Total Cost Assessment (TCA)

Romein, Arie; Louw, Erik; Stead, Dominic

Publication date
2016

Document Version
Final published version

Published in
Urban Strategies for Waste Management in Tourist Cities

Citation (APA)
Romein, A., Louw, E., & Stead, D. (2016). Total Cost Assessment (TCA). In R. Ramusch, I. Gruber, & G. Obersteiner (Eds.), *Urban Strategies for Waste Management in Tourist Cities: D2.2 – Methodology framework document as guidance for accompanying assessment* (pp. 86-88). Urban-Waste.

Ref. Ares(2016)6730521 - 01/12/2016

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

URBANWASTE – 690452 – D2.2

URBANWASTE

Urban strategies for Waste Management in Tourist Cities

D2.2 - Methodology framework document as guidance for accompanying assessment

Grant Agreement No:	WASTE-6b-2015 690452	**Project Acronym:**	URBANWASTE
Project Title:	Urban Strategies for Waste Management in Tourist Cities		
Funding scheme :	Horizon 2020: Eco-Innovative Strategies: "Waste: a resource to recycle, reuse and recover raw materials"		
Project Coordinator:	Gobierno de Canarias (Government of Canary Islands)		
Start date of the project :	01/06/2016	**Duration of the project:**	36 months
Contractual delivery date:	30/11/2016		
Actual delivery date:	30/11/2016		
Contributing WP:	2		
Dissemination level:	Public		
Authors:	Roland Ramusch Iris Gruber Gudrun Obersteiner		
Contributors:	See next page		

Abstract

This report gives a comprehensive overview of commonly used methodologies for a sustainability assessment which were reviewed and evaluated based on certain criteria in order to identify a suitable methodology for the subsequent accompanying sustainability assessments of waste prevention and management activities within the URBANWASTE project. The results of this deliverable will be fed into task 2.3 and task 2.4 in order to define which input data will be necessary being collected for the actual sustainability assessment. The main methods identified, were the combination of MFA and LCA for the subset environmental assessment, eco-efficiency (EE) under partly consideration of CBA and LCC for

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

economic assessment, and a set of individually developed indicators reflecting social assessment (under consideration of SLCA).

Contributors

NAME	COMPANY	CONTRIBUTIONS INCLUDE
Roland Ramusch Iris Gruber Gudrun Obersteiner	Universität für Bodenkultur Wien (BOKU)	Chapter 1, Chapter 2, Chapter 3.4, 3.5, 3.11, 3.13, 3.16, 3.17, 3.18, Chapter 4
Arie Romein Erik Louw Dominic Stead	Technische Universiteit Delft (Delft University of Technology)	Chapter 3.15, 3.24
Mattias Eriksson	Sveriges Lantbruksuniversitet – Swedish University of Agricultural Sciences	Chapter 3.6, 3.21, 3.22
Christian Fertner Juliane Große	University of Copenhagen	Chapter 3.10, 3.12, 3.20
Trine Bjørn Olsen	Aarhus University - AU Herning	Chapter 3.1, 3.9
Claudia de Luca	CE - CONSULTA EUROPA PROJECTS AND INNOVATION SL	Chapter 3.8, 3.26
Pilar Zapata Aranda	BIOAZUL	Chapter 3.2, 3.3, 3.7, 3.19, 3.25
William Hogland Juris Burlakovs	Linnaeus University	Chapter 3.14, 3.23

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

List of abbreviations

CH4	Methane
CO2	Carbon dioxide
HFCs	Hydrofluorocarbons
N2O	Nitrous oxide
PFCs	Perfluorocarbons
SF6	Sulphur hexafluoride
Information regarding abbreviations of the names of methodologies that have been reviewed can be found in Table 2.	

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

Table of Contents

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

List of Figures

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

List of Tables

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Summary

Within WP 2 of the URBANWASTE project waste-related, tourism-related and socio-economic data are collected in the 11 pilot cases. Based on the collected data, the status-quo situation is assessed with the methodologies selected in this Deliverable (D2.2). In addition, future scenarios for the pilot cases will be developed (WP 4) and, partly, selected innovative strategies for waste prevention and management will be implemented (e.g. at hotel level). The impacts of future scenarios for the pilot cases will be assessed (environmental, economic and social assessment) as well at a later stage of the project within WP 7, again applying the methodologies selected in this deliverable. This second assessment aims at providing strategies and implementation activities that are environmentally sound, economically feasible and socially acceptable.

In D 2.2, a set of methods was identified that is suitable to answer URBANWASTE specific questions. This was done by reviewing 26 methods with the goal to identify methods that are allowing a comprehensive sustainability assessment and that fit to the objectives of URBANWASTE. 6 methods were identified that fit to URBANWASTE and 8 methods that fit partly. As there is no methodology covering all assessments that are necessary in order to meet the project's objectives a modular assessment approach was chosen for URBANWASTE applying several suitable methodologies in combination. Which methodology or methodologies will be used for assessing the environmental, economic and social impacts of the status-quo situation in the pilot cases as well as the changes after implementing selected waste prevention and management measures or strategies in the pilot cases is presented in the following table:

Assessment part	Selected method	Additional considered method
Structuring data and visualization of waste and material flows	Material Flow Analysis (MFA)	----
Environmental assessment	Life Cycle Assessment (LCA)	----
Economic assessment	Ecological Efficiency (EE)	Cost Benefit Analysis (CBA) and Life Cycle costing (LCC)
Social assessment	Individual indicators	Social Life Cycle Assessment (SLCA)
Structuring / ranking of results of sustainability assessment	Analytical Hierarchy Process (AHP)	Driving forces – Pressures – States – Impacts – Responses Framework (DPSIR)
Scenario building	Urban and Industrial Symbiosis (UIS) approaches	----

MFA will provide an inventory of material / waste flows and thus will lay the basis for the subsequent environmental assessment. LCA will be applied for environmental impact assessment. For assessing economic impacts, the method of Ecological Efficiency (EE) will be applied together with other cost-related methods such as CBA and LCC. For assessing social impacts only individual parameters will be selected and analysed within URBANWASTE, but under consideration of general aspects of one methodology (SLCA). Chapter 4 of this deliverable (D2.2) also provides information on impact categories and indicators used by the selected assessment methodologies.

How these methods will be applied in practice, meaning for example, which data is necessary to be collected from the pilot cases in Task 2.5 in order to be able to calculate the indicators and impact categories related to the selected methodologies will be defined in Task 2.3 and reported in D2.3.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

1. Introduction

In comparison with other cities, tourist cities have to face additional challenges related to waste prevention and management due to their geographical and climatic conditions, the seasonality of tourism flow and the specificity of tourism industry and of tourists as waste producers. One major objective of the URBANWASTE project is to support policy makers in answering these challenges and in developing strategies that aim at reducing the amount of municipal waste production and at further support the re-use, recycle, collection and disposal of waste in tourist cities.

Within this project, the concept of urban metabolism will be used to understand and analyse how cities that are influenced by tourism use their resources and how touristic activities are linked to waste management and resource conservation. Therefore, URBANWASTE will perform a metabolic analysis of the state of the art of urban metabolism in 11 pilot cases.

Within the project waste-related, tourism-related and socio-economic data are collected in the 11 pilot cases. Firstly, the status-quo situation is assessed with the methodologies selected in this Deliverable. In addition, future scenarios for the pilot cases will be developed and partly selected, innovative strategies for waste prevention and management will be implemented (e.g. at hotel level). The impacts of future scenarios for the pilot cases will be assessed (environmental, economic, social) with the selected methodologies.

Within work package (WP 2) three procedural steps are envisaged to meet the project's objectives: As first procedural step the **development of a proper methodology (Task 2.2) and the adjustment and definition of data requirements** is envisaged. Metabolism indicator sets and a database for the selected pilot tourist cities (Task 2.4) shall be developed. The database focusses on the touristic processes and the link to resource use, waste generation, prevention, recycling, waste treatment and disposal activities. The database will provide the information necessary to analyse how tourism is responsible for positive and negative impacts considering the three pillars of sustainability (environment, society and economy). In a second step, a **baseline assessment** will be carried out (Task 2.6), **applying MFA and LCA to assess the current situation in selected URBANWASTE pilot cases**. The third procedural step within WP 2 to meet the project's objectives will be the **identification of best waste management practices** and **options for optimization** of waste management strategies in the selected pilot cases.

This report refers to URBANWASTE Work Package 2, Task 2.2, Deliverable 2.2: Methodology Framework. The main aim of this Work Package is to provide background data and to assess waste related impacts of tourism using a Life Cycle approach.

To meet the main goal of Task 2.2, the **development of a suitable methodology for the subsequent accompanying sustainability assessments** of waste prevention and management activities within the URBANWASTE project, within this deliverable D2.2 existing methodologies for a sustainability assessment are reviewed in order to identify the best methodologies for the assessment of environmental impacts as well as social and economic aspects suitable within the scope of URBANWASTE. This review will provide knowledge on the underlying concepts and assessed impacts of different commonly used methodologies as well as on their suitability to meet the project's objectives.

Together with the results from Task 2.1, which gives a comprehensive literature review on previous urban metabolism studies in order to provide knowledge on which indicator sets and background data are suitable for linking tourism activities with waste and use of resources, the results from Task 2.2 will subsequently be fed

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

into Task 2.3 in order to operationalize the concept of Urban Metabolism. In Task 2.3 a final list of indicators will be developed based on selected touristic processes. Based on the results of Task 2.3, a database template will be developed within Task 2.4 for the subsequent collection of the input data (Task 2.5) that is necessary to calculate the indicators selected in Task 2.3.

In order to identify methodologies suitable to answer specific URBANWASTE questions, several methodologies were reviewed by the project partners and evaluated based on specific criteria. For those methodologies that are considered suitable for this project, a set of suitable impact categories and indicators (and the underlying data needed) was defined. Practicable impact categories[1] for environmental, economic and social assessment were selected in reference to the ILCD handbook. A more detailed description of the procedure for Task 2.2 is given in Chapter 2.

[1] Impact categories are logical groupings of results related to specific issues of interest. In the context of environmental assessments, for example, impact categories represent environmental issue of concern such as climate change, acidification or ecotoxicity (eplca.jrc.ec.europa.eu/uploads/ILCD-Recommendation-of-methods-for-LCIA-def.pdf). In the context of social assessments impact categories such as human rights, working conditions, health and safety or cultural heritage, for example, can be used (UNEP & SETAC, 2009).

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

2. Approach

In order to identify a methodology that meets the project's objectives, a **methodology review** was carried out by the project partners. URBANWASTE aims to quantitatively describe the current situation (status-quo or baseline) of the pilot cases. This includes, amongst others, an evaluation of waste streams related to touristic activities (WP 2, Task 2.6) to provide a basis for the development of eco-innovative, inclusive and gender sensitive waste prevention and management strategies (WP 4, Task 4.1).

Procedure for review of methodologies

First of all, a list of commonly used assessment methodologies was compiled by the project partners. Those methodologies have been reviewed according to their suitability to URBANWASTE needs. In order to meet the URBANWASTE project's objectives, the chosen method has to allow a quantitative assessment of the current situation regarding the touristic impact on waste generation, waste types and waste management as well as to allow an assessment of environmental impacts and social and economic aspects related to touristic activities. In order to select a suitable methodology, a set of five criteria was developed. These criteria (described in Table 1) shall help to identify suitable methodologies.

Table 1: Criteria used in the methodology review

CRITERION	NAME OF CRITERION	SHORT DESCRIPTION
I	... based on a life cycle perspective	Methodology considers upstream and downstream processes.
II	... considers or at least allows the consideration of quantitative material flows	Methodology considers or at least allows the consideration of quantitative material flows.
III	Suitability for social, economic and environmental assessment	Method allows assessing the three main issues of sustainability (meaning social, economic and environmental impacts).
IV	Suitability for URBANWASTE	According to the project partners' opinion the reviewed methodology is suitable to answer specific URBANWASTE questions.
V	Suitability for assessment of changes on hotel level or on municipality level	Methodology allows assessment of changes based on implementing waste prevention and management measures on either the hotel level or on municipality level.

In total, 26 methods were selected for the methodology review and distributed among project partners for the actual reviewing process (Table 2). Based on the criteria presented in Table 1, each of those methods was described and evaluated by the project partners according to the following structure:

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

- "Underlying Concepts": In this section the underlying concepts of the reviewed method was described. Special attention was paid on aspects related to criteria I and II, meaning if the reviewed methodology considers upstream and downstream processes and considers or at least allows the consideration of quantitative material flows.
- "Assessed impacts": In this section the reviewed method was described in relation to the assessment of sustainability aspects (meaning social, economic and environmental impacts) it allows.
- "Suitability for URBANWASTE": This section contains an evaluation if the reviewed method can be considered as being suitable to answer specific URBANWASTE questions (criterion IV).
- "Suitability for assessment of changes on hotel level or on municipality level": In this section the reviewed method was evaluated in relation criterion V, describing if it allows the assessment of changes due to the implementation of waste prevention and management measures on either the hotel level or on municipality level.

An overview of the results of evaluating the selected methods against the abovementioned five criteria is presented in Chapter 0. This chapter further contains a description of which methodologies are considered as being suitable to meet the URBANWASTE project's objectives including the justification of that selection.

In a second step of the methodology review **suitable impact categories and indicators (and the underlying data needed) were selected and defined for those methodologies that have been identified as being suitable for this project**. The choice of suitable indicators and impact categories was again tailored to meet central URBANWASTE issues, meaning the subsequent impact assessments of the current situation ("baseline") in the pilot cases (within WP 2, Task 2.6) and the development of strategies (within WP 4, Task 4.1). The set of practicable impact categories for environmental, economic and social assessment was selected in reference to the ILCD handbook. The choice of impact categories and selected indicators is presented in Chapter 4.4.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

3. Review of Methodologies

In total 26 commonly used assessment methodologies were reviewed by the project partners in order to identify a set of methodologies suitable to be applied in the URBANWASTE project. Applying the criteria described in Chapter 2 to judge on the suitability of a reviewed method a set of suitable methods will be identified. The allocation of the methodologies included in the review is shown in Table 2.

Table 2: Allocation of methodology review amongst partners

	Name of Methodology	Responsible partner
1	Activity-Based Costing (ABC)	AU
2	Analytical Hierarchy process (AHP)	Bioazul
3	Balanced Scorecard Approach (BSC)	Bioazul
4	Carbon Footprint (CF, Corporate Carbon Footprint, Product Carbon Footprint)	BOKU
5	Comparative Risk Assessment (CRA)	BOKU
6	Corporate Social Responsibility (CSR)	SLU
7	Cost-Benefit Analysis (CBA)	Bioazul
8	Drives-Pressures-State-Impact-Response (DPSIR)	CE
9	Eco-Efficiency (EE)	AU
10	Ecological Footprint (EF)	UCPH
11	Economic Input-Output (EIO)	BOKU
12	Energy Flow Analysis (EFA) and Material Flow Analysis (MFA)	UCPH
13	Environmental Impact Assessment (EIA)	BOKU
14	Environmental Profit and Loss (EP&L)	LU
15	Industrial Symbiosis (IS)	TUD
16	Life Cycle Assessment (LCA)	BOKU
17	Life Cycle Costing (LCC)	BOKU
18	Life Cycle Working Environment (LCWE)	BOKU
19	Multi-Criteria Decision Making (MCDM)	Bioazul
20	Multi-Scale Integrated Analysis of Societal and Ecosystem Metabolism (MuSIASEM)	UCPH
21	Social Life Cycle Assessment (s-LCA)	SLU
22	Strategic Environmental Assessment (SEA)	SLU
23	Sustainability Assessment (SA)	LU
24	Total Cost Assessment (TCA)	TUD
25	Urban and Industrial Symbiosis (UIS)	Bioazul
26	Water Footprint (WF)	CE

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

3.1 Activity-Based Costing (ABC)

Activity-Based Costing (ABC) was first introduced in the late 1980ies to provide more-accurate ways of assigning costs of indirect and support activities, and business processes to products, services and customers in industrial organizations. Unlike traditional costing, the method recognizes that a considerable part of an organisation's resources is required for indirect costs to provide activities that support the actual production. The ABC method aims to improve the accuracy of **assigning indirect costs to the production**, by tracing the use of resources in all the activities performed, and then linking the cost of these activities to the cost objects (e.g. products, services and/or customers). In this way the ABC method can serve not only as a tool for costing and budgeting, but also for more accurate profitability analysis of products/services and for supporting strategic managerial decisions within the organisation (Kaplan & Atkinson, 1998).

Unlike costs for raw materials and manufacturing labour, the costs for support activities cannot be assigned directly to the production; they are indirect costs. In traditional costing an overall average of indirect costs (overhead costs) is assigned uniformly to all products/services. However, this may give a false impression of cause-and-effect which may lead to under- or overestimating costs and cross-subsidisation. This could have an adverse effect on the company profitability and competitiveness, and consequently the company's survival on the market. To avoid this, the ABC method aims to provide a more accurate way of measuring, differentiating and assigning indirect costs to the support activities (Horngren et al., 2002).

In practice, the assignment of costs through the ABC method occurs in two stages (Figure 1):

1. Resource costs are assigned to various identified activities by creating resource drivers (e.g. administration working hours, driven kilometres, square meters, orders, energy consumption etc.). Each type of resource traced to an activity becomes a cost element of an activity cost pool (e.g. administration, logistics, maintenance, sales, purchase etc.). An activity cost pool therefore represents the total costs identified with an activity or activity centre, which is usually clustered by function or process.
2. The costs in each activity cost pool are allocated to cost objects by an activity cost-driver (e.g. *number* of orders, *number* of square meters, *number* of driven km etc.) which is used to measure the consumption of activities by the cost objects (Bukh & Israelsen, 2004: Tsai et al., 2010).

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

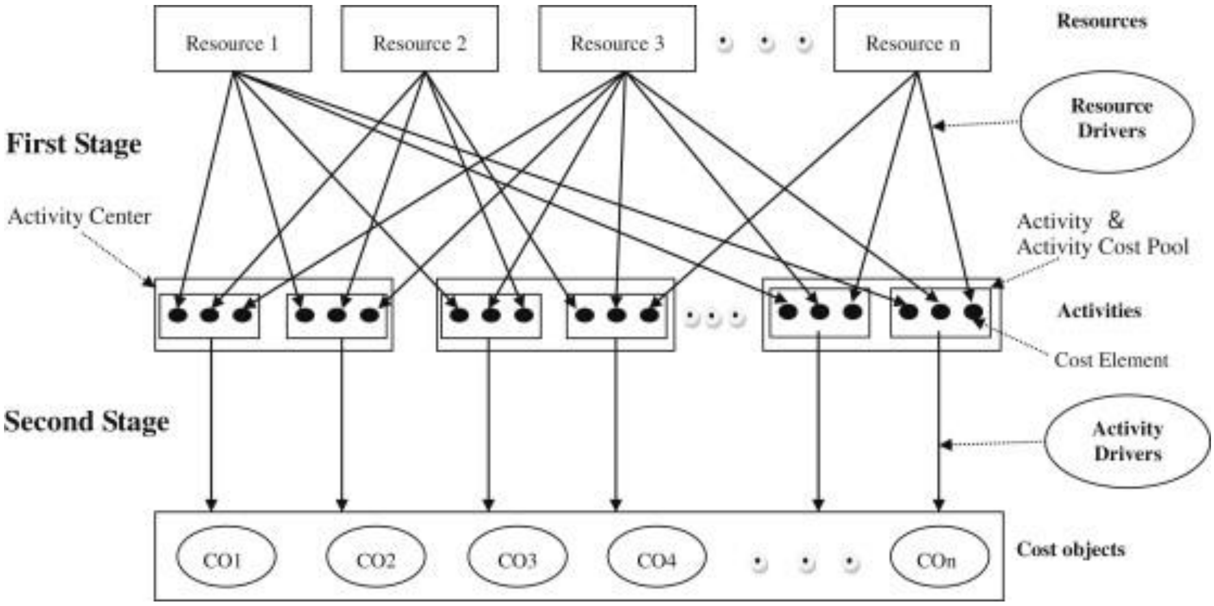


Figure 1: Detailed cost assignment view of ABC (Source: Tsai, 2010).

Even though it may pay off in the longer perspective, ABC is a complicated and time consuming method, that requires support both from the management and the respective departments involved, to operate successfully. Bukh & Israelsen (2004) suggest, that **companies should only apply ABC if indirect costs account for a large percentage of the total costs**, and particularly if indirect costs are increasing (Bukh & Israelsen, 2004). The current trend of rising indirect costs, there seems to be an increasing need for application of ABC in modern enterprises.

Since the 1990ies ABC has been one of the most discussed and debated costing methods. Numerous surveys have been conducted and mathematical models for implementation of ABC at company level have been developed, particularly in the health sector, but **only few studies been made in the tourism industry** (Stefano & Freitas, 2013). Several studies point to the importance of the involvement of company employees in the implementation of ABC and the possible environmental and work-related social improvements that a broad-based ownership of the system can provide. However, only few models have been developed that combines ABC and environmental and/or social assessment methodologies. No examples were found on ABC in relation to assessment of social impacts on citizens.

Underlying Concepts

Surveys and interviews with company managers who use ABC indicate, that the method is used to support a wide range of economic activities, including environmental management, and other activities in the environmental field (Tsai et al. 2012).

In contrast to **conventional accounting**, which has been **criticized for not including environmental impacts**, Tsai et al. (2012) proposed an ABC approach, at company level in the Taiwanese paper industry which is able to track pollutants created by each product and calculate their cost. By tracing costs through activities the ABC approach not only produces more accurate estimations of the **environmental costs** of the cost objects, but it

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

also represents a specific environmental cost structure which can be used strategically in policy making, pricing and process improvement decisions.

Environmental policies along with regulations and legislative requirements such as monitoring of emissions, waste and environmental costs have become increasingly important to both governments and companies. **Linking cost accounting and environmental management procedures** through an integrated management system, can help companies meet both internal and external policy objectives. There is, however, **no standardized system** which means that the **ABC method has to be individually designed** to fit the context and strategic objectives of each organisation (Tsai et al., 2012).

ABC and Life-Cycle Assessment (LCA)

The relevant literature typically does not use profitability analysis as an input into production decisions even though such integrated models can offer a competitive edge in terms of reducing environmental impact during the product's life cycle, and is able to overcome one of the obstacles to a more sustainable society (Tsai et al., 2015).

Tsai et al. (2015) developed the mathematical programming model LINGO for decision-making in an electrical and electronic equipment (EEE) industry in Taiwan based on a calculation model that combines ABC and LCA in order to maximise the company's profits and minimise environmental impact. The study gives an insight into environmental management in a highly competitive industry (similar to the tourism industry).

Figure 2 outlines a flowchart of different life-cycle activities in a production process. Each activity requires different resources, and includes a range of associated resource costs. By recognising each activity's pattern of resource use and selecting the most appropriate measure of resources consumed (resource drivers) the company can begin to allocate the proper resource costs to each activity.

