Waste Management 33 (2013) 1290–1301

Contents lists available at SciVerse ScienceDirect

Waste Management

journal homepage: www.elsevier.com/locate/wasman

Review

Decision support models for solid waste management: Review and game-theoretic approaches

Athanasios C. Karmperis [a,b,*], Konstantinos Aravossis [a], Ilias P. Tatsiopoulos [a], Anastasios Sotirchos [a]

[a] *Sector of Industrial Management and Operational Research, School of Mechanical Engineering, National Technical University of Athens, Iroon Polytechniou 9, 15780 Athens, Greece*
[b] *Army Corps of Engineers, Hellenic Army General Staff, Ministry of Defence, Greece*

ARTICLE INFO

Article history:
Received 22 April 2012
Accepted 17 January 2013
Available online 23 February 2013

Keywords:
Cost–benefit
Decision support framework
Game theory
Life-cycle
Multi-criteria decision-making
Waste management bargaining game

ABSTRACT

This paper surveys decision support models that are commonly used in the solid waste management area. Most models are mainly developed within three decision support frameworks, which are the life-cycle assessment, the cost–benefit analysis and the multi-criteria decision-making. These frameworks are reviewed and their strengths and weaknesses as well as their critical issues are analyzed, while their possible combinations and extensions are also discussed. Furthermore, the paper presents how cooperative and non-cooperative game-theoretic approaches can be used for the purpose of modeling and analyzing decision-making in situations with multiple stakeholders. Specifically, since a waste management model is sustainable when considering not only environmental and economic but also social aspects, the waste management bargaining game is introduced as a specific decision support framework in which future models can be developed.

1. Introduction

Over the last decades, a traditional research topic in the waste management field has been focused on developing tools and methods to help decision makers with tactical decisions over waste management systems. This paper reviews the most widely used decision support frameworks, which are the life-cycle assessment (LCA), the cost–benefit analysis (CBA) and the multi-criteria decision-making (MCDM), and further presents other frameworks that can include decision support models for solid waste management (SWM). Initially, the paper presents the LCA, CBA and MCDM frameworks analyzing their strengths and weaknesses as well as their similarities and possible combinations. However, taking into consideration that any SWM system should be socially acceptable (Petts, 2000; Morissey and Browne, 2004; Weng and Fujiwara, 2011), the main challenge is to develop decision support models considering cooperative interactions among stakeholders, who are any group or individual that can affect or be affected by SWM systems. Therefore, the major objective of this paper is to introduce a game-theoretic approach to the analysis of decision-making for SWM, through:

(1) The presentation of the waste management bargaining game (WMBG) as a specific decision support framework, within which various models regarding negotiations over any aspect of SWM systems can be developed, and
(2) The analysis of the commonly applied solutions to cooperative games and the discussion of cooperative and non-cooperative game-theoretic approaches, as an effective way to consider the preferences of all stakeholders included in SWM systems.

2. Decision support frameworks for solid waste management (SWM)

Over the past decade, there are many scholars proposing different definitions of the term "decision support" (see Bardos et al., 2001; Sullivan, 2002; Bohanec, 2003), in which a common approach is that decision support provides a collection of techniques aimed at supporting people faced with complex decisions. Additionally, a model is defined as: *a representation of an object, system or idea in some form, other than that of reality itself* (Qureshi et al., 1999). Following these approaches, we define the term "decision support framework" as: *a broad outline of interlinked items supporting stakeholders in a decision-making approach for specific objectives and also serving as a guide that can be modified as required by adding or deleting items.* In other words, a decision is the output of a decision-making process that is followed by decision-makers, who develop decision support models using specific assumptions and constraints in order to achieve the desired objectives. Moreover,

* Corresponding author at: Sector of Industrial Management and Operational Research, School of Mechanical Engineering, National Technical University of Athens, Iroon Polytechniou 9, 15780 Athens, Greece. Tel.: +30 210 7721066; fax: +30 210 7723571.
E-mail address: athkarmp@mail.ntua.gr (A.C. Karmperis).

0956-053X/$ - see front matter © 2013 Elsevier Ltd. All rights reserved.
http://dx.doi.org/10.1016/j.wasman.2013.01.017

Table 1
List of abbreviations.

AHP	Analytic Hierarchy Process
B/C	Benefit/cost
CBA	Cost benefit analysis
CV	Contingent valuation
CEA	Cost-effectiveness analysis
DCFA	Discounted cash flow analysis
EASEWASTE	Environmental assessment of solid waste systems and technology
EC	European commission
EIA	Environmental impact assessment
ELECTRE	Elimination et choix traduisant la réalité (Elimination and choice expressing reality)
EPA	Environmental protection agency
IEAs	International Environmental Agreements
ISO	International organization for standardization
LCA	Life-cycle assessment
MCDM	Multi-criteria decision-making
PV	Present value
PROMETHEE	Preference ranking organization method for enrichment evaluations
RA	Risk assessment
RBMCA	Risk-based multi-criteria assessment
SWM	Solid waste management
WMBG	Waste management bargaining game
WTA	Willingness to accept
WTP	Willingness to pay

it is clear that each decision support framework includes a basic model serving as a guide, while there are alternative models that can be developed within a framework, simply by changing (adding or deleting) either the assumptions and constraints, or the objectives set by decision-makers in the basic model. It should be mentioned here that decision support frameworks should not be confused with "decision support systems", which are defined as: *computer technology solutions that can be used to support complex decision making and problem solving* (Shim et al., 2002; Abeliotis et al., 2009).

Since the late 1960s, several decision support models have been applied to the waste management field aiming to support decision-makers to evaluate complex waste management systems. The recent literature includes various papers examining waste management strategies, e.g. Berge et al. (2009) evaluate the impact of various operational and construction bioreactor landfill strategies on project economics and Parthan et al. (2012) review current models (unit-cost, benchmarking and cost modeling), which are used to estimate costs of SWM in industrial regions. However, most decision support models applied to the SWM area are mainly developed within three decision support frameworks, which are the LCA, the CBA and the MCDM. We refer readers to Morissey and Browne (2004) for a brief history of the development of decision support models in the waste management field as well as for a complete list of software tools available for MCDM and LCA. The abbreviations used throughout this paper are presented in the following Table 1.

2.1. Life-cycle assessment (LCA) framework

2.1.1. The basic model

The origins of Life Cycle Assessment (LCA) method can be tracked back to the late 1960s and early 1970s, when several concerns raised over the limitations of raw materials and energy resources (Royston, 1979) as well as over the pollution damage (EPA, 2006). Since the late 1990s, when solid waste became a worldwide issue, the LCA is a tool among others, e.g. Environmental Impact Analysis (EIA), which is mainly used for determination of the environmental impacts of products (goods or services), throughout their design, production, usage and disposal activities. As mentioned by Tukker (2000), there is no fundamental contradiction between the LCA and the EIA, as they both evaluate the environmental impacts arising through a specific societal sub-system's activities. According to the ISO 14000 environmental management standards that is the ISO 14040 and 14044 (ISO, 2006), the basic model included in the LCA framework consists of four steps, which are illustrated in Fig. 1. Rather than provide an exhausted description of the LCA's basic model, we briefly present its steps and we refer readers to Rebitzer et al. (2004) and Pennington et al. (2004) for more in-depth analysis of these steps.

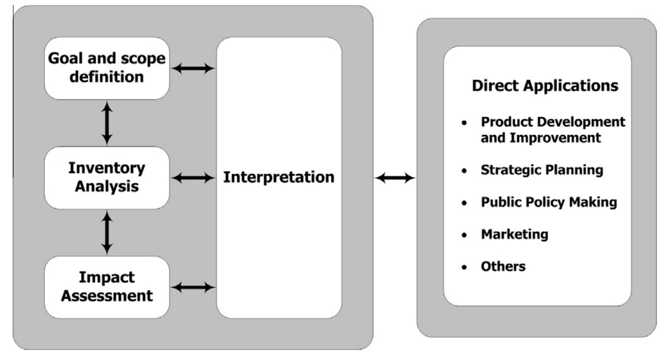

Fig. 1. The basic model of the LCA framework.

(1) In the 1st step of a LCA model that is the goal and scope definition, the system of the examined product is described, while a functional basis for comparison between alternative products is chosen and the level of analysis is also defined.
(2) According to the 2nd step, all extractions and emissions are identified and placed in an inventory list that includes all inputs and outputs of the examined products.
(3) The 3rd step of the LCA evaluates the significance of potential environmental impacts arising through the inventory analysis. Specifically, inventory data are sorted and converted to common equivalence units for each impact category (e.g. global climate change, human toxicity, acidification, etc.), in order to be summed for calculation of each category's indicator. A basic formula that can be used for these calculations is presented in the following equation (Tukker, 2000):

$$S_i = \sum_{1-j} (E_j)(e_{ij}) \tag{1}$$

where S_i denotes the score on impact category, E_j denotes the magnitude of environmental intervention j, and e_{ij} is the equivalency factor indicating the contribution of a single unit of intervention j to each impact category i.

(4) In the 4th step that is the interpretation, results arising through the three steps are reported informatively and the necessity and opportunities in minimizing the product's impacts on the environment are discussed.

The literature includes several decision support models that are developed within the LCA framework (den Boer et al., 2007; Gentil et al., 2010). Moreover, Winkler and Bilitewski (2007) examine some widely used LCA models and analyze their advantages and limitations mentioning that most efficient modeling of LCA for waste management systems can be achieved by using collective experience in a common software platform, which could be updated providing best practices in the waste management field. Furthermore, some remarkable models that have been recently developed within the LCA framework are the EASEWASTE (Kirkeby et al., 2006; Riber et al., 2008; Manfredi et al., 2009; Chen and Christensen, 2010), the

IWM-2 (McDougall et al., 2001), the WRATE (UK Environment Agency, 2012) and the US EPA (US EPA, 2006).

