussion section.

EcoIndicator and EDIP, schemes with complex models, cover less than 10% of current TRI substances. These models require a great deal of data to produce values for a substance. In addition, these schemes were developed in Europe and were thus not targeted toward TRI substances, as were schemes such as RSEI and HTP.

Comparative Analysis

Table 3 presents pair-wise Pearson correlation coefficients between the schemes' values and the number of observations used to calculate each correlation coefficient. Correlations among all values common to both schemes are presented in columns labeled "a." The "b" columns present correlations for only consistently reported TRI substances. By examining Pearson correlation coefficients, we measure the extent to which the relationship between any two schemes is linear. The lower the correlation, the greater the chance that using the two different schemes is likely to yield different results. For example, the first panel in table 3 shows that RSEI CHHI inhalation toxicity factors (used in the RSEI hazard model) are correlated with HTP cancer air values (which TRACI uses to assess human health impacts), suggesting that using either scheme may yield similar results. In contrast, the middle panel illustrates that inverse RQ values are not correlated with EcoIndicator99's or HTP's cancer water values, suggesting that these schemes are

Table 2 Scheme complexity and realism

Scheme[a]	*Release media*	*Toxicity*[b]	*Fate and transport modeling*	*Exposure pathways*	*Location-specific exposure modeling*	*Values per substance*	*Values specific to release media?*	*Relative value or units*	*Model or Score?*[c]
HTP	Air, water	C, NC	Yes	Ingestion, inhalation, and dermal absorption	No	Two: cancer and chronic noncancer	Yes	Relative	Model
IRCHS	Air, water, soil	n/a	No	No	No	Two: environmental hazard value and worker exposure hazard value	No	Relative	Score
RSEI	Air, water, transfers to POTWs[d]	C, NC	Yes	Inhalation and ingestion[e]	Yes	One per facility-year-release medium-exposure pathway	Yes	Relative	Model
EI99	Air, water, soil	C, NC, E	Yes	Inhalation and ingestion	No	Up to 11[f]	Yes	Absolute[g]	Model
EDIP	Air, water, soil	A, C, NC, E	Yes	Ingestion and inhalation[h]	No	Up to 16[i]	Yes	Relative	Model
TRACI	Air, water	C, NC, E	Yes	Ingestion, inhalation, and dermal absorption	Yes[j]	Up to 7[k]	Yes	Relative	Model
Inverse RQ	Air, water	A, C, NC, E[l]	No	None	No	One	No	Relative	Score

[a]HTP = Human Toxicity Potential; IRCHS = Indiana Relative Chemical Hazard Score for Environment; RSEI = Risk-Screening Environmental Indicators; EI99 = EcoIndicator99; EDIP = Environmental Design of Industrial Products; TRACI = Tool for the Reduction and Assessment of Chemical Impacts; RQ = Comprehensive Environmental Response, Compensation, and Liability Act Reportable Quantity.

[b]Toxicity types: A = acute; C = cancer; NC = chronic noncancer; E = eco-toxicity.

[c]IRCHS scores are based on the regulatory attention given to a chemical. RQs sort chemicals to one of five categories based on release reporting requirements. The other five schemes use models to estimate each chemical's impact.

[d]POTWs = publicly owned treatment works.

[e]For releases to air (stack and fugitive) and surface water, RSEI models both (1) inhalation exposure from air releases and (2) ingestion of drinking water and fish from surface water releases. For releases to land or underground injection, RSEI does not explicitly model exposures, but rather uses toxicity to estimate environmental hazard.

[f]Six relate to human health: carcinogenic effects on humans, respiratory effects on humans caused by organic substances, respiratory effects on humans caused by inorganic substances, damages to human health caused by climate change, human health effects caused by ionizing radiation, and human health effects caused by ozone layer depletion. Three relate to different causes of ecosystem quality damage: eco-toxic emissions, the combined effect of acidification and eutrophication, and land occupation and land conversion. Two relate to different causes of damage to resources: extraction of minerals and extraction of fossil fuels.

[g]Human health damage is measured using the disability-adjusted life years scale. Eco-toxicity is measured using the potentially affected fraction of species as a function of the concentration of toxic substances. Acidification and eutrophication are measured using 1 minus the probability that a plant species still occurs in an area to calculate the potentially disappeared fraction.

[h]In addition to inhalation, six ingestion exposure routes are modeled, two of which are direct (groundwater and soil) and four of which are indirect (fish/shellfish, plants, animals, and milk).

[i]Substances may have as many as eight eco-toxicity values and nine human toxicity values. The eco-toxicity values are water chronic and air chronic for emissions to air, water acute and chronic and soil chronic for emissions to water, chronic water and chronic soil for emissions to soil, and eco-toxicity to sewage treatment plant microorganisms for emissions to wastewater treatment plants. The nine human toxicity values are for exposure via air, surface water, and soil to substances emitted to air, water, or soil.

[j]U.S. region-, state-, and /or county-level values are available for acidification, photochemical smog, eutrophication, human health criteria, and land use.

[k]Substance may have one or more values reflecting their relative damage with respect to human health cancer, human health noncancer, eco-toxicity, acidification, eutrophication, global warming, and photochemical smog.

[l]RQ values are based on aquatic toxicity, acute and chronic toxicity, ignitability, reactivity, and potential carcinogenicity.

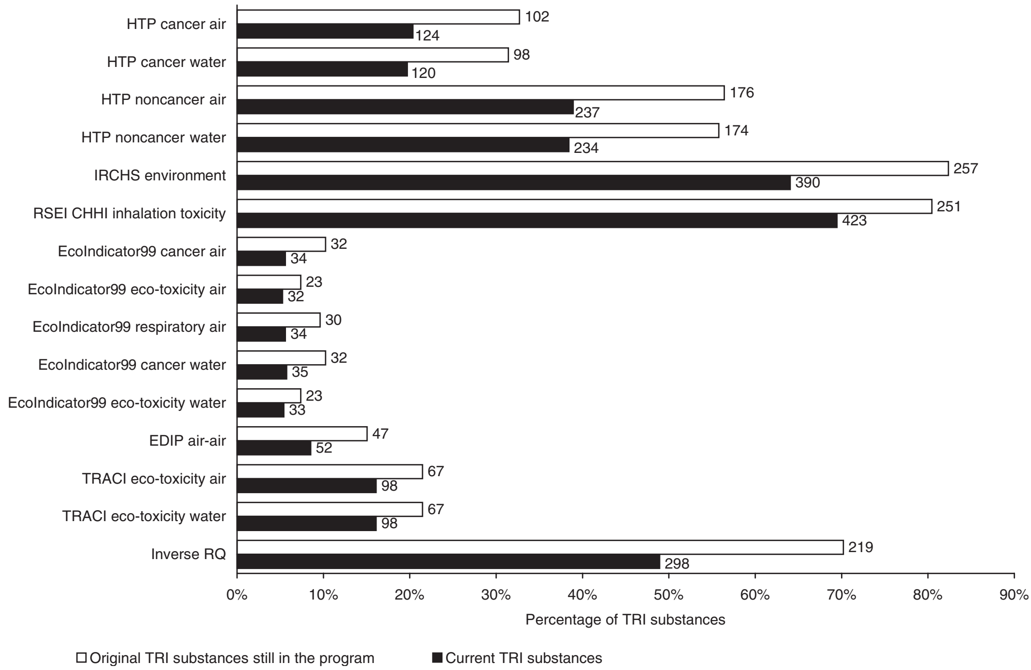

Figure 1 The number and proportion of Toxic Release Inventory (TRI) substances for which values are available from each scheme varies widely. The white bars indicate the percentage of the original 312 substances still in the TRI program for which each scheme has values. The labels refer to the number of values the scheme has pertaining to this set of chemicals. The black bars and labels convey the analogous information pertaining to the 609 TRI substances currently in the TRI program. For example, the Human Toxicity Potential cancer scores for releases to air (HTP cancer air) include 102 values pertaining to the original TRI substances still in the program. This constitutes 33% of those 312 substances. This weighting method includes 124 values that pertain to the 609 current TRI substances, representing 20% coverage. Abbreviations are as follows: HTP = Human Toxicity Potential; IRCHS = Indiana Relative Chemical Hazard Score for Environment; RSEI = Risk-Screening Environmental Indicators; CHHI = Chronic Human Health Indicator; EDIP = Environmental Design of Industrial Products; TRACI = Tool for the Reduction and Assessment of Chemical Impacts; RQ = Comprehensive Environmental Response, Compensation, and Liability Act Reportable Quantity.

likely to yield dissimilar results. Eco-toxicity values are presented in the bottom panel for EcoIndicator99, IRCHS, EDIP, TRACI, and inverse RQ. Among the ten comparisons in the bottom panel, only TRACI and EDIP are correlated.