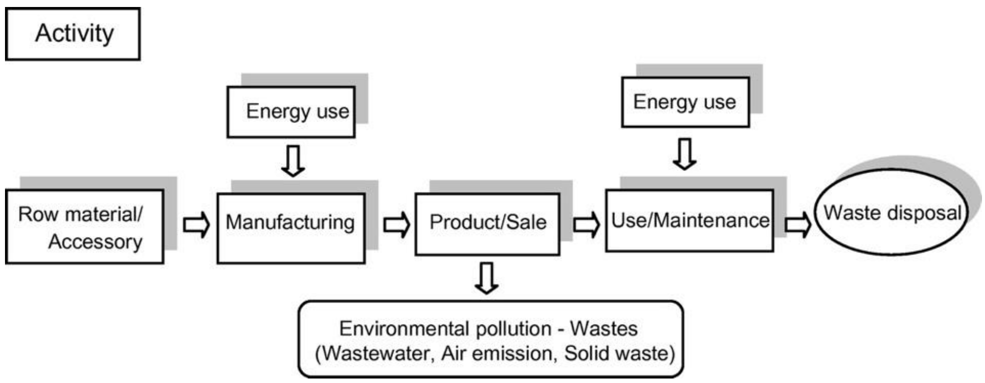


Figure 2. Lifecycle flowchart of product activities (Source: Tsai, 2015).

Figure 3 illustrates the following step where the life-cycle costs are incorporated and assigned to the aspects of the ABC system; activity centres, activity contents, activity drivers and cost objects. Each activity centre is composed of related activities. The activities for the disposal of general solid waste vary in accordance with the amount of general solid waste produced by each product. Activities for monitoring environmental impacts vary with the amount of internal auditing required by each of these products, and so forth. From this allocation a mathematical program model for profit maximisation was developed with the LCA activities included in the ABC system.

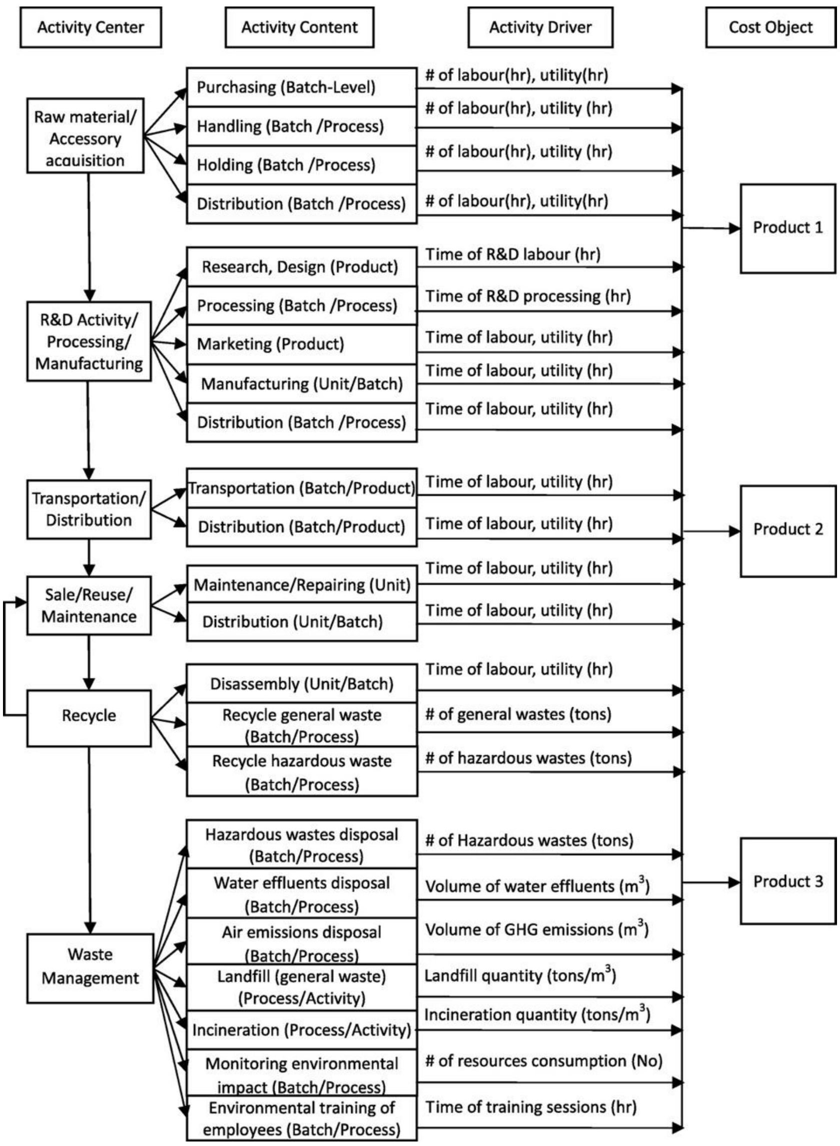


Figure 3: ABC system for products life cycle environmental assessment activity (Source: Tsai, 2015).

ABC and Material Flow Analysis (MFA)

MFA is an approach that helps to track physical flows of resources through systems of production from inputs, processing and various kinds of outputs of a process. MFA is a scientific approach that can help the identification and selection of environmentally friendly input material and energy components in production processes for reducing pollution and global warming. The MFA approach tracks materials, energy and pollution in physical units, whereas business organizations are mostly driven by monetary performance measures.

According to Deo (2015), choosing which costing model should be used along with the MFA approach to identify and select mixes of material and energy components in a production process is a problem, that needs

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

to be addressed. So far, there has not been made a synthesis of MFA with a specific costing model to measure the environmental performance of inputs along with costs of operations. However, environmentally sustainable materials and energy supply need to be selected in such a way that they are less costly, too. There is a need for the development of an operational costing approach, that can be easily used with MFA to identify and select materials and energy forms that are more sustainable, both in terms of environment and economy (Deo, 2005).

Assessed Impacts

Economic Assessment

ABC was developed specifically for economic assessment. In its starting point the **method aims to allocate indirect costs** in order to give a more accurate view of how support activities require resources in an organization.

Environmental Assessment

Businesses often resist doing what is good for ecosystems because it is unprofitable or because they lack the knowledge about more sustainable solutions. The prevention of environmental damage needs the reconciliation of the stages in the whole supply chain. As suggested by Tsai et al. (2009) a combination of LCA and ABC may be used as a basis for decision making in situations where an organisation wants to include economic assessment for comparison of alternative environmental scenarios.

In many cases pure profit maximisation makes recycling limited to the high-value components or the portions that can be easily recycled, which is not necessarily ideal for ecosystems. However, along with the prevalence of environmental awareness and the implementation of more and more new environmental laws, modern enterprises are faced with pressures and obligations and need new perspectives and decision models to find their solutions. For example, with the implementation of the WEEE-directive in the EU it is now required that the producers take the responsibility of collection, treatment, recovery and disposal of WEEE, whether the environmental activity is profitable or not.

A separation of non-profit and for-profit processes can make decision objectives more flexible and applicable in return logistics management particularly when considering the issues of extended producer responsibility. In the non-profit model the environmental criterion can be given a higher weight. The non-profit model is necessary because the environmental problems of waste treatment not only involve individual finance and interest but also public safety and sustainable development (and possibly legal requirements). In addition, integrating ABC can help decision-makers to obtain more precise information about value-added and non-value-added costs by the identification of cost drivers (Tsai & Hung, 2009).

Pollutant-based taxes enable firms to estimate the marginal damages and marginal costs of different taxation levels in calculating the optimal range of product prices. To better manage waste and facilitate implementation of a pollution tax, the effective calculation of environment-related costs becomes increasingly important. The ABC method can help managers to make better decisions by enabling them to clearly identify the costs, by product, of environmental compliance and responsibility. (Tsai et al., 2012)

Social Assessment

ABC does not consider social impacts separately from other overhead costs; thus they are hidden among other production and service processes. In a study in Australian non-service manufacturing companies Percherat & Mula (2012) designed a conceptual model based on the sustainability management accounting system (SMAS)

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

in combination with ABC that enables an allocation of i.a. social impacts and providing an opportunity to involve the economic aspect in the choice of alternative social scenarios (Petcharat & Mula, 2012). It must be emphasized, though, that this case refers to **social impact costs expenditure relating to the support of employees' health and safety, training, working conditions and not social impacts on citizens**.

Suitability for URBANWASTE

URBANWASTE may adapt ABC especially in interconnection with LCA to provide eco-innovative waste management strategies in the pilot areas. The ABC method was developed as a tool for economic costing and strategic management decisions at organizational level where also co-ownership and the involvement of stakeholders is essential. The method can be a valuable supplement for a decision making basis for the assessment of alternative environment-improving scenarios in the pilot areas because it will highlight the economic impacts.

Municipal waste systems are rooted in organizations with a **wide scope of activities** to support the core services and thus they have a **high proportion of indirect costs**. In the interest of the customers (businesses and citizens) it is important to estimate as accurately as possible how the individual services strain on resources in order to set fair waste fees. In this context it will also be appropriate to distinguish between profit-making and non-profit-making processes.

Suitability for assessment of changes on hotel level or on municipality level

The ABC method is geared towards implementation at company level or within a defined organizational framework, i.e. not a geographical area. Studies show, that the method **can be applied to individual hotels** and, as suggested above, it may also be **suitable for implementation in municipal waste management departments**. Furthermore, ABC in combination with LCA can complement the assessment of the environmental impacts of different scenarios with economic considerations. However, **ABC is a time and resource consuming system** and, combined with other assessment methods, a **rather complex task**. Since there is no such thing as a standard ABC system it has to be tailored to fit the individual organization's context, objectives and strategies.

Fathi & Dozahiri (2015) made an empirical investigation of ABC implementation in a hotel in Iran. The survey concluded that organizational, technological, individual and environmental (i.e. working environment, *not* external environment) factors influence the implementation of ABC in hotel industry with individual factors, including allocation and responsibility, being more important than other factors. The gender aspect was also surveyed, but did not show any meaningful impact in the particular study.

By comparing traditional costing with ABC in the Greek hotel business by Vazakidis & Karagiannis (2008), it was indicated, that bad cost information will result in too high room prices and possibly cause a decrease in tourist influx. The study also recognizes, that although rewarding in terms of better basis for managerial decisions and a way to improve performance, the ABC is also an ongoing system that needs constant attention and adjustment.

The URBANWASTE project may well provide guidelines for the methodologies to be applied in the preparation of municipal waste strategies in the pilot areas. For implementing measures at the level of waste generators (hotels, restaurants etc.) it has to be clarified when the pilot implementation cases are chosen, whether ABC approaches can be applied.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Summary / Conclusions

ABC is a **costing method that aims to improve the accuracy of assigning indirect costs to production (products/services) and eliminate under-/overestimation of costs and cross-subsidisation in order to maximize profitability**. This makes the ABC system a vital strategic tool for policy and decision making in an organization. Using an ABC system is resource and time consuming, and the question is, whether the quality of an ABC system is noticeably better compared to how many resources it consumes. Some companies opt out ABC because the improvements are not commensurate with the price. It may be necessary to make some compromises and delimitations because the current framework of the organization does not allow a full implementation of the ABC method. However, the rising share of indirect costs in modern enterprises have increased the need for the application of ABC.

Due to the use of **traditional cost accounting, many organisations do not estimate their environmental and social costs precisely**. In contrast to conventional accounting, an ABC system allows combination with other assessment methods to provide more accurate estimates of environmental and social impacts at company level. Tsai et al. (2009: 2012: 2015) proposed ABC approaches that are able to track pollutants and waste fractions created by each product and calculate their cost. Percharat & Mula (2012) also designed a merged methods model for assigning both environmental and social costs.

ABC provides a more accurate and sophisticated way to allocate costs. By tracing costs through activities, the ABC approach not only produces more accurate estimations of e.g. environmental and social costs but it also represents a specific cost structure which can be used in policy making, pricing and process improvement decisions. However, combinations of ABC with environmental and social assessment methodologies has only been applied in very few industrial organisations and therefore the experiences to draw upon are rather limited. **No combination of ABC with MFA have been tested or developed so far**.

Critics of ABC have said, that it is decisions and not activities that cause costs (Haladu, 2016). If the organization wants to use the ABC system over a longer period of time and it has to remain a valuable financial tool it must be ensured, that the model is rooted in the organization. This can only happen if the information that the system provides is relevant. At the same time, it is vital that the organization has incorporated a procedure for ownership to update the model. It is also vital that the involved departments have an influence and can control the way activities are measured. Otherwise there will be no incentive for the employees to maintain the system.

For implementation of ABC to be successful, a top management commitment will be needed so that all objectives are in accordance with management strategies, quality and performance assessment and awareness of the time that is required for implementation (Stefano & Freitas, 2013). Any ABC system should be designed specifically for the organizational context in which it operates, which can be relatively complex and laborious. Also the system risks becoming too static and difficult to validate e.g. after an organizational change. Therefore, it needs to be updated regularly. Currently there are **no pre-designed software programs** that can perform continuous ABC calculations. The ABC system cannot stand alone, but must be considered as an additional system to support management decisions (Bukh & Isrealsen 2004).

Despite the fact, that ABC generally fits to the URBANWASTE project, it can be concluded, that **it is too complex in the implementation and therefore time and resources consuming**. **It is recommended to include not the methodology itself, but to consider the fact of including indirect costs as good as possible in order to have real costs. Of course the extent of including indirect costs depends on the scenarios and strategies for implementation.**

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

3.2 Analytical Hierarchy Process (AHP)

Developed by Thomas L. Saaty in the early 1970ies, the Analytical Hierarchy Process (AHP) is a structured technique for organizing and analyzing complex decisions, based on mathematics and psychology.

When using the AHP to model a problem, one needs a **hierarchic or a network structure to represent that problem and pairwise comparisons to establish relations within the structure**. Therefore, problems are decomposed into a hierarchy of criteria and alternatives. That way, the information can be arranged in a hierarchical tree with different levels of criteria and sub-criteria. Both quantitative and qualitative criteria can be compared using informed judgments to derive **weights and priorities** (Saaty 1987).

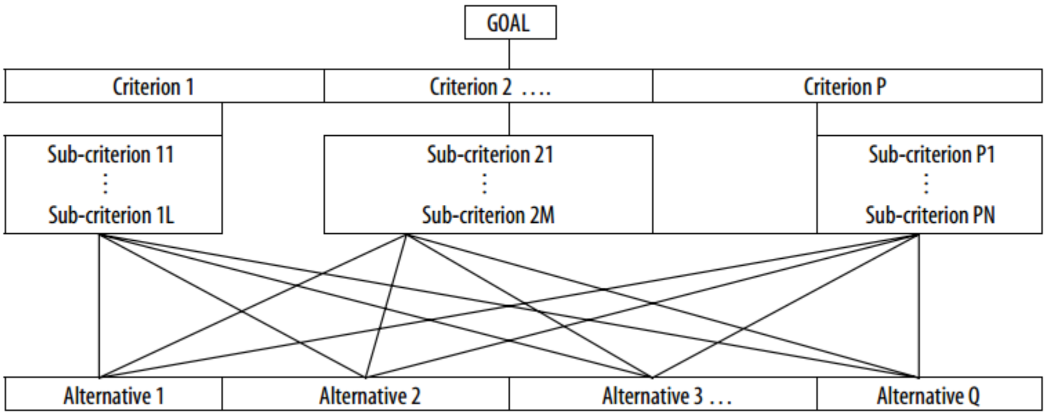


Figure 4: Generic hierarchic structure (Source: Bhushan & Rai, 2004)

In order to make decisions in an organised manner and to generate priorities, the decision needs to be decomposed (see also Figure 4) into the following steps (Saaty 2008):

- Define the problem and determine the kind of knowledge sought.
- Structure the decision hierarchy from the top with the goal of the decision, then the objectives from a broad perspective, through the intermediate levels (criteria on which subsequent elements depend) to the lowest level (which usually is a set of the alternatives).
- Construct a set of pair-wise comparison matrices. Each element in an upper level is used to compare the elements in the level immediately below with respect to it. To make comparisons, a scale of numbers is required and it will indicate how many times more important or dominant one element is over another element with respect to the criterion or property with respect to which they are compared (e.g. 1 = equal importance, 9 = extreme importance).
- Use the priorities obtained from the comparisons to **weigh the priorities** in the level immediately below. This needs to be done for every element. Then for each element in the level below the weighed values must be added and the overall or global priority will be obtained. Continue this process of weighing and adding until the final priorities of the alternatives in the bottom most level are obtained.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

The AHP can be used for a wide variety of applications: multi-criteria decision making, strategic planning, business/public policy decisions, benchmarking, forecasting, resource allocation, source selection, conflict resolution, programme selection, and many more (Saaty, 1987; Bhushan & Rai, 2004).

Underlying Concepts

The AHP can be viewed as a formal method for rational and explicit decision making. It possesses seven fundamental properties or pillars (Schmoldt et al. 2013):

- *Normalised ratio scales* are central to the generation and synthesis of priorities.
- *Reciprocal paired comparisons* are used to express judgments semantically, and to automatically link them to a numerical and fundamental scale of absolute numbers.
- *Sensitivity of the principal right eigenvector* to perturbation in judgements limits the number of elements in each set of comparisons to a few and requires that they be homogeneous.
- *Homogeneity and clustering* are used to extend the fundamental scale gradually from cluster to adjacent cluster.
- *Synthesis that can be extended to dependence and feedback* is applied to the derived ratio scales to create a uni-dimensional ratio scale for representing the overall outcome.
- *Rank preservation and reversal* can be shown to occur without adding or deleting criteria.
- *Group judgements* must be integrated one at a time carefully and mathematically.

AHP and Life Cycle Assessment (LCA)

The **AHP has been considered and implemented in conjunction with the Life Cycle Assessment** (LCA) **methodology in many situations** (Tolle et al. 1998; Eagan & Weinberg, 1999; Seppälä, 2003; Hermann et al. 2007; Qian, et al. 2007). The integration of AHP and LCA provides a framework for robust decision making that is consistent with sustainable practices. For instance, the authors Qian et al. (2007) used a systematic approach for the life cycle design of a chemical product in which the life cycle cost of a product was analysed together with the environmental impact (see Figure 5). The AHP proves to be a very useful tool when several LCA analyses are performed and a consistent interpretation and comparison of the results is needed.

In addition, The U.S. Environmental Protection Agency (EPA) has used the AHP in LCA case studies in order to provide a basis for the product alternatives (Tolle et al. 1998).

There is also a software tool called "BEES" for LCA which includes an option to use AHP for weighting (National Institute of Standards and Technology, 2000).

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

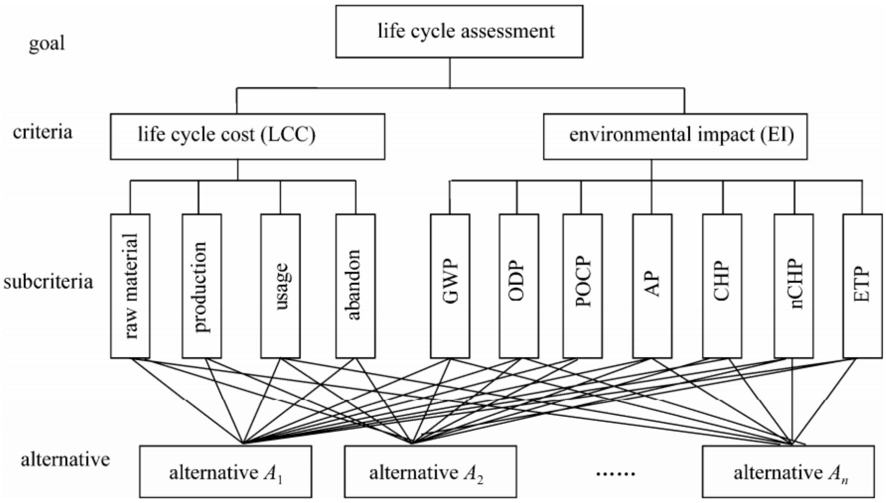


Figure 5: Integrated assessment of environmental and economic performances using the AHP model (Source: Qian et al. 2007)

AHP and Material Flow Analysis (MFA)

The number of references of AHP integrated into Material Flow Analyses is **rather limited**. However, Zhou and Zheng (2010) reported about a comprehensive evaluation indicator system which unified physical and value information from a MFA in an enterprise by using the AHP. It was stated, that compared with other evaluation indicator systems for cleaner production and evaluation indicators in environmental accounting, the use of AHP resulted more comprehensive and provided more information for recycling economy management decision-making.

Assessed Impacts

Social Assessment

The AHP is a **suitable methodology as it allows assessing social impacts**. In general, a hierarchical model of some societal problem might be one that descends from a focus (an overall objective), down to criteria, down further to sub-criteria (which are subdivisions of the criteria) and finally to the alternatives from which the choice is to be made (Saaty, 1987).

Economic Assessment

Economic aspects can also be considered when using the AHP. There are several references confirming the feasibility of this methodology for economic assessments and life cycle costs of a product (Bhushan & Rai, 2004; Qian et al. 2007).

Environmental Assessment

The AHP has been proved to be particularly useful in different environmental issues such as natural resource management and decision making, biodiversity conservation assessment and habitat restoration (Schmoldt et al. 2013), as these aspects involve selecting or prioritising among a finite set of alternative courses of action. Qian et al. (2007) also applied the AHP to adopt a multi-attribute decision-making in a trade-off consideration of technical economical evaluation and environmental impacts assessments.

Suitability for URBANWASTE

The AHP would be a **suitable methodology for the URBANWASTE** project as it is designed to cope with complex decisions including those related to social, economic and environmental aspects. Moreover, the AHP allows for group decision making, being possible to aggregate individual judgements in a group into a single representative judgement for the entire group and also to construct a group choice from individual choices (Saaty, 2007).

Suitability for assessment of changes on hotel level or on municipality level

After analysing the different settings where the AHP has been applied, this technique turns out to be **a feasible methodology for different scales**. It has been used to prioritise strategic enhancements for several governing bodies in the US; to allocate resources within the Department of Defense in the US; by British Airways to choose the entertainment system vendor for its entire fleet of airplanes; by Ford Motor Company to establish priorities for criteria that would improve customer satisfaction; and by the parliament of Finland to decide what type of power plant to build and how the new plant would affect Finland's national economy, the health, safety and environment for Finish citizens (Saaty 1987; Saaty 2008). It has also been applied in regional and urban planning and R&D management (Bhushan & Rai, 2004).

Summary / Conslucions

It can be concluded, that the **AHP is an effective tool to deal with complex decision making processes**, **helping the decision maker to set priorities and make the best decision**. By **reducing complex decisions to a series of pairwise comparisons**, and then synthesizing the results, the AHP helps to capture both subjective and objective aspects of a decision. This methodology has found use in business, government, social studies, R&D, defence and other domains involving decisions in which choice, prioritization or forecasting is needed. Due to its simplicity and ease of use, the AHP has found ready acceptance by busy managers and decision-makers and it has proved a methodology capable of producing results that agree with perceptions and expectations (Bhushan & Rai, 2004).

Its use is **therefore recommended in combination with other methodologies such as LCA and MFA**.

3.3 Balanced Scorecard Approach (BSC)

The Balanced Scorecard Approach was first introduced by Robert S. Kaplan and David Norton in a 1992 Harvard Business Review article (Kaplan & Norton, 1992). That article was based on a multi-company research project to study performance measurement in companies whose intangible assets played a central role in value creation.

The concept was then adopted by thousands of private, public, and non-profit enterprises around the world, and Kaplan and Norton extended the concept into a management tool for describing, communicating and implementing strategies (Kaplan 2010).

