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The use of valuation and weighting sets in environmental impact assessment

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ABSTRACT

In environmental impact assessment of policies and product design results need to be presented in a comprehensible way to make alternatives easily comparable. One way of doing this is to aggregate results to a manageable set by using weighting methods. Valuing the environmental impacts can be a challenging task that can also be quite time-consuming. To the aid of practitioners, several weighting sets with readily available weights have been developed over the last decade. The scope and coverage of these sets vary, and it is important to be aware of the implications of using different valuation methods and weighting sets.

The aim of this paper is to map valuation and weighting techniques and indicate the methods that are suitable to use, depending on the purpose of the analysis. Furthermore, we give an overview over sets of generic values or weights and their properties, and give an illustration of how different sets may influence the results. It is very useful to use several weighting sets, and discuss the results thoroughly. It is often a very interesting and fruitful exercise to see if and how the results differ, why they differ, and which one seems to be the best alternative to base any recommendation on.

The example provided in this article demonstrates that looking at aggregate results is not enough. Since many weighting sets are not sufficiently transparent as to how they are constructed and what their impact categories actually include, a general recommendation is to provide weighting sets with a declaration of content, providing a clear picture of what is included and what is not, and a recommendation of suitable uses of the weighting set.

1. Introduction

Today there is a widespread awareness that most of our actions may have consequences on the environment. Decision makers need to take environmental impacts into account when making decisions about projects and investments. Results from impact assessments can however be both extensive and diverse, which makes comparison of alternatives difficult. To help decision makers to interpret the results, many impact assessment tools include a possibility to aggregate results to an index or a few indicators, by translating them into a common unit. In economic tools like cost-benefit analysis, impacts on non-marketed goods are monetized to make them comparable to monetary costs and benefits. In other tools, e.g. life-cycle assessment (Udo de Haes et al., 2002) and strategic environmental assessment (Brown and Therivel, 2000), weighting in order to aggregate the results is often made by monetary valuation, but need not be. Valuation can be done in monetary terms or just as a value judgement, expressed as weights. To avoid confusion, we will henceforth follow the shorthand often used in economics and use the term valuation when we mean monetary valuation. Weighting will be used as a general term, the weights being in any unit, monetary or non-monetary.

There are several methods for valuing environmental goods, each with its advantages and disadvantages. The scope of the methods varies significantly: some cover pure economic losses (e.g. damage costs using market prices), some impute values by using different types of costs (imputed willingness to pay methods) and some attempt to measure welfare losses (expressed willingness to pay methods). The latter may include both use and non-use values. Non-use values refer to the value that people derive from goods independent of any use, present or future, that they might make use of those goods, in contrast to use values, which people derive from direct use of the good (Mitchell and Carson, 1989). Non-use values may include quasi-option values (the value of preserving options for future use given some expectation of expanding knowledge), existence values (the value of knowing that an amenity exists) and values to future generations.

The purpose of this paper is to map valuation and weighting techniques, and give an overview over sets of generic values or

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weights that are readily available for use in life cycle assessments and cost benefit analyses.

The first section maps methods for assessing environmental impacts, followed by a discussion on which methods are suitable to use, depending on the purpose of the analysis.

In the next section, an overview of available weighting/valuation sets is given. Lastly, a comparison of results from using some of the weighting sets is done.

2. Methods for weighting of environmental impacts and potential uses

In this section, an overview of valuation/weighting methods is given. We use the taxonomy developed by Ahlroth et al. (2011), where a more comprehensive list and description of each method can be found.

As pointed out in Section 1, valuation can be done in both monetary and non-monetary terms. Both can give a cardinal ranking, but of course only the monetary methods are fully comparable to costs and benefits of marketed goods and services.

2.1. Monetary valuation

Monetary valuation provides added information to non-monetary weighting in two ways: it makes it possible to

(1) rank impacts from a welfare perspective,
(2) estimate whether benefits of a certain policy or action exceed the costs.