Below, we present three pair-wise comparisons that may be of interest to the reader. First, we compare HTP air to inverse RQ values. We provide this comparison because of the recent trend among management scholars to weight TRI data using inverse RQ values and because HTP values incorporate multimedia fate and transport. Second, we compare the two environmental impact schemes offering the most comprehensive coverage of TRI: TRACI and IRCHS environment values. Third, we compare RSEI's two scoring systems to explore how its risk scores, which incorporate both relative population exposure and relative toxicity, differ from the hazard scores that only incorporate the latter.

Comparing HTP Air and Inverse RQ Values

Box-and-whisker plots illustrating the distribution of HTP noncancer air (NCA) and cancer

Table 3 Pair-wise Pearson correlations among schemes

	1		2		3		4		5	
	a	*b*	*a*	*b*	*a*	*b*	*a*	*b*	*a*	*b*
Human toxicity schemes, releases to air										
1 HTP cancer air	1.00	1.00								
	157	102								
2 HTP noncancer air	1.00	−0.02	1.00	1.00						
	121	82	314	176						
3 EcoIndicator99 cancer air	−0.05	0.01	0.04	0.04	1.00	1.00				
	35	30	32	30	43	33				
4 RSEI CHHI inhalation toxicity	0.73	0.73	0.01	0.01	0.16	0.16	1.00	1.00		
	115	94	230	172	35	33	425	251		
5 Inverse RQ	0.17	0.32	0.16	0.21	0.31	−0.03	0.32	0.38	1.00	1.00
	122	94	217	142	36	31	245	189	770	219
Human toxicity schemes, releases to water										
1 HTP cancer water	1.00	1.00								
	149	94								
2 HTP noncancer water	0.999	0.89	1.00	1.00						
	113	74	309	174						
3 EcoIndicator99 cancer water	0.38	0.92	0.22	0.22	1.00	1.00				
	31	26	32	30	41	33				
4 Inverse RQ	0.17	0.29	0.16	0.27	0.34	0.19	1.00	1.00		
	118	90	214	140	36	31	770	219		
Eco-toxicity schemes, releases to air										
1 EcoIndicator99 ecotoxity air	1.00	1.00								
	46	23								
2 IRCHS environment	−0.13	−0.10	1.00	1.00						
	35	22	1122	257						
3 EDIP air-air	−0.04	0.01	0.11	0.10	1.00	1.00				
	10	8	77	47	100	47				
4 TRACI eco-toxicity air	−0.05	−0.05	0.05	−0.01	0.997	0.01	1.00	1.00		
	31	19	115	65	29	25	161	67		
5 Inverse RQ	0.09	−0.18	0.17	0.28	0.43	0.08	0.20	0.31	1.00	1.00
	30	21	734	218	59	42	101	61	770	219

Note: The “a” columns present pair-wise Pearson correlation coefficients above the number of observations for all weighting values promulgated by both schemes, even for substances not in the U.S. EPA TRI program. The “b” columns present these figures for the subset of weighting values that pertain to original TRI substances still in the program. The high correlations between HTP cancer and noncancer values, and between EDIP and TRACI values, are attributable to one substance (2,3,7,8-TCDD, Chemical Abstracts Service #1746016), whose weighting values are orders of magnitude larger than all others in each of these schemes.

air values for each inverse RQ value are presented in figures 2 and 3, respectively.

Figure 2 illustrates an increasing value of the median HTP NCA value for each consecutive inverse RQ value, although the correspondence is much weaker when viewing the box or whisker values. Figure 3, which presents the relationship between HTP cancer air values and inverse RQ values, shows less correspondence than figure 2. For example, substances with inverse RQ values of 0.0002 (i.e., RQ values of 5,000), the U.S. EPA’s least stringent value, are considered more

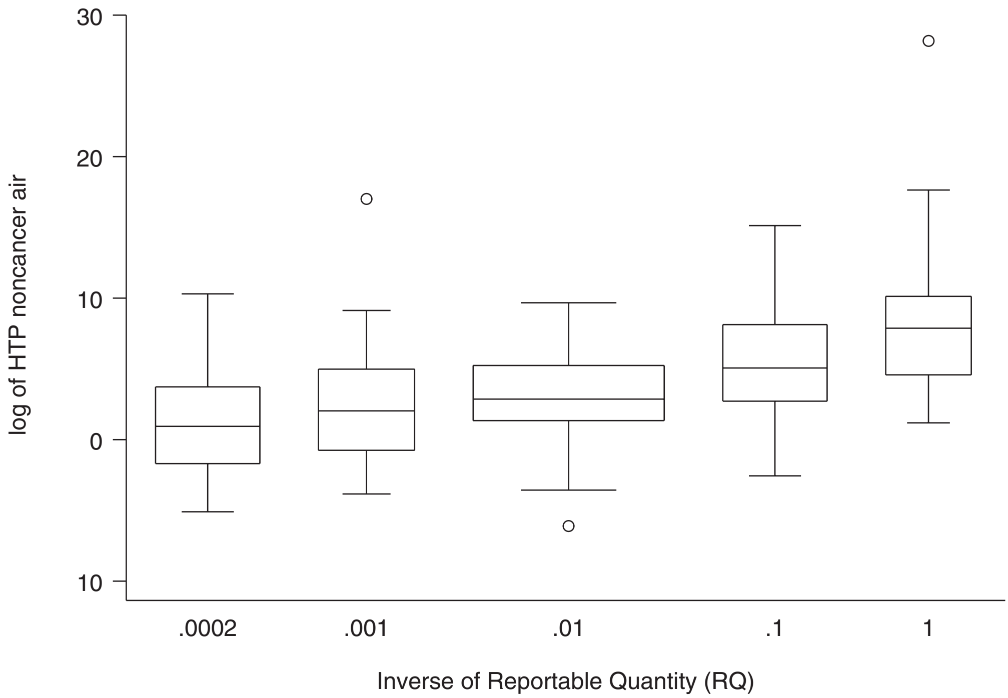

Figure 2 Plot of Human Toxicity Potential (HTP) noncancer air values against inverse of Reportable Quality (inverse RQ) values. The horizontal line within each box represents the median HTP noncancer air value for each inverse RQ value. The plot reveals a positive correlation between these median values and inverse RQ values, although the relationship is weak when considering the box-and-whisker values. The box ranges from the twenty-fifth percentile (X_{25}) to the seventy-fifth percentile (X_{75}), the interquartile range (IQR). The upper vertical line extends up from the top of the box to the largest data point that is less than or equal to X_{75} plus 1.5 times the IQR. The lower vertical line extends down from the bottom of the box to the smallest data point that is greater than or equal to X_{75} minus 1.5 times the IQR. Any points beyond the range of the vertical lines are presented as points above or below the lines. The width of each box is proportional to the number of observations.

relatively damaging by the HTP scheme than substances with inverse RQ values of 0.001, 0.01, and 0.1. In other words, substances considered to have a low impact according to the RQ values have a high impact according to the HTP values.

Comparing TRACI and IRCHS Environmental Values

As illustrated in figures 4 and 5, IRCHS environment scores are uncorrelated with either TRACI eco-toxicity air ($r = 0.010$) or water ($r = 0.073$) values. This result is likely due to the very different methods employed to generate their values. As discussed earlier, IRCHS values are based on the number of regulations to which a particular substance is subjected. On the other hand, TRACI models the potential harm to plant and animal species from chemicals released into the environment. TRACI eco-toxicity values are more scientifically rigorous than IRCHS environmental values. As such, the low correlation between the two schemes undermines the credibility of using IRCHS as a toxicity-weighting scheme.