2.1.2. Strengths of LCA

The primary strength of a LCA model is the simplicity of its framework. Especially for waste management system analysis, LCA provides a comprehensive view of the processes and impacts involved (Finnveden et al., 2007). More specifically, the LCA methodology considers the overall life-cycle of different products identifying their impact in multiple categories and evaluating their overall environmental behavior. The strengths arising from LCA models that are applied to SWM systems can be summarized as follows:

- There are long-term benefits in environmental protection from different options, e.g. waste recycling instead of landfilling.
- A LCA model allows for both environmental improvements and economic benefits.
- All emissions into land, water and air can be quantified.
- All effects arising through material consumption on humans and eco-systems are estimated and evaluated.
- Alternative scenarios of an examined waste management strategy can be identified and compared, in order to distinguish the most suitable scenario.

2.1.3. Weaknesses of LCA

On the other hand, LCA models may include some weaknesses, e.g. application of LCA models to waste management systems tends to produce quite diverging and even conflicting results (Heijungs and Guinee, 2007). According to Villanueva and Wenzel (2007), the literature in the waste management field includes non-comparative LCA models, i.e. there is no comparison between waste management alternatives. Moreover, the complex reality may not be included in a waste management LCA model, in which the economic and dynamic linear (and non-linear) aspects are usually simplified (Ekvall et al., 2007). In general, some basic weaknesses of LCA models can be recognized in the following:

- Development of a comprehensive LCA model is a time-consuming process.
- Even though most LCA models evaluate some identified scenarios, there is always space for additional scenarios that might be considered.
- The assumptions made in a LCA model, e.g. boundary conditions, data sources, impact assessment criteria and weights might be subjective, even arbitrary.
- In cases with limitations in the available data, reliability of a LCA model's result cannot be achieved.
- LCA does not specifically quantify impacts on eco-systems and species diversity.

2.1.4. Critical issues in LCA models

In the literature, there are many papers reviewing the critical issues included in LCA models (Cleary, 2009; Gentil et al., 2010; Villanueva and Wenzel, 2007). Taking into consideration that all the assumptions used in a LCA can be criticized (Finnveden, 2000; Finnveden et al., 2007), we conclude that the most critical issues in a LCA model are the assumptions used by analysts. Specifically, since different analysts may use different assumptions in a LCA model for the same product, these assumptions may have a significant impact on the results provided by each analyst. Therefore, following the suggestion of Rebitzer et al. (2004), it is important to specify clear rules in the LCA framework, in order to use it in national and international policymaking support. As mentioned by Villanueva and Wenzel (2007), comparative waste LCAs should include three assumption categories, which are the system delimitation, the identification of secondary services (energy production, forestry, disposal capacity, fertilizing value) and the time perspective. Additionally, in order to verify that the inputs used are correct as well as to handle with efficiency the uncertainty included in a LCA model, it is also suggested that scenario analysis (Finnveden et al., 2009), or/and sensitivity analysis (Ong et al., 2012), or/and risk assessment (RA) (Kirkeby et al., 2006; Rentizelas et al., 2009) should be performed by analysts.

2.2. Cost–benefit analysis (CBA) framework

2.2.1. The basic model

As early as 1808 the basic concept of comparing cost and benefits in water related projects has been first proposed in the USA, while the CBA framework has been established in the early 1900 and has been developed rapidly since the 1950s (Hammond, 1966; Hanley and Spash, 1993). CBA is regarded as the main investment evaluation technique, through the quantitative summation of the investment's anticipated impacts on consumption benefits and resource costs (Almansa and Martínez-Paz, 2011). In general, projects are examined in a case-by-case basis, as the environmental benefits and costs are correlated with the project scope. For instance, most benefits of investments in the SWM area usually arise through environmental protection, e.g. saving of groundwater resources and limitation of CO_2 emissions (Karmperis et al., 2012a). Moreover, as mentioned by Lavee (2010) and Nahman (2011), the decision depends primarily upon the valuation of the non-market costs and benefits as the decision-making on waste management options should be based on the overall net benefits and costs to society, e.g. although recycling costs more than the conventional landfilling method, however it should be preferred as provides more environmental benefits. Fig. 2 illustrates the basic model of the CBA framework (EC, 2008), which has six basic steps as follows:

(1) In the 1st step, the socio-economic context is discussed, while the project objectives are clearly defined. It should be mentioned that these objectives should be in accordance with the National framework, a fact that should be also analyzed.
(2) In the 2nd step that is the clear identification of the project, a CBA should include a description of the project's life cycle phases (design – implementation – operational – closure), in which the connectivity with other projects should be identified and analyzed. In this context, all indirect and network effects as well as the level of analysis should be also discussed, i.e., it should be clear whose costs and benefits are going to be considered in the specific model.
(3) The 3rd step includes the study of the feasibility of the project, in which the availability of the appropriate technology and personnel skills as well as the adequacy of the demand are demonstrated. Furthermore, this step includes the evaluation of two different scenarios, which are the scenario with the investment and the scenario without it (Beria et al., 2012). In the SWM area, there is usually a set of project alternatives, which should be identified and examined by the investment analysts (see applications in Dijkgraaf and Vollebergh, 2004; Aye and Widjaya, 2006; Lavee, 2010; Karmperis et al., 2012a).
(4) In the 4th step, financial analysis is implemented by following the discounted cash flow analysis (DCFA) using a financial discount rate. More specifically, the spreadsheets that are commonly used include estimations of investment and operational costs as well as of project revenues. These spreadsheets take into consideration the financing sources and compute the project financial indicators, in order to examine the project's financial sustainability (Karmperis et al., 2012b) and the project financial impact on the national (public or/and private) investors.

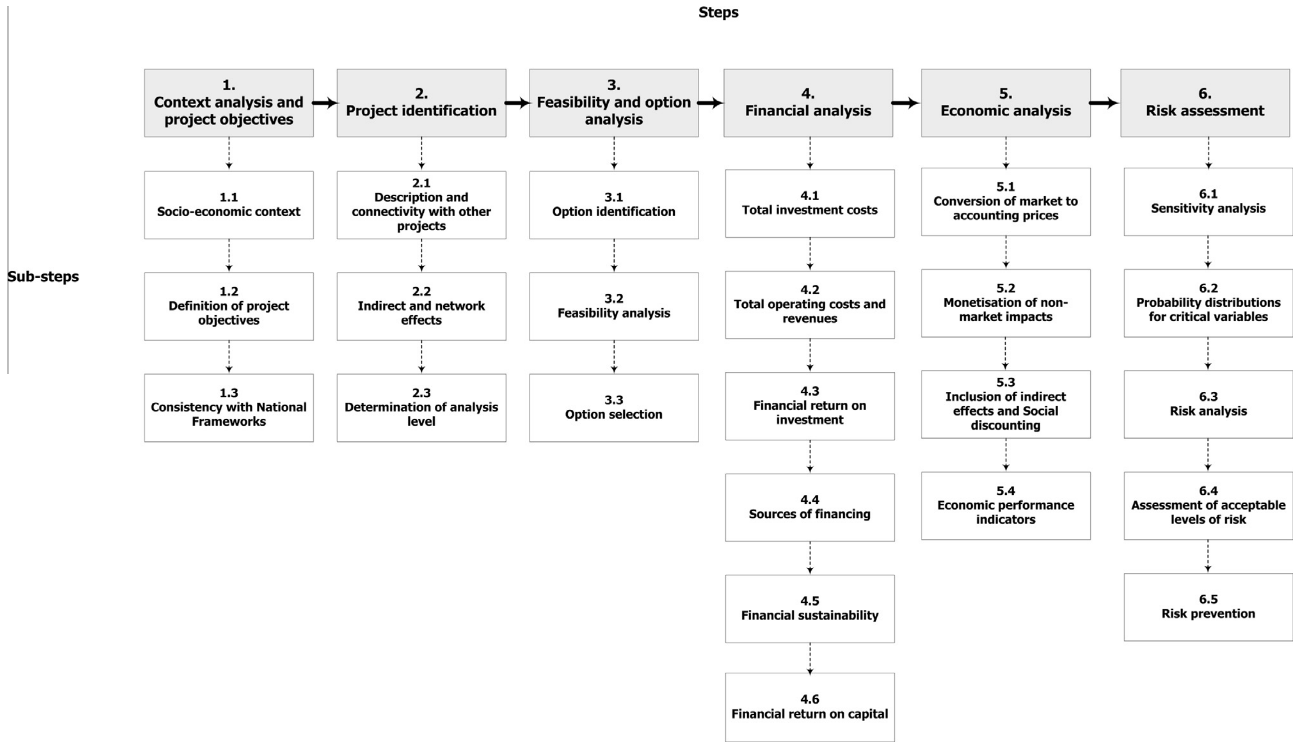

Fig. 2. The basic model of the CBA framework.

(5) Economic analysis is the 5th step of the basic CBA model. In this step, market prices are converted to accounting ones, while all project externalities and nonmarket impacts are monetized. However, the discount rate used in computations is the social discount rate reflecting the social view on how net future benefits should be valued against present ones. According to this procedure, the analysis calculates the economic performance indicators, which are the economic net present value, the economic rate of return and the benefit–cost (B/C) ratio. It should be mentioned that in investment assessment, there are three different types of B/C ratios that can be used, which are the conventional B/C, the modified B/C and the Lorie-Savage B/C ratio (see Remer and Nieto, 1995; Biezma and San Cristobal, 2006). Main difference among these ratios is that there are different variables considered in the numerator and denominator and thus there are different evaluation criteria, i.e. B/C > 1, or B/C > 0, etc. However, taking into consideration that the most common and widely applied type is the conventional B/C ratio that includes all investment benefits in the numerator and the relative costs in the denominator, i.e. B/C > 1 implies positive project evaluation, it is suggested that this specific type should be used in future CBA models. Specifically, conventional B/C ratio is the ratio of the present value (PV) of project inflows (summary of the total revenues and positive externalities), which is divided by the PV of project outflows (summary of the total costs and negative externalities), over the investment's time horizon (EC, 2008; Nahman, 2011; Karmperis et al., 2012a):

$$B/C = PV_{inflows}/PV_{outflows} \tag{2}$$

According to the basic model of the CBA framework, the criterion for evaluation result to be positive for a specific project, the value of the B/C ratio is to be higher than unity i.e. 1. If B/C > 1, then the project is feasible, as the benefits, measured by the PV of the total inflows, are greater than the relative costs, measured by the PV of the total outflows.