This means that monetary valuation has a wider field of potential uses than non-monetary weighting. On the other hand, there are several caveats attached to the monetary valuation methods, which are added to the uncertainties inherent in the non-monetary weighting methods. In both cases, it is useful to see the weighting results as indicative and to use them for further discussion of the merits of the analyzed alternatives.

In Fig. 1, methods for monetary valuation are listed. A market price is what people are willing to pay for a certain good at the current level of supply. Environmental damages can be valued by the loss of production that the damages infer – often called damage cost valuation. One example of this is decreased crop yield due to tropospherical ozone. To value goods and services that are not sold on a market, we can try to simulate markets or to deduce the willingness to pay for a good from the price of related marketed good. The latter is called *revealed willingness to pay*, since people's preferences are revealed from what they pay for the related good (Champ et al., 2003). In the *travel cost method*, willingness to pay (WTP) for e.g. visiting a nature area is elicited from the costs to travel there, as well as other costs incurred, e.g. food and equipment needed. *Hedonic pricing* is most often used to value real estates, by trying to identify different qualities that influence the price. Environmental amenities valued might be proximity to swimmable water, good fishing water or a nature park. Disamenities may also be valued, such as contaminated sites. The *production function approach* is applicable in cases where the environmental goods/services are some (or one) of the inputs to produce a marketed good (Champ et al., 2003). An appropriately specified production function may indicate the contribution of these inputs to the output. From this information one may deduce the benefit due to the inputs. All these three methods give a lower bound of the value, since they can only capture part of the value of a certain good or service (Hanley et al., 2007).

In the *expressed willingness to pay methods*, hypothetical markets are constructed, where people are asked for their preferences and what they would be willing to pay to have access to environmental amenities. These methods are the most comprehensive, in that they capture the total value to the relevant population, including non-use values (Hanley et al., 2007). In the most frequently used method, *contingent valuation*, the respondents are asked to state their willingness to pay for an increase in environmental quality, contingent on a carefully structured hypothetical market (Hanley and Spash, 1993). *Choice modelling* includes a range of methods, e.g. contingent ranking, paired comparisons and choice experiments (Louviere et al., 2000). In choice experiments, respondents are asked to choose between alternative goods, defined in terms of their attributes, one attribute being a monetary cost. This allows the analyst to derive a monetary value of each of the attributes.

Imputed WTP methods include several cost methods such as *damage cost avoided*, *replacement cost*, and *substitute cost method* (Mishra, 2006). These cost methods are based on the assumption that, if people incur costs to avoid damages caused by lost ecosystem services, or to replace the services of ecosystems, then those services must be worth at least what people paid to replace them. The advantage of these methods is that they reflect actual costs that may be imposed on households and businesses. Cost approaches are however problematic (depending on the application) since the costs are not linked to the extent of the damages, and are thus not related to the perceived severity of the problem.

Political willingness to pay is similar to revealed WTP, but in this case it is the political decisions that reveal the preferences. Finnveden et al. (2002) note that societal values may be different from the sum of individual values, and that it may therefore be reasonable to deduce values from the behaviour of e.g. governments. The costs for reaching established targets can be interpreted as society's willingness to pay, mediated by the political process. The targets should be enforced by a political decision, i.e. there is an explicit will to pay the costs (Kopp et al., 1996). Another way is to use environmental taxes, which can be interpreted as a price on environmental damage, e.g. incurred by a certain emission (Finnveden et al., 2002).

As we will see in the next section, another way to monetize impacts is to use the cost of reducing either the pressure (e.g. the cost of reducing emission) or the impact (e.g. liming of acidified lakes). The former are usually labelled *avoidance costs* or *prevention costs*, the latter *restoration costs* (UN, 2003). These approaches are not willingness to pay measures, since there is no relation to any decision to actually enforce the measures and take on the costs.