Basically the BSC is a **concept to measure, document and control the activities of a company / organisation related to its vision and strategy**.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

The **generic BSC model consists of four interrelated quadrants**, each one containing objectives and measures from a distinct perspective. The selected perspectives are: Financial, Customer, Internal Processes, and Learning and Growth (see Figure 6). The scope of these perspectives is designed to cover the whole of the organisation's activities both internally and externally, both current and for the future (Mackay 2005).

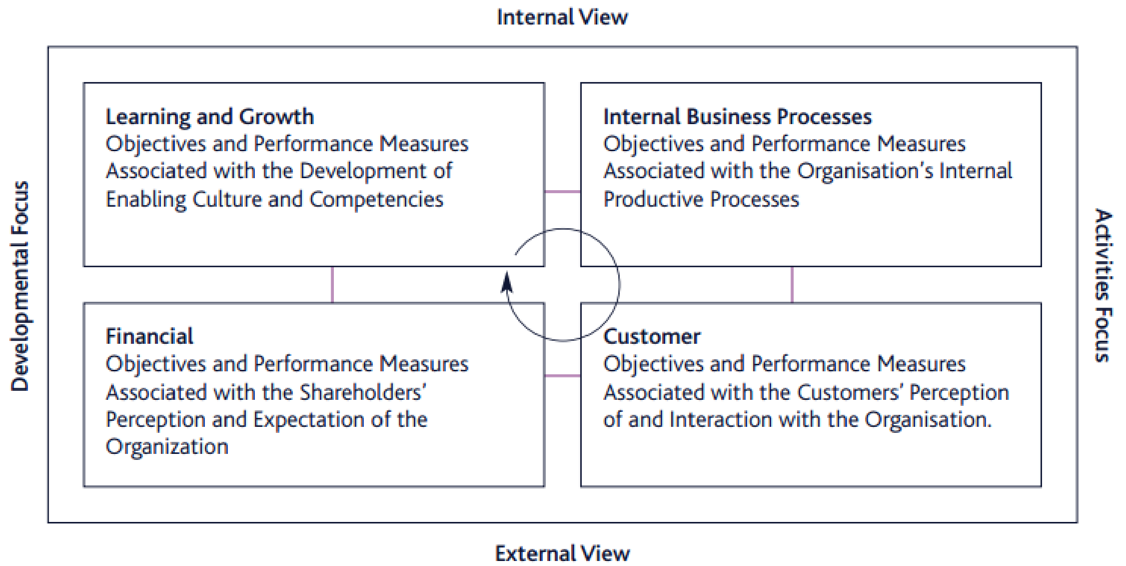


Figure 6: The Balanced Scorecard Quadrants (Source: Mackay, 2005)

After the quadrants are formulated, the organisation's strategy is translated into specific objectives that can be classified within each of the perspectives. Once the objectives have been identified, appropriate quantitative measures are conceived to report and monitor the success in achieving these objectives. The following table (Table 3) shows examples of objectives and measures for the quadrant "Customer".

Table 3: Examples of objectives and measures for the quadrant "Customer" (Source: own elaboration extracted from Mackay, 2005)

Objectives	**Measures**
To dominate our major markets	Market Share
To delight our targeted customers	Customer Satisfaction Survey Results
To increase revenue through repeat purchases	Customer Retention Over Time
To grow our business in a selected target group	Customer Acquisition From Target Group
To add margin through image or fashion	Marketing Spend as a Percentage of Sales
To build customer recognition	Corporate Image or Brand Awareness Polls

The real power of a properly developed Balanced Scorecard is, that it links the performance measures to the organisation's strategy, and organisations have the freedom to use whatever quadrants or perspectives that best suit their environment and strategy. Organisations implementing a BSC process are forced to think clearly about their purpose or mission, their strategy and who the stakeholders in their organisation are and what their requirements might be.

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

Underlying Concepts

BSC and Life Cycle Assessment (LCA)

The fact that the BSC has already been applied to supply chain management along its different stages (Wittstruck & Teuteberg, 2011; de Sousa et al. 2014) might turn this management tool into a potential and suitable approach that could be combined with LCA**. But as it does not consider quantitative material flows which are the basis for LCA it cannot be used as standalone methodology following the Life cycle approach.** According to several authors, the four perspectives included in the BSC are appropriate for overcoming the problems related to performance assessment in supply chains (de Sousa et al. 2014).

BSC and Materials Flow Analysis (MFA)

According to literature reviewed and case studies where the BSC approach has been applied, **this methodology does not consider quantitative material flows** and therefore there is **no clear relation to MFA**.

Assessed Impacts

Social Assessment

The BSC approach **can certainly be extended** and used to assist the **measurement the social impact of implementing specific strategies**. For this purpose, social aspects could be integrated into the four existing perspectives or, alternatively, be included as new perspectives (Wittstruck & Teuteberg, 2011). A combination of the two previous options would be possible too.

Social aspects have been assessed with the BSC in many occasions. For instance, Dias-Sardinha & Reijnders (2005) and Wittstruck & Teuteberg (2011) have reported on the suitability of such methodology to assess social and environmental performance – which is highly interlinked – at a company level.

Economic Assessment

The BSC approach was originally created to supplement traditional financial measures with criteria assessing performance from different perspectives. Therefore, it is of great use when assessing economic aspects of an organisation. In this sense, existing processes that are normally run by different parts of the organisation (e.g. budgeting by finance, process management by operations, etc.) must be modified and coordinated to create strategic alignment, as they must work as a system (Kaplan 2010).

Environmental Assessment

The specific perspectives or quadrants selected to develop a BSC approach at any organisation are **rarely related to environmental aspects**. However, some studies show the use of the BSC to evaluate the environmental impact of different activities. For instance, Wittstruck and Teuteberg (2011) developed a BSC for Sustainable Supply Chain Management (SSCM). The use of the BSC allowed the assessment of the environmental and economic benefits of sustainability investments for the partners within a recycling supply chain.

Another study, carried out by Wati and Koo (2011), integrated the measurement of environmental aspects into the BSC and offered a new possibility to sustainable businesses, creating a Green-IT Balanced Scorecard.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Suitability for URBANWASTE

The BSC approach is **mainly intended for companies**, **public sector agencies** and **non-profit organisations** working on the implementation of strategy execution systems. Due to the company-oriented nature of this approach and as it requires simultaneous coordination among all organisational line and staff units, the suitability of such approach to **evaluate innovative solutions for waste management in tourist cities remains unclear**. In addition, the BSC approach does not seem to match the MFA methodology, which would hinder its applicability to potential defined strategies.

Suitability for assessment of changes on hotel level or on municipality level

The use of the BSC **approach on municipal level has not been clearly identified in literature**, as it is a tool intended to describe, communicate and implement different strategies in companies, public sector agencies and non-profit organisations.

With regard to its implementation on a hotel level, Denton and White (2000) reported about a BSC developed to improve the effectiveness of operations in a hotel, taking into account the objectives of owners and managers in assessing the hotel's success.

Summary / Conclusions

Although the BSC approach has been introduced as a very useful tool to achieve a balance between different perspectives on the basis on targets, key performance indicators (KPIs) and measures; and evaluate the economic, social and environmental aspects of an organisation, **this tool might not be the right method to be used in URBANWASTE)**.

3.4 Carbon Footprint (CF) of products (PCF) and corporate (CCF)

More than any other concept or method, the "carbon footprint" (CF) has gained widespread popularity over the last years. In contrast to other assessment methodologies, carbon footprinting has not been driven by research but rather has been promoted by nongovernmental organizations (NGOs), companies and various private initiatives. This has resulted in many definitions and suggestions of methods to calculate the carbon footprint (Weidema et al., 2008). A review of Wiedmann and Minx (2007) showed that **most currently used definitions focus on greenhouse gas (GHG) emissions instead of solely including carbon dioxide (CO_2) emissions and use carbon dioxide equivalents (CO_2) indicators**. The Carbon Trust (2012), for examples, defines carbon footprinting is a methodology to estimate "***the total greenhouse gas (GHG) emissions caused directly and indirectly by an individual, organisation, event or product***". The calculated carbon footprint (CF) is expressed as carbon dioxide equivalents (CO_2e), a unit that allows comparing the radiative forcing of different greenhouse gases (CO_2, CH_4, N_2O, HFCs, PFCs, SF_6) to carbon dioxide (BSI, 2011).

In general, the Carbon Footprint (**CF**) can be applied to organisations and products. The **organisational or corporate carbon footprint (CCF)** covers the direct and indirect GHG emissions from all activities across an

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

organisation, including energy used in buildings, industrial processes and company vehicles. The **product carbon footprint (PCF)** covers the GHG emissions **from all stages of the life cycle** of the "product" (meaning both goods or services), from the extraction and transport of raw materials, the manufacturing and distribution of goods or provision of services to its use and final re-use, recycling or disposal (Carbon Trust, 2012). For assessing the overall emissions of a product both GHG emissions to the atmosphere as well as removals of GHG gases from the atmosphere have to be accounted for (BSI, 2011).The scope of the assessment of a product's climate impact due to GHG emissions can be cradle-to-gate[2] or cradle-to-grave[3] (BSI, 2011). The different boundaries of organisational/corporate and product carbon footprint are illustrated in Figure 7Figure 7.

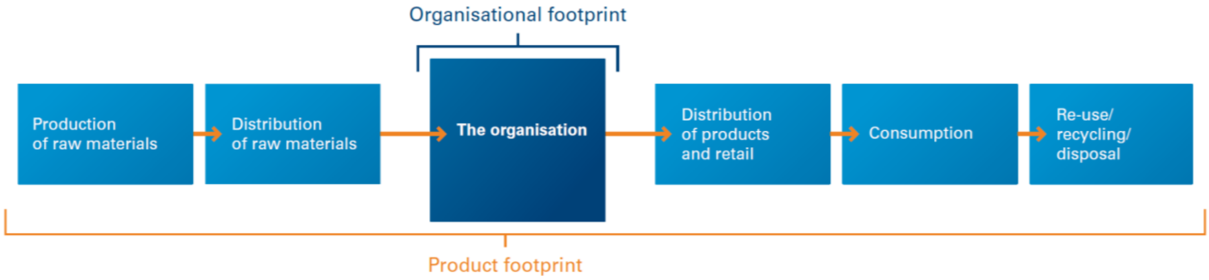


Figure 7: The different boundaries of organisational/corporate and product carbon footprint (Carbon Trust, 2012)

Carbon footprinting (CF) has a wide range of application (Carbon Trust, 2012). Compared to more comprehensive assessment methodologies such as Life Cycle Assessment (LCA), CF keep things simple by focussing only on one environmental impact and are easy to calculate online (Weidema et al., 2008). Furthermore, the use of a common measurement unit (CO_2e) simplifies the analysis and makes the results (calculated values) easily understandable also outside the scientific community (Weidema et al., 2008). On the other hand, relying entirely on one indicator could be misleading due to oversimplifying the environmental impacts (Weidema et al., 2008).

In order to produce reliable footprints that also allow comparison it is important to follow a uniform way of calculation. To standardise the accounting of carbon footprint (CF) some international standards exist:

- **PAS 2050 – Assessing the Life Cycle Green House Gas Emissions of <u>Goods and Services</u>:** This Publicly Available Specification (PAS) provides a consistent method (requirements for assessment) to assess the GHG emissions of goods and services resulting from all stages of the product's life cycle (including production, transport, storage, use, recycling and disposal of goods and services). The term "products" covers both goods and services (BSI, 2011).
- **ISO 14067:2013 Greenhouse gases -- Carbon footprint of products -- Requirements and guidelines for quantification and communication:** ISO 14067:2013 specifies principles, requirements and guidelines for the quantification and communication of the carbon footprint of a product (CFP) (ISO, 2013).

[2] A cradle-to-gate assessment covers all *"life cycle stages from the extraction or acquisition of raw materials to the point at which the product leaves the organization undertaking the assessment"* (BSI, 2011).
[3] A cradle-to-grave assessment covers all "*life cycle stages from the extraction or acquisition of raw materials to recycling and disposal of waste"* (BSI, 2011).

- **ISO 14064 standards for greenhouse gas accounting and verification:** This group of ISO standards provides tools (validation or verification protocols) to account GHG emissions and removals for organization- and project-level (e.g. for programmes aimed at reducing greenhouse gas emissions). These standards can be applied by governments, business corporations and voluntary initiatives (ISO, 2006), but not for assessing products.

- **Greenhouse Gas Protocol (GHG Protocol):** The Greenhouse Gas Protocol, created in 2001 developed by World Resources Institute (WRI) and World Business Council on Sustainable Development (WBCSD), sets the global standard for how to measure, manage, and report greenhouse gas emissions (WRI and WBCSD, 2016). This standard is widely used by businesses and governments. Emissions are categorised into three groups or 'scopes' (Carbon Trust, 2012):

    - Scope 1: Direct emissions that result from activities within an organisation's control (e.g. on-site fuel combustion, manufacturing and process emissions, refrigerant losses, company vehicles).

    - Scope 2: Indirect emissions from any electricity, heat or steam purchased and used by an organisation.

    - Scope 3: Any other indirect emissions from sources outside the direct control of an organisation (e.g. employee commuting, business travel, outsourced transportation, waste disposal, water consumption).

Underlying Concepts

Carbon footprinting considers upstream and downstream processes and, thus, is based on life cycle thinking (Weidema et al., 2008). **It can be seen as a subset** of the concepts of Ecological Footprinting which was developed by Rees and Wackernagel in the 1990s (Wiedmann and Minx, 2007) and the **more comprehensive Life Cycle Assessment (LCA)**. In contrast to a complete LCA, **CF focusses only on the impacts related to GHG emissions**. According to PAS 2015, the assessment of the life cycle GHG emissions is carried out by using LCA techniques and principles (BSI, 2011).

Carbon footprinting **also includes aspects of Material Flow Analysis (MFA)** as information on quantitative material flows has to be considered. For calculating the total (direct and indirect) GHG emissions data such as energy and materials needed for and by a product (covering all life cycle stages of the product from production to disposal) or for and by activities of an organization has to be included.

Assessed Impacts

Carbon footprinting is **assessing only one single environmental issue**, namely GHG emissions and their contribution to climate change. The **only impact category addressed** by this method is the **Global Warming Potential (GWP)**. Other environmental impacts (e.g. non-GHG-emission, acidification, eutrophication, toxicity, biodiversity) or economic and social impacts (e.g. labour standards) are not addressed (BSI, 2011).

Suitability for URBANWASTE

In order to be suitable for the URBANWASTE project, **a method has to allow an accompanying assessment of environmental, economic and social impacts of the current situation** ("baseline") in the URBANWASTE pilot

cases and support the development of selected strategies aiming at reducing the amount of municipal waste production and at further support the re-use, recycle, collection and disposal of waste in tourist cities.
Generally, CF could be used to assess the waste related impacts of various tourist establishments as the carbon footprint of an organization encompasses a wide range of emission covering not only direct emissions but also indirect emissions from sources outside the direct control of an organisation up and down the supply chain (e.g. waste disposal). Nevertheless, this assessment would only cover the environmental aspects of tourism related waste generation, but not the economic and social impacts as these are not addressed with this method. Thus, the method of carbon footprinting (CF) is not suitable to meet the complex demands of the URBANWASTE project. If the method is used for environmental assessment, it has to be considered, that

Suitability for assessment of changes on hotel level or on municipality level

The method of carbon footprinting (CF) **allows an assessment of changes on both hotel and municipality level**.

To assess the environmental impact of tourism enterprises such as **hotels** or other tourist accommodation establishments the method of **organisational/corporate carbon footprinting (CCF)** can be applied. As Filimonau et al. (2011) demonstrate even outsourced laundries and breakfast services can be taken into account in the assessment. The aspect of solid waste – which is in focus of the URBANWASTE project - is covered by measuring also the indirect GHG emissions (i.e. emissions that are labelled "Scope 3" emission in the Greenhouse Gas Protocol).

A superficial web search revealed that **worldwide many hotels already assess their carbon footprints**. Online tools such as **www.hotelfootprints.org** are available for hotels to facilitate benchmarking and comparison of carbon footprints (ITP and GreenView, 2016). Furthermore, some online booking platforms such as "**www.bookdifferent.com**" show the carbon footprint of every hotel, thus, giving consumers the choice to decide for an accommodation with low carbon emissions (BookDifferent Foundation, 2016).

Assessing the carbon footprint in the life cycle of accommodation services is also discussed in scientific literature. Studies (e.g. Hu et al., 2015; Rosselló-Batle et al., 2010) show that CF can be used to investigate the difference between the carbon emissions of the baseline and, for example, after a programme aiming at reducing GHG emissions to understand the effects of carbon emission reduction in individual hotels as well as on the level of the hotel industry. In order to calculate the carbon footprint attributed to a hotel's operational phase often LCA or simplified versions of LCA such as Life Cycle Energy Analysis (LCEA) are used. From the results, recommendations for hotel management and policy-making can be developed to reduce the energy and carbon intensity of individual hotels or the hotel industry (Filimonau et al., 2011).

Following the definition of Carbon Trust or standards like PAS 2050, ISO 14064 or ISO 14067 the assessment of the environmental impact due to GHG emission of a region is not in focus. Nevertheless, a web search reveals that calculations of carbon footprints on municipality level have been carried out. Larsen and Hertwich (2010), for example, calculated the carbon footprint of public services provided by all municipalities in Norway. These services included amongst others also the collection and treatment of solid waste. Thus, CF could also be applied for an assessment of changes on the municipal level.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

3.5 Comparative Risk Assessment (CRA)

Comparative Risk Assessment (CRA) is a **tool used to base regulatory and political decisions on a thorough and rational analysis of risks and decision options** (Schütz et al., 2006). According to the definitions of Morgenstern et al. (2000) and Brown (2015), CRA can be described as a **systematic and objective procedure to measure, compare and rank (environmental) problems** in terms of their consequences, thus evaluating the risk a given problem poses to the natural environment, to human health or the quality of life. Originally, this method was developed by the U.S. Environmental Protection Agency (US EPA) to compare environmental problem areas by systematically generating informed judgements among experts. Since the end of the 1980ies CRA has been widely applied in the USA. In Europe, though, CRA is by far less prominent (Schütz et al., 2006).

Comparative Risk Assessment (CRA) can be applied to characterize environmental priorities on the regional as well as on the national level. Micro studies using CRA usually compare the interrelated risks involved in a specific policy choice (e.g. drinking water safety: chemical versus microbial disease risks) and focus often on one or a few types of environmental problem. CRAs at a larger scale (e.g. nationwide) usually consider multi-risks the society faces by comparing different types of environmental problems (Andrews et al., 2005).

According to Morgenstern et al. (2000) the most common reasons for performing a CRA are:

- To involve the public in the priority-setting process and to identify and incorporate their concerns;
- To identify and rank the greatest environmental threats;
- To establish priorities and develop action plans and strategies to reduce risks (e.g. for how to allocate limited resources to different activities reducing or preventing environmental risks).

CRAs typically consist of a risk assessment phase, followed by a risk management phase (Morgenstern et al., 2000). In the first phase (risk assessment phase) the (environmental, societal, etc.) problems that should be addressed and analysed are determined and ranked based on their severity. The risk ranking process can be done based on "informed judgements" or by employing economic methodologies. In the second phase of CRA (risk management phase) the list of problems ranked in terms of relative risk is transformed into a set of priorities for action by incorporating risk management factors into the overall process. An international review by Morgenstern et al. (2000) showed that while all CRAs include the first risk assessment phase, only some also include an explicit risk management process.

Underlying Concepts

The underlying concept of CRA is to **combine science, policy and economic analysis as well as stakeholder participation to identify, address and prioritise topics of great importance due to the risks they pose to human health, the natural environment and quality of life** (Brown, 2015). Therefore, CRA is often used by government agencies, communities or other political bodies as an environmental decision making tool (Schütz et al., 2006).

<u>CRA and LCA</u>

In connection with LCA CRA can be used to enable comparison of risks of different decision options (e.g. for comparing the toxic impacts between different substances). But CRA cannot replace LCA or be used instead of LCA.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

CRA and MFA

In literature there is no mentioning of CRA being applied for the analysis of material flows.

Assessed Impacts

The most commonly considered assessment categories in CRAs are **risks to human health or environmental risks** (Brown, 2015; Morgenstern et al., 2000). An assessment of risks to **quality of life, welfare or to economic well-being** is also possible. International studies mainly focus on public health issues (Morgenstern et al., 2000).

Suitability for URBANWASTE

CRA is not suitable to answer specific URBANWASTE questions related to the impact of changes resulting from implementing certain waste management measures in the URBANWASTE pilot cases as this tool usually is applied to evaluate and rank the risks a given set of problems poses to the natural environment, to human health or to the quality of life and not for assessing the impacts of certain measures such as waste prevention activities.

Suitability for assessment of changes on hotel level or on municipality level

CRA is **not suitable for assessing changes on hotel level or municipality level** as this method is not designed for assessing the impact of certain measures (e.g. waste prevention activities) but for identifying priorities by ranking a defined set of problems based on and, thus, reflecting their relative health risks, ecological risks and risks to quality of life.

3.6 Corporate Social Responsibility (CSR)

Corporate Social Responsibility (CSR) is a standardised working methodology (since 2010) intended to **assist organizations in contributing to sustainable development** (ISO, 2014). It encourages the organisations to go beyond legal compliance, recognizing that compliance with the law is a fundamental duty of any organization and an essential part of their social responsibility programme. The standard, as it is described in ISO 26,000, seeks to promote a common understanding of social responsibility while complementing, but not replacing, other existing tools and initiatives. Organizations applying ISO 26,000 should consider societal, environmental, legal, cultural, political and organizational diversity as well as differences in economic conditions, while being consistent with international norms of behaviour (ISO, 2010). The scope of CSR can therefore be considered much wider than the social, economic and environmental considerations that are in the focus in this report.

Since CSR does not include any criteria that an organisation can fulfil, the organisations cannot be certified against this standard. Therefore, the organisations can only claim that they are working with CSR, which means that it will be more suitable as a strategy for increased sustainability (including waste reduction) for organisations like hotels and municipalities rather than a methodology to assess this increased sustainability. However, CSR could include qualitative descriptions of what has been done in the CSR work, and this descriptive data could be used as a way to assess the environmental performance of an organisation.

Underlying Concepts

The underlying concept of CSR is **not a scientific methodology for assessing sustainability**, but a working process for organisations how to become more sustainable. Even though it is described and defined by an ISO-standard it does not contain any requirements and therefore it cannot be used as a certification. Since there are no requirements included, it is **difficult to use this method for quantitative assessments**, but the method could still be used by organisations (e.g. hotels or municipalities) as a strategy to improve their sustainability.

Assessed Impacts

When applying ISO 26,000, organizations should consider societal, environmental, legal, cultural, political and organizational diversity as well as differences in economic conditions, while being consistent with international norms of behaviour. However, since CSR is a work methodology for organisations it does not include criteria on how to assess any of the considered parameters.

Suitability for URBANWASTE

This **methodology does not include assessments of sustainability** and is **therefore not suitabl**e to answer specific URBANWASTE questions. However, the implementation of CSR in organisations could be used as a strategy to improve sustainability, but these improvements should then be assessed with a supplementary methodology.