Comparing RSEI Hazard Scores to Risk Scores

Because RSEI calculates CHHI values specific to each facility-year for each substance released to air, one cannot directly compare RSEI to other schemes. Therefore, we used the following two equations to generate two weighting values for each substance; one value incorporates toxicity, the other value incorporates both toxicity and

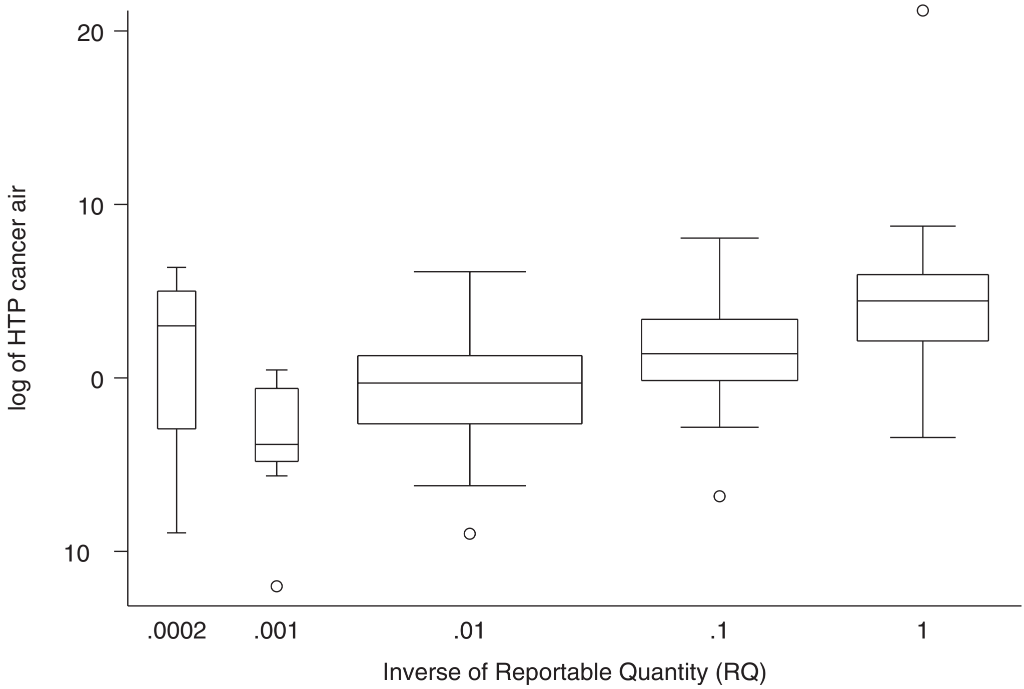

Figure 3 Plot of Human Toxicity Potential (HTP) cancer air values against inverse of reportable quantity (inverse RQ) values. This plot reveals little correspondence between these two schemes. The box-and-whisker graphing format is the same as in figure 2.

year-1999 population exposure. We summed the product of every facility's 1999 toxicity-weighted releases to air of chemical *c* and the model's exposure and population estimates and then divided this figure by the total pounds of chemical *c* released to air by all facilities reporting TRI data in 1999.

Average substance hazard weight$_{c,\text{air},1999}$

$$= \frac{\sum_{f=1}^{F} (\text{REL}_{c,\text{air},f,1999} * \text{TOX}_{c,\text{air}})}{\sum_{f=1}^{F} \text{REL}_{c,\text{air},f,1999}}$$

Average substance risk weight$_{c,\text{air},1999}$

$$= \frac{\sum_{f=1}^{F} (\text{REL}_{c,\text{air},f,1999} * \text{TOX}_{c,\text{air}})(*\text{EXP}_{c,\text{air},f,1999} * \text{POP}_{\text{air},f,1999})}{\sum_{f=1}^{F} \text{REL}_{c,\text{air},f,1999}}$$

where $\text{REL}_{c,f,\text{air},1999}$ is the mass of chemical *c* released to air by facility *f* in 1999 in pounds, $\text{TOX}_{c,\text{air}}$ is the toxicity weight of chemical *c* released to air, $\text{EXP}_{c,\text{air},f,1999}$ is the exposure weight of chemical *c* released to air by facility *f* in 1999, and $\text{POP}_{\text{air},f,1999}$ is the population affected by releases to air by facility *f* in 1999.

We then plotted these average risk weights against the toxicity weights to assess the degree to which their values corresponded (figure 6). The cluster of data points indicates an overall correspondence between these two weighting values. For most toxicity values, however, the corresponding average risk values range across 2 orders of magnitude. Thus, applying RSEI weights that incorporate both toxicity and population exposure may yield significantly different results from using a toxicity-weighting scheme.

Discussion

The Importance of Medium-Specific Values

A chemical's impact to human health and the environment depends on the release medium.

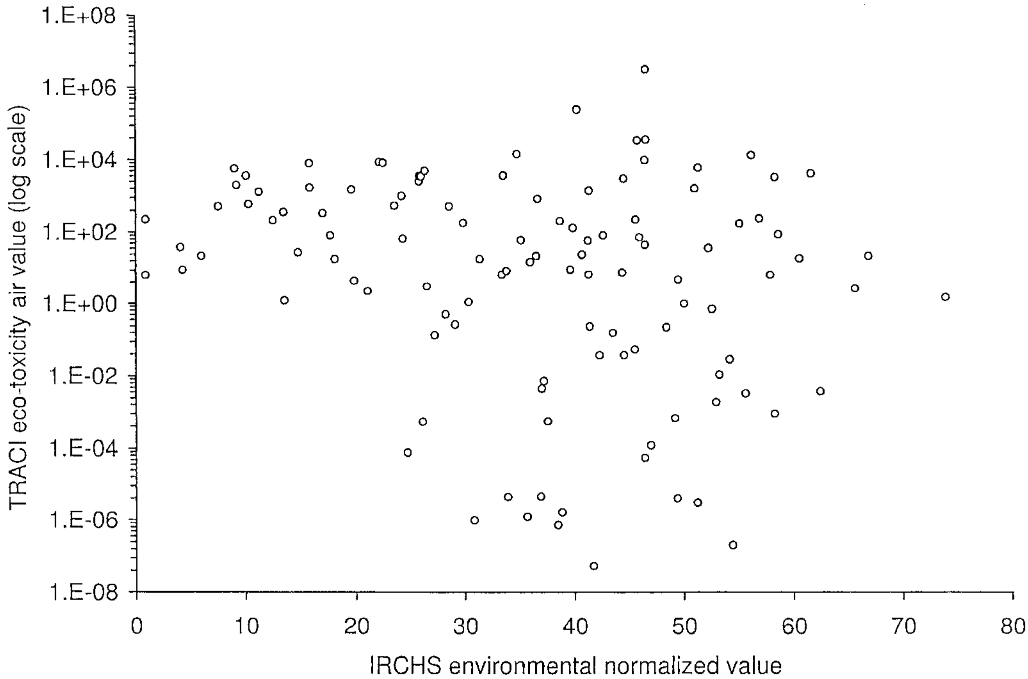

Figure 4 Plot of Tool for the Reduction and Assessment of Chemical Impacts (TRACI) eco-toxicity air values against Indiana Relative Chemical Hazard Score (IRCHS) environmental values. Little correspondence exists between these two schemes.

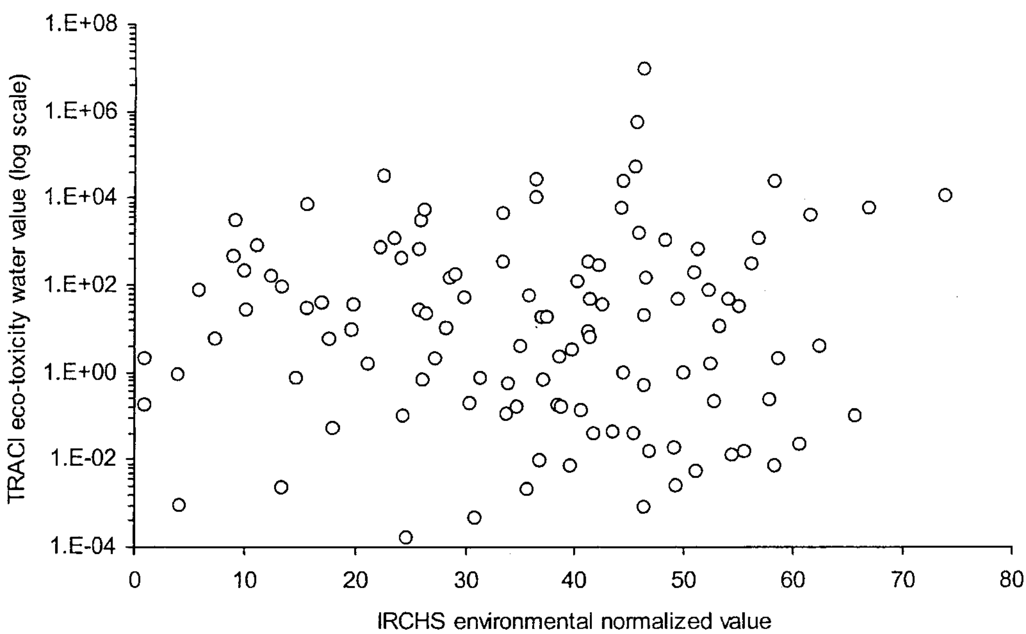

Figure 5 Plot of Tool for the Reduction and Assessment of Chemical Impacts (TRACI) eco-toxicity water values against Indiana Relative Chemical Hazard Score (IRCHS) environmental values. Little correspondence exists between these schemes.