2.2. Non-monetary weighting

Non-monetary weights are typically used to show the relative importance of different types of environmental impacts, according to experts, the general public or a specific population. *Proxy methods* in our classification scheme (Fig. 2) use one or a few quantitative measures stated to be indicative for the total environmental impacts (Lindeijer, 1996). An example of this is an approach sometimes used in Environmental Management Systems, where each environmental aspect is rated on a scale of 1–3 on the basis of a few criteria. There are also specific methods developed with the purpose of displaying environmental impact, e.g. TMR (total material requirement) (Adriansee et al., 1997) and Ecological Footprints (Rees and Wackernagel, 1994).

Weights in non-monetary units are often derived by some form of panel weighting method (Ascher and Steelman, 2006). This is similar to the expressed WTP methods, with the difference that monetary values are not included in the parameters. Using panels for eliciting preferences and judgments can be done in many different ways. Panels can consist of experts, stakeholders or lay people, and the elicitation process can be organized in many ways (Seppälä, 1999). Ad hoc methods are used in many instances, and there are

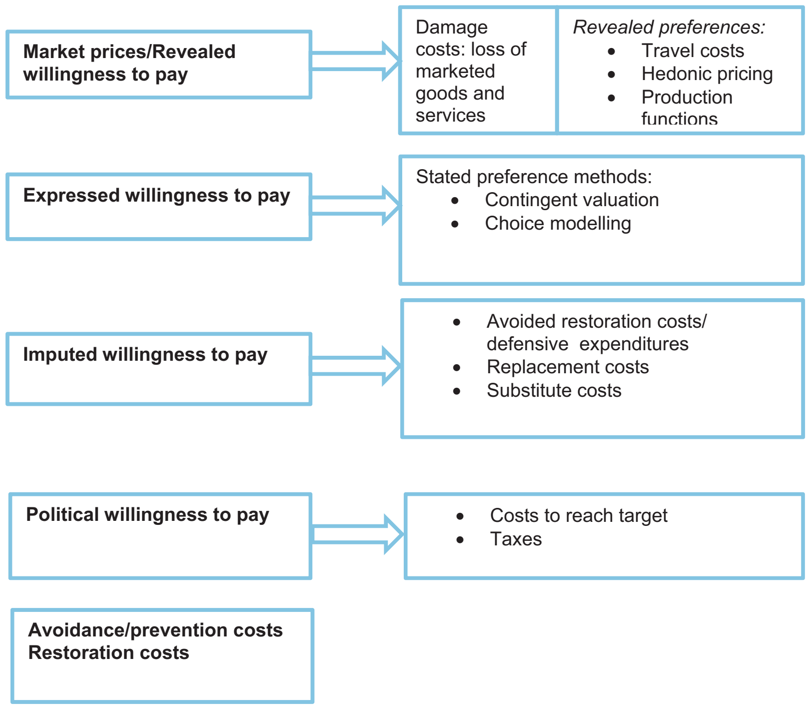

Fig. 1. Monetary methods for valuing/weighting environmental impacts.

also sets of generic weighting factors being developed (Goedkoop and Spriensma, 2000).

Multicriteria analysis is a set of methods that are designed to handle problems characterized by multiple objectives. The methodologies typically aim at quantifying trade-offs among attributes. How this is done and the requirements for elicitation of weighting factors vary between methods (Belton and Stewart, 2002).

In *distance-to-target methods*, the target-setting provides the weighting. As the name suggests, the weights are derived by calculating the distance to a target, which may be of different kind. Most often, the targets are political decisions, e.g. regulations, emission reduction targets, but depending on the purpose of the analysis, targets set by any other relevant decision maker may apply as well. If the targets represent a good environmental quality the weights reflect how large the gap is to reach a good environment; if the targets are set lower for political or economic reasons, the weights reflect the current political preferences, similar to the political willingness-to-pay methods (Weiss et al., 2007). When targets are set, there is normally no requirement that they should be of equal importance. Thus some weighting between impacts needs to done. However, in many applications the targets are just assumed to have equal weights, implying that inter-effect weighting is lacking (Finnveden et al., 2002).

Physical damage functions are cause–effect functions that describe the link between an environmental pressure, e.g. emissions, and damage incurred on environment or human beings. An example is exposure-response functions for air emissions and their impact on human health, or dose–response function for acidifying emissions and acidification of soil and water, as well as corrosion. They can be used to derive weights for the relevant substances according to how toxic they are. Physical damage functions can also be the basis for calculating damage costs.