Suitability for assessment of changes on hotel level or on municipality level

CSR could be used as a tool for improving sustainability in organisations like hotels and municipalities, and through this work descriptive data of what has been done and how it could be achieved.

3.7 Cost Benefit Analysis (CBA)

The history of the Cost-Benefit Analysis (CBA) shows how its theoretical origins date back to issues in infrastructure appraisal in France in the 19th century. The theory of welfare economics developed along with the "marginalist" revolution in microeconomic theory in the later 19th century, culminating in Pigou's Economics of Welfare in 1920 which further formalised the notion of the divergence of private and social cost, and the "new welfare economics" of the 1930ies which reconstructed welfare economics on the basis of ordinal utility only. Theory and practice remained divergent, however, until the formal requirement that costs and benefits be compared entered into water-related investments in the USA in the late 1930ies. After World War II, there was pressure for "efficiency in government" and the search was on for ways to ensure that public funds were efficiently utilised in major public investments. This resulted in the beginnings of the fusion of the new welfare economics, which was essentially CBA, and practical decision-making. Since the 1960ies CBA has enjoyed fluctuating fortunes, but is now recognised as the **major appraisal technique for public investments and public policy** (Pearce et al. 2006).

Thus, **CBA is an analytical tool for judging the economic advantages or disadvantages of an investment decision by assessing its costs and benefits in order to measure the welfare change attributable to it** (Sartori

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

et al. 2015). A CBA evaluates the costs and benefits to society of a project, policy or programme, for example implementing a given waste policy or building a treatment facility. If the net benefit is positive, the project should, as a general rule, be implemented. A clear advantage of the CBA is that the **result is expressed in a well-known measure: money** (Villanueva et al. 2006).

A CBA starts with a description of the system, its inputs and outputs, and an inventory of these, and includes all effects related to the implementation of the project or policy (investments, manpower, employment, and effects on human health and safety). Subsequently, a **value is ascribed to all these effects, converting them into monetary units**. The **various costs and benefits may occur at different times in the future**, which is why a so-called **net present value** is calculated to illustrate the aggregated value in today's prices. For this calculation, a **discount rate** is used which reflects the degree to which society prefers to consume and gain benefits today rather than tomorrow. The higher the discount rate, the lower the weight assigned to future costs and benefits compared to costs and benefits experienced today.

An *actual* CBA differs from an *ideal* CBA. This is, for example, seen in the geographical coverage of CBAs, which most often cover the activities and emissions inside the national or regional borders of interest. Emissions and resulting impacts in other countries are usually ignored, mostly because traditionally a government is only concerned with maximising the socio-economic benefits for its own citizens. There can be exceptions when an international treaty has been joined, or ethical reasons recommend it.

This methodology has a first phase of goal and scope definition, followed by the assembly of an inventory of inputs and outputs, as well as a stage of evaluation of impacts, and an interpretation of the results (Villanueva, et al. 2006).

A "standard" CBA is structured in seven steps (see Figure 8):

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

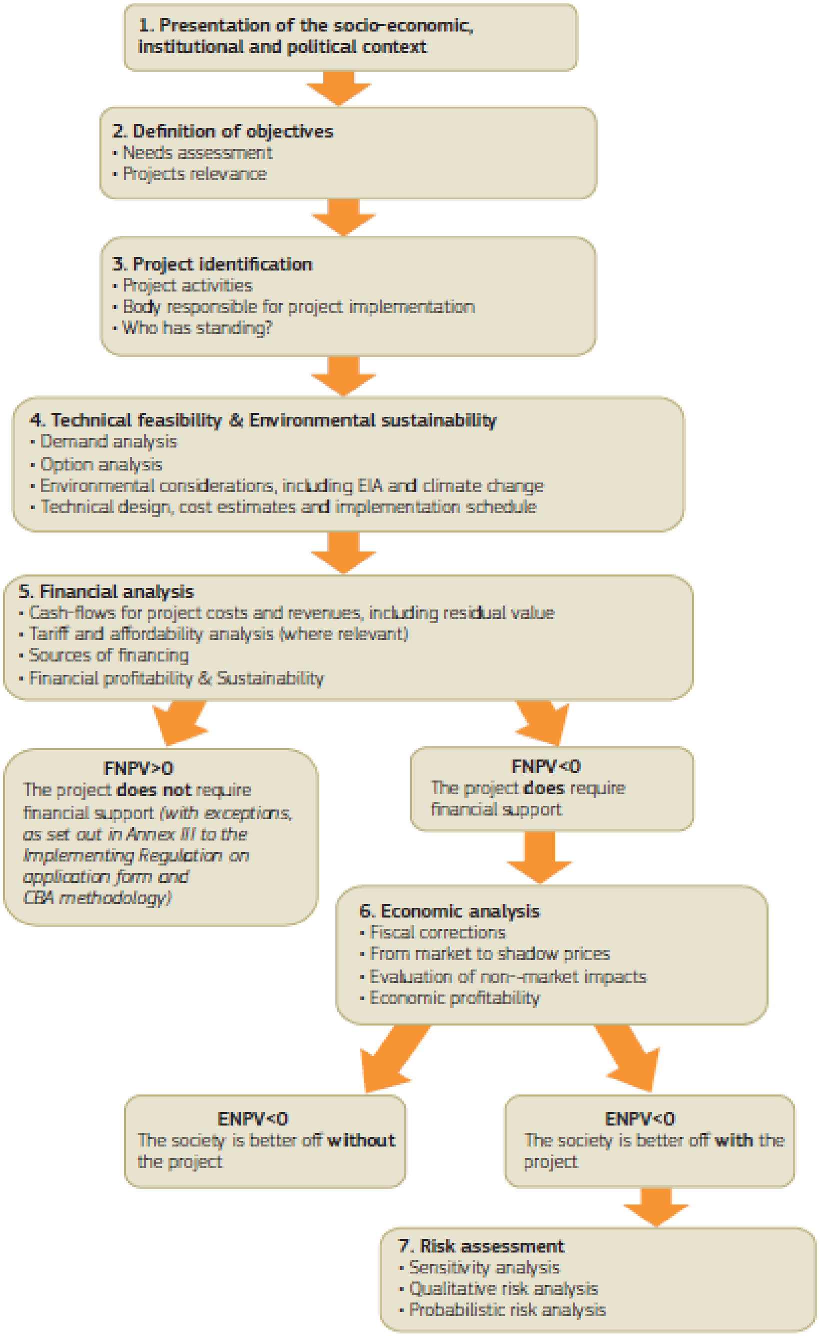


Figure 8: CBA steps (Source: Sartori et al. 2015).

Underlying Concepts

The analytical framework of CBA refers to a list of underlying concepts (Sartori et al. 2015):

- **Opportunity cost.** The opportunity cost of a good or service is defined as the potential gain from the best alternative forgone, when a choice needs to be made between several mutually exclusive alternatives. The rationale of CBA lies in the observation that investment decisions taken on the basis of profit motivations and price mechanisms lead, in some circumstances (e.g. market failures such as asymmetry of information, externalities, public goods, etc.), to socially undesirable outcomes. On the contrary, if input, output (including intangible ones) and external effects of an investment project are valued at their social opportunity costs, the return calculated is a proper measure of the project's contribution to social welfare.

- **Long‑term perspective.** A long‑term outlook is adopted, ranging from a minimum of 10 to a maximum of 30 years or more, depending on the sector of intervention. So, it is needed to set a proper time horizon, **forecast future costs and benefits**, adopt appropriate discount rates to calculate the present value of future costs and benefits and take into account uncertainty by assessing the project's risks.
- **Calculation of economic performance indicators expressed in monetary terms.** CBA is based on a set of predetermined project objectives, giving a monetary value to all the positive (benefits) and negative (costs) welfare effects of the intervention. These values are discounted and then totalled in order to calculate a net total benefit. The project overall performance is measured by indicators, namely the Economic Net Present Value (ENPV), expressed in monetary values, and the Economic Rate of Return (ERR), allowing comparability and ranking for competing projects or alternatives.
- **Microeconomic approach.** CBA is typically a microeconomic approach enabling the **assessment of the project's impact on society as a whole via the calculation of economic performance indicators**, thereby providing an assessment of expected welfare changes. While direct employment or external environmental effects realised by the project are reflected in the ENPV, indirect (i.e. on secondary markets) and wider effects (i.e. on public funds, employment, regional growth, etc.) should be excluded. This is due to two main reasons:
  - Most indirect and/or wider effects are usually transformed, redistributed and capitalised forms of direct effects, so there is a need of limiting the potential for benefits double‑counting.
  - There remains little practice on how to translate them into robust techniques for project appraisal, thus the need to avoid the analysis relies on assumptions whose reliability is difficult to check.

  It is recommended, however, to provide a qualitative description of these impacts.
- **Incremental approach.** CBA compares a scenario with‑the‑project with a counterfactual baseline scenario without‑the‑project. The incremental approach requires that:
  - A **counterfactual scenario is defined as what would happen in the absence of the project**. For this scenario, projections are made of all cash flows related to the operations in the project area for each year during the project lifetime. In cases where a project consists of a completely new asset, e.g. there is no pre‑existing service or infrastructure, the without‑the‑project scenario is one with no operations. In cases of investments aimed at improving an already existing facility, it should include the costs and the revenues/benefits to operate and maintain the service at a level that it is still operable (Business As Usual[4] (BAU)) or even small adaptation investments that were programmed to take place anyway (do‑minimum[5]).

    The choice between BAU or do‑minimum as counterfactual should be made case by case, on the basis of the evidence about the most feasible, and likely, situation. If uncertainty exists, the BAU scenario

[4] *For example, a scenario that ensures: (i) basic functionality of the assets, (ii) service provision under similar quality levels, (iii) limited asset replacements and (iv) minimum cost recovery to ensure financial sustainability of operations.*

[5] *For example, when limited amount of capital investments are necessary to avoid interruption of service or any other catastrophic scenario.*

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

shall be adopted as a rule of thumb. If do‑minimum is used as counterfactual, this scenario should be both feasible and credible, and not cause undue and unrealistic additional benefits or costs.

– Secondly, projections of cash‑flows are made for the situation with the proposed project. This takes into account all the investment, financial and economic costs and benefits resulting from the project. In cases of pre‑existing infrastructure, it is recommended to carry out an analysis of historical costs and revenues of the beneficiary (at least three previous years) as a basis for the financial projections of the with‑project scenario and as a reference for the without‑project scenario, otherwise the incremental analysis is very vulnerable to manipulation.

– Finally, the CBA only considers the difference between the cash flows in the with‑the‑project and the counterfactual scenarios. The financial and economic performance indicators are calculated on the incremental cash flows only.

CBA and Life-Cycle Assessment (LCA)

LCA and CBA are two methodologies that can be used to assess the environmental and/or socio-economic consequences of a decision, for instance whether the citizens in a community or a country should or should not recycle their waste paper, glass bottles, and beverage cans. However, these two tools provide the decision-makers with different information. A CBA's objective is to maximise the overall utility to society, whereas an LCA aims at modelling the overall environmental impacts of a product or service. **Since the methodologies are answering different questions, they should not be considered as competing, but rather as complementary tools** (Villanueva, Kristensen & Hedal, 2006).

LCAs and CBAs are decision *support* tools and not decision *making* tools, because they provide information that normally needs to be complemented with legal, social, economic or technical information before decisions are made.

An LCA evaluates all known environmental impacts of a product, material or service throughout its entire lifetime. It is an ambitious methodology, and it covers all stages in the life cycle of the products or systems investigated, from the cradle (material extraction) to the grave (final disposal). It attempts to cover all physical exchanges in the life cycle of a product with its surroundings, be it inputs of auxiliary materials and energy consumption, or outputs of emissions, waste and usable energy. The results are collected in a so-called life cycle inventory, which for a 40 g paper sheet, for example, may be expressed as the consumption of 100 g of wood and of 0.25 kWh energy, generation of 0.1 kWh energy, emission of 64 g CO_2 and 0.1 g SO_2, along with a host of other inputs and outputs (Villanueva, Kristensen & Hedal, 2006).

In an *ideal* LCA, all inputs and outputs should be covered in the inventory. In *practice*, the completeness of the inventory is limited by the analyst's knowledge of the interactions of the waste management system with society and the environment (for details see Chapter 3.16).

A CBA also starts with a description of the system, its inputs and outputs, and an inventory of these. A CBA includes all effects related to the implementation of the project or policy, i.e. the same inputs and outputs as the LCA, but in addition it includes investments, manpower, employment, and effects on human health and safety.

Both methodologies have a first phase of goal and scope definition, followed by the assembly of an inventory of inputs and outputs. Both have a stage of evaluation of impacts, and with an interpretation of the results. However, it is in the detail of the methodologies such as in their focus, perspective, scope and basis for

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

comparison that the differences are observed, justifying the remark that these methodologies are not perceived as competing, but as complementary.

The similarities in the stages of LCA and CBA are illustrated in Figure 9:

	LCA	CBA
	Goal and scope definition	**Goal and scope definition**
Objective	Minimise the environmental impact	Maximise the utility to society
Focus	Environmental impacts	Economic impacts
Time scope	Different impact horizons for different impact categories	The project's time horizon
Geographical scope	Global	Most offen national, but can be broader if there are international treaties or ethical reasons to do so
System scope	All known interactions in the life cycle	Interactions with known economic impact
Impact scope	All known physical exchanges	All known physical exchanges and non-physical impacts
	Inventory	**Inventory**
	Direct and indirect exchanges with the physical surroundings are followed iteratively	Mainly direct effects
	Evaluation of impacts	**Evaluation of impacts**
Basis of comparison	Based on functionality	Based on market relation (individual's preferences)
Intermediate steps	Characterisation, Normalisation, Weighting	Monetisation of inputs and outputs, aggrega-tion, definition of discount rate, calculation of net present value and indirect known effects.
Result	Set of environmental impacts	Net benefit
Sensitivity analysis	Compulsory	Recommended. Lack of data is represented by zero
	Interpretation	**Interpretation**

Figure 9: Comparison between CBA and LCA (Source: Villanueva, Kristensen & Hedal, 2006).

LCAs and CBAs translate the environmental (CBAs also the non-environmental) benefits and costs of the different options into measurable physical or economic units. Waste policy makers and waste management experts can use LCA and CBA for several purposes. Most frequently, to compare two or more alternative options for the treatment of a waste stream, such as the disposal or recycling of paper. Following this paper

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

example, an LCA would analyse all known physical interactions of the paper system with its surroundings, while a CBA would analyse all known physical interactions of the paper system and would also include all relevant effects which can be monetised (Villanueva, Kristensen & Hedal, 2006).

CBA and Materials Flow Analysis (MFA)

Materials Flow Analysis (MFA) consists of systematic assessments of the flows and stocks of materials within a system defined in a space and time. MFA diverges somewhat from the traditional SWM boundary to focus on product consumption patterns, waste generation, recycling, recovery and reuse (Boelens and Olsthoorn, 1998; Brunner and Rechberger, 2003; Chanchampee and Rotter, 2007).

The combination of CBA and MFA has been further developed and applied to problems mainly in waste management by Schönbäck and colleagues from the Vienna University of Technology and the GUA (consultants in Vienna) (Brunner & Rechberger, 2004).

Assessed Impacts

Economic Assessment

CBA is basically an economic assessment tool; it is devoted to the evaluation of the advantages and disadvantages of investing on a project, policy or programme, contrasting its costs and benefits to measure the welfare change that the project development can bring.

Environmental Assessment

The environment often displays the characteristics of a public good; in these cases, there is open access to the good, and is not apparently depleted. These public goods aspects of the environment are obvious sources of social utility, but they appear to command a price of zero in the market. Even where the environment can be depleted and the public can be excluded, the prices that result often reflect administered powers rather than market forces.

Conventional CBA operates on the premise that market efficiency is a pointer to social efficiency if one can simply spot the market's failures and correct for them. Environmental impacts are one source of missing markets (i.e. externalities). In evaluating a project or policy, the environment can be treated as a free factor of production, even though real costs may be involved. An example is the use of the air or nearby rivers, as waste sinks – emissions are simply released through smoke stacks or outfalls. The less the constraints on these emissions, the lower the production costs involved. There are two key points that emerge: (a) Both the true value of the environmental services provided, and the external costs imposed on others, have to be counted in an economic CBA; and (b) The values mentioned above have to be included without any double counting.

There is increasing pressure on current EIA practice to place a real value on the environment, and force public and private enterprises to take cognisance of it. The costs of the EIA appear in the financial CBA. The true value of the damage done, however, only appears at the level of economic CBA. Environmental regulations may impose costs on polluters, but may not put pressure on them to emit the socially 'optimal' amount of any pollutant (Leiman, & Tuomi, 2004).

Some environmentalists oppose the monetary valuation of natural resources that have 'immeasurable' intrinsic and aesthetic values. However, in today's monetised global economy, valuing resources (even if the figure attained is imprecise) can suggest the worth of protecting them.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Fortunately, this issue can often be circumvented. Valuation of environmental impacts is both costly and controversial. Assuming negative ecological impacts, if a project fails the cost benefit tests before these impacts are taken into account, there is obviously no need to proceed with valuation. Similarly, if the project is viable, and the environmental externalities are positive, there is no need to measure them. Where a project is positive but the environmental impacts negative, inspection of the EIA reports may indicate the relative magnitude of the problem, and whether or not it can be mitigated. Again it may be possible to avoid valuation of impacts. Only if valuation of environmental impacts is likely to influence the outcome of the analysis, it should be carried out (Leiman, & Tuomi, 2004).

Social Assessment

Generally speaking, most of the CBA impacts are expressed in monetary form. A problem that can be identified later is that some of their prices have been distorted by market failure, while others have no prices at all. If the prices are accepted at face value and no estimates are made of the unpriced impacts, the result is a simple financial CBA. If the distorted and missing prices are corrected for, the result is a social or extended or economic CBA (Leiman, & Tuomi, 2004).

The difference between the two types of CBA is important. For any given project, ***financial* CBA** looks at the costs and benefits to an individual stakeholder. ***Social* CBA** looks at the costs and benefits to society as a whole, trying to determine whether the project will make society better or worse off. An example will help to clarify the difference. Say a hotel chain was thinking about building a luxury hotel in a wilderness area. The hotel chain would effectively perform a *financial* CBA, looking at the direct costs and benefits to the chain. They would consider the capital, *financial* and labour costs incurred during the actual construction of the hotel, as well as the expected costs of running the hotel. On the benefit side they would consider the expected revenue stream from the new hotel as well as benefits such as increased prestige etc. If they considered environmental aspects at all, these would most likely be the costs of adhering to current government standards. They may also factor in an 'insurance cost' in the event of an environmental lawsuit. They would discount the CBA using the opportunity cost of their *financial* capital (i.e. the market rate of interest).

A CBA performed by the government would necessarily go further in order to consider the implications of the project on the whole of society. The prices of land, labour and capital would be corrected to address any implicit subsidies, distorting taxes or market imperfections. The exchange rate would be checked to ensure that over or under valuation was not generating a spuriously high or low net earnings flow. It would consider all stakeholders, looking not only at the direct costs incurred in the new hotel's construction and running, but also at the costs to the rest of society. This could include things like an estimation of the cost of environmental destruction, the effect of noise pollution, the impact on local communities etc. They would also have to consider a broader range of benefits (e.g. increased tax revenue, an increase in foreign exchange, a reduction in unemployment, etc.). A complete *social* CBA should also consider the needs of future generations and adjust the discount rate accordingly.

The two types of CBA can therefore yield to substantially different outcomes (Leiman, & Tuomi, 2004).

Suitability for URBANWASTE

CBA is meant to be one of the URBANWASTE relevant methodologies, **specially run in parallel with LCA** (as it is usually applied), as it evaluates the costs and benefits to society of a project, policy or programme. Decision makers can use its results, expressed as costs, as a very strong argument to go for the project evaluated or give it up.

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

Together with LCA, it can assess the environmental and socio-economic consequences of a decision, being the CBA focused on finding out how useful is the project for the society while LCA checks out the harmful impacts on the environment.

However, as mentioned, CBA (and also LCA) only support decision makers, as the information and conclusions they provide have to be complemented with legal, social, economic and/or technical information. Once all these data are compiled, a decision can be made.

Suitability for assessment of changes on hotel level or on municipality level

CBA is conceived for assessing the financial viability, as well as the implementation consequences at different levels, of a wide variety of projects, policies or programmes. It could be therefore applied at different levels, i.e., for a single hotel, in which those in charge of any decision making process can decide what kind of measures should be applied in their businesses, or at municipal level, being therefore local authorities the ones in charge of evaluating the results of the CBA performed and deciding if a concrete measure should be registered as a local regulation and, consequently, adopted by the businesses belonging to the concerned sector.

Summary / Conclusions

In the private sector, economic profit is often used as an indicator of economic efficiency. This needs not to be valid; the market may be distorted or the decision may involve non-profit projects or the introduction of new government policies. CBA offers an alternative test of efficiency for such situations (Leiman, & Tuomi, 2004).

As its name suggests, CBA simply compares all the expected present and future benefits of a project or policy with its present and future costs. In general future costs and benefits appear less important than present ones, for this reason CBA attaches a progressively lower weight to costs and benefits the further in the future they appear. It is this practice of **discounting** that forms the basis of the opposition to CBA by some environmentalists.

CBA has a number of advantages, which includes:

- The decision rules it uses are standard and well known.
- It provides non-prescriptive information in a standard format that informs decision makers and stakeholders.
- It is adaptable and flexible enough to reflect income distributional impacts, intergenerational sustainability, financial efficiency and the effects of externalities.
- It can be extended to match the EIA process. A CBA report can be adapted to include a stakeholder analysis showing a project's downstream impacts on interested and affected parties. Where projects are large enough to affect macroeconomic variables (e.g. wage levels and exchange rates), the CBA accounts for these. The report can be further enhanced to include other economy wide effects such as multiplier based impacts on employment and GDP (using input-output tables or computable general equilibrium models).
- It has a logical place in the Integrated Environmental Management (IEM) process. In Environmental Impact Assessments (EIA), for example information on project impacts are generated. The economic and social relevance of these is not always clear. Also the data are not always comparable and easily

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

integrated. CBA can reduce most impacts to a single number which describes either a benefit/cost ratio, an internal rate of return or a Net Present Value (NPV). The CBA format establishes a clear link between data collection and the information provided for decision-making.

One of the **key weaknesses of CBA is that it can oversimplify, reduce complex cause and effect linkages to a single number like the NPV or the Benefit/Cost ratio**. This potential problem can be overcome by either ensuring that the sensitivity analysis performed captures the effects of variations in key variables (such as discount rates and income distributional weights) or by combining CBA with one of the multi-criteria decision analysis methods which allow weights to be attached to concerns and impacts identified by specific stakeholders as significant (Leiman, & Tuomi, 2004).