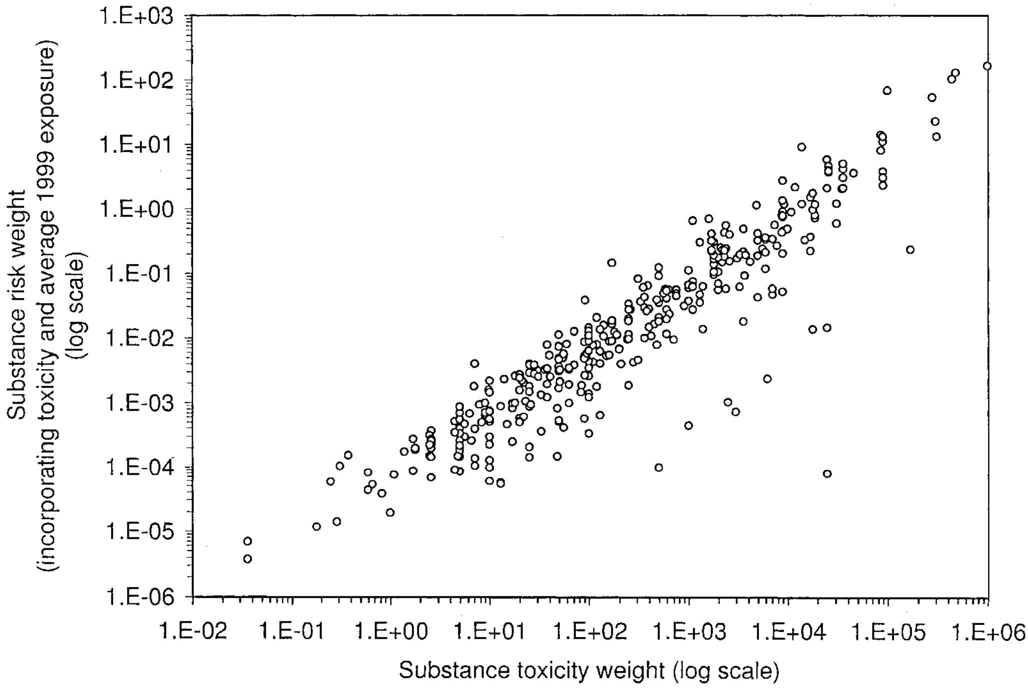

Figure 6 Plot of Risk-Screening Environmental Indicators (RSEI) model's average substance risk weight values against substance toxicity weight values on a log-log scale. Average substance risk weight values were calculated using all facilities' 1999 toxicity-weighted releases to air and the model's exposure estimates. Although the plot indicates an overall correspondence, there is significant scatter in the data, and values deviate from a straight-line relationship by 2 or more orders of magnitude. This scatter indicates that analyses using the risk weight values may differ significantly from analyses using the toxicity weight values.

For example, the same quantity of a specific chemical might have vastly different impacts if released to air or to water. More realistic weighting schemes account for this difference and provide multiple sets of values for the various possible release media. For example, HTP provides distinct toxicity values for releases to air and to water, whereas TLVs are only for releases to air. Media-specific weighting values should only be used for the medium to which those values apply (e.g., toxicity values for releases to air should not be applied to water releases).

Three types of problems can arise with weighting schemes related to this issue of release medium. First, some schemes, such as RQs and IRCHS, generate only one value per substance regardless of release medium, implicitly indicating that their values are equally appropriate for any release medium. Although this type of scheme is straightforward to use, it is also less realistic than media-specific schemes. Second, some schemes combine media-specific values to yield an aggregate value for each chemical. Although evaluating media-specific inputs adds realism to a scheme, the single, aggregated values are less useful in the TRI context where emissions data are media-specific.

To quantify how release medium can influence toxicity, we compared HTP values for releases to air to values for releases to water of original TRI substances. We found low correlation between release media ($r = 0.32$, $n = 94$) for cancer toxicity and high correlation between release media ($r = 0.95$, $n = 174$) for noncancer toxicity. This comparison suggests that when assessing relative risks between chemicals, release medium may be more important for cancer toxicity than for noncancer toxicity. For certain chemicals, there can be enormous differences between the HTP-cancer-air and HTP-cancer-water values. For example, the HTP cancer value for releases of benzoic trichloride to air is 10,000 times larger than the value pertaining to releases to water.[10] On the other hand, the HTP cancer air value for releases of methyl tert-butyl ether to air is 2/1,000 of the value pertaining to releases to water.[11]

Missing Weighting Values

One of the practical difficulties facing an analyst who wishes to use a TRI weighting scheme is the issue of missing values. Weighting schemes often involve information-intensive modeling, and it is not surprising that the data required to produce weighting values are not available for all chemicals of interest.

Lack of comprehensive coverage is particularly important in two cases: (1) if the substances for which values are missing would be heavily weighted and (2) when an analyst suspects the omitted values might systematically bias results. The latter case requires a thorough understanding of how the authors of each scheme determined which substances they would create values for and is beyond the scope of this article. One way to evaluate the former cases is to examine these substances based on their values provided by other schemes.

For example, consider the 222 HTP NCA values and the 298 RQ values that pertain to the current list of TRI substances. Of these, 173 substances have both HTP NCA and RQ values, 125 substances have RQ values but not HTP NCA scores, and 49 substances have HTP NCA scores but not RQ values. We evaluated the average toxicity for these three groupings of chemicals (i.e., those with both HTP NCA and RQ values, those with only HTP NCA values, and those with only RQ values). Our analysis, shown in table 4, suggests that for both weighting methods, chemicals with missing values are more toxic than chemicals with values. The RQ values suggest that chemicals without HTP NCA values are approximately 80% more toxic than chemicals with HTP NCA values. The HTP NCA values suggest that chemicals without RQ values are approximately 60% more toxic than chemicals with RQ values. For any weighting scheme, there may be a systematic difference between chemicals with and without weighting values. Therefore, analysts should carefully consider how missing values are treated.

No single, broadly accepted method exists for evaluating emissions that lack a weighting value. Four possible methods include the following:

1. Restrict the analysis to those chemicals with a weighting value (i.e., omitting chemicals that lack weighting values).
2. Assign the mean or median weighting value to all chemicals that lack weighting values.
3. Estimate surrogate weighting values with a second weighting scheme. For example, if an analyst using scheme A wants to use scheme B to estimate a surrogate value for chemical *c*, he can first calculate the average ratio for weighting values for these two schemes, based on all chemicals that the two schemes have in common. Then, the weighting value in scheme B for chemical *c* can be multiplied by the average ratio of A to B to estimate a surrogate value for chemical *c*.
4. Estimate values based on the chemical structure, the mechanism of toxicity, or other a priori information.

None of these four methods is perfect. Although the fourth method is the most grounded scientifically, time and budget constraints often preclude its use.[12] When it is not possible to implement the last method, determining which among the remaining methods is most accurate is not straightforward. It depends on how similar the distribution of the (unknown) missing values would be to the distribution of known values. If the analyst has reason to believe that these two distributions are similar, then the best proxy for the unknown values is the mean of the known values. If, however, there is reason to suspect that the distributions differ significantly (e.g., if the missing values pertain to less toxic chemicals), then it is unclear which method is more accurate. In most cases, the analyst has no information about the distribution of the missing values. Consequently, analysts may wish to employ all three remaining methods to determine the sensitivity of the results to this choice.

Table 4 Analysis of missing values: Reportable Quantity (RQ) and Human Toxicity Potential (HTP) schemes for current Toxic Release Inventory (TRI) substances

	Average HTP NCA value	*Average inverse RQ value*	*Number of substances*
1. Substances with *both* RQ and HTP noncancer air (NCA) values	441,222	0.129	173
2. Substances with an RQ value but no HTP NCA value	n/a	0.204	125
3. Substances with an HTP NCA value but no RQ value	799,492	n/a	64
4. Ratio of average value for substances in only one scheme (row 2 or 3) to average value for substances in both schemes (row 1)	1.8	1.6	n/a

Conclusions

Understanding the fate and transport of pollutants, and their impact on human health and the environment, is an area of ongoing research. No single best weighting method exists to evaluate chemical release inventories, and choosing a method involves trade-offs, such as between scientific sophistication and comprehensiveness of TRI chemicals.

The 13 schemes we evaluated can be divided into four categories, according to what the weighting values represent:

1. Toxicity to workers and/or the general public (TLV, MRLs, STEL, PEL, REL, unit risk potency factor)
2. Regulatory attention (IRCHS, RQ)
3. Human toxicity, based on a multimedia fate and transport model (HTP, RSEI)
4. Human toxicity and ecosystem impacts, based on a multimedia fate and transport model (EcoIndicator99, EDIP, TRACI)

Recognizing the inherent trade-offs in choosing a weighting scheme, we make the following recommendations.