2.3. Choosing weighting method

Which weighting method to use depends on the purpose of the analysis and the tool you are using in the analysis. For an overview, see Ahlroth et al. (2011). Cost–benefit analysis (CBA) requires valuation; in both life-cycle assessment (LCA) and life-cycle cost analysis (LCC), valuation/weighting is optional. While CBA is most often used for assessing public policies, LCA and LCC are often used by

Table 1
Properties of valuation and weighting methods.

Monetary valuation methods	Type of value	Value by whom?	What does the value include?
Revealed willingness to pay	Market price	Consumers and producers	Part of value of environmental amenity reflected in price for marketed goods and services
Expressed willingness to pay (stated preference methods)	Willingness to pay, including consumer surplus	Households	Total economic value, including non-use values
Imputed willingness to pay	Market prices	Households, companies, public sector	Alternative costs to reducing environmental impact
Political willingness to pay	Market prices	Politicians, constituents	Costs to society for reducing environmental impact

Non-monetary weighting methods	Type of weight	Weights by whom?
Proxy methods	Ordinal or cardinal weights	Companies, experts, scientists
Distance-to-target methods	Cardinal weights	Politicians, constituents
Panel weighting methods	Cardinal weights	Experts, scientists, stakeholders

companies to assess the impact of their products. Since LCA and LCC typically concern products, they are best served by generic weights that are not derived for a certain geographical area, since the products can be used in many different environments. CBA may concern a project in a certain site, and in that case it may be more accurate if values are derived for that specific site or similar ones.

The tools mentioned above can be used with a number of valuation methods. In Table 1, the scope and coverage of different valuation/weighting measures are shown. In the second column, we can see the type of value in the monetary methods. A market price is the marginal price for a good or service at a certain point in time, excluding the consumer surplus. In contrast, the values elicited from expressed WTP methods include the consumer surplus. Also, the scope of the valuation is wider: so called non-use values or existence values are included. The revealed WTP methods give a lower bound of the value of an environmental good or service, since only parts of the value are included. Furthermore, it only pertains to a limited population, e.g. tourists and real estate owners.

The imputed and political WTP methods do not estimate the value of a environmental good or service, but yields different cost measures that may be relevant when assessing which action that should be taken, both for public actors and private companies. The political willingness to pay measures gives information about political preferences that may be valuable for informing decisions at company level.

The non-monetary methods all try to reflect the pressure on the environment. The difficulty often lies in constructing a measure that includes all different kinds of environmental impacts, and to weigh between different kinds of impacts, e.g. between global warming, eutrophication and air quality.

3. Weighting sets

When using weighting to display your results, it is important that the weights or values you use are consistent; i.e. they are derived with the same method for all the environmental impacts involved. As we have seen, the scope and coverage of different methods vary considerable, so using one method for e.g. greenhouse gases and another for eutrophying pollutants will bias the result and make it less transparent. In many cases, there is neither time nor money to make valuation studies. For this reason, so-called benefit transfer is often used. In benefit transfer, estimates from one or more original studies are transferred to a new context or site, usually after adjusting them to the new circumstances (Hanley et al., 2007).

To help practitioners evaluate the environmental performance of products or projects, sets of generic weights or values have been derived over the years. The aim is often to build easy-to use, consistent sets of weights or values, based on transparent methods. The results are however not always as consistent and transparent as desired, often due to data and knowledge gaps. Knowledge about causes and effects, feasibility of deriving generic values instead of site-specific values and availability of data all differ between impacts. Thus, mixing valuation methods and using different units for different impacts is done in several weighting sets for pragmatic reasons.

Many of the sets are based on quite complicated models. Even though there are in most cases short descriptions of how the sets are derived as well as more detailed descriptions on websites and in reports, it is usually not straight-forward to fully understand their scope and limitations. A common structure to describe the scope and coverage of the weighting sets would make it easier for users to understand and compare different weighting schemes. In the following, an attempt to describe a number of weighting sets and their properties is done.