3.8 DPSIR Framework

The DPSIR Framework has been introduced in a technical report of the European Environmental Agency (EEA) in 1999 with the aim of simplifying the understanding of existing environmental indicators for policy makers, given the number and diversity of indicators in use. The earliest antecedent for DPSIR was the Pressure–State–Response (PSR) framework developed by the Organization for Economic Co-operation and Development (OECD 1994), itself an extension of Rapport and Friend's (1979) stress–response model. The PSR focus on anthropocentric pressures and responses in its evaluation of environmental problems proved problematic, in that it tended to push aside natural variability, as there was no place for it in the PSR classification scheme. The UN Commission on Sustainable Development (1997) attempted to address this problem by expanding upon PSR with a Driving Force–State–Response (DSR) framework. In this model, a driving force is a pressure expanded to include not only the social, political, economic and demographic pressures in the PSR model, but also pressures that resulted from the natural system. Both these methodologies were not able to investigate the underlying reasons of the pressure, while this issue was indeed included in the final formulation of the DPSIR framework.

This conceptual framework helps to organize and structure indicators in the context of a so-called causal chain that links indicators of the environmental driving forces, to pressure indicators, to environmental state indicators, to impact indicators and finally to indicators of societal response (Niemeijer et Groot, 2006).

The use of conceptual frameworks based on causality has important benefits. Such frameworks, through a clearly structured organization of the indicators, enable clear and concise communication to decision-makers (see Figure 10). They help to expose how the information provided by the indicators is related to various processes and how specific policy or management actions can address human-induced environmental problems. Additionally, a uniform approach to indicator reporting helps to link up different but related assessment areas (e.g., transport and environment, agriculture and environment).

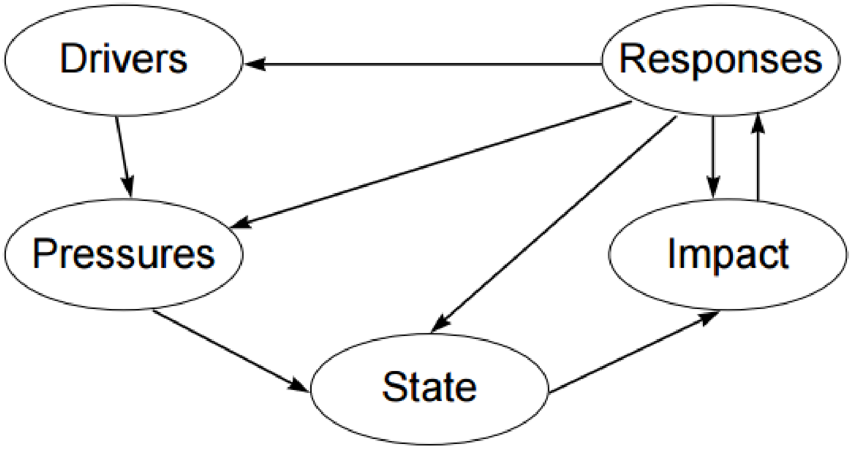


Figure 10: DPSIR Framework (EEA, 1999)

According to the DPSIR framework, **social and economic developments are driving forces** that **exert pressure on the environment**, which leads to **changes in the state of the environment**. In turn, these **changes lead to impacts on human health, ecosystems** and materials that **may elicit a societal response** that feeds back on the driving forces, pressures, or on the state or impacts directly (Smeets and Weterings, 1999). Obviously, the real world is far more complex than can be expressed in simple causal relations in systems analysis. There is arbitrariness in the distinction between the environmental system and the human system.

And, moreover, many of the relationships between the human system and the environmental system are not sufficiently understood or are difficult to capture in a simple framework. Nevertheless, from the policy point of view, there is a need for clear and specific information on:

(i) Driving forces that refer to fundamental social processes;

(ii) the resulting environmental pressures are both the specific human activities that result from driving forces which impact the environment, and the natural processes that have a similar impact on the environment;

(iii) the State of the Environment corresponds to the condition of the environment;

(iv) the Impacts are the results of changes on environmental quality and on human well-being;

(v) the societal Response to these changes in the environment that mostly refer to institutional actions to change the state, acting on the pressures.

The DPSIR framework is useful to describe the relationships between the origins and consequences of environmental problems, but in order to understand their dynamics it is also useful to focus on the links between DPSIR elements. For instance, the relationship between the 'Driving Forces' and the 'Pressures' by economic activities is a function of the eco-efficiency of the technology and related systems in use, with less 'Pressures' coming from more 'Driving Forces' if eco-efficiency is improving. Similarly, the relationship between the Impacts on humans or eco-systems and the 'State' depends on the carrying capacities and thresholds for these systems. Whether society 'Responds' to impacts depends on how these impacts are perceived and evaluated; and the results of 'Responses' on the 'Driving Forces' depends on the effectiveness of the Response (EEA, 1999).

Underlying Concepts

DPSIR and Life-Cycle Assessment (LCA)

According to Khajuria et al. 2012, LCA and DPSIR can be integrated in an effective and dynamic way. In particular, they applied DPSIR framework integrated with the LCA model for municipal solid waste management system in developing countries. On one side the authors used the DPSIR model to evaluate the driving forces of municipal solid waste generation, pressure on flora-fauna, state and impact on environment and responses of government and non-government sector which helps to make new strategies of integrated municipal solid waste management of India. On the other side, they used LCA to assess waste collection, transfer stations, recovery, composting, combustion and landfill. Additionally, they considered within the LCA model energy production, transportation and recycling (see Figure 11).

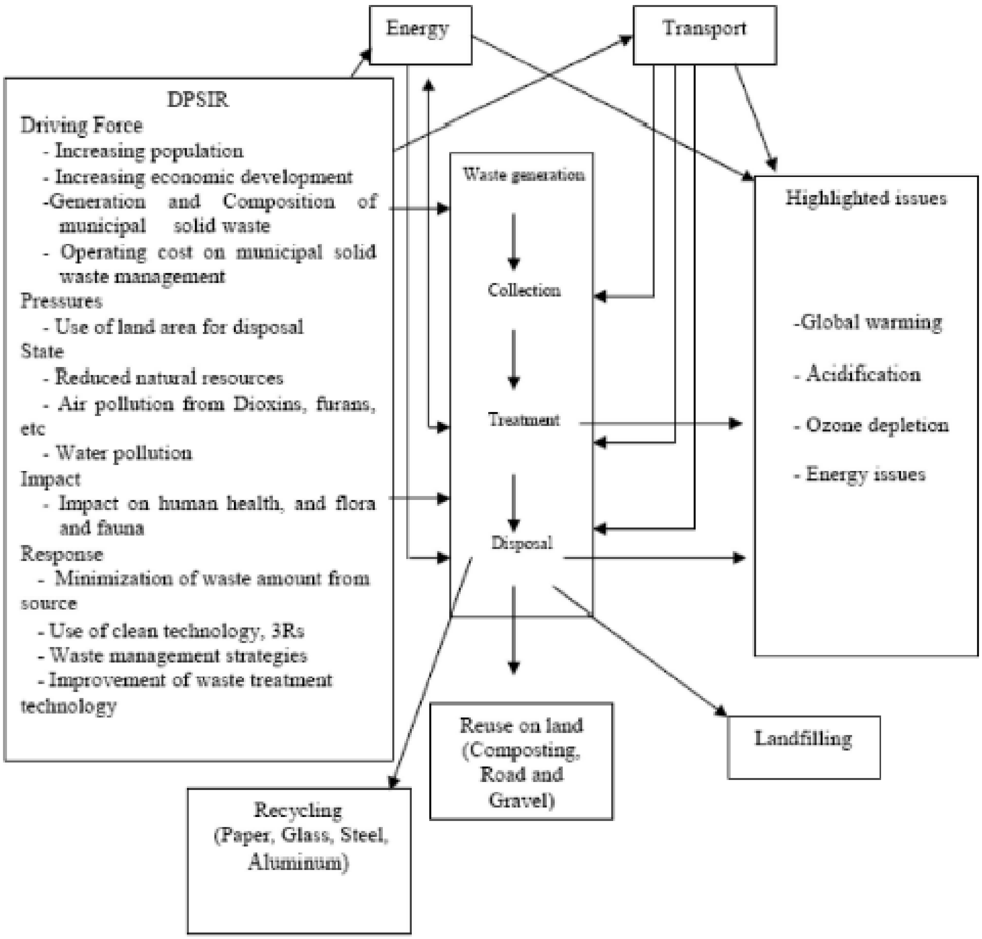


Figure 11: Conceptual Structure of DPSIR-LCA Model (Source: Khajuria et al., 2012)

Using the framework shown in Figure 11 the authors argued that the DPSIR model integrated with LCA model, can be used as a best tool for assessing the environmental impact of municipal solid waste management system. The conceptual model as an environmental tool can be successfully applied in an Integrated Solid Waste Management System (ISWMS) as a decision support tool which helps to develop better and effective strategy of municipal solid waste in developing countries.

DPSIR and Materials Flow Analysis (MFA)

According to Hinterberger et al. 2003, within the internationally harmonised classification systems for environmental indicators, like the pressure-state-response (PSR) framework of the OECD (1994) or the extended Driving Forces-Pressures-State-Impact-Response (DPSIR) system, **material-flow based indicators are part of the pressure indicator group**. These indicators identify and describe socio-economic activities, which cause pressures on the environment. However, their ability to provide information on the actual environmental impacts is very limited.

Assessed Impacts

Economic Assessment

Economic aspects are included in several steps of the DPSIR framework.

In many studies for instance, the term Driving Forces is employed for describing economic sectors that create pressures, such as industry, agriculture and transport (Maxim et al., 2009). Some others describe Driving Forces:

- through mixes of descriptors for the economic sectors, structural features of the economic system, demography and social characteristics (EEA, 2002),
- through demography, developments in economic, socio-political, science and technology domains, cultural and religious factors (MEA, 2003).

These economic aspects are then considered in the process of the evaluation of the impacts. Impacts can be considered as changes in the environmental functions affecting the social, economic and environmental dimensions. These impacts can be then monetized and the results can be presented to the policy and decision makers in economic terms.

Environmental Assessment

Within the DPSIR framework the environmental assessment is described through the State category. State is indeed defined as a result of pressures, that affects the state of the environment; the quality of the various environmental compartments (air, water, soil, etc.) in relation to the functions that these compartments fulfil. The 'state of the environment' is thus the combination of the physical, chemical and biological conditions (Kristensen, 2004). In this sense this framework can provide a comprehensive picture of the environmental responses to certain pressures.

Social Assessment

Concerning the social assessment, according to Maxim et al., 2009, the DPSIR could consider the social sphere at three different levels:

- Impacts: the change of the state of the environment could lead to socio economic impacts that would then need to tailored socio-political responses;
- Responses: responses could be formulated as policy action, initiated by institutions or groups (politicians, managers, consensus groups, etc.) which are directly or indirectly triggered by impacts and which attempt to prevent, eliminate, compensate, reduce or adapt to them and their

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

consequences. The social perception of the existence of relevant Impacts is one of the main reasons for developing political Responses;

- Driving forces: Driving Forces belong to the "whole" context (social, economic, political) that defines an environmental problem. If Responses act on the social component of the Driving forces, social changes are foreseen.

Suitability for URBANWASTE

As mentioned in the previous section, URBANWASTE could use the DPISR framework together with an LCA approach to assess and evaluate the municipal solid waste management system of the pilot cases involved in the project. But overall it has to be concluded that DPSIR is more a causal framework for describing the interactions between society and the environment than an assessment methodology. But it could **support the understanding of the driving forces** and the **pressures that the tourism industry has on the entire waste management system**, thus helping in the development of tailored strategies. Moreover, the possibility to include in the study economic and social factors could contribute to create inclusive, gender sensitive and effective innovative waste management strategies.

Suitability for assessment of changes on hotel level or on municipality level

The DPSIR framework is normally used to assess environmental problems and their relationships with the socio-economic domain in a policy-meaningful way (Maxim et al.,2009). Several examples exist in literature (Maxim et al., 2009, Holman et al. 2005, Khajuria et al. 2012, etc.) regarding the implementation of the DPSIR framework to assess several environmental related issues (biodiversity, climate change) at a regional or a municipal level. For instance, the DPSIR framework allowed the Regional Climate Change Impact and Response Studies in East Anglia and North West England methodology to be built around a consistent structure of linked assessments (Holman et al., 2005), that facilitates the use of the approach in other regions.

Summary / Conclusion

Overall it has to be concluded that DPSIR is more a causal framework for describing the interactions between society and the environment than an assessment methodology. The DPSIR framework has turned out as a useful framework to integrate the **four different dimensions of sustainable development: social, economic, environmental and political**. The possibility to divide the whole analysed process in Driving Forces, Pressures, States, Impact and Responses, allows to develop and bring a smoother and more effective communication to policy makers.

3.9 Eco-Efficiency (EE)

Eco-efficiency (EE) assessment is a quantitative management tool which enables the study of environmental influences of a product system along with its values for the stakeholders. EE was formally developed by the World Business Council for Sustainable Development (WBCSD) as a business concept in a report for the Rio Earth Summit in 1992. It was expressed as *"the ratio of an output divided by an input: the "output" being the* *value of products and services produced by a firm, sector, or the economy as a whole, and the "input" being the sum of environmental pressures generated by the firm, sector or economy*" (OECD, 1998).

The WBCSD defined EE as "*achieved by the delivery of competitively-priced goods and services that satisfy human needs and bring quality of life, while progressively reducing ecological impacts and resource intensity throughout the life-cycle to a level at least in line with the Earth's estimated carrying capacity.*" (OECD, 1998).

Later the European Environmental Agency (EEA) also, but slightly differently, defined EE as "*a concept and strategy enabling sufficient delinking of the use of nature from economic activity, needed to meet human needs (welfare), to keep it within carrying capacities; and to allow equitable access to, and use of the environment, by current and future generations. Eco-efficiency is only a relative measure, a necessary, but not sufficient condition for achieving sustainability, as, in some case, absolute reductions in some environmental pressures are needed*." (EEA, 1998).

Whereas the original definition by WBCSD is absolute, the definition by EEA is relative and does not say which production method is good or bad for the environment or economy, only whether it is better or worse; i.e. products/processes are compared to each other and not to the carrying capacity of the Earth (Huppes & Ishikawa, 2005).

Defined as the **ratio between economic performance and environmental influence**, EE can either be written as (Huppes & Ishikawa, 2005: Zhao & Zhao, 2011: Yang et al., 2014: Yuan et al, 2016);

$$\mathbf{Eco-efficiency} = \frac{\mathbf{economic\ performance}}{\mathbf{environmental\ influence}}$$

or as (Huppes & Ishikawa, 2005: Yuan et al., 2016);

$$\mathbf{Eco-efficiency} = \frac{\mathbf{environmental\ influence}}{\mathbf{economic\ performance}}$$

The higher the value of the EE indicator, the more beneficial the effect.

EE can also be measured as

- environmental intensity and environmental productivity in the realm of value creation, or
- environmental improvement cost and environmental cost-effectiveness in the realm of environmental improvement measures.

This results for the four basic types is shown in Table 4. The choice of which EE type to apply depends on the context, and whether the managerial focus is on the creation of maximum value with minimum environmental impact or is dedicated to concrete environmental improvements.

Table 4: Four basic types of eco-efficiency (Source: Huppes & Ishikawa, 2005)

	Product or production primary	***Environmental improvement primary***
Economy divided by environment	Production value per unit of environmental impact, or *environmental productivity*	Cost per unit of environmental improvement or *environmental improvement cost*
Environment divided by economy	Environmental impact per unit of production value, or *environmental intensity*	Environmental improvement per unit of cost, or *environmental cost-effectiveness*

In businesses EE is mainly understood as an indicator of performance, or as a strategic tool for sustainable business development combining economic and environmental performance of a specific system. The practical uses of EE in industry has been classified as (Koskela & Vehmas, 2012)

- a **measure for resource productivity**; more value added with less use of resources;
- a part of the management strategy as an **indicator for continuous improvement**, life cycle perspective, innovative procedures and/or improving economic and environmental performance;
- a measure for improvement from bringing a life cycle perspective into environmental management, reducing the energy intensity of the production, enhancing recyclability and/or maximizing the use of renewable raw materials.

EE should be seen not as a specific methodology but as an overarching general concept with variants residing under this umbrella (Huppes & Ishikawa, 2005).

Underlying Concepts

EE and Life-Cycle Assessment (LCA)

For policy and strategy development **EE is mostly used in relation with LCA to make a linking of economic costs and environmental impacts in a systematic manner** (Jancovic et al., 2011), thus providing the necessary values to support a decision-making process (Vercalsteren et al., 2010).

However, the resulting EE indicators are highly dependent on how the chosen LCA approach is operated. This is illustrated by Frischknecht (2010) in a study comparing two ISO-compliant LCA approaches. The results indicate that EE is very much influenced by both values, policy and cultural perspectives:

- *The recycled content (or cut-off) approach* accounting for environmental impacts at the time they occur meaning that if a product is made of primary material, the environmental impacts of primary production are attributed to this product: This approach resulted in higher EE of recycling compared to primary production.
- *The end of life recycling approach* according to which it is assumed that used materials are probably recycled in the future to avoid primary production: This approach resulted in higher EE of primary production compared to secondary production.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

These different results imply that clear statements regarding preferences, choice and application of LCA concept as well as transparency in the data inventory is crucial to ensure consistency between the management strategy and the used indicators (Frischknecht, 2010).

EE and Materials Flow Analysis (MFA)

MFA is based on the flows and stocks of material in a defined system by means of the law of mass conservation and material balances. **Though EE is mostly attributed to LCA, EE indicators can also be derived from a combination of MFA factors and economic indexes. Economic data from the defined system is used to derive the EE indicators for material and energy inputs, consumption and emission outputs.**

Several studies and model developments have been conducted with EE based on MFA (Wang et al., 2015: Zhao & Zhao, 2011: Walker et al., 2009). For example, Wang et al. (2015) constructed a systematic material flow model integrating the production, usage and waste management in cement and concrete production on a national level in China. The model was developed not only for assessment of status and potential improvement level of EE indicators, but also as a tool for monitoring and predicting the resource consumption and emissions of the industry, i.e. making implications for further policymaking.

However, from an environmental point of view maximization of EE indictors cannot stand alone in conjunction with the MFA in a strategy towards greater sustainability, because MFA relates to material flow rather than environmental impacts. Furthermore, the realization of municipal policies and decisions are often time consuming processes. An increase in consumption due to population growth may reduce the effect of a planned improvement of EE indicators, unless there is an ongoing adjustment to reflect the development (Walker et al., 2009).

Assessed Impacts

Economic Assessment

The concept of EE in economic assessment is the link between monetary and physical environmental management accounting for decision-making in a systematic manner. Establishing a sustainable interrelation between environmental and economic goals can reduce overall costs (Jankovic et al., 2011). However, a change in practice in one part of the system could possibly cause secondary economic effects elsewhere in the system. Therefore, the analysed system should be thoroughly described to provide true and fair indicators for economic assessment.

Environmental Assessment

Like economic performance, environmental influence is an integral part of EE and can be assessed on the basis of this method. However, with reference to the model shown in Table 4, the environmental assessment will depend on the chosen type of EE, whether it relates to value creation or environmental improvement measures (Huppes & Ishikawa, 2005).

It has been criticized that relative EE indicators (based on the EEA definition) can be calculated as positive even if development becomes more unsustainable, by an increase in the use of ecological resources to meet human needs. EE would increase even with higher environmental impact as long as the economic value increases even faster (Mickwitz et al., 2006).

Another critical issue to EE is the risk of incomparability in measurement units. It should be assured that measurements attribution always refer to the boundary where the criterion is assessed (Sun & Pratt, 2014).

Social Assessment

EE has been widely used in the evaluation of regional economic systems, industry, enterprises, production and services. The definition of EE stresses both environmental and economic benefits of production and services (Zhao & Zhao, 2011). Zhao & Zhao (2011) have also alleged that EE effectively embodies the target of social sustainable development.

Nonetheless, it has to be acknowledged that the materialization of any structural change or technical innovation is subject to the acceptance of society (Walker et al., 2009). Social acceptance is multidimensional and is dependent on the prevalence of a number of factors related to both policy, market and community (Sovacool & Gross, 2015).

Although **social aspects** are an essential part of sustainable development and also EE in a broad sense, they **are not embedded in the concept of EE in practical applications**. On municipal level social aspects are part of the authority's responsibility portfolio and must either be included, or the concept of EE must be limited to the efficiency with which ecological resources are used to provide economic welfare instead of to "meet human needs" (Mickwitz et al., 2006).

Suitability for URBANWASTE

EE indicators have proven suitable for decision makers to evaluate both tourism and waste processes on various scales, based on either flow models that delivers information on economic performance and environmental pressure, or by means of LCA combined with economic data. Assessing different scenarios by means of EE makes it possible to demonstrate principle pathways, combine parameters and draw more complex pictures of future scenarios (Kytziaa et al., 2011).

The **EE method is highly flexible and adaptable, and relatively simple to communicate which makes it an appropriate tool to meet the objectives of URBANWASTE**, both in terms of fostering and structuring a stakeholder participatory framework for policy-making in waste management and for application and integration of an urban metabolic approach for waste policies, although the use in relation to social assessment is somewhat unclear.

Suitability for assessment of changes on hotel level or on municipality level

EE is a flexible and relatively simple tool for assessing the environment influence against economic performance. It can be adapted and applied in various types and scales of systems, both at company level, within sectors, at municipal level and other defined systems. It can also be used across sectors and stakeholders and help to build a consensus around policy, objectives, decisions and achievement of targets in both waste management and in the integral product framework of tourist destinations (Jankovic et al., 2011).

Cooperation with customers is essential for achieving environmental goals in the hotel industry, given that resource consumption by operations activities and by customers directly has been shown to affect environmental sustainability in hotel operation. Research addressing consumer goods has shown that the practices stipulated in the eco-certification guidelines can bring both operational and pricing advantages through improvements in process and product quality (Zhang et al., 2014).

However, divergent interests exist between the hotel industry and the municipality since the perception of economic performance relative to environmental influence has a different meaning depending on whether the objective is profit maximization or "more environmental value for money". For tourism destination planners and managers, the challenge is to select and implement EE strategies that appeal to the tastes and interests of tourists while meeting the needs of community stakeholders (Kelly et al., 2007).

Figure 11 shows the principle of how different scenarios can be compared on both environmental impact and economic costs, this providing decision-makers with an informed basis to prioritize the more appropriate option in terms of both environment and economy. The diagram is reprinted from a study by Yang et al. (2015) who explored the EE analysis for municipal solid waste management to optimize waste disposal systems with the aim of reducing carbon emission and improving cost effectiveness per unit investment cost during its life cycle processing. The model integrates the environmental impact and economic costs of each process, from collection to treatment, into the entire disposal system and then analyses the contribution of each process individually to determine the EE optimization measure. Although from a study in a municipal waste management system the model shown in the diagram could equally well represent an assessment and decision-making process in another sector, e.g. the hotel industry.