1. Schemes based on toxicity to workers (e.g., PEL) or the number of regulations that govern a chemical (IRCHS) are not well suited to weight chemical releases to the environment. Instead, analysts should choose a scheme from categories 3 or 4, as these schemes are more sophisticated and provide a more realistic description of the impacts caused by emissions to the environment.
2. Analysts only interested in impacts to human health should use a scheme from category 3. Because of the greater coverage of TRI chemicals and the greater realism gained from incorporating site-specific data, we recommend RSEI over HTP for developing metrics of facility performance. RSEI offers significantly more values pertaining to current TRI substances than does HTP (69% for RSEI CHHI compared to 19% for HTP cancer and 36% for HTP noncancer). RSEI estimates site-specific impacts for each TRI facility in the United States. HTP is based on exposure in a generic environment, but it incorporates pathways not considered in RSEI (e.g., deposition of air emissions onto agricultural plants and subsequent ingestion). Analysts especially concerned with multimedia pollutant transfer, or seeking a second weighting method as a sensitivity analysis, may wish to use HTP.
3. Analysts interested in impacts to human health and the environment should choose a scheme from category 4. The choice among EcoIndicator99, EDIP, and TRACI is significantly impacted by coverage of TRI substances. Although the TRI coverage is poor for TRACI (between 16% and 36% for the six TRACI schemes we evaluated), TRI coverage for EcoIndicator99 and EDIP is too small (less than 10% each) to be useful.

4. Analysts may wish to use multiple weighting schemes to determine the extent to which their results depend on which scheme is chosen.
5. Because no scheme has 100% coverage of TRI emissions, analysts may wish to explore the extent to which results are sensitive to missing weighting values. This article has described several methods to do so.

On a final note, many of the methods discussed in this article are under ongoing development, and new methods are emerging. Improvements are likely, and analysts should consult the latest literature to obtain the most recent values and methods.

Acknowledgments

We thank three anonymous reviewers for their excellent feedback.

Notes

1. The U.S. EPA inspects a few hundred facilities per year. Facilities that reported TRI emissions are inspected to assess the accuracy of reported data. Facilities that are in a TRI industry, but did not report TRI emissions, are inspected to assess whether they should have reported. In U.S. EPA region 9 (primarily California, Arizona, Hawaii, and Nevada), the U.S. EPA annually inspects 25 to 40 facilities that did not report, which results in roughly ten enforcement actions against facilities that were required to report. From its annual inspections of five to ten facilities in region 9 that reported TRI data, the U.S. EPA typically brings approximately three enforcement actions based on data inaccuracies (Browning 2002).
2. In LCA terminology, "weighting" assumes a completely different meaning, defined as "a step of impact assessment in which the (normalized) indicator results for each impact category are assigned numerical factors according to their relative importance, multiplied by these factors and possibly aggregated; weighting is based on value-choices (e.g. monetary values, standards, expert panel)" (Guinée 2002, 99). In other words, in LCA methodology, weighting involves assigning relative weights to aggregate disparate impact categories to a single value. Various methods are discussed elsewhere (e.g., Bengtsson and Steen 2000; Guinée 2002; Pennington et al. 2000; Seppälä et al. 2001; Udo de Haes 2000; Udo de Haes et al. 2002).
3. For example, nearly 50 hazard/risk assessment methodologies are listed in a database jointly developed by the Organisation for Economic Co-operation and Development (OECD) and the World Health Organization's International Programme on Chemical Safety (OECD/IPCS 2000).
4. The IRCHS scheme builds on and extends a similar scheme for aquatic ecosystem impacts produced by the University of Tennessee (UTN). Because IRCHS incorporates the UTN method, we have not considered the UTN method here.
5. RSEI version 2.0 beta 2.0 was used in our analysis; 425 inhalation toxicity weights (ITWs) and 419 oral toxicity weights (OTWs) are provided in a file named "chemical.db" in the ITW and OTW columns, respectively.
6. In EcoIndicator99 terms, we employ the hierarchist characterization rather than individualist or egalitarian (see Goedkoop and Spriensma 2000).
7. See *The Emergency Planning and Community Right-to-Know Act* (EPCRA), Section 304 (U.S. EPA 1986). For the substances covered under this act, see the *Code of Federal Regulations* (U.S. CFR 2003a).
8. For the relevant sections, see the *Code of Federal Regulations* (U.S. CFR 2003a). Also, see Designation of Additional Hazardous Substances and Establishment of Reportable Released Quantities; Regulations (U.S. EPA 1980).
9. Current research may extend HTPs to incorporate site-specific data (DeSimone et al. 2002a, 2002b).
10. HTP cancer values for benzoic trichloride are 277.65 for releases to air and 0.023 for releases to water.
11. HTP cancer values for methyl tert-butyl ether are 6.1×10^{-6} for releases to air and 3.6×10^{-3} for releases to water.
12. This fourth method is time consuming when implemented by hand. Desktop programs such as TOPKAT, MultiCase, and DEREK automate the process, but they are expensive, their output often requires further processing (e.g., converting animal toxicity estimates into human toxicity estimates), and their licensing agreements often prohibit sharing results with third parties. The U.S. EPA's Ecological Structure Activity Relationships program is easy to use and free, but it only estimates aquatic ecosystem toxicity (U.S. EPA 2003a). (For further information, see, e.g., Cro-

nin et al. [2003a, 2003b], Moore et al. [2003], Patlewicz et al. [2003], and Russom et al. [2003]).

References

Adriaanse, A., S. Bringezu, A. Hammond, Y. Moriguchi, E. Rodenburg, D. Rogich, and H. Schütz. 1997. *Resource flows: The material basis of industrial economies*. Washington, DC: World Resources Institute.

Alexeeff, G. V., J. D. Budroe, J. F. Collins, R. H. F. Lam, D. C. Lewis, M. J. Lipsett, M. A. Marty, and T. R. Parker. 1999. *Determination of acute reference exposure levels for airborne toxicants*. Sacramento, CA: Office of Environmental Health Hazard Assessment, California Environmental Protection Agency.

Alexeeff, G. V., J. D. Budroe, J. F. Collins, D. E. Dodge, J. R. Fowles, J. B. Faust, R. F. Lam, D. C. Lewis, M. A. Marty, F. J. Mycroft, H. Olsen, J. S. Polakoff, J. Rabovsky, and A. G. Salmon. 2000. *Determination of noncancer chronic reference exposure levels*. Sacramento, CA: Office of Environmental Health Hazard Assessment, California Environmental Protection Agency.

Antweiler, W. and K. Harrison. 2003. Toxic release inventories and green consumerism: Empirical evidence from Canada. *Canadian Journal of Economics* 36(2): 495–520.

Associated Press Newswires. 2002. Report: Coastal Chem is state's top polluter. 31 May.

ATSDR (Agency for Toxic Substances and Disease Registry). 2002. *Minimal risk levels (MRLs) for hazardous substances*. ⟨www.atsdr.cdc.gov/mrls.html⟩. Accessed 23 August 2002.

Bare, J. C., G. A. Norris, D. W. Pennington, and T. McKone. 2002. TRACI: The tool for the reduction and assessment of chemical and other environmental impacts. *Journal of Industrial Ecology* 6(3–4): 49–78.

Bengtsson, M. and B. Steen. 2000. Weighting in LCA: Approaches and applications. *Environmental Progress* 19(2): 101–109.

Bouwes, N. W. and S. M. Hassur. 1997. *Toxics release inventory relative risk-based environmental indicators methodology*. Washington, DC: Office of Pollution Prevention and Toxics, U.S. Environmental Protection Agency.

Brentrup, F., J. Kusters, H. Kuhlmann, and J. Lammel. 2001. Application of the life cycle assessment methodology to agricultural production: An example of sugar beet production with different forms of nitrogen fertilisers. *European Journal of Agronomy* 14(3): 221–233.

Browning, A. 2002. Personal communication with A. Browning, toxics section, region 9. U.S. Environmental Protection Agency, San Francisco, CA, 23 October 2002.

Cal/EPA (California Environmental Protection Agency). 2000. *Strategic vision*. Sacramento, CA: Office of the Secretary, Cal/EPA.

Caudill, R. J., Y. Luo, P. Wirojanagud, and M. Zhou. 2000. *Exploring the environmental impact of e-commerce on electronic products: An application of fuzzy decision theory and life-cycle studies*. Newark, NJ: New Jersey Institute of Technology.

CECS (Commission for Environmental Cooperation Secretariat). 2002. *Taking stock 1999: North American pollutant releases and transfers*. Montreal: Commission for Environmental Cooperation of North America.

Clift, R. and L. Wright. 2000. Relationships between environmental impacts and added value along the supply chain. *Technological Forecasting and Social Change* 65: 281–295.