Table 2 lists ten weighting sets, used mainly in life cycle assessments and cost–benefit analyses. The table is an adaptation and development of work within the project getting the prices right (CPM – Swedish Life Cycle Center, 2012). In column two of the matrix, the type of value is noted, according to the classification laid out above. Next, the impact categories covered by the valuation sets are shown. They are divided into two columns, depending on where the environmental impacts valued are found in the cause–effect chain. Early in the cause–effect chain are typically chemical and physical changes, e.g. changes in concentrations in the atmosphere or changes in infrared radiation (in the case of climate change). Later in the cause–effect chain are typically biological changes, e.g. changes in ecosystems or human health. Methods that are based on changes early in the cause–effect chain may be called mid-point methods in contrast to end-point or damage methods which are based on changes later in the cause–effect mechanism (Bare et al., 2000; Hauschild, 2005). The midpoint level typically represents a mechanism where several substances may contribute to the same impact, e.g. acidification or global warming. The endpoint level typically represents issues that may be experienced and observed

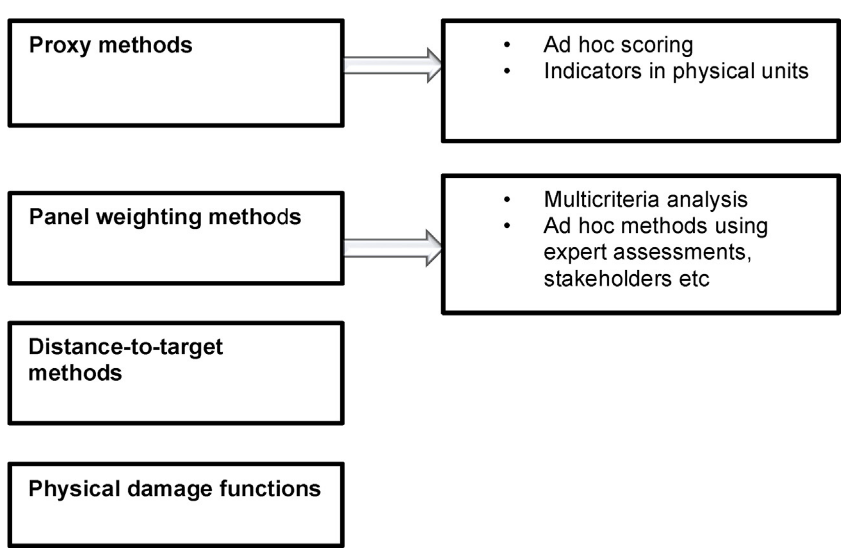

Fig. 2. Non-monetary methods for weighting environmental impacts.

Table 2
Weighting sets.