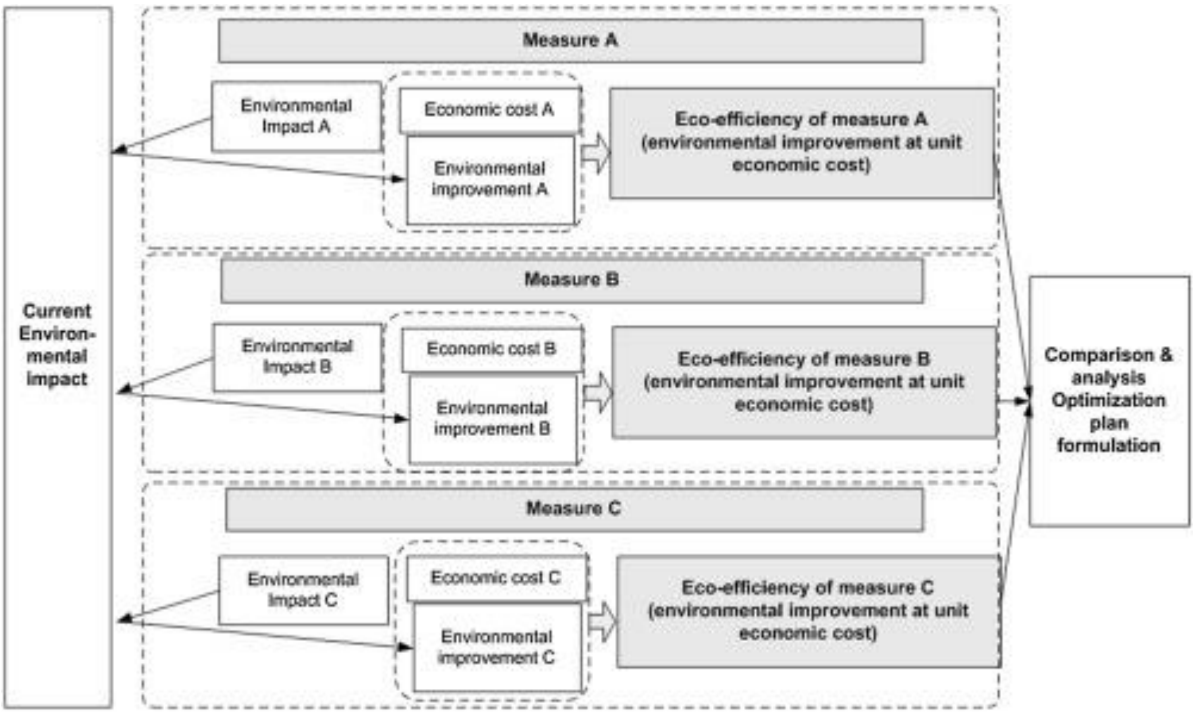


Table 5: Diagram for the optimization concept of municipal waste systems (Source: Yang et al. 2015)

Central to the EE concept is the identification of resource consuming and waste generating processes at different scales, this makes the method highly flexible to be applied both at municipal level, in single organisations and other defined systems. It also enables identification of interfaces and interconnections between different processes and across sector borders (Byström, 2012).

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

The properties make EE an appropriate tool to meet the objectives of the URBANWASTE project. Furthermore, EE is in line with the environmental and waste policy in the EU and can be seen as a means for decoupling economic development from environmental impact and waste generation.

From a municipal solid waste management perspective EE is a valuable tool for operationalising the waste hierarchy, and also a concept that encourages focus on resource management with process optimisation, waste utilization and recovery of materials and energy.

From a commercial point of view, the use of EE can also offer valuable benefits to image building and competitive advantages, as environmental optimisation of processes are not only beneficial from a sustainability point of view but also makes more profitable business models.

However, this divergence between the commercial and non-commercial perception of economic performance in relation to environmental influence as being either profit maximization or "more environmental value for money", together with other cultural and value-borne differences, means that EE is not immediately suitable as a standard of reference between different systems. EE is an ambiguous term interpreted in various ways, depending on the current situation and should predominantly be seen as a value-based management tool. This should be acknowledged and articulated in any practical application of EE.

Summary / conclusions

EE is a relatively **simple tool for assessing environmental influences against economic performance**. Central to the EE concept is the **identification of resource consuming and waste generating processes at different scales**, this makes the method highly flexible to be applied both at municipal level, in single organisations and other defined systems. It also enables identification of interfaces and interconnections between different processes and across sector borders (Byström, 2012).

The properties make EE an **appropriate tool to meet the objectives of the URBANWASTE project**. Furthermore, **EE is in line with the environmental and waste policy in the EU and can be seen as a means for decoupling economic development from environmental impact and waste generation**.

From a municipal solid waste management perspective EE is a valuable tool for operationalising the waste hierarchy, and also a concept that encourages focus on resource management with process optimisation, waste utilization and recovery of materials and energy.

From a commercial point of view, the use of EE can also offer valuable benefits to image building and competitive advantages, as environmental optimisation of processes are not only beneficial from a sustainability point of view but also makes more profitable business models.

However, this divergence between the commercial and non-commercial perception of economic performance in relation to environmental influence as being either profit maximization or "more environmental value for money", together with other cultural and value-borne differences, means that EE is not immediately suitable as a standard of reference between different systems. EE is an ambiguous term interpreted in various ways, depending on the current situation and should predominantly be seen as a value-based management tool. This should be acknowledged and articulated in any practical application of EE.

3.10 Ecological Footprint (EF)

Underlying Concepts

Ecological Footprint (EF) **analysis is not based on LCA or MFA**, but, however, dealing with some of the critiques, namely that MFA can hardly be used to evaluate the sustainability of an urban system (Zhang, 2013, p. 467; Zhang et al., 2015, p. 11255). Also, **EF can be an indicator in LCA**, evaluating the impact on land of a certain product or process. **Also other Footprint analysis as e.g. Carbon Footprint or Water Footprint can be indicators within and LCA.**

The concept of Ecological Footprint was introduced in the 1990s by Wackernagel and Rees (1995). EF measures the **land area necessary to sustain humankind's, a city's or also a person's resource consumption and waste discharge**. The advantages of EF are that **it combines socioeconomic development demands with ecological capacity** and that, as mentioned above, can therewith reveal ecologically unsustainable situations. A problem of the concept is though, that land is not considered to provide multiple functions, and that, because of incomplete descriptions of resource provision (and waste discharge) by the natural system, it underestimates human impact (Zhang, 2013). Via an expert survey, Wiedmann and Barrett (2010) found that EF

"*(a) is seen as a strong communication tool,*

(b) has a limited role within a policy context,

(c) is limited in scope,

(d) should be closer aligned to the UN System of Environmental and Economic Accounting and

(e) is most useful as part of a basket of indicators." (Wiedmann & Barrett, 2010)

According to Wackernagel and Rees (1995) the ecological footprint is "*the **total area of productive land** and **water ecosystems required** to **produce the resources that the population consumes, and assimilate the wastes that the population produces**, wherever on earth the land and water may be located using prevailing technology*." This means on the one hand all direct land-use is considered, like area for mining, land for growing of plants, etc., and on the other hand the indirect land-use for instance needed for CO_2 sequestration is assessed. In this way fossil fuels are accounted for in their emissions. The advantage of the EF is that all inputs and outputs are aggregated to just one aspect – area of land (Unger et al. (2008)).

Assessed Impacts

Social Assessment

There is **no social assessment** in the original ecological footprint. Additional indicators should be applied in parallel to EF.

Economic Assessment

There **is no economic assessment** in the original ecological footprint. Additional indicators should be applied in parallel to EF.

Environmental Assessment

The main purpose of the EF is the **assessment of human consumption's impact on the environment**, i.e. the use of natural capital and more specific **the use of global hectares land (gha)**. This is typically juxtaposed a

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

measurement of the biocapacity (BC) of a region and therewith gives the possibility to measure a potential overshoot in resource use.

However, additional indicators which e.g. consider biodiversity should be used to avoid transformation of wilderness and areas with high biodiversity but low productivity into monoculture agricultural areas which have higher biocapacity in terms of productivity. Adapted EF methodologies exist to cope with this.

Suitability for URBANWASTE

Waste is part of the EF as **the assessment includes the area of nature needed to cope with human waste**. EF could thereby be used to estimate the biocapacity needed to treat the city's / region's waste. The concept is also very illustrative (see Figure 12) and would be appropriate for dissemination and communication campaigns but also for more visions or strategies for sustainable urban development (Kuzyk & Rockley, 2014). As written above, it is most useful in combination with other indicators which consider other aspects of sustainable development.

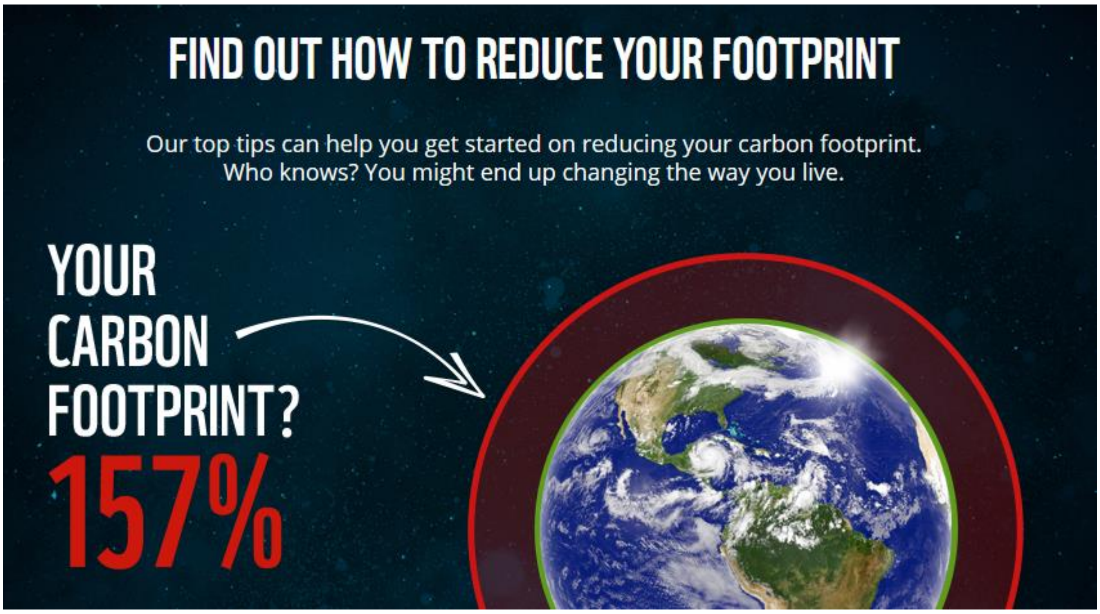


Figure 12: Illustration from WWF on carbon footprint, exceeding the earth capacity to cope with CO_2 which is part of WWF's 'Ecological footprint of consumption' (WWF, 2016)

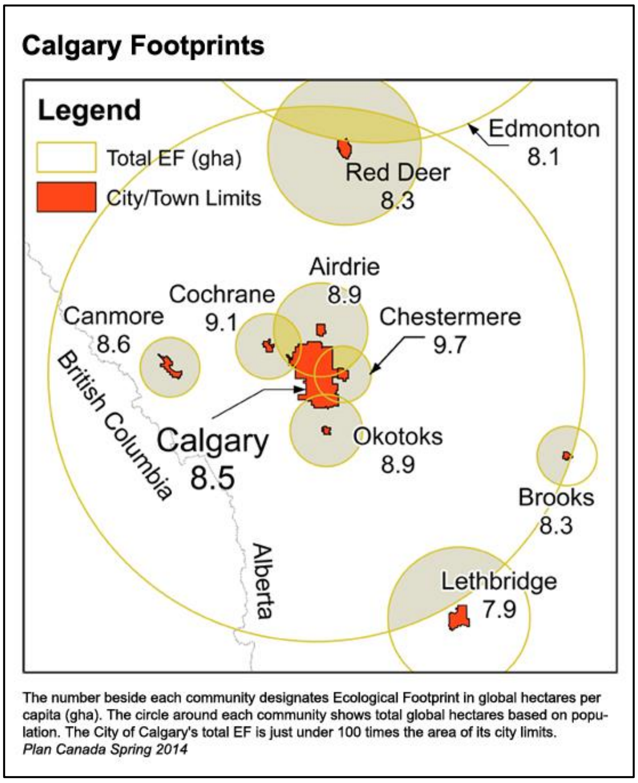


Figure 13: Application of EF in Calgary and surrounding towns (Kuzyk & Rockley, 2014)

Suitability for assessment of changes on hotel level or on municipality level

EF is typically used on a global scale, while there is some criticism to apply it at region or city scale. This regards mainly the fact that EF argues for self-sufficiency, which seems unrealistic for large cities which naturally rely on large hinterlands, and does not consider the benefits of trade. Besides that**, footprints can be calculated on every scale, even on hotel or individual scale**.

3.11 Economic Input-Output (EIO)

According to the Carnegie Mellon University (http://www.eiolca.net/Method/index.html) the "Economic Input-Output Life Cycle Assessment (EIO-LCA) method **estimates the materials and energy resources** required for, and the **environmental emissions resulting from activities in our economy**. It is one technique for performing a life cycle assessment, an evaluation of the environmental impacts of a product or process over its entire life cycle. The **method uses information about industry transactions** - **purchases of materials by one industry from other industries**, and the **information about direct environmental emissions of industries, to estimate the total emissions throughout the supply chain**."

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Underlying Concepts

Economic Input-Output Life Cycle Assessment (EIO-LCA) is an approach that applies a **mathematically defined procedure** using economic and environmental data **to determine the effect of changing the output of a single sector**. The method can be applied to any economy defined by the transactions between sectors. Each application of the method defines an EIO-LCA model.

This method uses **data that are aggregated at sector level** in order **to quantify environmental impacts that can be allocated to the sector**. Basically it is calculated how much each sector purchases from other sectors in producing its output. For this method, **sector-level averages** are used, but they **may or may not be representative.**

Hendrickson et al. (2006) state, that the "*economic input-output analysis was developed by the Nobel Prize-winning economist Wassily Leontief. It quantifies the interrelationships among sectors of an economic system, enabling identification of direct and indirect economic inputs of purchases. This concept was extended by including data about environmental and energy analysis from each sector to account for supply chain environmental implications of economic activity.*"

Economic input-output (EIO) models represent the monetary transactions between industry sectors in mathematical form. EIO models indicate what goods or services (or output of an industry) are consumed by other industries (or used as input). As an example, consider the industry sector that produces automobiles. Inputs to the automobile manufacturing industry sector include the outputs from the industry sectors that produce sheet metal, plate glass windshields, tires, carpeting, as well as computers (for designing the cars), electricity (to operate the facilities), etc. In turn, the sheet metal, plate glass windshield tire, etc. industry sectors require inputs for their operations that are outputs of other sectors, and so on. Each of these requirements for goods or services between industry sectors is identified in an EIO model. The traditional economic input-output model indicating economic transactions between industries can be appended with information on emissions to the environment. In effect, this creates an additional column representing "the environment" sector, and the value in each row represents the pollutant "output" from an industry sector that is "input" to "the environment" sector.[6]

Hendrickson et al. (2006) describe the difference between a "traditional" process-based LCA and the EIO-LCA approach (see Table 6).

[6] http://www.eiolca.net/Method/LCAapproaches.html

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Table 6: Methodological differences between process-based LCA and EIO-LCA (Hendrickson et al. (2006))

	Process-Based LCA	**EIO-LCA**
Advantages	results are detailed, process specific	results are economy-wide, comprehensive assessments
	allows for specific product comparisons	allows for systems-level comparisons
	identifies areas for process improvements, weak point analysis	uses publicly available, reproducible results
	provides for future product development assessments	provides for future product development assessments
		provides information on every commodity in the economy
Disadvantages	setting system boundary is subjective	product assessments contain aggregate data
	tend to be time intensive and costly	process assessments difficult
	difficult to apply to new process design	must link monetary values with physical units
	use proprietary data	imports treated as products created within economic boundaries
	cannot be replicated if confidential data are used	availability of data for complete environmental effects
	uncertainty in data	difficult to apply to an open economy (with substantial non-comparable imports)
		uncertainty in data

Assessed Impacts

Generally spoken, the idea of this mathematical model is to calculate demands from sectors. The units used are usually monetary units. On the one hand these are monetary amounts purchased by specific sectors to make their product (what the "buy" from other sectors to produce their product). If data are available on a particular emissions release (or other attribute of interest) from each sector of the economy, then it is possible to link this information with the economic (monetary data), yet this is displaying information at whole sector level, using a lot of aggregated information.

Suitability for URBANWASTE

EIOLCA provides an LCA method, combined with economic data. In general terms, the use of EIOLCA does not present special advantages, since the method it offers is also integrated in other tools. At the same time, **this method presents less transparency, and requires more expertise for use and interpretation of results**.

The method has several problems for implementation in URBANWASTE. The method uses the calculation of **whole sectors, this means on the one hand aggregates are used**, that are presenting "averages" within the sector. The more diverse a sector is, the less representative the results might be for specific parts of the sector. Tourism on the one hand might be considered uniform in the products and services delivered, **but results cannot be derived at local level (cities) or at lower levels (accommodation types)**. Secondly, it is **difficult to allocate the environmental impacts to the EIO models**. Therefore, the **method is considered not suitable for the project**, especially due to the fact, that it is a whole sector analysis and that both the inventory data and emissions / impacts are likely to be missing at pilot case (city) level.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Suitability for assessment of changes on hotel level or on municipality level

Method is considered not suitable due to the reasons stated above.

3.12 Material Flow Analysis (MFA) and Energy Flow Analysis (EFA)

"Material Flow Analysis" (MFA) and "Energy Flow Analysis" (EFA) are two "*accounting and assessment methods for urban metabolism [that] are based on an analysis of material and energy flows*" (Zhang, 2013, p. 465; Zhang, Yang, & Yu, 2015, p. 11249). **MFA** or "mass balance" has the **goal to provide a system level understanding** of how a city, region or nation functions (Holmes & Pincetl, 2012, p. 8). **EFA or "energy balance" as a modification of the MFA framework** was developed to provide a more detailed understanding of urban metabolic processes (Zhang, 2013, p. 466).

Underlying Concepts

Material- (MFA) and Energy-Flow-Analysis (EFA) are – as their name suggests – based on MFA and account for material or energy flows. **MFA traces the input, storage, transformation, and output processes** and it allows **following the material flows throughout the life cycle within an urban system**, **based on the physical principle that matter can neither be created nor destroyed** (Holmes & Pincetl, 2012, p. 8). MFA also allows for comparisons across cities and inputs (Pincetl, 2012). EFA allows integrating material flows with different measurement units (Zhang, 2013, p. 466). "*Emergy*", a further development of MFA, provides a method for studying the energetic flows in a socio-economic system.

Assessed Impacts

A **main critique of MFA** and similar approaches is, that **they can hardly be used to evaluate the sustainability of an urban system as they are not (directly) relating to the impact of material (and non-material) flows** (Zhang, 2013, p. 467; Zhang et al., 2015, p. 11255). According to these shortcomings the **method does not allow an integrated assessment of social, economic and environmental impacts**.

As the method is a physical assessment of material or energy use, neither social, nor economic or environmental assessment is directly conducted. However, **based on the accounted material or energy flows, an economic or environmental assessment might be possible by converting the accounted masses or energy into respective parameters** (prices, CO_2 emissions etc.).

Suitability for URBANWASTE

The reviewed method, particularly MFA, is **very suitable to address questions of waste management** or to deal with problems of waste management, as for instance a case study of Oahu, Hawaii (Eckelman & Chertow, 2009) shows. However, the method does not allow a comprehensive assessment of social, economic and environmental impacts; n**either does the method provide a particular solution to deal with tourism or touristic waste**.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Therefore, the suitability of the method for URBANWASTE depends on the particular purpose or aim that has to be fulfilled. If the purpose is limited to questions of waste management in tourist regions, related to mass management, then MFA is very suitable; if, however, a more integrated perspective should be applied, assessing questions of sustainability, then the method is of limited suitability for URBANWASTE.

It can be stated, that MFA aims at displaying material flows in a system allowing for considering also inputs / outputs and stocks of a system. Especially in terms of waste management it is important to map the flows of different waste streams in an (end of life) system. This serves as entry point for subsequent sustainability assessments.

Suitability for assessment of changes on hotel level or on municipality level

MFA/EFA is suitable on different scales, but it can be questioned what contribution the application of the method on hotel level could make for the questions of the URBANWASTE project.

3.13 Environmental Impact Assessment (EIA)

Environmental assessment according to is a procedure that ensures that the environmental implications of decisions are taken into account before the decisions are made[7]. Environmental assessment can be undertaken for individual projects, such as a dam, motorway, airport or factory, on the basis of Directive 2011/92/EU (known as 'Environmental Impact Assessment' – EIA Directive) or for public plans or programmes on the basis of Directive 2001/42/EC (known as 'Strategic Environmental Assessment' – SEA Directive). The common principle of both Directives is to ensure that plans, programmes and projects likely to have significant effects on the environment are made subject to an environmental assessment, prior to their approval or authorisation. Consultation with the public is a key feature of environmental assessment procedures.

Underlying Concepts

EIA is a procedure that must be followed for certain types of development before they are granted development consent. The requirement for EIA in the EU comes from the EIA Directive 85/33/EEC as amended three times and codified by Directive 2011/92/EU of 13 December 2011. Directive 2011/92/EU has been amended in 2014 by Directive 2014/52/EU.

Project in the sense of the EIA Directive means: the execution of construction works or of other installations or schemes, as well as other interventions in the natural surroundings and landscape including those involving the extraction of mineral resources.

The procedure requires the developer to compile an **Environmental Statement** (ES) describing the likely significant effects of the development on the environment and proposed mitigation measures.

The ES must be circulated to statutory consultation bodies and made available to the public for comment. Its contents, together with any comments, must be taken into account by the competent authority (e.g. local planning authority) before it may grant consent. In essence, the process involves an analysis of the likely effects

[7] *http://ec.europa.eu/environment/eia/*

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

on the environment, recording those effects in a report, undertaking a **public consultation** exercise on the report, taking into account the comments and the report when making the final decision and informing the public about that decision afterwards. The EIA Directive outlines which project categories shall be made subject to an EIA, which procedure shall be followed and the content of the assessment.

The particular components, stages and activities of an EIA process and the application of the main stages is a basic standard of good practice. Typically, the EIA process begins with screening to ensure time and resources are directed at the proposals that matter environmentally and ends with some form of follow-up on the implementation of the decisions and actions taken as a result of an EIA report (see Figure 14).