CMTI (Clean Manufacturing Technology and Safe Materials Institute). 2001. *Indiana relative chemical hazard score*. West Lafayette, IN: Indiana CMTI. ⟨www.ecn.purdue.edu/CMTI/Pollution_Prevention_Progress_Measurement_Method⟩. Accessed 28 September 2001.

Cohen, M. A., S. A. Fenn, and S. Konar. 1997. *Environmental and financial performance: Are they related?* Nashville, TN: Vanderbilt Center for Environmental Management Studies.

Costa, S., M. Palaniappan, A. K. Wong, J. Hays, C. Landeiro, and J. Rongerude. 2002. *Neighborhood knowledge for change: The West Oakland environmental indicators project*. Oakland, CA: Pacific Institute for Studies in Development, Environment, and Security. ⟨www.pacinstit.org/reports⟩. Accessed April 2, 2004.

Cronin, M. T. D., J. S. Jaworska, J. D. Walker, M. H. I. Comber, C. D. Watts, and A. P. Worth. 2003a. Use of QSARs in international decision-making frameworks to predict health effects of chemical substances. *Environmental Health Perspectives* 111(10): 1391–1401.

Cronin, M. T. D., J. D. Walker, J. S. Jaworska, M. H. I. Comber, C. D. Watts, and A. P. Worth. 2003b. Use of QSARs in international decision-making frameworks to predict ecologic effects and environmental fate of chemical substances. *Environmental Health Perspectives* 111(10): 1376–1390.

Cutter, S. L., M. S. Scott, and A. A. Hill. 2002. Spatial variability in toxicity indicators used to rank chemical risks. *American Journal of Public Health* 92(3): 420–422.

Dasgupta, S., B. Laplante, and C. Meisner. 1998. *Accounting for toxicity risks in pollution control: Does it matter?* Washington, DC: World Bank.

Davis, H., M. Hancock, T. Johnson, P. Smith, C. Storms, C. Sullivan, K. Teliha, and D. Wintz. 2000. *1999 pollution prevention and toxic release inventory annual report*. Indianapolis, IN: Indiana Department of Environmental Management.

DeSimone, M. C., T. E. McKone, and J. D. Marshall. 2002a. Health impact calculations for LCIA based on source location, population distribution, and characteristic travel distance. Paper presented at Society of Environmental Toxicology and Chemistry (SETAC) Europe 12th Annual Meeting, 12–16 May, Vienna.

DeSimone, M. C., T. E. McKone, and J. D. Marshall. 2002b. How source location, population distribution, and pollutant travel distance affect exposure estimates for pollution prevention. Paper presented at Joint Annual Meetings of the International Society of Exposure Analysis and the International Society of Environmental Epidemiology, 10–15 August, Vancouver.

Dooley, R. S. and G. E. Fryxell. 1999. Are conglomerates less environmentally responsible? An empirical examination of diversification strategy and subsidiary pollution in the U.S. chemical industry. *Journal of Business Ethics* 21(1): 1–14.

DTSC (Department of Toxic Substances Control). 1993a. *CalTOX: A multimedia total-exposure model for hazardous-wastes sites, part I—Executive summary*, prepared for the state of California. Report ucrl-cr-111456pti. Livermore, CA: DTSC, Lawrence Livermore National Laboratory.

DTSC. 1993b. *CalTOX: A multimedia total-exposure model for hazardous-wastes sites, part II—The dynamic multimedia transport and transformation model*, prepared for the state of California. Report ucrl-cr-111456ptii. Livermore, CA: DTSC, Lawrence Livermore National Laboratory.

Dyer, R. E. 2003. Personal communication with R. E. Dyer, director of the Office of Innovation and Assistance. Maine Department of Environmental Protection, Augusta, ME, 23 September 2003.

Environmental Defence Canada. 2002. *Airwatch*. ⟨www.edcanada.org/airwatch⟩. Accessed 23 October 2002.

Environmental Defense. 2001. *Important assumptions made in assigning risk scores*. ⟨www.scorecard.org/env-releases/def/tep_assum.html⟩. Accessed 26 September 2001.

Environmental Science Center. 2003. Persistent, bioaccumulative, and toxic profiles estimated for organic chemicals on-line. ⟨http://www.pbtprofiler.net/⟩. Accessed April 2004.

Eskeland, G. S. and A. E. Harrison. 1997. *Moving to greener pastures? Multinationals and the pollution haven hypothesis*. Policy research working paper 1744. Washington, DC: World Bank.

European Environmental Agency. 2003. The European pollutant emission register. ⟨http://www.eper.cec.eu.int⟩. Accessed 26 April 2004.

Fava, J., F. Consoli, R. Denison, K. Dickson, T. Mohin, and B. Vigon, eds. 1992. *A conceptual framework for life-cycle impact assessment*. Pensacola, FL: SETAC Press.

Feldman, S. J., P. A. Soyka, and P. Ameer. 1997. Does improving a firm's environmental management system and environmental performance result in a higher stock price? *Journal of Investing* 6(4): 87–97.

Frey, H. C. and M. J. Small. 2003. Integrated environmental assessment, part 1: Estimating emissions. *Journal of Industrial Ecology* 7(1): 9–11.

Goedkoop, M. J. 1998. *The Eco-Indicator 95, final report*. Amesfoort, The Netherlands: Pré Consultants.

Goedkoop, M. J. and R. S. Spriensma. 2000. *The Eco-Indicator 99: A damage oriented method for life cycle impact assessment, methodology report*, second edition. Amesfoort, The Netherlands: Pré Consultants.

Guinée, J. B., ed. 2002. *Life cycle assessment: An operational guide to the ISO standards*. Boston: Kluwer Academic.

Hamilton, J. T. 1999. Exercising property rights to pollute: Do cancer risks and politics affect plant emission reductions? *Journal of Risk and Uncertainty* 18(2): 105–124.

Harrison, K. and W. Antweiler. 2003. Incentives for pollution abatement: Regulation, regulatory threats, and non-governmental pressures. *Journal of Policy Analysis and Management* 22(3): 361–382.

Harrison, N. 2002. Do customers matter? The impact of customer influence on corporate environmental behavior. Paper presented at 2002 Academy of Management Conference, 11–14 August, Denver, CO.

Hassur, S. 2003. Personal communication with S. Hassur, Office of Pollution Prevention and Toxics. U.S. Environmental Protection Agency, Washington, DC, 11 September 2003.

Hauschild, M. and D. Pennington 2002a. Indicators for ecotoxicity in life cycle impact assessment. In *Life cycle impact assessment: Striving towards best practice*, edited by H. A. Udo de Haes, G. Finnveden, M. Goedkoop, M. Hauschild, E. Hertwich, P. Hofstetter, W. Klöpffer, W. Krewitt, E.

Lindeijer, O. Jolliet, R. Müller-Wenk, S. I. Olsen, D. Pennington, J. Potting, and B. Steen. Pensacola, FL: SETAC Press.

Hauschild, M. and D. W. Pennington. 2002b. 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. Z., S. I. Olsen, and H. Wenzel. 1998. Human toxicity as a criterion in the environmental assessment of products. In *Environmental assessment of products*. Scientific background, Vol. 2, edited by M. Hauschild and H. Wenzel. New York: Chapman and Hall.

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

Hazards Research Lab. 1997. *Calculation of TRI chemical toxicity*, University of South Carolina. ⟨www.cla.sc.edu/geog/hrl/sctrap/toxfaqs/toxicity.htm⟩. Accessed 23 August 2002.

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. *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(5): 138A–144A.

Hertwich, E. G., S. F. Mateles, W. S. Pease, and T. E. McKone. 2001. Human toxicity potentials for life-cycle assessment and toxics release inventory risk screening. *Environmental Toxicology and Chemistry* 20(4): 928–939.

Hertwich, E. G., O. Jolliet, D. W. Pennington, M. Hauschild, C. Schulze, W. Krewitt, and M. Huijbregts. 2002. Fate and exposure assessment in the life-cycle impact assessment of toxic chemicals. In *Life cycle impact assessment: Striving towards best practice*, edited by H. A. Udo de Haes, G. Finnveden, M. Goedkoop, M. Hauschild, E. Hertwich, P. Hofstetter, W. Klöpffer, W. Krewitt, E. Lindeijer, O. Jolliet, R. Müller-Wenk, S. I. Olsen, D. Pennington, J. Potting, and B. Steen. Pensacola, FL: SETAC Press.

Hofstetter, P., A. Braunschweig, T. Mettier, R. Muller-Wenk, and O. Tietje. 2000. Dominance analysis in the mixing triangle: Graphical decision support for comparisons with LCA. *Journal of Industrial Ecology* 3(4): 97–115.