Risk assessment: The last step of the CBA is the risk assessment (RA) and consists of studying the probability that the selected option will achieve an accepted performance. Initially, this assessment includes the sensitivity analysis, in which the critical variables impacting mostly the financial/economic performance indicators are identified and different realistic scenarios are analyzed. Further, a probability distribution is assigned in each critical variable, in order to calculate the cumulative distributions of the performance indicators, to analyze the results and to propose actions for risk mitigation.

2.2.2. Strengths of CBA

The main strength of a CBA model is that satisfies the axiom of rational behavior (Tol, 2003), by measuring the social worth of investment projects and policies. According to the basic CBA model, main concept is that costs present the losses, while benefits present the relative gains (welfare and utility) in human wellbeing. More specifically, benefits and costs are measured by how much an individual is willing-to-pay (WTP) to secure gains or willing-to-accept (WTA) in compensation to tolerate losses (Pearce, 1998; Aravossis and Karydis, 2004; Pearce et al., 2006). Moreover, other strengths of CBA models can be found in the following:

- Both direct and indirect long-term impacts (either positive or negative) are taken into account through their summation.
- The uncertainty included in the project's performance can be handled with efficiency through a comprehensive risk assessment.
- A CBA model allows for identification and evaluation of different technical options for the examined project implementation (for instance in a SWM project, the incineration option with and without energy recovery).

- CBA models examine the project performance on behalf of both the project operator and the society through the financial analysis and the economic analysis, respectively.

2.2.3. Weaknesses of CBA

Moreover, there are some weaknesses identified in models developed within the CBA framework. Particularly, when applying CBA models to SWM systems, common problems can be summarized in the following:

- Valuing non-market goods can be complicated, e.g. in landscape and wildlife.
- A comprehensive CBA model for a SWM system can be time-consuming.
- There is much difficulty in measuring the project's benefits and costs regarding impacts in ecosystems, due to complexity of these systems.
- Values of the variables used in the financial/economic analysis may have a high non-forecasted variation throughout the waste programme life-cycle, changing the preferred option, e.g. changes in landfill costs may impact on how much waste is recycled (Morissey and Browne, 2004).

2.2.4. Critical issues in CBA models

In the literature, several papers include reviews of the CBA framework, e.g. Bleichrodt and Quiggin (1999) highlight under which conditions CBA models give the same results with the relative models developed within the cost-effectiveness analysis (CEA) framework. Prest and Turvey (1965) survey CBA methods and mention that measuring benefits is much more complicated than the relative measuring of costs. However, this procedure is much easier, when the goods provided are in the nature of private goods and provision is through public sale, so that benefits can be measured by market prices (Musgrave, 1969). More recently, Almansa and Martínez-Paz (2011) examine how CBA models should discount or, in effect, weight environmental impacts, and Pickin (2008) reviews some waste policy's CBA applications mentioning their variations in critical areas. Indisputable, one critical issue of a model implemented within the CBA framework is the discount rate used (Prest and Turvey, 1965), as a decision based on a specific social discount rate may violates the rights of future generations (see example in Hanke and Walker, 1974). Another critical issue is the level of analysis considered in a CBA model. Specifically, the level of analysis should be clear enough, in order to measure with the highest accuracy the environmental benefits and costs. For instance, if the level of analysis in a CBA model is limited in a specific area and the project has a global impact (either positive or negative) on the environment, e.g. through climate change, then the project environmental benefits (or costs) will be considered proportionally to the local level (Campbell and Brown, 2005), i.e. the project externalities in other areas may not be accounted. Conclusively, although some scholars indicate that it is impossible to price the priceless values of life, health and nature (Ackerman and Heinzerilng, 2002), CBA models can be useful in organizing public investments (Hanke and Walker, 1974). As mentioned by Turner (2007), who reviews the role of CBA in UK and European environmental policy appraisal, the basic model of the CBA framework has a number of indisputable limitations; however it still plays an important role in environmental policy assessment, while it can be a useful component in a wider decision support model that will be developed.

2.3. Multi-criteria decision-making (MCDM) framework

2.3.1. The basic model

In most cases, models developed within the MCDM framework consist of processes with specific steps that can be followed by decision makers, in order to analyze and evaluate alternative solutions of a problem. The environmental literature includes various MCDM models classified in three basic approaches, which are the Multi Attribute Utility Theory (see Wallenius et al., 2008), the Analytic Hierarchy Process (AHP) (Saaty, 1978) and the analytic network process (see Promentilla et al., 2006) as well as the Outranking, e.g. the PROMETHEE (Brans and Vincke, 1985) and the ELECTRE (Roy and Vanderpooten, 1996) models. We refer readers to Linkov et al. (2004) and Huang et al. (2011) for detailed discussions over the similarities and differences among these models. Over recent years, these models have been widely applied to problems regarding waste management systems (see for example Aravossis et al., 2001; Hung et al., 2007; Ohman et al., 2007; Khan and Faisal, 2008; Hsu et al., 2008; Garfi et al., 2009; Sener et al., 2010; Karagiannidis et al., 2010; De Feo and De Gisi, 2010a; Lin et al., 2010). However, the basic model included in the MCDM framework can be presented in the following steps (Mourits and Oude Lansink, 2006), which are illustrated in Fig. 3:

(1) In the 1st step, the objectives of the MCDM should be clearly defined, in order to establish the decision context. These objectives have to be specific, realistic and measurable, in order to identify all options satisfying them.
(2) According to the 2nd step, all options that satisfy the project/program objectives are identified, e.g. a municipal waste treatment project can be implemented through landfilling, or/and recycling and landfilling, or/and recycling and incineration without energy production, or/and incineration with electricity or/and heat production, etc.
(3) In the 3rd step, decision makers define the evaluation criteria reflecting performance in meeting the objectives and further they assign the weight values (priorities) in each criterion.
(4) In the last step, the alternatives' scores are computed so as decision-makers can have a comparable view of the alternatives, in order to evaluate them and to select the preferred one (the solution is selected according to the criteria used as well as their weight values).

2.3.2. Strengths of MCDM

The strengths of models that are developed within the MCDM framework can be stated as:

- There are multiple conflicting criteria that can be formally incorporated into the management planning process (Kou et al., 2011).
- Models developed within the MCDM framework can use not only quantitative but also qualitative criteria for the evaluation of a SWM project's alternatives (Linkov et al., 2006).

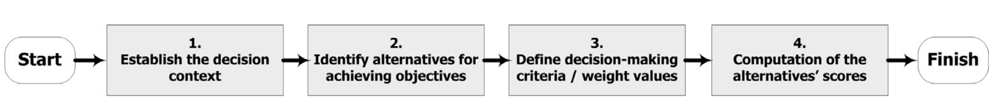

Fig. 3. The basic model of the MCDM framework.

- In most cases, MCDM models are flexible, as there are alternative categories of evaluation criteria that can be used, such as economic, environmental and technical criteria (Linkov et al., 2009).
- The majority of MCDM models that are used by a single decision-maker, can be easily expanded in group decision-making models, simply by assigning weights in each decision-maker's outcome, so as the overall result will take into account all participants' preferences.
- Even though application of different MCDM models to the same problem may result in different prioritization of management alternatives, the top few alternatives are the same no matter which MCDM model is used (Huang et al., 2011).

2.3.3. Weaknesses of MCDM

Moreover, the main weaknesses of MCDM models when applying to SWM systems are the following:

- There are few models developed within the MCDM framework that take into consideration the impact of risks that may be posed in the project alternatives (Karmperis et al., 2012a).
- In the waste management field, MCDM models evaluate only the alternative solutions and do not provide any information for waste minimization and waste prevention (Morissey and Browne, 2004).
- Since the evaluation result can be changed when different criteria or/and criteria weight values are used, the selection of evaluation criteria to be used and especially the assignment of weights in each criterion can be highly subjective.

2.3.4. Critical issues in MCDM models

From the above, it is clear that the most critical issues in a MCDM model are the evaluation criteria set by the decision makers and specifically the weight assigned in each criterion. Furthermore, it should be mentioned that according to the flexibility of the MCDM framework, MCDM models can be very simplistic if they do not consider all aspects of the problem and on the contrary can be very complicated when they take too many aspects into account. Conclusively, in order to be useful to decision-makers, models developed within the MCDM framework should be supported with the contribution of experts when needed, while they should be flexible enough to consider preferences from different stakeholders, in order to become as transparent as possible. Moreover, since there is additional research needed to acquire further knowledge and understanding of different types of uncertainty inherent in environmental decision-making (Ascough et al., 2008), it is suggested that future models developed within the MCDM framework should take into account the risks that may be posed in the alternatives examined.

3. Combinations and extensions of LCA–CBA–MCDM

One similarity of the aforementioned frameworks is that the decision can be based not only on economic terms, since the environmental impacts can be also taken into account; however, no framework considers environmental, economic and social aspects together in the application of the model (Morissey and Browne, 2004). Moreover, a common weakness is that all assessment techniques require contribution of experts in several disciplines. Another similarity is that in models developed within the LCA and MCDM frameworks, the selection of the assessment criteria and particularly the weights assigned in each criterion are highly subjective, because different analysts in a LCA model or in a MCDM model would probably select different alternatives. On the other hand, subjective can also be the valuation method used in a CBA model, in order to monetize non-market values. Furthermore, similarly with the LCA, in which a product is examined in a specific time horizon, a project evaluated within the CBA framework is examined in a specific time horizon too, namely the project's life-cycle. In both cases, every product/project has a "life" starting with the development/design phase, followed by the production/implementation phase, the use/operational phase, and the product end of life/project close phase.