Method/feature	Type of value	Impact categories (midpoint)	Impact categories (endpoint)	Elementary flows (emissions and resources)	Unit	Population/ geographical area	Intended application	Transparency of results and publication status	Created/ last updated	References (links)
Eco-Costs'99	Marginal prevention costs	Acidification, eutrophication, carcinogenesis, dust, smog, global warming, ecotoxicity. Depletion of wood, metals and fossil fuels		All that have equivalency figures in used impact categories	€	Europeans	Product design	Published, model and data available at website	2012	www.ecocostvalue.com
Ecoindicator99	Physical damage functions	Acidification, eutrophication, climate change, ecotoxicity, ozone layer depletion, ionizing radiation, respiratory effects, carcinogenesis, regional and local effects on plant species, surplus energy for future extraction	Human health, ecosystem quality and resources	CO_2, NO_x, SO_x, NH_3, pesticides, heavy metals, HCFC, nuclides, PM, VOC, PAH, land use and extraction of minerals and fossil fuels and all equivalent substances	DALYs, % species disappeared in a certain area due to the environmental load, quality indicator for resources	Europe	Product design	Model and data available at website	1999	Ministry of Housing, Spatial Planning and the Environment (2000). www.pre-sustainability.com/impact-assessment-methods
Ecotax 2002	Political willingness to pay: taxes	Abiotic resources, biotic resources, global warming, depletion of stratospheric ozone, photochemical oxidation, acidification, eutrophication, fresh water aquatic ecotoxicity, marine aquatic ecotoxicity, terrestrial ecotoxicity, human toxicity		All that have equivalency figures in used impact categories	SEK	Swedish inhabitants	Swedish LCA	Published. Data traceable	2002	
Ecovalue08	Stated preference methods	Depletion of abiotic resources, global warming, forming of tropospherical ozone, acidification, eutrophication, human toxicity		All that have equivalency figures in used impact categories	SEK	Swedish inhabitants (eutr and acid.). Global market values for other IC	Swedish CBA, LCA, SEA and other tools. Both generic point estimates and site-specific analyses	Published. Background report with detailed information on the calculations, assumptions, etc.	Is currently being updated to also include ecotoxicity	Ahlroth (2009) and Ahlroth and Finnveden (2011)
EPS2000d	Expressed WTP		Human health, bioproductivity, biodiversity, abiotic resources, aesthetic and recreational values	All with known impacts on defined categories as global averages	ELU (=EUR in WTP)	OECD inhabitant 1998	Design and product development	All data are traceable	1999	http://lifecyclecenter.se

ExternE/NEEDS/EcoSense Web	Damage costs and avoidance costs		Human health, biodiversity, crop yield, material damage, land use	Air emissions from energy and transport sectors	€	Europeans	Policy making in energy sector	Websites with reports and on-line databases	Methodology update 2005, continuously developed	http://www.needs-project.org, www.externe.info
LIME	Expressed WTP		Human health, social welfare, biodiversity, primary productivity	1000 substances	YEN	Japan	Product design, environmental efficiency analysis, environmental accounting	Published	2004	Itsubo and Inaba (2004). LIME – A Comprehensive Japanese LCIA Methodology based on endpoint modeling. In proceedings of the 6th International Conference on Ecobalance in 2004
PUMAs Environmental Profit and Loss Account	Mixed	Not specified		Greenhouse[a] gases, water use, land use, (PM, NH_3, SO_2, CO, NO_x, VOC, waste	€	An average of several studies	CSR reporting	Not published. Using averages from many studies	2010	http://about.puma.com/. puma-completes-first-environmental-profit-and-loss-account-which-values-impacts-at-e-145-million/
ReCiPe	Physical damage functions	Clematis change, acidification, eutrophication, ozone depletion, human toxicity, ecotoxicity, ozone formation, PM, land use. Consumption of fossil fuels, minerals and water	Human health, ecosystems, resources	CO_2, NO_x, SO_2, P, CFC, Cd, PAH, VOS, DDT land use and extraction of minerals and fossil fuels and all equivalent substances	DALYs, species/year, surplus cost for resources	Europe	Product design	Published, model and data available at website	2012	Goedkoop et al. (2009). www.lcia-recipe.net
Stepwise2006[a]	Damage costs	Acidification, ecotoxicity, eutrophication, global warming, human toxicity, injuries, ionizing radiation mineral extraction, nature occupation ozone layer depletion photochemical ozone – vegetation respiratory inorganics respiratory organics		All that have equivalency figures in used impact categories	€ (monetizaton of QALYs and BAHYs)	Europeans	Product design, policy support	Published. Documented on website	2006	Weidema (2009). www.lca-net.com/projects/stepwise_ia

[a] Based on impact models from Impact2002+ and EDIP 2003 (revised versions of Ecoindicator99 and EDIP1997).

Table 3
Environmental burden from waste management scenarios – absolute values in million SEK.