Horvath, A., C. T. Hendrickson, L. B. Lave, F. C. McMichael, and T.-S. Wu. 1995. Toxic emissions indices for green design and inventory. *Environmental Science and Technology* 29(2): 86–90.

Horvath, A., C. Hendrickson, L. Lave, F. McMichael, and T.-S. Wu. 1997. *Toxic emissions indices: A tool for green design and performance measurement*. Carnegie Mellon University Green Design Initiative. ⟨www.ce.cmu.edu/GreenDesign/research/cmu-et.html⟩. Accessed 23 August 2002.

ICI. 1997. *Environmental burden: The ICI approach*. Second edition. London: ICI.

ILO (International Labor Organization). 1991. *Occupational exposure limits for airborne toxic substances: Values of selected countries*. Geneva: ILO.

IRRC (Investor Responsibility Research Center). 2002. *IRCC's corporate environmental profiles database key to elements*. Washington, DC: IRRC.

ISO (International Standards Organization). 1997. *ISO 14040 environmental management: Life cycle assessment—Principles and framework*. Geneva: ISO.

ISO. 2000. *ISO 14042 environmental management: Life cycle assessment—Impact assessment*. Geneva: ISO.

Jia, C. Q., A. Di Guardo, and D. Mackay. 1996. Toxics release inventories: Opportunities for improved presentation and interpretation. *Environmental Science and Technology* 30(2): 86A–91A.

Kay, J. 2002. Refineries top polluters on EPA list in Bay area: Discharges taint air, water and land. *San Francisco Chronicle*, 24 May, section A: 11.

Khanna, M. and W. R. Q. Anton. 2002. Corporate environmental management: Regulatory and market-based incentives. Working paper, University of Illinois at Urbana-Champaign.

King, A. and M. Lenox. 2000. Industry self-regulation without sanctions: The chemical industry's responsible care program. *Academy of Management Journal* 43(4): 698–716.

King, A. and M. Lenox. 2001a. Does it *really* pay to be green? An empirical study of firm environmental and financial performance. *Journal of Industrial Ecology* 5(1): 105–116.

King, A. and M. Lenox. 2001b. Lean and green? An empirical examination of the relationship between lean production and environmental performance. *Production and Operations Management* 10(3): 244–256.

King, A. A. and J. M. Shaver. 2001. Are aliens green? Assessing foreign establishments' environmental conduct in the United States. *Strategic Management Journal* 22: 1069–1085.

Kleijn, R. 2001. Adding it all up: The sense and nonsense of bulk-MFA. *Journal of Industrial Ecology* 4(2): 7–8.

Konar, S. and M. A. Cohen. 2001. Does the market value environmental performance? *Review of Economics and Statistics* 83(2): 281–289.

Krewitt, W., D. W. Pennington, S. I. Olsen, P. Crettaz, and O. Jolliet. 2002. Indicators for human toxicity in life cycle impact assessment. In *Life cycle impact assessment: Striving towards best practice*, edited by H. A. Udo de Haes, G. Finnveden, M. Goedkoop, M. Hauschild, E. Hertwich, P. Hofstetter, W. Klöpffer, W. Krewitt, E. Lindeijer, O. Jolliet, R. Müller-Wenk, S. I. Olsen, D. Pennington, J. Potting, and B. Steen. Pensacola, FL: SETAC Press.

Lee, S. G., S. W. Lye, and M. K. Khoo. 2001. A multiobjective methodology for evaluating product end-of-life options and disassembly. *International Journal of Advanced Manufacturing Technology* 18(2): 148–156.

Lenox, M. 2001. Overcoming agency and information problems in the diffusion of best practice within firms. Paper presented at Wharton Technology Miniconference, 30–31 March, Philadelphia.

Lenox, M. J. and J. Nash. 2003. Industry self-regulation and adverse selection: A comparison across four trade association programs. *Business Strategy and the Environment* 12: 343–356.

Lifset, R. 2001. Moving from mass to what matters. *Journal of Industrial Ecology* 4(2): 1–3.

Luo, Y., P. Wirojanagud, and R. J. Caudill. 2001. Comparison of major environmental performance metrics and their application to typical electronic products. Paper presented at 2001 International Symposium on Electronics and the Environment, 7–9 May, Denver, CO.

Matthews, E., C. Amann, S. Bringezu, M. Fischer-Kowalski, W. Hüttler, R. Kleijn, Y. Moriguchi, C. Ottke, E. Rodenburg, D. Rogich, H. Schandl, H. Schütz, E. v. d. Voet, and H. Weisz. 2000. *The weight of nations: Material outflows from industrial economies*. Washington, DC: World Resources Institute.

McKone, T. E. and K. G. Enoch. 2002. *CalTOX™: A multimedia total exposure model spreadsheet user's guide, version 4.0 (beta)*. Berkeley, CA: Environmental Energy Technologies Division, Lawrence Berkeley National Laboratory.

McKone, T. E. and E. G. Hertwich. 2001. The human toxicity potential and a strategy for evaluating model performance in life cycle impact assessment. *International Journal of Life Cycle Assessment* 6(2): 106–109.

McKone, T. E., D. H. Bennett, and R. L. Maddalena. 2002. *CalTOX, multimedia, multipathway exposure model technical support document*. LBNL-47254. Berkeley, CA: Lawrence Berkeley National Laboratory.

Moore, D. R. J., R. L. Breton, and D. B. MacDonald. 2003. A comparison of model performance for six quantitative structure-activity relationship packages that predict acute toxicity to fish. *Environmental Toxicology and Chemistry* 22(8): 1799–1809.

Mosovsky, J., D. Dickinson, and J. Morabito. 2000. Creating competitive advantage through resource productivity, eco-efficiency, and sustainability in the supply chain. Paper presented at 2000 IEEE International Symposium on Electronics and the Environment, 8–10 May, San Francisco.

Murray, C. and A. Lopez. 1996. *The global burden of disease*. Boston: WHO, World Bank, and Harvard School of Public Health.

Noah, T. 1996. EPA proposal expands list of industries subject to disclosing toxic releases. *Wall Street Journal*, 27 June, section B: 12.

Norris, G. A. 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): 79–101.

NOSB (National Organic Standards Board). 2000. *Ethylene crops*. NOSB materials database. ⟨www.omri.org/ethylene_pineapples.pdf⟩. Accessed 23 August 2002.

OECD (Organisation for Economic Cooperation and Development). 2001a. *Pollutant release and transfer registers: Status of PRTR systems*. ⟨www.oecd.org⟩. Accessed 3 August 2003.

OECD. 2001b. *Why pollutant release and transfer registers (PRTRs) differ: A review of national programmes*. ENV/jm/mono(2001)16. OECD environment, health, and safety publications, series on pollutant release and transfer registers no. 4. Paris: OECD.

OECD/IPCS (OECD and International Programme on Chemical Safety). 2000. *The OECD/IPCS database on hazard/risk assessment methodologies*. ⟨appli1.oecd.org/ehs/ipcs.nsf⟩. Accessed 23 October 2002.

Patlewicz, G., R. Rodford, and J. D. Walker. 2003. Quantitative structure-activity relationships for predicting mutagenicity and carcinogenicity. *Environmental Toxicology and Chemistry* 22(8): 1885–1893.

Pennington, D. W. and J. C. Bare. 2001. Comparison of chemical screening and ranking approaches: The waste minimization prioritization tool versus toxic equivalency potentials. *Risk Analysis* 21(5): 897–912.

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.

Pennington, D. W., G. Norris, T. Hoagland, and J. C. Bare. 2000. Environmental comparison metrics for life cycle impact assessment and process design. *Environmental Progress* 19(2): 83–91.

Pratt, G. C. 2002. Personal communication with G. C. Pratt, environmental outcomes division. Minnesota Pollution Control Agency, St. Paul, MN, 30 September 2002.

Pratt, G. C., P. E. Gerbec, S. K. Livingston, R. Oliaei, G. L. Bollweg, S. Paterson, and D. Mackay. 1993. An indexing system for comparing toxic air pollutants based upon their potential environmental impacts. *Chemosphere* 27: 1359–1379.

Rubin, E. 1999. Toxic releases from power plants. *Environmental Science and Technology* 33(18): 3062–3067.

Russo, M. V. 2002. Institutional change and organizational strategy: ISO 14001 and emissions in the electronics industry. Paper presented at Academy of Management Conference, 11–14 August, Denver, CO.

Russom, C. L., R. L. Breton, J. D. Walker, and S. P. Bradbury. 2003. An overview of the use of quantitative structure-activity relationships for ranking and prioritizing large chemical inventories for environmental risk assessments. *Environmental Toxicology and Chemistry* 22(8): 1810–1821.