In general, an effective way to overcome weaknesses of the basic models developed within specific frameworks, is to combine them developing extended models, in order to maintain their strengths, e.g. see Xu et al. (2009), Li et al. (2011), and Sun et al. (2012) with applications of the coupled fuzzy flexible programming and the internal-parameter linear programming models to waste management systems. According to Tudela et al. (2006), a CBA model can be effectively combined within the MCDM framework, while Linkov and Seager (2011) suggest that LCA and MCDM combination provides a flexible strategy for decision-makers. In recent literature, there are several papers combining the LCA, CBA and MCDM frameworks (Duinker and Greig, 2007; Benedetti et al., 2008; Carter and Keeler, 2008; Lee et al., 2009; Karmperis et al., 2012a). Moreover, a MCDM model can be effectively applied to weighting and summing results of a LCA model into a single index (Hermann et al., 2007), while decision support models for SWM can be linked with economic, engineering and risk assessment models (see examples in Nakanishi et al., 2003; Eriksson and Bisaillon, 2011). As mentioned by Gouldson et al. (2009), there is a need to develop reliable and responsive models of risk assessment and monitoring, in order to embed these models within organizations and making them fit for purpose (Pollard et al., 2004, 2008).

Conclusively, it is difficult to distinguish one decision support framework as the most suitable for all SWM systems. Since all waste management strategies are examined in a case by case basis, it is crucial to identify and to develop a decision support model that fits best in the objectives of each decision-making process. Therefore, it is clear that the selection of the most appropriate framework and further the choice of an existed model or the development of a new model combining the basic models included in the main decision support frameworks, should be based on the scope of evaluation.

4. Game-theoretic approaches in decision support models for SWM

4.1. Stakeholders in waste management systems

Over the last decade, the waste management literature includes several decision support models developed within the aforementioned frameworks. However, a decision-making problem has more than one objective to achieve, while there is always a trade-off between the different objectives, advocated by different interest groups or stakeholders. Specifically, a stakeholder is any group or individual that can affect, or is affected by the achievement of a SWM system's objectives. The MCDM is proposed as a useful framework for the introduction of the stakeholder concept (Banville et al., 1998), e.g. Garfi et al. (2009) and De Feo and De Gisi (2010a) develop MCDM models considering the stakeholder preferences for assigning values in the criteria weights. In the CBA, stakeholders' preferences can be taken into account through the WTP/WTA surveys; however, as mentioned by Weng and Fujiwara (2011), these surveys should be conducted routinely, in order to reflect the dynamic changes of the public consciousness in the analysis outcomes.

Moreover, since there are environmental, economic or social externalities arising from a SWM system operating in a specific area, or from the respective systems that are used in the neighboring areas, it is concluded that decision-making over SWM systems should be expanded among several groups of stakeholders, who have to be included from the very beginning of a waste management process (BDI, 1999). Environmental protection within waste management systems at present to meet sustainability goals in the future (Pires et al., 2011), requires collective actions between various stakeholders, while collaboration occurs within the context of cooperative work and is defined as: *multiple individuals working together in a planned way in the same production process or in different but connected production processes* (Wilson, 1994). However, as mentioned by Thorneloe et al. (2007) and De Feo and De Gisi (2010b), communities may have different preferences over environmental protection, e.g. one may have greater concern over air quality issues whereas others may value water quality more. In recent papers examining the attitude of communities towards SWM systems, Yau (2010) analyzes the behavior of domestic waste recycling from a rational choice perspective and Kurisu and Bortoleto (2011, 2012) compare the waste prevention behaviors among three megacity regions demonstrating the different preferences among different groups of stakeholders. Specifically, these stakeholders can be different nations, or different public authorities, or municipal authorities and citizens, in a regional, national, and local level, respectively. Cooperation between two or more stakeholders implies a voluntarily arrangement, in which they engage having mutually beneficial exchange. In such multi-stakeholder cooperative situations where the outcome depends on the choice made by every party, game theory can be effectively applied (Nagarajan and Sosic, 2008).

In the literature, several game-theoretic models are applied to negotiations regarding environmental problems and particularly to International Environmental Agreements (IEAs) (see Weikard and Dellink, 2010; McGinty et al., 2012). Moreover, cooperative game-theoretic approaches are used to modeling negotiations over the sharing of costs and benefits in brownfields redevelopment (Wang et al., 2007, 2011), in water resource sharing (Ozelkan and Duckstein, 1996; van den Brink et al., 2012) and in hydropower licensing (Madani, 2011). Kołodziej (2007) analyze the water resource systems when there is disagreement of the interests of users, who seek minimizing the costs connected with the responsibility for the pollution of the natural environment. In the SWM area, Hideshima et al. (1999) present a game-theoretic approach analyzing coalition formation among Japanese cities, which cooperate in the reusing of waste soil for construction and Jørgensen (2010) studies a waste disposal game among three neighboring regions. Taking into consideration that cooperative game theory can provides the basis for choosing satisfactory SWM alternatives (Cheng et al., 2002), cooperative game-theoretic models can be used to analyze the stability of agreements through the fair distribution of the costs and benefits arising from environmental policies (Bahn et al., 2009), and particularly from the solid waste policy (Moretti, 2004; Johansson-Stenman and Konow, 2010). Clearly, a game-theoretic model can improve the sustainability of a SWM system, since it considers the fair distribution of benefits and costs among stakeholders, namely the government, the local authorities, the technical experts and the community.

4.2. Waste management bargaining game (WMBG) framework

Generally, even though insights from game theory are recorded in two Plato's texts, namely the ''Laches'' and the ''Symposium'' (see Ross, 2010), the mathematical context of game theory has been established by von Neumann and Morgenstern (1944). In its current form, game theory is a collection of mathematical models to analyzing situations of conflict and cooperation and thus, it is divided in two categories that is the non-cooperative and cooperative game theory. The main difference among these types is that in non-cooperative games players make decisions independently, while in cooperative games the basic unit of analysis is on sets of players (Maskin, 2011). Herein, we focus on the cooperative part of game theory and specifically on the waste management bargaining game (WMBG) framework, in which alternative decision support models can be developed.

4.2.1. The basic model

This section presents the basic model of the WMBG framework. This model addresses the problem where a finite set of stakeholders, namely players $N = \{1, 2, 3, \dots, n\}$, negotiate over a surplus yielded through SWM systems. In such cases, the participants' objectives are partially cooperative, as they aim at reaching an agreement regarding a SWM system, and partially conflicting, because each player has its own utility function regarding the negotiation outcome. Indicatively, some examples of bargaining over different aspects of SWM systems are following presented:

- In the case where the set N consists of public authorities who negotiate over the division of the national SWM system's costs, then each player wants to maximize her own utility by minimizing the cost allocated to her.
- Another example is the case where a set of nations negotiate over their common strategies in management and disposal of toxic substances, which can affect ecosystems and human populations far from the point of use or disposal. On one hand, the utility for a nation that uses toxic substances for industrial purposes is increased, as far as the negotiation outcome does not prohibits the use of these substances. On the other hand, the relative utility for the neighbored nations which may not use toxic substances is increased, in proportion to the strictness of the rules for their disposal.

More specifically, taking into consideration that any rational player seeks to maximize her own utility by proposing a specific strategy in a negotiation, it is concluded that there is conflict of interests and alternative bargaining outcomes may be realized. Therefore, for the purpose of modeling and analyzing decision-making in such multi-player bargaining cases, WMBG framework can be a natural choice. The following assumptions are used in the basic model of the WMBG:

(1) Each player is faced with a set S of feasible bargaining outcomes, any of which presents the result if all participants agree upon.
(2) It is assumed that S, (which is illustrated in Fig. 4 as a sub-set of $\mathbb{R}^2$ for the two-player bargaining case), is closed, convex, non-empty and bounded.
(3) The set of players N is called the grand-coalition, while any subset in which the N can be divided, is called coalition (Renna and Argoneto, 2011), and a coalition with just one player is called singleton (Karmperis et al., 2012c,d). Specifically, if players do not agree in a bargaining outcome, then sub-coalitions of N may form and each player gets a specific payoff that is the disagreement point, i.e. each player receives what it could obtain through the non-cooperative option. Let $d \in S$ denotes the disagreement point.

The basic model of the WMBG framework can be defined by a pair (N, p), where $p : 2^N \rightarrow \mathbb{R}$ with $p(\emptyset) = 0$, is the characteristic function representing the collective payoff for players forming the grand-coalition (Guardiola et al., 2007). Moreover, the solution provided by this model is a vector $x \in \mathbb{R}^N$ representing the allocation of the overall profit/cost $p(N)$ to each player.

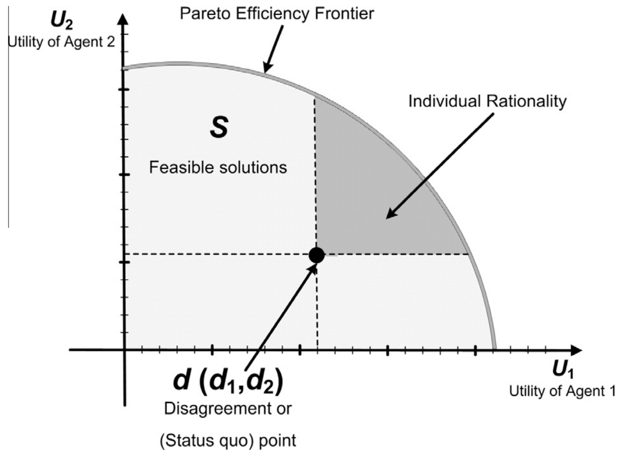

Fig. 4. Illustration of feasible, individual rational and Pareto-efficient solutions.

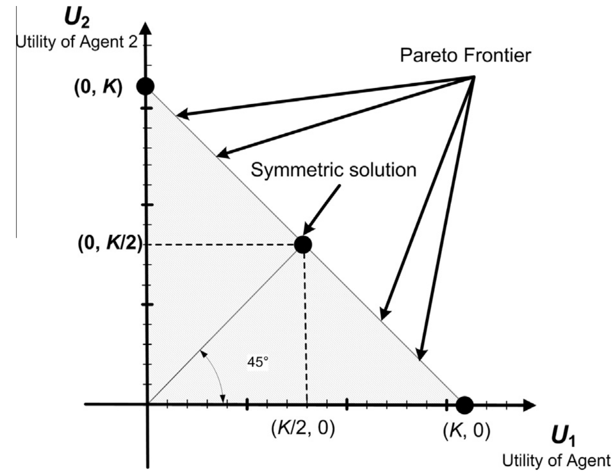

Fig. 5. Illustration of the symmetric solution.