	Ecotax min				Ecovalue min			
	Alt 0	Alt 1	Alt 2a	Alt 2b	Alt 0	Alt 1	Alt 2a	Alt 2b
Abiotic resources	0	0	0	0	−65	−76	−205	−214
Biotic resources	0	0	0	0	0	0	0	0
Global warming	1040	932	−191	−319	165	148	−30	−51
Ozone layer depletion	0	0	0	0	0	0	0	0
Photochemical oxidation	−24	−28	−108	−98	−7	−8	−30	−28
Acidification	−22	−32	−185	−196	−36	−53	−309	−327
Eutrophication	12	9	−7	−13	89	71	−55	−100
Fresh water aquatic ecotoxicity	−1735	−1801	−3457	−3037	0	0	0	0
Marine water aquatic ecotoxicity	0	0	−1	−1	0	0	0	0
Terrestrial ecotoxicity	2082	2947	−199	0	0	0	0	0
Human toxicity	−2880	−3180	−6840	−6690	−8	−9	−19	−18
Total	−1527	−1153	−10,989	−10,355	139	75	−649	−738
Percent change		−25	620	578		−46	−567	−632

by laymen in everyday life, such as mortality and decreased harvests.

Column six shows which emissions and resources that are included in the set and the fifth column shows which unit is used. The next two columns show which population the values are representative for and the intended use of the weighting set. As is clear from the table, the sets included are primarily derived for Europe or single European countries, the exception being EPS2000 (OECD countries) and LIME (Japan). In column nine an assessment of the traceability of the results and whether or not it has been subject to peer review, i.e. published in a scientific journal is provided. Thus "published" in the table means published in a peer reviewed journal. The tenth column shows when the set was last updated, and lastly references where more information can be found are given.

The comparability of the sets varies. The sets focusing on midpoints include the same categories to a large extent, whereas the sets focusing on endpoints are less homogenous. While all of them include human health, other categories are more varied. Also, the human health category does not necessarily include the same health endpoints.

Several of the sets use equivalency figures developed within LCA methods, so that all substances that contribute to a certain impact category can be included in the weighting.

It is evident that the result from the weighting will differ depending on which weighting set you use. In the next section, we will compare the weighted results of a few case studies using different weighting sets.

4. Comparing different weighting sets

In this section, we will use case study on waste management policy proposals from Nilsson et al. (2005) to illustrate the difference between some of the weighting sets listed above. In Nilsson et al., an LCA of a waste incineration tax proposal for Sweden was performed. Impacts from four alternative policy packages were analyzed: Alt 0 is the no-action alternative. Alt 1 represents a waste incineration tax of SEK 400 per tonne. Alternative 2a and 2b explore the potential to achieve more far-reaching environmental goals than achieved by the tax alternative as described in the committee report. The design of these two alternatives is based on experience from environmental systems analyses of waste management systems (Finnveden et al., 2007). The main difference between Alt 1 and these two alternatives is in increased source separation rates (90%), and as a consequence of that, reduced landfilling and incineration.

To show how different weighting sets, using different valuation methods can influence the results when using weighting to interpret the outcome of an LCA, we have applied four different weighting sets to the results from the life-cycle analysis described above. The impacts calculated by Nilsson et al. have been weighted with four different weighting methods, Ecotax02, EcoIndicator99, EPS2000 and EcoValue08. In Fig. 3, the difference to the no-action alternative is shown in percentage points. In all methods, alternative 2a and 2b reduce total environmental burden significantly more than alternative 1. With Ecotax02, Alternative 1 gives a slightly higher net environmental impact than the no-action alternative. This is primarily due to higher impacts on terrestrial ecotoxicity. Alternatives 2a and 2b are both significant improvements to the no-action alternatives regardless of which weighting set is used. However, the relation between alternative 2a and 2b varies between weighting methods. In Ecotax02, Alt 2a reduces total environmental impact more than alternative 2b. In EPS2000 and EcoValue08, Alt 2b reduces impacts slightly more than 2a, while the two alternatives are equal according to Ecoindicator99. The different result for Ecotax02 is due to increased impacts on freshwater ecotoxicity. Despite these differences, the weighting sets all point to the conclusion that alternative 2a or 2b. This result thus seems rather robust. To see which one of these would be preferable, one needs to look at the results for the individual impact categories, and decide which impact category and which perspective that is most important for the analysis in question.