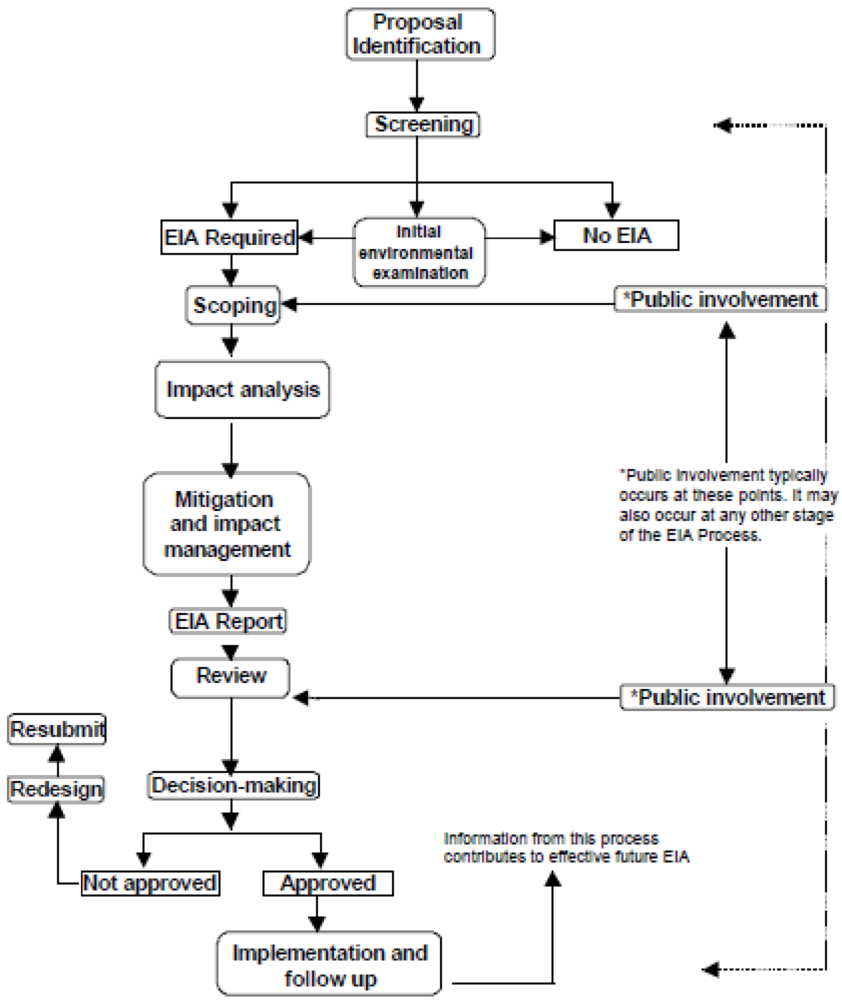


Figure 14: Flow chart of an environmental assessment process

EIA is a procedural tool for ensuring that the likely effects of a new development on the environment are fully understood and taken into account before the development is allowed to go ahead.

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

As such, it is used to:

- Identify potential environmental impacts;
- Examine the significance of environmental implications;
- Assess whether impacts can be mitigated;
- Recommend preventive and corrective mitigating measures;
- Inform decision makers and concerned parties about the environmental implications.

Assessed Impacts

According to Article 3 of Directive 2011/92/EU by Directive 2014/52/EU "1. The environmental impact assessment shall identify, describe and assess in an appropriate manner, in the light of each individual case, the direct and indirect significant effects of a project on the following factors:

(a) population and human health;

(b) biodiversity, with particular attention to species and habitats protected under Directive 92/43/EEC and Directive 2009/147/EC;

(c) land, soil, water, air and climate;

(d) material assets, cultural heritage and the landscape;

(e) the interaction between the factors referred to in points (a) to (d).

Suitability for URBANWASTE

Environmental Impact Assessment **sets a methodological framework for assessing environmental impacts of a project**. Therefore, as a tool for measuring and monitoring environmental impacts of projects, it can be of use. But per definition EIA "projects" refer to much bigger issues than concerned within URBANWASTE. Additionally, real projects like the construction of a motorway are main topics for EIA. Within URBANWASTE the tourist sector as service sector is the main issue. As a full EIA process includes e.g. participation and is very extensive the **approach seems too broad in the sense of URBANWASTE**. Nevertheless, the general concept can be used. EIA sets a methodological framework for assessing environmental impacts of a project. Therefore, it could be a useful tool for measuring and monitoring the environmental impacts.

Suitability for assessment of changes on hotel level or on municipality level

EIA is a methodological framework and foreseen for much bigger projects than the hotel level. On municipality level SEA would be a more appropriate methodology. Therefore, it can be concluded that EIA is not suitable for URBANWASTE.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

3.14 Environmental Profit & Loss (EP&L)

Underlying Concepts

An environmental profit and loss account (E P&L) is a **company's monetary valuation and analysis of its environmental impacts including its business operations and its supply chain from cradle-to-gate** (WBCSD, 2011).

Assessed Impacts

Environmental profit and loss are for obvious reasons focused on specific entities impact and can be seen as narrower methodological part than LCA - derived partly from Ecosystem services valuation and LCA might be used here as a wider concept combined with this specific technique as it can be found on various companies business reviews or consultancy companies proposed studies for clients (WBCSD, 2011; PUMA, 2016; Kering on EP&L, 2016).

Suitability for URBANWASTE

Relevance with URBANWASTE project as relevant method is only given if all data from companies about everything linked from tourism business as used materials, amount of tourist waste, impacts of fuel emissions might be necessary to be calculated as this is very precise model requiring methodology. It shall be used for distinct companies to measure ecological footprint and gains on sacrifice of the former. It means that **the use of this method is doubtful as then data from each singular company unit must be used in model performance** and established boundaries of the "LCA impact" that is hard task if recreation zones have many tourists and multiple / many companies working. The method shall be advised on later stages of projects when main strategies are established and single unit analyses might be performed to elaborate tactic plans of singular entities.

The methodology was developed within the company PUMA and aims at internalizing externalities and monetizing the cost of business to nature by accounting for the ecosystem services a business depends on to operate in addition to the cost of direct and indirect negative impacts on the environment. This aims at improving the visibility, where in the value chain of a product negative environmental impacts occur. As this is a very extensive task, and also tourism as service is difficult to assess, it is recommended not to use this approach.

Suitability for assessment of changes on hotel level or on municipality level

Municipality level is hardly to be approached and in general the EP&L is better for products neither services nor a group of services and interactions like the tourism is. The EP&L analysis provides monitoring the footprint of the company's operations and identifying new opportunities to enhance the sustainability of a company's products - product/impact in metric values. Therefore, project URBANWASTE performance **shall not use this methodology in full manner**, however some scientific idioms are useful for defining monetary environmental expenditures in specific cases.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

3.15 Industrial Symbiosis (IS)

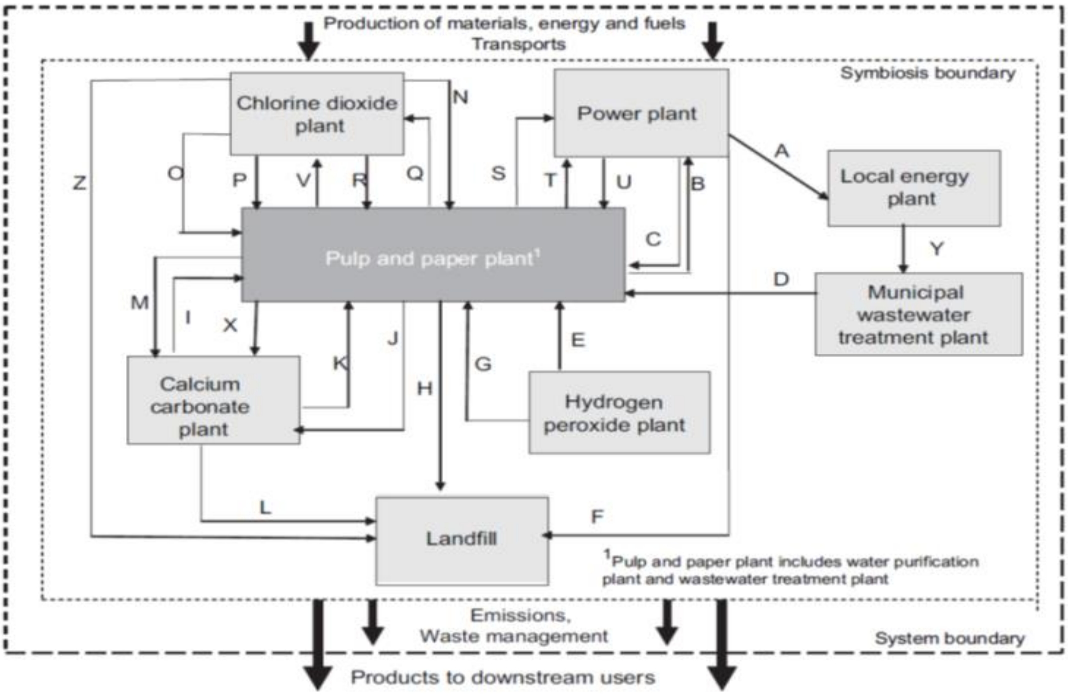
The concept of industrial symbiosis (IS) branches from industrial ecology (I-Ecol), a discipline that arose around 1990 (Yazan et al., 2016). In analogy to ecological systems, the wastes from some is converted into useful products and energy for others (Smith et al., 2015: 317). Rooted in I-Ecol, IS is **based on the principles of system approach, mass balance, circular economy and life cycle thinking**.

To define IS, several authors refer to Chertow (2000; cited by Paquin et al., 2015: 96): "*An inter-firm collaboration involving the exchange and reprocessing of wastes and other excess resources from one firm into valuable inputs for another*". Exchanged are solid materials, energy, water, and a variety of smaller amounts of other by-products. By 'roundput' of both material and energy between companies, as opposed to actions of individual firms, businesses aim at improving their environmental performance and profit margins. Concomitantly, the efficiency of material and energy use of the whole system increases. These two aims are interwoven, Paquin et al. (2015: 97) observe an "*economically-driven environmental focus of I-Sym*". This can be seen for instance in cases where growing costs of environmental inappropriate waste disposal are an ecomonic driver of I-Sym (Ferrer et al., 2012; Paquin et al., 2015). Referring to the UK, Paquin et al. (2015: 95) attribute "*escalating costs […] to the enforcement of environmental regulations such as increasing landfill taxes and additional environmental regulations*". Ren et al. (2016) consider I-Sym something that is being designed to make an industrial system more sustainable, environmentally and economically.

Geographical proximity of firms is key to opportunities for synergistic by-products exchange and, hence, a major concept in I-Sym. Most often proximity is considered economically beneficial due to savings on transport costs, but that is also environmentally beneficial by savings on emissions. In some cases, proximity is even required by the nature of the by-product to be exchanged. Heat for instance, can be transferred over a maximum distance of about 10 to maximum 20 km (Korhonen, 2001: 373). An optimal scale for IS is hardly to define (Yazan et al., 2016: 537) but it can nevertheless be commented that it works best on local or, at the most, sub-regional scale.

A frequently studied showcase of industrial symbiosis is the **eco-industrial park** Kalundborg, Denmark. It is described by Jacobsen (2006: 241) as "*a complex web of interactions among five collocated companies and the local community. The symbiotic companies include a power-plant, an oil-refinery, a biotech and pharmaceutical company, a producer of plasterboard and a soil remediation company*". The 'web of interactions' in I-Sym is not by definition formed by such large-scale, capital intensive plants. The shoe-making cluster in Brazil in a case study of I-Sym by Ferrer et al. (2012) is "*characterised by a large number of small manufacturing firms*" (op. cit: p. 142).

Figure 15 presents a scheme of flows in the Kymi eco-industrial park of forest industry in Finland as an example of IS (Sokka et al., 2011: 287). It distinguishes between no less than 26 different types of energy-carriers, solid waste materials, water and other by-products that are 'put round'. This example illustrates that the by-product exchange that is essential of IS is more common in multi-industry than in single-industry clusters (Ferrer et al., 2012) due to the importance of complementarity of types of by-product for exchange. It also shows that IS in practice is not by definition a closed system based on mutual proximity within clear boundaries. The IS in this eco-industrial park is extended with boundary-crossing upstream suppliers and downstream users.



A District heat and electricity
B Sewage sludge
C Wastewater
D Sewage sludge
E Wastewater
F Ash
G Hydrogen peroxide (H_2O_2)
H Miscellaneous inert waste
I Wastewater
J Water
K Calcium carbonate (CaCO3)
L Miscellaneous waste
M Carbon dioxide (CO_2)
N Chlorine (Cl_2)
O Wastewater
P Chlorine dioxide (ClO_2)
Q Water
R Sodium hydroxide (NaOH)
S Sodium hydroxide (NaOH)
T Biomaterials used as fuel
U Steam, electricity, heat
V Steam
X Steam and electricity
Y Electricity
Z Waste

Figure 15: Energy and material flows in the Kymi eco-industrial park.

Underlying Concepts

In essence, IS **is not by itself a methodology to measure quantity, qualitative composition or treatment of waste materials**. This would make it of little relevance to URBANWASTE. However, it offers opportunities to assess the performance of symbiosis in industrial clusters from both the environmental and economic point of view. In fact, quite a few studies analyse the supply and demand (use) sides of flows of waste products in both quantitative and qualitative terms. Many of these studies apply apply MFA, LCA, or both types of analysis. In general, two different approaches to assess the performance of these symbiotic systems exist. The first one takes the 'perfect' IS as a baseline. In a perfect IS, "*there will be no waste to dispose of and no primary input to purchase from external [outside the system, author's addition] suppliers*" (Yazan et al. 2016: 538). The perfect I-Sym is a hypothetical, practically non-existent ideal-type. The larger the deviation from that hypothetical situation, the worse the performance. The second approach: it compares the performance of IS with the opposite baseline, i.e. the hypothetical situation of no actual exchange of by-products at all [e.g. Sokka et al. 2011; Korhonen, 2001; Eckelman & Chertow, 2013).

IS *and Materials Flow Analysis (MFA)*

'Flow' is an essential concept in the general opinion of what IS is, i.e. import and export flows of firms within the defined industrial system. The exchanged materials in the flows are of energy, water and less sizeable amounts of other sorts of by-products. As the name itself makes clear, **MFA is a very dedicated methodology for IS**. **It tracks and quantifies the flows of products as outputs and inputs of firms in symbiotic industrial systems**. It is a **descriptive, systematic [and quantitative, author's addition] approach to assess the metabolism of a defined system**. Its outcomes can be used to "*minimise disposal through source reduction*" (Eckelman et al., 2014: 307).

IS *and Life-Cycle Assessment (LCA)*

Although it is a conclusion based on a review of a limited amount of literature, **LCA appears to be more often implemented than MFA, to evaluate the performance of symbiotic industrial systems** (e.g. Mohammed et al, 2016; Smith et al., 2015; Strazza et al., 2015). In these systems, LCA 'follows' the life cycle of material until its end in order to assess its environmental impacts. Reading Turner et al., (2016) carefully, that assessment appears a more explicit objective in LCA than in MFA. LCA is usually applied to one single by-product only or to a very few types of similar by-products, for instance a few different types of GHG. Notably, in **most reviewed studies the objective of LCA is comparing a few alternative methods of by-product exchange, or of treatment at the end of their life cycle**, as a basis for decision-making on the environmentally least damaging and economically most profitable method.

Studies that apply LCA to a symbiotic system in which tourist waste is a by-product are rare. One example is presented by Strazza et al. (2015). They compare the environmental and economic impacts of **recycling food waste produced at a cruise ship for use in aquaculture** (salmon) with the conventional system of feeding. On the selected indicators for comparison - global warming potential, non-renewable energy demand and water scarcity index – the recycling system scores better than the conventional one.

Assessed Impacts

Economic Assessment

In principle, economic gains of IS are achieved at company level. The more perfect IS is, the more firms can save on costs for waste disposal and purchase of resources from elsewhere. For one thing, waste materials and excess energy as by-products within symbiotic industrial systems are usually cheaper than virgin materials. Moreover, local supply within these geographical clusters saves on costs for transport, i.e. vehicles, fuel and networks of pipes and cables. Economic gains also include reduction of investments in own energy supplying installations. At the supply side, symbiosis of industrial clusters can result in opportunities for co-generation of different types of energy. A type of co-generation in large clusters of energy intensive industries is the combined heat and power (CHP) plant. Heat as and excess by-product is often supplied to district heat systems of nearby towns or urban districts.

Environmental Assessment

Paquin et al. (2015: 97) present a rather broad spectrum of environmental impacts of IS: "*reduced material intensity, reduced energy intensity, reduced dispersal of toxic substances, enhanced recyclability, maximized use of renewables, extended product life, and increased service intensity*". Again, this enumeration illustrates the close connection between environmental and economic returns of IS. The use of recycled waste materials and energy by firms for use in their buildings and production processes, rather than dumping in the environment, reduces the emittance of GHGs generated by chemical decomposition of solid waste in landfills. However, investments in proper engineering structures may turn the landfill in a source of useful by-products, for instance methane (Eckelman et al., 2014). Environmental and economic benefits of IS go more indirectly together in case economic benefits can be achieved from a green image and green markets potentials of the cluster production. Furthermore, using inputs of by-products from neighbouring firms saves money to be spend on upstream imports of virgin sources, but also implies reduction of air pollution by fuel combustion for transport and, in case of mining of virgin sources, reduction of adverse landscape alteration and the use of, sometimes enormous quantities of water (Kaza & Curtis, 2014).

Social Assessment

Economic and environmental benefits are explicit and closely interwoven objectives of theory and practice of IS. Social benefits on the other hand are rarely included explicitly in the concept. This does not mean that IS is 'non-social', but that the social component concerns pre-conditions and effects of IS rather than the symbiosis itself. Hence, assessment of social values is not included in MFA and LCA. 'The social' as precondition to IS touches qualities of (the networks of) actors and organisations involved in strategic thinking of symbiotic industrial system. Dedicated management skills, mutual trust, inter-organizational culture, knowledge sharing, and social and cognitive learning capacities of firms, mostly evolved through historical pathways, are crucial for cooperation of successful energy-efficient industrial eco-systems (Baas & Boons, 2004; Ferrer, 2012; Luciano et al., 2016; Yazan et al., 2106). Regarding the effects of IS, Paquin et al. (2015: 97) refer to case studies in China, Australia and Europe to underpin the observation that IS "*show that new job and business opportunities can result from exchanges, while simultaneously reducing resource and energy use, and environmental discharge*".

Suitability for URBANWASTE

The question if IS is suitable for the project URBANWASTE not as a standalone methodology but as a 'vehicle' to apply MFA and LCA to determine quantities and types of tourist waste and to assess the status-quo and future (to be developed) scenarios, can be approached in two different ways: is tourism as a separate industry suitable to realize a certain, or higher level of symbiosis, or can tourism be plugged in broader industrial systems of firms in other economic sectors to add more by-products into symbiotic relations.

The **first approach is highly unlikely**. **Tourist industry generates much waste, but that does consist of sizeable flows of single or 'pure' types of waste.** Instead, its output is a flow that is a composite of small quantities of many different types of waste (see D 2.1). If it should be only about the quantity, only the amount of **food waste**, the single largest type that "*can account for more than 50% of the hospitality waste*" (Pirani and Arafat (2014: 321), **might be sizeable enough to be converted into cost-effective input for other firms**, if sorted out from other types. However, no firms in tourist industry are interested in food waste as input for their 'production process'. Other types of waste may be re-used within tourist industry, but re-use does not fit in the definition of IS. In fact, as mentioned above, the by-product exchange that is essential of IS is much more common in multi-industry than in single-industry clusters.

The **second approach provides some opportunities**. **Food waste can be a valuable resource** to be used in symbiotic industrial systems, but only under certain conditions: **it has to be sorted out from other types of tourist waste** – which is not yet a general practice in that industry itself -, it has to be supplied in cost-effective quantities, and it has to be processed into other by-products first. Food waste as such is of little use for industries – we do not consider aquaculture an 'industry' – but chemical decomposition (gasification) may yield useful by-products. Furthermore, other types of solid waste by the tourist industry, if combustible, can be incinerated in power plants or in CHP plants that are occasionally part of symbiotic industrial systems. If proximate to urban municipalities, these can purchase the produced heat or electricity that is generated in these industrial systems.

Since social benefits of industrial symbiosis is not part of IS as a concept, there is no need to formulate indicators and to find data for IS in URBANWASTE. Economic and environmental benefits on the other hand are crucial impact categories of I-Sym. Indicators of both types of impacts of IS have to be first and foremost traced back to the application of MFA or LCA, i.e. related to measuring flows and life cycles of materials. In our opinion, URBANWASTE should first decide on the selection of relevant methods to be used in the project before thinking about appropriate indicators; for not selected methods that would be a waste of time.

It is proposed to consider the cascaded utilisation of certain waste types within the tourism sector or that certain waste types might be re-used or serve as input into other sectors. This means from a conceptual point of view, the idea of industrial symbiosis and circular economy is used in the project.

Suitability for assessment of changes on hotel level or on municipality level

IS is **hardly appropriate to assess changes on the hotel or municipality level**, let alone on the level of the individual hotel or municipality. If tourist waste is being supplied to symbiotic industrial systems for conversion into inputs for firms in other sectors, that will always be mixed up with quantities of the same waste from other sources to achieve cost-effective quantities for treatment. Ironically, it is most likely that tourist waste will be put together with municipal waste, making assessment of these two respective single types impossible.

3.16 Life Cycle Assessment (LCA)

LCA is seen as the main instrument for realising life cycle thinking. The ISO 14040 series of international standards defines LCA as: "a method for detecting environmental relevance of products, processes or services in their life cycle". The life cycle comprises raw material acquisition, production, manufacturing use, EoL treatment, recycling and disposal. Environmental impacts are measured by different techniques. Results are clustered and weighted to get significant values. LCA helps to identify environmental performance and supports decision-makers (ISO 14040).

In order to avoid misconceptions, the borderlines to similar terms used in scientific literature have to be defined:

- "Life cycle analysis" is often used as a synonym for LCA, especially in papers published before standardisation in 1997. Yet, no accorded definition exists in standards or scientific literature. Therefore, this term will not be used further;
- "Life cycle cost analysis" (LCC) is an assessment of all costs associated with the life cycle of a product that are directly covered by one or more of the actors in the product life cycle The integration of economic analysis with LCA is addressed in this innovative method. However, major differences concerning objectives and methodology exist (see Norris, 2001); and
- Likewise, "social life cycle assessment" (SLCA) is a new methodology that is based on principles of LCA, though has to be seen as an independent approach. SLCA analyses social impacts on people caused by the activities in the life cycle of their product (Dreyer et al., 2006).

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

LCA is an ambitious tool used for a wide variety of purposes. However, barriers for further dissemination of this approach have to be borne in mind. It has been repeatedly criticised for being too complex, time consuming and costly for industrial use (Rex and Baumann, 2008).

Underlying Concepts

In this chapter, the overall framework (Figure 16) of LCA is described according to ISO 14,040 and 14,044 as well as to the guidance books of the International Reference Life Cycle Data System (ILCD) (European Commission 2010). Key issues will be further discussed and accompanied with examples of literature.

Goal and scope

The goal and scope of an LCA have to be defined very thoughtfully. As a goal the intended application, the intended audience or the intended use of the results can be arranged (ISO 14,040). The scope includes the product system, the functions of the product system, the method and impact categories, the functional unit, system boundaries and data quality requirements as well as the type of critical review (ISO 14,040).

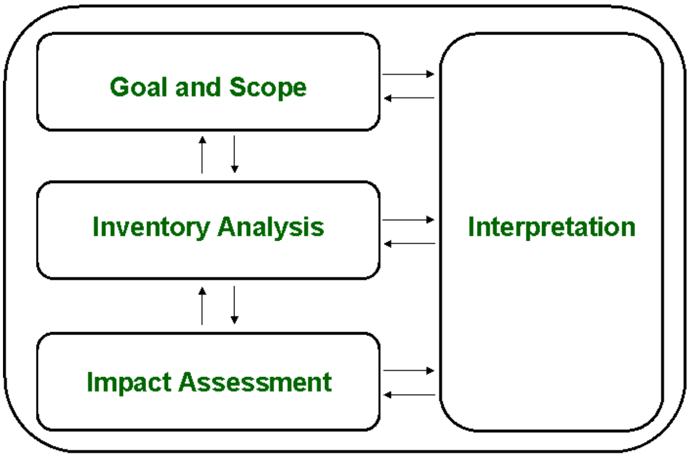


Figure 16: Overall framework of an LCA (ISO 14040).