4.2.2. Axioms – Nash bargaining solution and Shapley value

In cooperative game theory, solutions are mainly characterized by using axiomatic approaches, i.e. in terms of natural axioms that are satisfied. The most widely used solutions, which can be used in a model developed within the WMBG framework, are the Nash bargaining solution and the Shapley value. These solutions are defined by specific axioms, as follows:

Axiom 1 (*Feasibility*). As presented in Fig. 4, a solution provided through a WMBG model is feasible, if the payoff vector x allocates the value yielded through cooperation, according to following inequality:

$$\sum_{i=1}^{n} x_i \leqslant p(N) \tag{3}$$

Axiom 2 (*Pareto efficiency*). Moreover, in the case that the payoff vector x allocates exactly the overall value, then the solution is Pareto-efficient:

$$\sum_{i=1}^{n} x_i = p(N) \tag{4}$$

Pareto efficiency implies that there are no Pareto improvements that can be made in the allocation, i.e. for any other allocation $y \neq x$, with which at least one player $i \in N$ is better off, there is at least one other player $j \in N$ worse off (see Pareto frontier in Fig. 4).

Axiom 3 (*Individually Rationality*). As can be seen in Fig. 4, each player $i \in N$, should gets at least as it could obtain through the non-cooperative option, i.e. $\forall\ i = 1, 2, \ldots, n$:

$$x_i \geqslant p(i) \tag{5}$$

Axiom 4 (*Symmetry*). According to the axiom of symmetry, if players forming a grand-coalition have symmetric utility functions, then they should receive equal allocations of the profit/cost. Fig. 5 illustrates the symmetric solution for the two-person model, while for models with more than two players where all players $i, j \in N$, are identical, we have the following equation:

$$x_i = x_j \tag{6}$$

Axiom 5 (*Independent of Irrelevant Alternatives*). This axiom implies that if the solution $x \in \mathbb{R}^N$ is the bargaining outcome among the set of feasible outcomes S, then for all other sub-sets of S containing x, the specific vector x will be again the bargaining outcome, as presented in the following Fig. 6.

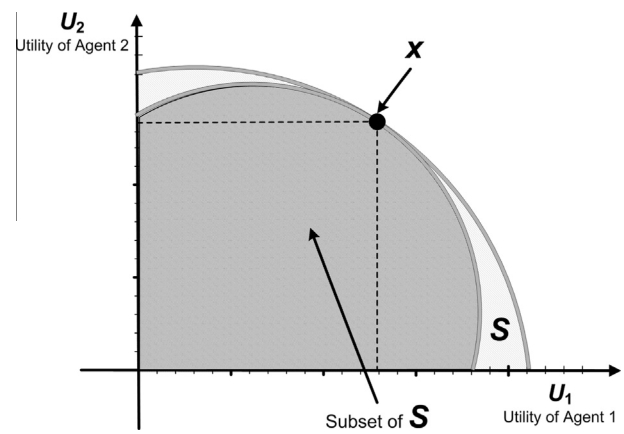

Fig. 6. Illustration of the axiom independence of irrelevant alternatives.

Axiom 6 (*Scale Invariance*). This implies that the solution x is independent of the scale that is used in measuring the players' utilities, i.e. if we multiply the utilities of all players $i = 1, 2, 3, \ldots, n$ by constants $(a_1, a_2, a_3, \ldots, a_n)$, then we have the feasible set S' and we get the relative solution x' through the multiplication of the players' coordinates by these constants (Jain and Mahdian, 2007).

Axiom 7 (*Coalitionally Rational*). This axiom implies that there is no coalition which has an incentive to quit the grand-coalition, as it is worse off to act independently. Specifically, a payoff vector $x = (x_1, x_2, x_3, \ldots, x_n)$, where $x_i \geqslant 0$ and $\sum_{i=1}^{n} x_i = p(N)$, is coalitionally rational, if for all sub-coalitions $M \subset N$, there are Eqs. (7.1) and (7.2), for the profit and the cost game, respectively:

$$\sum_{i \in M} x_i \geq p(M) \tag{7.1}$$

$$\sum_{i \in M} x_i \leq p(M) \tag{7.2}$$

Axiom 8 (*Additivity*). If for a given set of players N we have more than one cooperative bargaining games defined by pairs: (N, p_1), (N, p_2), $\ldots$, (N, p_m), then for the single game $(N, p_1 + p_2 + \cdots + p_m)$ that is defined by: $(p_1 + p_2 + \cdots + p_m)\ (M) = (p_1)\ (M) + p_2\ (M) + \cdots + p_m\ (M)$ for each coalition M, the value allocated in each player i is given:

$$x_i(N, p_1 + p_2 + \cdots + p_m) = x_i(N, p_1) + x_i(N, p_2) + \cdots + x_i(N, p_m).$$

Axiom 9 (*Dummy player*). This axiom implies that if one player i is dummy that is: $p(M \cup i) - p(M) = 0, \forall M \subset N$, then no value will be allocated to her: $x_i\ (p) = 0$.

A formal solution, which can be used in models developed within the WMBG framework, was first proposed by Nash (1950), and it is called the Nash bargaining solution. This solution is unique in satisfying Axioms 1–5, i.e. feasibility, Pareto optimality, symmetry, independence of irrelevant alternatives and independence of equivalent utility representations. According to Roth (1979), who presents in detail all axioms used by Nash, the specific solution is the function that maximizes the geometric average of the players' gains through the agreement, instead of settling for the disagreement point d. Specifically, as illustrated in Fig. 7, the Nash bargaining solution for a two-person bargaining game, is the solution maximizing the product of the excesses: $(u_1 - d_1)\ (u_2 - d_2)$, subject to constraints: $u_1 \geqslant d_1$, and $u_2 \geqslant d_2$.

For example, let's consider a case where a local authority representing the citizens and the manager of the local waste treatment plant negotiate over the service fee ($/ton of waste treated in the plant). It is assumed that the monetary cost for the plant operator is 100 $/ton and the WTP of the citizens is estimated through a survey at 150 $/ton. Following the WMBG model, the local authority knows that the cost is less than 150 $/ton and the plant manager knows that the value is greater than 100 $/ton. So long as the players differ over which service fee is most suitable, there is a need for negotiation over which outcome should be agreed upon. If players are rational then they seek to maximize their own utility. That is, the plant manager and the local authority seek 150 and 100 $/ton as a service fee, respectively, i.e. there is a surplus of 50 $/ton, which should be divided among them. This surplus is generated when both players agree at a specific service fee, and thus the disagreement outcome is $(d_1,\ d_2) = (0,\ 0)$. As presented in Fig. 8, if players have symmetric utility functions, then the Nash bargaining solution for the surplus division is $(u_1,\ u_2) = (25,\ 25)$, i.e. the service fee that will be paid is 125 $/ton.

Moreover, the WMBG can be effectively applied to negotiations between a local authority and citizens regarding the location of a waste disposal site (see example in Binmore, 2005). It is mentioned that on the contrary with other modeling approaches (LCA, CBA, MCDM) that can be followed in these cases, the WMBG approach should be considering as more sustainable, since it takes into account not only environmental and economic but also social aspects.

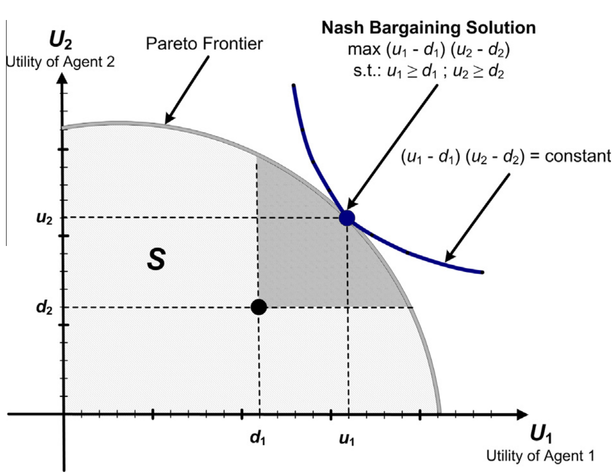

Fig. 7. Illustration of the Nash bargaining solution.

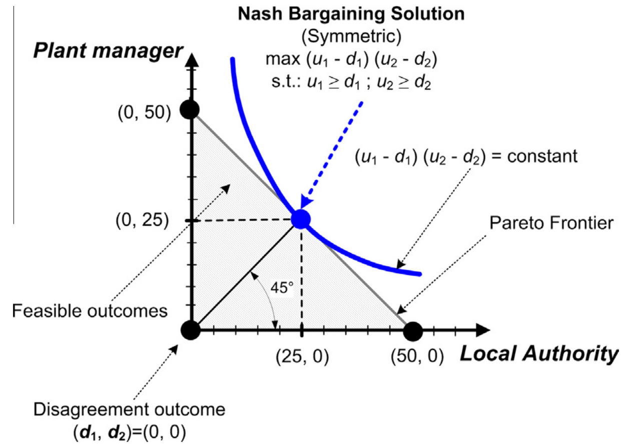

Fig. 8. Solution with a game-theoretic approach.

Another solution that can be used in WMBG models is the Shapley value (Shapley, 1953), which provides a specific concept in analyzing how coalitional powers impact all the possible cooperative game outcomes. Specifically, in a bargaining problem, the Shapley value allocates a specific value to all players included in the grand-coalition, by taking into consideration the marginal contribution of each player i to the worth of each coalition M that is a subset of N. Specifically, Shapley showed that the Axioms 2, 4, 8 and 9, i.e. efficiency, symmetry, additivity and dummy player, are satisfied by a unique value for each player, which is computed through the following equation:

$$x_i(p) = \sum_{M \subset N\{i\}} \frac{|M|!(n - |M| - 1)!}{n!} (p(M \cup \{i\}) - p(M)) \tag{8}$$

4.3. Future models within cooperative and non-cooperative game approaches

As mentioned above, models including cooperative behavior among stakeholders seem to be a natural choice to model decision-making in bargaining over any aspect of SWM systems. Taking into consideration that the Nash bargaining solution can be easily expanded in n-stakeholders, with $n > 2$ (Harsanyi, 1959, 1963), while the main advantage of the Shapley value is that it provides a "fair" solution that always exists and is unique, we conclude that both solutions concepts can be very useful if applied to models developed within the WMBG framework. Other fair solution concepts included in the literature are the solution of von Neumann and Morgenstern (1944), the core (Gillies, 1959) and the Nucleolus (Schmeidler, 1969). Specifically, due to the fact that the basic notion in SWM is sustainability, this can be achieved when decision support models developed within the WMBG framework provide solutions that are considered to be fair for all stakeholders (see application in Karmperis et al., 2012c).