Ecotax02 and Ecovalue08 include the same impact categories, and are based on political WTP – represented by tax rates – and expressed WTP by households, both for Sweden. Both these sets present intervals for the estimates, and are available in a Min and a Max variant. Here, we use the Min variant of both sets. In Table 3, the absolute estimates by impact category are shown. In total, the difference in percent is not that different between the two sets.

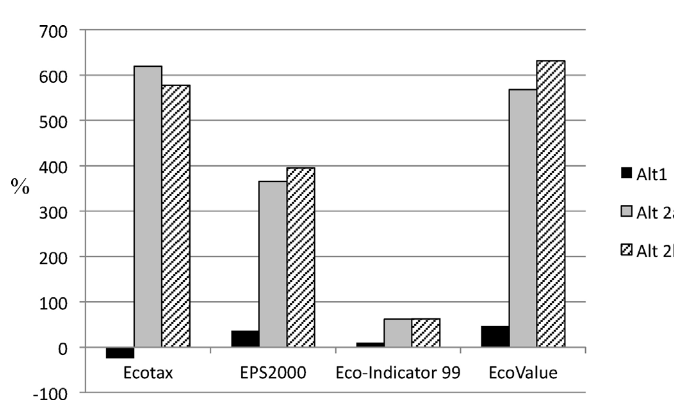

Fig. 3. Waste management scenarios. Reduction in environmental burdens relative to baseline scenario. Percent difference from Alt 0.

However, which impact categories that influence the result is quite different. The ecotoxicity categories, which are not included in Ecovalue, are large contributors according to Ecotax. In Ecovalue, the difference between the first and the last two alternatives is largely due to the reduction in acidifying substances and abiotic resources. In Ecotax, the large differences lie in the global warming, ecotoxicity and human toxicity categories. What needs to be considered when drawing conclusions from the analysis is thus (1) which weighting set that best reflects the important issues of the policy, (2) which perspective, and thus which valuation method, that is most relevant for the decision maker.

5. Conclusions

Using weighting and valuation of environmental impacts is a useful way to display results from e.g. life-cycle analyses in a comprehensive manner. As is often pointed out (Finnveden et al., 2002), it is vital to be aware of the properties of the method(s) used, and to present and discuss the weighting result in a transparent manner. The choice of weighting method should be made based on the purpose of the analysis.

As pointed out above, all weighting, be it monetary or non-monetary, is based on a number of assumptions that are more or less robust. The weighting results should therefore be seen as indicative. It is often wise to use intervals, either directly from the weighting sets if they provide them, or from several weighting sets using the same type of valuation or weighting. When doing a cost–benefit analysis, it is sometimes enough to see the magnitude of the costs and benefits, and to compare the difference with the uncertainty interval inherent in the calculations.

There is a growing flora of weighting sets with readily available weights. There are many advantages with this: use of well-known and well documented weighting sets makes results from different studies comparable and provide a certain transparency. The downside is that the calculations behind the weights are not always easy to follow, despite extensive documentation. Thus there is a risk that the weights are used uncritically and maybe in a context that they are not suited for.

It is very useful to use several weighting sets, and discuss the results thoroughly. It is often a very interesting and fruitful exercise to see if and how the results differ, why they differ, and which one seems to be the best alternative to base any recommendation on. The example provided in this article demonstrates that looking at the aggregate results is not enough. Although the weighting sets point towards the same two alternatives, indicating that this result is rather robust, the impact categories that have the largest influence on the results in the different weighting sets are different. Furthermore, to decide between the two alternatives that get similar rankings, it would be necessary to look into which impact categories influence the results in the different weighting sets and decide which categories are the most important for the analysis in question. Since many weighting sets are not sufficiently transparent as to how they are constructed and what their impact categories actually include, a general recommendation is to provide weighting sets with a declaration of content, providing a clear picture of what is included and what is not, and if possible also a recommendation of suitable uses of the weighting set. The aim of most weighting sets is to be inclusive, and caveats are not always highlighted in the documentation. However, transparency is just as important and very helpful to both the analysts and the policy makers.

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