Seppälä, J., L. Basson, and G. A. Norris. 2001. Decision analysis frameworks for life-cycle impact assessment. *Journal of Industrial Ecology* 5(4): 45–68.

Sheppard, E., R. B. McMaster, H. Leitner, S. Elwood, and H. Tian. 1999. *Examining environmental equity in Hennepin County and Minneapolis*. Minneapolis, MN: University of Minnesota Center for Urban and Regional Affairs.

Shields, T. 1999. Pepco plant's toxic emissions rank near top: Amounts legal but concerns remain. *Washington Post*, 17 June, section M: 1.

Sørensen, P. 2002. *Survey on international recognition of the edip methodology for life cycle assessment*, working report no. 8. København, Denmark: Danish Environmental Protection Agency.

Steen, B. 1999. *A systematic approach to environmental priority strategies in product development (EPS), version 2000: Models and data of the default method*. CPM report 1999:5. Göteborg, Sweden: Chalmers University of Technology.

Tellus Institute. 1992. *CSG/Tellus packaging study, report no. 4: Impacts of production and disposal of packaging materials—Methods and case studies*. Boston: Tellus Institute.

Udo de Haes, H. A. 2000. LCA in Europe: Weighting in LCA, is there a coherent perspective? *Journal of Industrial Ecology* 3(4): 3–7.

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

U.S. Census Bureau. 2001. *Statistical abstract of the United States*. Washington, DC: U.S. Census Bureau.

U.S. CFR (U.S. Code of Federal Regulations). 2003a. *Designation of hazardous substances*. 40CFR302.4. Washington, DC: U.S. Government Printing Office. ⟨www.gpoaccess.gov/cfr/retrieve.html⟩. Accessed April 2004.

U.S. CFR. 2003b. *Determination of reportable quantities for hazardous substances*. 40CFR117.3. Washington, DC: U.S. Government Printing Office. ⟨www.gpoaccess.gov/cfr/retrieve.html⟩. Accessed April 2004.

U.S. CFR. 2003c. *Air contaminants*. 29CFR1910.1000. Washington, DC: U.S. Government Printing Office. ⟨www.gpoaccess.gov/cfr/retrieve.html⟩. Accessed April 2004.

U.S. EPA (U.S. Environmental Protection Agency). 1980. *Designation of additional hazardous substances and establishment of reportable released quantities; regulations*. 42USC9602. Washington, DC: U.S. EPA. ⟨www.epa.gov/superfund/programs/recycle/tools/cercla/9602.htm⟩. Accessed April 2004.

U.S. EPA. 1986. *Emergency Release Notification*. 42USC11001. Washington, DC: U.S. EPA. ⟨www4.law.cornell.edu/uscode/42/11001.html⟩. Accessed April 2004.

U.S. EPA. 1998a. *1996 toxic release inventory: Data quality report*. EPA-745-r-98-016. Washington, DC: U.S. EPA.

U.S. EPA. 1998b. *Extremely hazardous substances (EHS) chemical profiles and emergency first aid guides*. Washington, DC: U.S. EPA. ⟨www.epa.gov/swercepp/ehs/ehslist.html⟩. Accessed 23 August 2002.

U.S. EPA. 2002a. *Health effects notebook for hazardous air pollutants (Dec. 1994)*. Washington, DC: U.S. EPA. ⟨www.epa.gov/ttn/atw/hapindex.html⟩. Accessed 23 August 2002.

U.S. EPA. 2002b. *How does the reportable quantity adjustment methodology work?* Washington, DC: U.S. EPA. ⟨www.epa.gov/superfund/programs/er/

triggers/haztrigs/rqwork.htm⟩. Accessed 5 December 2002.

U.S. EPA. 2002c. *Integrated risk information system (IRIS)*. Washington, DC: U.S. EPA. ⟨www.epa.gov/iris/index.html⟩. Accessed 1 September 2002.

U.S. EPA. 2002d. *Ongoing efforts*. Research and Development, National Risk Management Research Laboratory, U.S. EPA. ⟨www.epa.gov/ORD/NRMRL/lcaccess/ongoing.htm⟩. Accessed 1 September 2002.

U.S. EPA. 2002e. *Sustainable technology systems analysis: TRACI—Tool for the reduction and assessment of chemical and other environmental impacts*. Cincinnati, OH: U.S. EPA.

U.S. EPA. 2002f. *Tool for the reduction and assessment of chemical and other environmental impacts (TRACI): User's guide and system documentation*. EPA/600/r-02/052. Cincinnati, OH: U.S. EPA, Office of Research and Development.

U.S. EPA. 2002g. *The toxics release inventory (TRI) and factors to consider when using TRI data*. Washington, DC: U.S. EPA. ⟨www.epa.gov/tri/tridata/tri00/press/overview.pdf⟩. Accessed 23 October 2002.

U.S. EPA. 2002h. *Toxics release inventory (TRI) program: TRI chemicals*. Washington, DC: U.S. EPA. ⟨www.epa.gov/tri/chemical/index.htm⟩. Accessed 15 August 2002.

U.S. EPA. 2002i. *TRACI software (version 1.0) and user's guide*. Washington, DC: U.S. EPA. ⟨www.epa.gov/ORD/NRMRL/std/sab/download2.htm⟩. Accessed 1 September 2002.

U.S. EPA. 2002j. *User's manual for RSEI version 2.0 beta 2.0 [1988–1999 TRI data]*. Washington, DC: U.S. EPA.

U.S. EPA. 2002k. *Variations between facilities can result from the use of different estimation methodologies*. Washington, DC: U.S. EPA. ⟨www.epa.gov/tri/tridata/tri00/press/overview.pdf⟩. Accessed 23 October 2002.

U.S. EPA. 2002l. *Waste minimization products and documents*. Washington, DC: U.S. EPA. ⟨www.epa.gov/epaoswer/hazwaste/minimize/docs.htm⟩. Accessed 22 August 2002.

U.S. EPA. 2003a. *Ecological structure activity relationships*. Washington, DC: U.S. EPA. ⟨www.epa.gov/oppt/newchems/21ecosar.htm⟩. Accessed 9 September 2003.

U.S. EPA. 2003b. *Risk-screening environmental indicators*. Washington, DC: U.S. EPA. ⟨www.epa.gov/oppt/rsei⟩. Accessed 20 September 2003.

U.S. EPA. 2003c. *Toxics release inventory (TRI): Case number lookup*. Washington, DC: U.S. EPA. ⟨http://oaspub.epa.gov/enviro/tri_lookups.tri_cas_lookup⟩. Accessed 16 April 2003.

U.S. GAO (U.S. General Accounting Office). 1994. *Toxic substances: Status of EPA's efforts to reduce toxic releases*. Report to the chairman, environment, energy, and natural resources subcommittee, committee on government operations. GAO/RCED-94-207. Washington, DC: U.S. GAO.

U.S. OSHA. (U.S. Occupational Safety and Health Administration) 2001. *Safety and health topics: Permissible exposure limits (PELs)*. U.S. OSHA. ⟨www.osha-slc.gov/SLTC/pel⟩. Accessed 23 August 2002.

Vogtlander, J. G., A. Bijma, and H. C. Brezet. 2002. Communicating the eco-efficiency of products and services by means of the eco-costs/value model. *Journal of Cleaner Production* 10(1): 57–67.

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

World Bank. 2002. *The toxint dataset: Toxic pollution intensities and risk by industry sector*. ⟨www.worldbank.org/nipr/data/toxint⟩. Accessed 23 August 2002.

Wright, M., D. Allen, R. Clift, and H. Sas. 1997. Measuring corporate environmental performance: The ICI environmental burden system. *Journal of Industrial Ecology* 1(4): 117–127.

Yang, J.-X. and P. Nielsen. 2001. Chinese life cycle impact assessment factors. *Journal of Environmental Sciences (China)* 13(2): 205–209.

Yarwood, J. M. and P. D. Eagan. 2001. Index of hazardous chemical pollutants. In *Design for the environment: A competitive edge for the future—Toolkit*. St. Paul, MN: Minnesota Office of Environmental Assistance.

Yu, Y., K. Jin, H. C. Zhang, F. F. Ling, and D. Barnes. 2000. A decision-making model for materials management of end-of-life electronic products. *Journal of Manufacturing Systems* 19(2): 94–107.

Zhang, Q., J. C. Crittenden, and J. R. Mihelcic. 2001. Does simplifying transport and exposure yield reliable results? An analysis of four risk assessment methods. *Environmental Science and Technology* 35(6): 1282–1288.

About the Authors

Michael Toffel is a doctoral candidate at the Haas School of Business and **Julian Marshall** is a doctoral candidate in the Energy and Resources Group, both at the University of California in Berkeley, California, USA.