On the other hand, there are several opportunities to develop decision-support models for SWM within non-cooperative game-theoretic approaches, where the concept of Nash equilibria (Nash, 1951) is of significant importance (Binmore, 2011). Specifically, as mentioned by Weibull (1996) for the mass-action interpretation, if a specific non-cooperative game is played repeatedly by individual players who are drawn at random from large populations, then their best reply strategies will vary over time by adjusting their preferences to whatever is currently being played in the population at large. In other words, a decision support model for SWM should aim to identify the waste management strategy that forms an equilibrium point in each population of stakeholders, in order to increase the sustainability of the overall SWM system.

However, as happens with the decision support models developed within the LCA framework, crucial in game-theoretic models are the assumptions used by analysts. It should be noted here that a decision support model that will be developed within a game-theoretic framework (either cooperative or non-cooperative) should include realistic assumptions, in order to avoid criticisms over its subjectivity and also to has wider application to different SWM systems.

5. Conclusions

The development of decision support models for sustainable SWM is one of the most widely examined topics over the last decades. This paper reviews the current literature focusing on three basic frameworks, in which alternative decision support models can be developed. Specifically, the basic models included in LCA, CBA and MCDM frameworks are presented, while their strengths and weaknesses as well as their similarities are also discussed. In summary, the following outcomes can be pointed out:

- The most critical issues, which are included in decision support models developed within these frameworks, and should be addressed by decision-makers are the following:
  (i) In LCA models, most critical issues are the assumptions used by analysts.
  (ii) In CBA models, crucial are the selection of the discount rate and the level of analysis.
  (iii) Most critical issues in models developed within the MCDM framework are the selection of evaluation criteria and further the selection of the criteria weight values.
- Since all frameworks have shortcomings, it is suggested that future models can be developed combining suitably the LCA, CBA and MCDM frameworks, in order to maximize their strengths or/and minimize their weaknesses.

Additionally, since there are various groups of stakeholders (groups or individuals with conflict of interests) who are involved in SWM systems, the game-theoretic approach is proposed as a sustainable way to model and analyze decision-making in multi-stakeholder situations. The WMBG is presented as a specific decision support framework, in which various decision support models can be developed. Particularly, in cases where different stakeholders negotiate over any aspect of a SWM system, future models can use the axiomatic approaches included in the Nash bargaining solution or in the Shapley value, in order to present the most sustainable solution for the gains or losses allocated among all stakeholders. Future papers can be focused on non-cooperative game-theoretic approaches by developing decision support models, which will identify the waste management strategies that form equilibrium points, in order to increase the sustainability of the overall SWM system.

Acknowledgments

The authors deeply appreciate the Editors and two anonymous reviewers for their insightful comments and suggestions.

References

Abeliotis, K., Karaiskou, K., Togia, A., Lasaridi, K., 2009. Decision support systems in solid waste management: a case study at the national and local level in Greece. Glob. Nest J. 11, 117–126.
Ackerman, F., Heinzerilng, L., 2002. Pricing the priceless: cost–benefit analysis of environmental protection. Univ. Pennsylvania Law Rev. 150, 1553–1584.
Almansa, C., Martínez-Paz, J.M., 2011. What weight should be assigned to future environmental impacts? A probabilistic cost benefit analysis using recent advances on discounting. Sci. Total Environ. 409, 1305–1314.
Aravossis, K., Anagnostopoulos, P., Koungolos, A., Vliamos, S., 2001. A new methodology approach for the technical–economical evaluation of alternative waste disposal methods by use of multi criteria analysis. In: Proceedings of the Seventh International Conference on Environmental Science and Technology, pp. 40–51.
Aravossis, K., Karydis, V., 2004. Combination of monetary valuation techniques and application to environmental impact receptors. Fresen. Environ. Bull. 13, 283–288.
Ascough II, J.C., Maier, H.R., Ravalico, J.K., Strudley, M.W., 2008. Future research challenges for incorporation of uncertainty in environmental and ecological decision-making. Ecol. Model. 219, 383–399.
Aye, L., Widjaya, E.R., 2006. Environmental and economic analyses of waste disposal options for traditional markets in Indonesia. Waste Manage. 26, 1180–1191.
Bahn, O., Breton, M., Sbragia, L., Zaccour, G., 2009. Stability of international environmental agreements: an illustration with asymmetrical countries. Int. Trans. Oper. Res. 16, 307–324.
Banville, C., Landry, M., Marte, L.J.-M., Boulaire, C., 1998. A stakeholder approach to MCDA. Syst. Res. 15, 15–32.
Bardos, R.P., Mariotti, C., Marot, F., Sullivan, T., 2001. Framework for decision support used in contaminated land management in Europe and North America. Land Contam. Recl. 9, 149–163.
BDI, 1999. Implementation of Life Cycle Assessments (LCAs) to Inform the Public and Politicians: Guidance for Handling Life Cycle Assessments (LCAs) in Companies and Industrial Associations. Bundesverband der Deutschen Industrie, Cologne, 1999.
Benedetti, L., Bixio, D., Claeys, F., Peter, A., Vanrolleghem, P.A., 2008. Tools to support a model-based methodology for emission/immission and benefit/cost/risk analysis of wastewater systems that considers uncertainty. Environ. Model. Softw. 23, 1082–1091.
Berge, N.D., Reinhart, D.R., Batarseh, E.S., 2009. An assessment of bioreactor landfill costs and benefits. Waste Manage. 29, 1558–1567.
Beria, P., Maltese, I., Mariotti, I., 2012. Multicriteria versus cost benefit analysis: a comparative perspective in the assessment of sustainable mobility. Eur. Transp. Res. Rev.. http://dx.doi.org/10.1007/s12544-012-0074-9.
Biezma, M., San Cristobal, J., 2006. Investment criteria for the selection of cogeneration plants—a state of the art review. Appl. Therm. Eng. 26, 583–588.
Binmore, K., 2005. Natural Justice. Oxford University Press, New York, p. 22–25.
Binmore, K., 2011. Commentary: Nash's work in economics. Games Econ. Behav. 71, 2–5.
Bleichrodt, H., Quiggin, J., 1999. Life-cycle preferences over consumption and health: when is cost-effectiveness analysis equivalent to cost–benefit analysis? J. Health Econ. 18, 681–708.
Bohanec, M., 2003. Decision support. In: Mladenić, D., Lavrač, N., Bohanec, M., Moyle, S. (Eds.), Data Mining and Decision Support: Integration and Collaboration. Kluwer Academic Publishers, pp. 23–35.
Brans, J.P., Vincke, P., 1985. A preference ranking organisation method: (the PROMETHEE method for multiple criteria decision-making). Manage. Sci. 31, 647–656.
Campbell, H.F., Brown, R.P.C., 2005. A multiple account framework for cost–benefit analysis. Eval. Program Plann. 28, 23–32.
Carter, T., Keeler, A., 2008. Life-cycle cost–benefit analysis of extensive vegetated roof systems. J. Environ. Manage. 87, 350–363.
Chen, D., Christensen, T.H., 2010. Life-cycle assessment (EASEWASTE) of two municipal solid waste incineration technologies in China. Waste Manage. Res. 28, 508–519.
Cheng, S., Chan, C.W., Huang, G.H., 2002. Using multiple criteria decision analysis for supporting decisions of solid waste management. J. Environ. Sci. Health, Part A: Toxic/Hazard. Substances Environ. Eng. 37, 975–990.
Cleary, J., 2009. Life cycle assessments of municipal solid waste management systems: a comparative analysis of selected peer-reviewed literature. Environ. Int. 35, 1256–1266.
De Feo, G., De Gisi, S., 2010a. Using an innovative criteria weighting tool for stakeholders involvement to rank MSW facility sites with the AHP. Waste Manage. 30, 2370–2382.
De Feo, G., De Gisi, S., 2010b. Public opinion and awareness towards MSW and separate collection programmes: a sociological procedure for selecting areas and citizens with a low level of knowledge. Waste Manage. 30, 958–976.
den Boer, J., den Boer, E., Jager, J., 2007. LCA-IWM: a decision support tool for sustainability assessment of waste management systems. Waste Manage. 27, 1032–1045.
Dijkgraaf, E., Vollebergh, H.R.J., 2004. Burn or bury? A social cost comparison of final waste disposal methods. Ecol. Econ. 50, 233–247.
Duinker, P.N., Greig, L.A., 2007. Scenario analysis in environmental impact assessment: Improving explorations of the future. Environ. Impact Asses. Rev. 27, 206–219.
EC, 2008. Guide to Cost–Benefit Analysis of Investment Projects. European Commission, Directorate General Regional Policy, Brussels, pp. 13–15 (41–42, 99, 233–239).
Ekvall, T., Assefa, G., Anna Bjorklund, A., Eriksson, O., Finnveden, G., 2007. What life-cycle assessment does and does not do in assessments of waste management. Waste Manage. 27, 989–996.
EPA, Environmental Protection Agency, 2006. Life Cycle Assessment: Principles and Practice. Scientific Applications International Corporation (SAIC), EPA/600/R-06/060, Ohio.

Eriksson, O., Bisaillon, M., 2011. Multiple system modelling of waste management. Waste Manage. 31, 2620–2630.
Finnveden, G., 2000. On the limitations of life cycle assessment and environmental systems analysis tools in general. Int. J. Life Cycle Asses. 5, 229–238.
Finnveden, G., Björklund, A., Ekvall, T., Moberg, Å., 2007. Environmental and economic assessment methods for waste management decision-support: possibilities and limitations. Waste Manage. Res. 25, 263–269.
Finnveden, G., Hauschild, M.Z., Ekvall, T., Guinee, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., Suh, S., 2009. Recent developments in life cycle assessment. J. Environ. Manage. 91, 1–21.
Garfi, M., Tondelli, S., Bonoli, A., 2009. Multi-criteria decision analysis for waste management in Saharawi refugee camps. Waste Manage. 29, 2729–2739.
Gentil, E.C., Damgaard, A., Hauschild, M., Finnveden, G., Eriksson, O., Thorneloe, S., Kaplan, P.O., Barlaz, M., Muller, O., Matsui, Y., Ryota, I., Christensen, T.H., 2010. Models for waste life cycle assessment: review of technical assumptions. Waste Manage. 30, 2636–2648.
Gillies, D., 1959. Solutions to general nonzero sum games. Ann. Math. Stud. 40, 47–85.
Gouldson, A., Morton, A., Pollard, S.J.T., 2009. Better environmental regulation—contributions from risk-based decision-making. Sci. Total Environ. 407, 5283–5288.
Guardiola, L.A., Meca, A., Timmer, J., 2007. Cooperation and profit allocation in distribution chains. Decis. Supp. Syst. 44, 17–27.
Hammond, R., 1966. Convention and limitation in benefit–cost analysis. Nat. Resources J. 6, 195–222.
Hanke, S.H., Walker, R.A., 1974. Benefit–cost analysis reconsidered: an evaluation of the mid-state project. Water Res. Res. 10, 898–908.
Hanley, N., Spash, C.L., 1993. Cost–Benefit Analysis and the Environment. A Sort History of Cost Benefit Analysis, Edward Elgar Publishing, pp. 4–5 (Chapter 1.2).
Harsanyi, J.C., 1959. A bargaining model for the cooperative *n*-person game. In: Tucker, A.W., Luce, D.R. (Eds.), Contributions to the Theory of Games. Princeton University Press, Princeton, NJ, pp. 324–356.
Harsanyi, J.C., 1963. A simplified bargaining model for the *n*-person game. Int. Econ. Rev. 4, 194–220.
Heijungs, R., Guinee, J.B., 2007. Allocation and 'what-if' scenarios in life cycle assessment of waste management systems. Waste Manage. 27, 997–1005.
Hermann, B.G., Kroeze, C., Jawjit, W., 2007. Assessing environmental performance by combining life cycle assessment, multi-criteria analysis and environmental performance indicators. J. Clean. Prod. 15, 1787–1796.
Hideshima, E., Koike, N., Yamamoto, K., 1999. Coalition Formation among Regions to the Reuse of Waste Soil for Construction. Systems, Man, and Cybernetics, 1999. In: 1999 IEEE International Conference on IEEE SMC '99 Conference Proceedings, pp. 961–966.
Hsu, P.F., Wu, C.R., Li, Y.T., 2008. Selection of infectious medical waste disposal firms by using the analytic hierarchy process and sensitivity analysis. Waste Manage. 28, 1386–1394.
Hung, M.L., Ma, H.W., Yang, W.F., 2007. A novel sustainable decision making model for municipal solid waste management. Waste Manage. 27, 209–219.
Huang, I.B., Keisler, J., Linkov, I., 2011. Multi-criteria decision analysis in environmental sciences: ten years of applications and trends. Sci. Total Environ. 409, 3578–3594.
ISO, 2006. Life Cycle Assessment, LCA, Discusses Pre-Production Planning and Environment Goal Setting, Series (14040–14049).
Jain, K., Mahdian, M., 2007. Cost sharing. In: Nisan, N., Roughgarden, T., Tardos, E., Vazirani, V.V. (Eds.), Algorithmic Game Theory. Cambridge University Press, pp. 385–409.
Johansson-Stenman, O., Konow, J., 2010. Fair air: distributive justice and environmental economics. Environ. Resource Econ. 46, 147–166.
Jørgensen, S., 2010. A dynamic game of waste management. J. Econ. Dyn. Cont. l34, 258–265.
Karagiannidis, A., Papageorgiou, A., Perkoulidis, G., Sanida, G., Samaras, P., 2010. A multi-criteria assessment of scenarios on thermal processing of infectious hospital wastes: a case study for Central Macedonia. Waste Manage. 30, 251–262.
Karmperis, A.C., Sotirchos, A., Aravossis, K., Tatsiopoulos, I.P., 2012a. Waste management project's alternatives: a risk-based multi-criteria assessment (RBMCA) approach. Waste Manage. 32, 194–212.
Karmperis, A.C., Sotirchos, A., Tatsiopoulos, I.P., Aravossis, K., 2012b. Environmental project evaluation: IRR-based decision support with a Monte Carlo simulation algorithm. Civ. Eng. Environ. Syst. 29, 291–299.
Karmperis, A.C., Aravossis, K., Tatsiopoulos, I.P., Sotirchos, A., 2012c. On the fair division of multiple stochastic pies to multiple agents within the nash bargaining solution. PLoS ONE 7 (9), e44535. http://dx.doi.org/10.1371/journal.pone.0044535.
Karmperis, A.C., Aravossis, K., Sotirchos, A., Tatsiopoulos, I., 2012d. Coordination of multi-agents with a revenue-cost-sharing mechanism: a cooperative game theory approach. Int. J. Math. Mod. Meth. Appl. Sci. 6, 209–221.
Khan, S., Faisal, M.N., 2008. An analytic network process model for municipal solid waste disposal options. Waste Manage. 28, 1500–1508.
Kirkeby, J.T., Birgisdottir, H., Hansen, T.L., Christensen, T.H., Bhander, G.S., Hauschild, M., 2006. Environmental assessment of solid waste systems and technologies: EASEWASTE. Waste Manage Res. 24, 3–15.
Kołodziej, J., 2007. Hierarchical evolutionary search method of solving non-cooperative game models of selected decision-making problems for water resource management. In: 21st European Conference on Modelling and Simulation (ECMS 2007), 3–7 June, Praga.
Kou, G., Miettinen, K., Shi, Y., 2011. Multiple criteria decision making: challenges and advancements. J. Mutli-Crit. Decis. Anal. 18, 1–4.
Kurisu, K.H., Bortoleto, A.P., 2011. Comparison of waste prevention behaviors among three Japanese megacity regions in the context of local measures and socio-demographics. Waste Manage. 31, 1441–1449.
Kurisu, K.H., Bortoleto, A.P., 2012. Corrigendum to "Comparison of waste prevention behaviors among three Japanese megacity regions in the context of local measures and socio-demographics" WM 31(7), 2011, pp. 1441–1449. Waste Manage. 32, 1055–1057.
Lavee, D., 2010. A cost–benefit analysis of a deposit–refund program for beverage containers in Israel. Waste Manage. 30, 338–345.
Lee, J., Kang, S., Kim, C.K., 2009. Software architecture evaluation methods based on cost benefit analysis and quantitative decision making. Empirical Softw. Eng. 14, 453–475.
Li, Y.P., Huang, G.H., Sun, W., 2011. Management of uncertain information for environmental systems using a multistage fuzzy-stochastic programming model with soft constraints. J. Environ. Inform. 18, 28–37.
Lin, C.-h., Wen, L., Tsai, Y.m., 2010. Applying decision-making tools to national ewaste recycling policy: an example of analytic hierarchy process. Waste Manage. 30, 863–869.
Linkov, I., Varghese, A., Jamil, S., Seager, T.P., Kikerm, G., Bridges, T., 2004. Multi-criteria decision analysis: framework for applications in remedial planning for contaminated sites. In: Linkov, I., Ramadan, A. (Eds.), Comparative Risk Assessment and Environmental Decision Making. Kluwer, Amsterdam.
Linkov, I., Satterstrom, F.K., Kiker, G., Batchelor, C., Bridges, T., Ferguson, E., 2006. From comparative risk assessment to multi-criteria decision analysis and adaptive management: recent developments and applications. Environ Int. 32, 1072–1093.
Linkov, I., Loney, D., Cormier, S., Satterstrom, F.K., Bridges, T., 2009. Weight-of-evidence evaluation in environmental assessment: review of qualitative and quantitative approaches. Sci. Total Environ. 407, 5199–5205.
Linkov, I., Seager, T.P., 2011. Coupling multi-criteria decision analysis, life-cycle assessment, and risk assessment for emerging threats. Environ. Sci. Technol. 45, 5068–5074.
Madani, K., 2011. Hydropower licensing and climate change: insights from cooperative game theory. Adv. Water Res. 34, 174–183.
Manfredi, S., Niskanen, A., Christensen, T.H., 2009. Environmental assessment of gas management options at the Old Ämmässuo landfill (Finland) by means of LCA-modeling (EASEWASTE). Waste Manage. 29, 1588–1594.
Maskin, E., 2011. Commentary: Nash equilibrium and mechanism design. Games Econ. Behav. 71, 9–11.
McDougall, F., White, P., Franke, M., Hindle, P., 2001. Integrated Solid Waste Management: A Life Cycle Inventory. Blackwell Science, London.
McGinty, M., Milam, G., Gelves, A., 2012. Coalition stability in public goods provision: testing an optimal allocation rule. Environ. Resource Econ. 52, 327–345.
Moretti, S., 2004. A model for cooperative intermunicipal waste collection: cost evaluation toward fair cost allocation. In: Game Practice and the Environment. Edward Elgar, pp. 125–144.
Morissey, A.J., Browne, J., 2004. Waste management models and their application to sustainable waste management. Waste Manage. 24, 297–308.
Mourits, M.S.M., Oude Lansink, A.G.J.M., 2006. Multi-criteria decision making to evaluate quarantine disease control strategies. In: Oude Lansink, A.G.J.M. (Ed.), New Approaches to the Economics of Plant Health. Springer, pp. 131–144.
Musgrave, R.A., 1969. Cost–benefit analysis and the theory of public finance. J. Econ. Liter. 7, 797–806.
Nagarajan, M., Sosic, G., 2008. Game-theoretic analysis of cooperation among supply chain agents: review and extensions. Eur. J. Opera. Res. 187, 719–745.
Nahman, A., 2011. Pricing landfill externalities: emissions and disamenity costs in Cape Town, South Africa. Waste Manage. 31, 2046–2056.
Nakanishi, J., Gamo, M., Iwasa, Y., Tanaka, Y., 2003. Environmental risk evaluation of chemicals: achievements of the project and seeds for future––development of metrics for evaluating risks. Chemosphere 53, 389–398.
Nash, J., 1950. The bargaining problem. Econometrica 18, 155–162.
Nash, J., 1951. Non-cooperative games. Ann. Math. 54, 286–295.
Ohman, K.V.H., Hettiarachi, J.P.A., Ruwanpura, J., Balakrishnan, J., Achari, G., 2007. Development of a landfill model to prioritise design and operating objectives. J. Environ. Monitor. Assess. 135, 85–97.
Ong, H.C., Mahlia, T.M.I., Masjuki, H.H., Honnery, D., 2012. Life cycle cost and sensitivity analysis of palm biodiesel production. Fuel 98, 131–139.
Qureshi, M.E., Harrison, S.R., Wegener, M.K., 1999. Validation of multicriteria analysis models. Agr. Syst. 62, 105–116.
Ozelkan, E.C., Duckstein, L., 1996. Analyzing water resources alternatives and handling criteria by multi criterion decision techniques. J. Environ. Manage. 48, 69–96.
Parthan, S.R., Milke, M.W., Wilson, D.C., Cocks, J.H., 2012. Cost estimation for solid waste management in industrialising regions – precedents, problems and prospects. Waste Manage. 32, 584–594.
Pearce, D., 1998. Cost–benefit analysis and environmental policy. Oxford Rev. Econ. Policy 14, 84–100.
Pearce, D., Atkinson, G., Mourato, S., 2006. Cost–Benefit Analysis and the Environment: Recent Developments. OECD Publishing, pp. 253–267.

Pennington, D.W., Potting, J., Finnveden, G., Lindeijer, E., Jolliet, O., Rydberg, T., Rebitzer, G., 2004. Life cycle assessment Part 2: Current impact assessment practise. Environ. Int. 30, 721–739.

Petts, J., 2000. Municipal waste management: inequities and the role of deliberation. Risk Anal. 20, 821–832.

Pickin, J., 2008. Representations of environmental concerns in cost–benefit analyses of solid waste recycling. Res. Conserv. Rec. 53, 79–85.

Pires, A., Martinho, G., Chang, Ni-Bin, 2011. Solid waste management in European countries: a review of systems analysis techniques. J. Environ. Manage. 92, 1033–1050.

Pollard, S.J.T., Strutt, J.E., Macgillivray, B.H., Hamilton, P.D., Hrudey, S.E., 2004. Risk analysis and management in the water utility sector: a review of drivers, tools and techniques. Proc. Safety Environ. Protect. 82, 453–462.

Pollard, S.J.T., Davies, G.J., Coley, F., Lemon, M., 2008. Better environmental decision making—recent progress and future trends. Sci. Total Environ. 400, 20–31.

Prest, A.R., Turvey, R., 1965. Cost–benefit analysis: a survey. Econ. J. 75 (300), 683–735.

Promentilla, M.A.B., Furuichi, T., Ishii, K., Tanikawa, N., 2006. Evaluation of remedial countermeasures using the analytic network process. Waste Manage. 26, 1410–1421.

Rebitzer, G., Ekvall, T., Frischknecht, R., Hunkeler, D., Norris, G., Rydberg, T., Schmidt, W.-P., Suh, S., Weidema, B.P., Pennington, D.W., 2004. Life cycle assessment Part 1: Framework, goal and scope definition, inventory analysis, and applications. Environ. Int. 30, 701–720.

Remer, D.S., Nieto, A.P., 1995. A compendium and comparison of 25 project evaluation techniques. Part 2: Ratio, payback, and accounting methods. Int. J. Prod. Econ. 42, 101–129.

Renna, P., Argoneto, P., 2011. Capacity sharing in a network of independent factories: A cooperative game theory approach. Robot. Comp.-Integr. Manuf. 27, 405–417.

Rentizelas, A., Tolis, A., Tatsiopoulos, I., 2009. Biomass district energy trigeneration systems: Emissions reduction and financial impact. Water, Air Soil Poll. Focus. 9, 139–150.

Riber, C., Bhander, G.S., Christensen, T.H., 2008. Environmental assessment of waste incineration in a life-cycle-perspective (EASEWASTE). Waste Manage. Res. 26, 96–103.

Ross, D., 2010. Game Theory. The Stanford Encyclopedia of Philosophy, May 2010. <http://plato.stanford.edu/entries/game-theory/>.

Roth, A.E., 1979. Axiomatic Models of Bargaining. Lecture Notes in Economics and Mathematical Systems. vol. 170. Springer Verlag.

Roy, B., Vanderpooten, D., 1996. The European school of MCDA: emergence, basic features and current works. J. Multi-Crit. Dec. Anal. 5, 22–37.

Royston, M.G., 1979. Pollution Prevention Pays. Pergamon Press, Oxford – UK, p. 34.

Saaty, T.L., 1978. Exploring the interface between hierarchies, multiple objectives and fuzzy sets. Fuzzy Sets Syst. 1, 57–68.

Schmeidler, D., 1969. The nucleolus of a characteristic function game. SIAM J. Appl. Math. 17, 1163–1170.

Sener, S., Sener, E., Nas, B., Karagüzel, R., 2010. Combining AHP with GIS for landfill site selection: a case study in the Lake Beysehir catchment area (Konya, Turkey). Waste Manage. 30, 2037–2046.

Shapley, L.S., 1953. A value for *n*-person games. In: Kuhn, H.W., Tucker, A.W. (Eds.), Contributions to the Theory of Games, vol. II. Annals of Mathematical Studies, vol. 28. Princeton University Press, pp. 307–317.

Shim, J.P., Warkentin, M., Courtney, J.F., Power, D.J., Sharda, R., Carlsson, C., 2002. Past, present, and future of decision support technology. Decis. Support Syst. 33, 111–126.

Sullivan, T., 2002. Evaluation of Environmental Decision Support Tools, vol. 11973. Environmental & Waste Technology Division, Brookhaven National Laboratory, Upton, NY, pp. 3–7.

Sun, W., Huang, G.H., Lv, Y., Li, G., 2012. Waste management under multiple complexities: Inexact piecewise-linearization-based fuzzy flexible programming. Waste Manage. 32, 1244–1257.

Thorneloe, S.A., Weitz, K., Jambeck, J., 2007. Application of the US decision support tool for materials and waste management. Waste Manage. 27, 1006–1020.

Tol, R.S.J., 2003. Is the uncertainty about climate change too large? Climatic Change 56, 265–289.

Tudela, A., Akiki, N., Cisternas, R., 2006. Comparing the output of cost benefit and multi-criteria analysis an application to urban transport investments. Transport. Res. Pol. Pract. 40, 414–423.

Tukker, A., 2000. Life cycle assessment as a tool in environmental impact assessment. Environ. Impact Asses. Rev. 20, 435–456.

Turner, R.K., 2007. Limits to CBA in UK and European environmental policy: retrospects and future prospects. Environ. Res. Econ. 37, 253–269.

UK Environment Agency, 2012. Waste and Resources Assessment Tool for the Environment (WRATE). http://www.environment-agency.gov.uk/research/commercial/102922.aspx>.

US EPA, 2006. Life cycle assessment: Principles and practices. Environmental Protection Agency. <http://www.cs.ucsb.edu/~chong/290N-W10/EPAonLCA2006.pdf>.

van den Brink, R., van der Laan, G., Moes, N., 2012. Fair agreements for sharing international rivers with multiple springs and externalities. J. Environ. Econ. Manage. 63, 388–403.

Villanueva, A., Wenzel, H., 2007. Paper waste – recycling, incineration or landfilling? A review of existing life cycle assessments. Waste Manage. 27, S29–S46.

von Neumann, J., Morgenstern, O., 1944. The Theory of Games and Economic Behavior. Princeton University Press, Princeton.

Wallenius, J., Dyer, J.S., Fishburn, P.C., Steuer, R.E., Zionts, S., Deb, K., 2008. Multiple criteria decision making, multiattribute utility theory: recent accomplishments and what lies ahead. Manage. Sci. 54, 1336–1349.

Wang, L., Fang, L., Hipel, K.W., 2007. A game-theoretic approach to brownfield redevelopment: negotiation on cost and benefit allocation. In: Proceedings of the 2007 IEEE international conference on systems, man, and cybernetics, Montreal, Canada, pp 1800–1805.

Wang, L., Fang, L., Hipel, K.W., 2011. Negotiation over costs and benefits in Brownfield redevelopment. Group Decis. Negot. 20, 509–524.

Weibull, J.W., 1996. The mass action interpretation. J. Econ. Theory 69, 153–185 (In: H. Kuhn et al. The Work of John Nash in Game Theory).

Weikard, H.-P., Dellink, R., 2010. Sticks and carrots for the design of international climate agreements with renegotiations. Ann. Oper. Res. 2010, 1–20.

Weng, Y.-C., Fujiwara, T., 2011. Examining the effectiveness of municipal solid waste management systems: an integrated cost–benefit analysis perspective with a financial cost modelling in Taiwan. Waste Manage. 31, 1393–1406.

Wilson, P., 1994. Introducing CSCW—what it is and why we need it. In: Scrivener, S.A.R. (Ed.), Computer-Supported Cooperative Work. Ashgate Publishing, Brookfield, VT.

Winkler, J., Bilitewski, B., 2007. Comparative evaluation of life cycle assessment models for solid waste management. Waste Manage. 27, 1021–1031.

Xu, Y., Huang, G.H., Qin, X.S., Huang, Y., 2009. SRFILP: a stochastic robust fuzzy interval linear programming model for municipal solid waste management under uncertainty. J. Environ. Inform. 14, 74–82.

Yau, Y., 2010. Domestic waste recycling, collective action and economic incentive: the case in Hong Kong. Waste Manage. 30, 2440–2447.