A good knowledge of the intended product system and their functions is a main requirement to perform an LCA. For this purpose, it is helpful to create a process flow diagram. Methods and impact categories taken for the LCA need to be chosen at this stage. The characterization of methods and impact categories are explained later in the section for life cycle impact assessment.

The functional unit is implemented to define what is being modelled (e.g. 1m² of solar panel, 1kg of iron). All inputs and outputs of each process are relative to this functional unit. The functional unit gives also the opportunity to catch the purpose of a product or a service (e.g. treating the wastewater of 1 habitant during a year). For instance, it would be useless to compare 1 litre of paint A with 1 litre of paint B. More important would be to compare the service provided by paints A and B by covering 1 m² of wall during a year.

The reference flow is the flow expressing the functional unit. LCA is moreover dealing with an iterative approach. All processes or phases of an LCA use results of other phases (ISO 14,040). System boundaries are set to cover all relevant life cycle stages, processes and flows. For example, raw material acquisition, main manufacturing and processing steps, transportation, production and use of fuels, electricity and heat, use

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

phase, disposal, recycling, manufacture, maintain and decommission of capital equipment should be taken into consideration. Additional operations like lighting and heating in manufacturing buildings may be included as well. It always needs to be defined which steps are most relevant considering the environmental impacts. The cut-off criteria are set, when a certain level of flows are excluded from the modelling. For example, if the cut-off criterion is 90% of the overall environmental impact then the included flows shall make up at least 90% of the environmental impact (European Commission, 2010).

Data requirements include allocation procedures, intended assumptions or limitations and data quality. Allocation procedures need to be set with multi-functional processes, where the input flows can't be allocated to the output flows. Several techniques are provided to solve this problem. ISO 14,044 proposes allocation by physical properties (e.g. mass) or by economic value (e.g. market value). Further allocation procedures can be found in Buxmann et al. (1998) and in European Commission (2010). According to the data quality reliability, consistency and representativeness should be achieved. Data should be prepared transparently so that traceability is possible (ISO 14,040).

A critical review is needed to verify the intended LCA. The type of review often depends on the intended audience and the costs. A critical review is particularly necessary when the LCA is used for communication purposes. Sometimes reviews from external experts are needed and sometimes reviews from internal staff are adequate. The ISO normally does not provide classification on which reviewer is taken with a certain LCA. In the ILCD documents (Del Borghi et al., 2009) a draft is provided where it is classified which LCA needs which review.

The goal and scope definition is important to outline the LCA. It may be possible that goal and scope are adjusted during the inventory analysis.

Inventory analysis

The life cycle inventory can be assessed by two main modelling frameworks: attributional and consequential modelling. Attributional modelling means to assess potential environmental impacts that can be attributed to a given, existing product system over its life cycle, i.e. upstream along the supply-chain and downstream following the products use and end of life (European Commission, 2010). Consequential modelling aims at identifying the consequences that a decision in the foreground system has for any other processes and product systems of the economy. It models the to-be-analysed product system around these consequences (European Commission, 2010). The big difference of the two modelling systems is that attributional modelling models the product system as it is whereas the consequential modelling considers how decisions affect other processes or products. For example, by modelling the use of biofuel with consequential modelling the consequences on the market of crops or on the land-use are included whereas with attributional modelling the system is considered as it is.

The inventory analysis comprises data collection, data calculation and allocation. Data collection needs to be performed through the whole life cycle. Primary or secondary data can be generated for each input and output flow. Primary data is difficult to obtain and very resource-intensive. Secondary data can be collected by questionnaires or with literature sources.

Furthermore, it has to be declared, if specific or generic/average data are used. As a general rule the main process step should be modelled with specific data (European Commission, 2010). Specific data means data from the represented process. Background processes can be represented by average or generic data (e.g. electricity mix). Average data is used, if the technology standard applies for a whole region/country/continent and a mix of that is advantageous. However, data can only be averaged if the same methodology, limitations and allocations are used (European Commission, 2010). If the modelled technology is very different from the standard, then specific data would be useful. The same applies for very relevant processes (European Commission, 2010).

The last step is allocation. In ISO 14,044, allocation procedures are divided into allocation by physical properties (e.g. mass) or by economic value (e.g. market value). In general, all calculation steps and allocation procedures have to be documented and consistent. Good documentation is a requirement to edit an LCA in a transparent and traceable way. When using allocation, the sum of the inventories which were allocated need to be equal to the inventory before making the allocation.

Assessed Impacts

Life Cycle Impact Assessment serves to aggregate the inventory data in support of interpretation. Optionally, normalisation and weighting may be applied to further support this. Life Cycle Impact Assessment (LCIA) according to the ILCD handbook (European Commission, 2010) is provided by various categories. These categories show environmental potentials in different areas: human health, natural environment and natural resources (Figure 17). **Impact categories** can be calculated by impact indicators with a certain inventory (see Table 7).

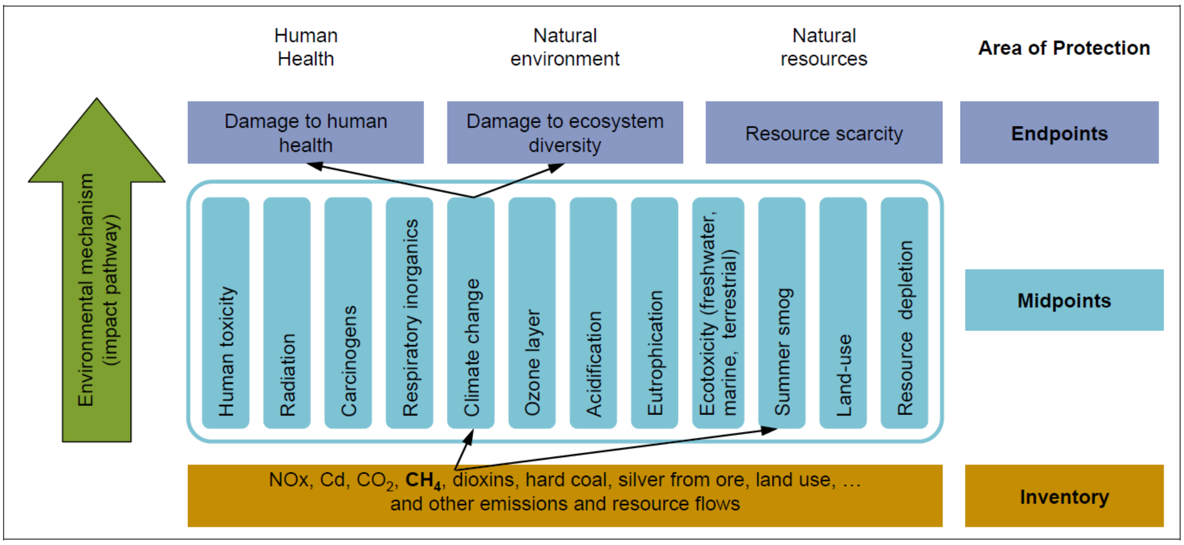


Figure 17: Life Cycle impact assessment, Schematic steps from inventory to category endpoints. (European Commission, 2010)

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Table 7: Impact categories and their characterizations (European Commission, 2010, Fleischer and Riebe, 2002).

Impact category	Impact indicators	End point	Inventory
Climate change	e.g. Global Warming Potential GWP	Increase of the global average temperature	Emissions of green-house gases
Ozone depletion	e.g. Ozone Depletion Potential ODP	Impact on human health e.g. skin cancer due to expansion of ozone hole	Emissions of chlorofluorocarbons (CFC) and halons
Human toxicity	e.g. Quality Adjusted Life Years QALY, Disability Adjusted Life Years DALY, Human Toxicity Potential HTP	Impact on the human health	Emissions of toxic substances
Respiratory inorganics/ particulate matter	e.g. Total Suspended Particulates, TSP, PM_{10}, $PM_{2,5}$		
Ionizing radiation		Impact on human health and ecosystems	Emissions of radioactive substances
Photochemical ozone formation (smog)	e.g. Photochemical Ozone Creation Potential POCP, Maximum Incremental Reactivity MIR	Impact on the human health e.g. asthma and natural environment	Emissions of VOC volatile organic compounds and NO_X
Acidification	e.g. Acidification Potential AP	Impact on vegetation and loss of aquatic biodiversity	Emissions of acids (NH_3, NO_X, SO_2, …)
Eutrophication	Terrestrial and aquatic eutrophication	Impact on vegetation and loss of aquatic life due to the lack of O_2	Emissions of nutrients like phosphor and nitrogen
Ecotoxicity	Terrestrial and aquatic, e.g. Potentially Disappeared Fraction of species PDF, Probability of Occurrence POO, Mean Extinction Time (MET)	Impact on natural environment (chemical's fate, species exposure, toxicological response)	Environmental persistence and ecotoxicity of chemicals
Land use	e.g. Land transformation, land occupation	Impact on ecosystems due to effects of occupation and transformation of land	Physical changes to soil surface, to soil and to flora and fauna
Resource depletion	e.g. Shortage of mineral resources	Decrease of resources, so that future generation will suffer	Input of non-renewable energy (coal, oil, bauxite,…)

To calculate environmental burdens elementary flows are classified, characterised and implemented with a factor. This classification and characterization can only be done by experts, who then provide complete sets of LCIA methods. Further details on the methods can be found in the ILCD background document "Framework and requirements for Life Cycle Impact Assessment (LCIA) models and indicators". LCIA results can also be normalised, if the value needs to be related to e.g. an average citizen. Weighting factors can also be implemented to point out certain relevance. They can be used to combine some impact categories to one overall indicator (e.g. Eco-Indicator 99).

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

The choice of indicators depends on the product or process which is modelled. The indicator sets are presented by different methods (e.g. CML 2002) and are related to time and region (e.g. GWP 100 global).

In dealing with LCA, various methodologies on environmental impact assessments are available. A short summary of some methodologies is provided in Table 8. For further details, refer to the ILCD background document "Analysis of existing Environmental Impact assessment methodologies for use in Life Cycle Assessment (LCA)" (European Commission, 2010).

Table 8: Environmental impact assessment methodologies (summarised from European Commission, 2010).

Method	Developer	MP	EP	Description
CML 2002	CML (NL)	x		One of the first methods and frequently used; separate normalisation factors for each indicator, in total 19 impact categories.
Eco-indicator 99	Pré (NL)		x	Three perspectives: Hierarchist, Individualist and Egalitarian. One fully integrated approach which covers all impact categories resulting in damage to human health, ecosystems and resources.
EDIP97 and EDIP2003	DTU (DK)	x		Classic emission-related impact categories as well as resources and working environment. Normalisation and weighting is based on political environmental targets.
EPS2000	IVL (SE)		x	Expression in monetary units derived on the Willingness To Pay (WTP) principle.
Impact 2002+	EPFL (CH)	x	x	14 midpoint categories are linked to four damage categories (human health, ecosystem quality, climate change, resources). Normalization either at midpoint or at damage level.
LIME	AIST (JP)	x	x	Expression in monetary units. One single index based on the midpoint to the endpoint. Weighting is based on environmental conditions of Japan.
LUCAS	CIRAIG (CAN)	x		Related to Canadian context based on TRACI and Impact 2002+.
ReCiPe	RUN + Pré + CML + RIVM (NL)	x	x	Follow up of Eco-indicator 99 and CML 2002. It harmonizes midpoint and endpoint approaches.
Swiss Ecoscarcity 07	E2 + ESU-services (CH)	x	x	Weighting is based on public policy targets and objectives. Originally developed for Swiss conditions, but already adapted to other European countries.
TRACI	US EPA (USA)	x		Related to conditions in the USA and in line with the EPA policy.
MEEuP	VhK (NL)	x		Evaluation of various energy-using products (EuP) and their extent of fulfilling certain criteria for implementing measures under the Ecodesign of EuP Directive 2005/32/EC.

Methodologies can be midpoint (MP) or endpoint (EP) related or a combination of both. Midpoint related impact categories are defined as a place where a common mechanism for a variety of substances within that specific impact category exists. Examples are climate change or cancer effects. Endpoint, also called damage-related, are methodologies which focus on indicators related to areas of protection of human health, natural

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

environment and natural resources in general. For instance, climate change is quantified at MP by providing a CO_2 equivalent of the GHG emissions. This gives an idea of the magnitude of the environmental pressure. At EP the consequence (sea rise, loss of species etc.) of these emissions and the damage (response) is evaluated.

Most of the methods are developed for European conditions. For some categories such as climate change, ozone layer depletion or resources global considerations are included. Some methods exist which can be used outside of Europe.

Interpretation

Interpretation, the last step of an LCA, is one of the most important stages. By interpreting the results of the LCIA the outcome becomes understandable for decision makers and interested parties. The interpretation must contain (ISO 14,040):

- Identification of significant issues;
- Evaluation; and
- Conclusions, limitations and recommendations.

To identify significant issues, firstly the LCIA results need to be analysed and then the overall LCA. Explanations for intended assumptions, allocations, cut-off decisions and selection of impact categories and indicators need to be provided. For evaluating the final results three checks are necessary:

- Completeness check;
- Sensitivity check; and
- Consistency check.

The outcome should be consistent with the goal and scope of the LCA. Therefore, the conclusions, limitations and recommendations shall be drawn in accordance with the goal definition and the intended application of the results (European Commission 2010).

Suitability for URBANWASTE

LCA is highly relevant to URBANWASTE as it is a necessary method for evaluating the targets of the project. Only with LCA environmentally-related targets such as cutting GHG and fresh water use can be measured. The methodology of LCA is convertible to a high extent as an **international standard method** exists. Technical and operational feasibility can be achieved by using standard LCA methods and **social and cost-related targets can also be evaluated by this approach**.

The potential to **display material flows and to estimate their impacts by using LCA is without controversy**. Improvements in environmental performance of products, services and processes have been widely documented. Economic benefits also arise, as environmental benefits are achievable, for instance, by reducing energy use or minimising waste for disposal. In each case, costs can be saved and more efficient production is reached. LCA can also play a major role in social impacts. Benefits for society occur when products or processes are evaluated by means of SLCA. Issues related to the working environment and to the whole society can be assessed and improved.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Furthermore, **LCA is compatible with EU policy**. The EC has introduced various initiatives to strengthen life-cycle thinking in policy and business. Activities such as the Green Paper on Integrated Product Policy (IPP) and the landmark communication on IPP have the objective of launching a debate on improving environmental performance of products, services or systems throughout their life-cycles. Life cycle thinking is further strengthened in the Commission's thematic strategies programmed by the 6th Environment Action Programme on the sustainable use of natural resources and on the prevention and recycling of waste. With the EU waste-related policies, the EC wants to highlight the importance of waste minimisation, the protection of the environment and human health (Del Borghi et al., 2009).

LCA is therefore suitable for the purpose of URBANWASTE. The method encapsulates environmental related issues, as well as economic and social aspects, in a life cycle approach, and will be a key tool for measuring the achievements in the project using standard methods and given sustainability indicators.

Suitability for assessment of changes on hotel level or on municipality level

Of course it is possible to use LCA as method at municipal level. Additionally, it is possible to use smaller scales for LCA considerations, e.g. company level (hotels).

3.17 Life Cycle Costing (LCC)

Life Cycle Costing (LCC), also called Whole Life Costing, is a technique to **establish the total cost of ownership, meaning the total costs that occur during the whole lifetime (e.g. of a product, service, investment or an asset)**. The definition of Blanchard and Fabrycky (1998) may serve as a common basis: "*Life-cycle cost refers to all costs associated with the system as applied to the defined life cycle*".

LCC is a structured approach that addresses all the elements of this cost and can be used to produce a spend profile or spend analysis of the product or service over its anticipated life-span. The results of an LCC analysis can be used to assist management in the decision-making process where there is a choice of options.

Life cycle costing is **an assessment of all costs associated with the life cycle of a product / service that are directly covered by one or more of the actors in the product life cycle (supplier, producer, user/consumer, EOL-actor), with complimentary inclusion of externalities that are anticipated to be internalised in the decision-relevant future** (Rebitzer and Hunkeler, 2003). This means, that both **different types of costs** as well as **different types of externalities** can be considered. Typical cost types can be related to different phases of the life cycle and may include costs occurring in the design and planning phase, in construction and operational / maintenance phase, in the end of life phase (including renewal, rehabilitation, disposal etc.). In addition, acquisition, depreciation, capital costs and the like may be considered.

More problematic is the fact, that **internalising environmental and social costs is very challenging** and often theses costs are hard to quantify.

LCC is seen as an essential link for connecting environmental concerns with core business strategies (Hunkeler and Rebitzer, 2003). Businesses are encouraged to incorporate good environmental performance to manufacturing and sales planning. However, the methodology of LCC differs from the methodology of Life Cycle Assessment (LCA).

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

Underlying Concepts

Hunkeler and Rebitzer (2003) describe LCC as being "*complementary to existing management structures*" and that it "*sits at the finance-technology-environment interface*." According to the authors Life Cycle Thinking "*aims at integrating environmental concerns into industrial and business operations by considering off-site, or supply chain, impacts and costs*."

As in life cycle assessment (LCA) the systems approach is the core concept of LCC (see e.g. Rebitzer (2002)). Swarr et al. (2011) state "*the code of practice is grounded in a conceptual framework for life-cycle sustainability assessment (LCSA) of products that uses distinct analyses for each of the three pillars of sustainability, environment, economy, and social equity. (…) Life-cycle assessment (LCA) is the only pillar that has been standardized to date*" and "*UNEP (2009) has published guidelines for social LCAs and is currently developing methodological sheets for impact subcategories*." In addition, the authors state that the "*code of practice provides guidance that builds on the four-phase structure of the ISO 14040 standard to facilitate definition and application of consistent system boundaries for complementary LCC and LCA studies of a given product system. Goal and scope definition is similar to that of an LCA*."

Norris (2001) describe the differences between LCA and LCC (see Table 9).

Table 9: Methodological differences of LCA and LCC (Norris (2001))

Aspect	LCA	LCC
Purpose	Compare relative environmental performance of alternative product systems for meeting the same end-use function, from a broad, societal perspective.	Determine cost-effectiveness of alternative investments and business decisions, from the perspective of an economic decision maker such as a manufacturing firm or a consumer.
Activities which are considered part of the 'Life Cycle'	All processes causally connected to the physical life cycle of the product; including the entire preusage supply chain; use and the processes supplying use; end of life and the processes supplying end of life steps.	Activities causing **direct costs or benefits** to the decision maker during the economic life of the investment, as a result of the investment.
Flows considered	Pollutants, resources, and inter-process flows of materials and energy.	**Cost and benefit monetary flows** directly impacting decision makers.
Units for tracking flows	Primarily mass and energy; occasionally volume, other physical units.	**Monetary units** (e.g. dollars, euro).
Time treatment and scope	The timing of processes and their release or consumption flows is traditionally ignored; impact assessment may address a fixed time window of impacts (e.g. 100-year time horizon for assessing global warming potentials) but future impacts are generally not discounted.	Timing is critical. Present valuing (**discounting**) of costs and benefits. Specific time horizon scope is adopted, and any costs or benefits occurring outside that scope are ignored.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Assessed Impacts

The assessed impacts within LCC can be diverse and depend on the scope and goal of an assessment. LCC uses monetary values of costs / revenues and also for external costs / revenues. For **almost all cost and revenue related parts of a waste management system data is available somewhere, yet often seen as company sensitive**. When considering **social and environmental impacts**, the situation is **more critical regarding quantifying these impacts, especially external effects**.

Suitability for URBANWASTE

Hunkeler and Rebitzer (2003) point out, that "*the terminology in the field of LCM/LCC, however, is not uniform, with various disciplines using different terms for similar problems or vice versa. For instance, the term life cycle might be referred to as the physical life cycle as in LCA (ISO 14,040) or the economic life cycle consisting of the market introduction-, growth-, and market saturation phases, etc. as it is commonly used in product planning*."

Swarr et al. (2001) state that "*one aspect that can be challenging is that LCC attempts to capture all costs across the life cycle, and some costs are borne by different actors with very different perspectives of the costs and potentially conflicting goals.*"

All this **requires a strict and consistence definition of cost types** included in assessment in order to avoid losing important information and double-counting. Also the **internalisation of external environmental and social costs may be a challenging task that might overwhelm the capacities in terms of time resources**, **data availability and quality of the involved partners**, especially the case study cities. The externalisation aspect also involves the definition of timeframes, e.g. the planning and implementation phase of a facility are different compared to average lifetime of the facility or the depreciation timeframe, not to speak from the **time lag of potential environmental impacts**.

In addition, it has to be clarified, whether revenues might be included in an LCC assessment, this could be of special interest in the case of waste management, leading to a "net cost" assessment. Nevertheless, the same problems occur as mentioned above.

Beside external costs, costs and revenues that are generally more easy available, e.g. at company level are often considered as important competitive information and therefore are often seen as "**company secret**", therefore often the information is not available.

Regarding the project URBANWASTE it might be possible to use a **very reduced form of LCC**, that cannot be called LCC anymore, as it **only covers specific parts of the waste management system**. One option could be to **focus on a clear set of defined costs / revenues in a system, that are available, in good quality and that are representable (e.g. for future scenarios in an assessment).** This is important to reduce the time efforts to a minimum, the drawback is that only parts of systems and parts of life cycles can be considered, this may lead to the effect, that important cost / revenue aspects are not included or important social and environmental impacts are not considered.

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

Suitability for assessment of changes on hotel level or on municipality level

The same as mentioned above holds true for the suitability at smaller geographical scales. If data from municipalities / hotels are not seen as company secret, then selected cost / revenue sets could be compiled and used. Again, this needs clear definitions of cost / revenue types included in the assessment. In addition, future scenarios should be possible being depicted in terms of cost / revenue types.

3.18 Life Cycle Working Environment (LCWE)

Life Cycle Working Environment (LCWE) **aims at integrating social aspects into Life Cycle Assessment**. In applying this method, process-specific data concerning social aspects for use in product LCA is obtained by using statistical sources. The following aspects are covered:

1. Amount and qualification level of work
2. Health and safety

Applying this approach, social indicators have to be collected for each process in the production chain. Here the problem occurs, that such data is barely available at the moment. Therefore, a top-down process is chosen for the LCWE methodology.

Assessed Impacts

The LCWE-Method (Life Cycle Working Environment) developed at the Department of Life Cycle Engineering of the University of Stuttgart **aims to asses and to quantify product or product system related social aspects** along the value-added chain. In analogy to the ecological profile of an LCA the LCWE-Method provides a **social profile of the product** with a possible level of detail up to individual process steps.

Statistical data concerning social issues is available for most of the highly developed countries. To prorate this data down to process level, the following assumptions are applied (Beck et al. N.N.):

1. The social impacts of a process are proportional related to the amount of human labour on the process;
2. The amount of human labour of a process is related to the effort made to add value by processing.

Those assumptions are valid within the same industry and the same country only. For the calculation based on statistical data the total working time per value added is calculated. This total working time to generate one Euro value added vary between industries. As second parameter the rate of **non-fatal injuries per value added** are included.

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

Suitability for URBANWASTE

As the **methodology is based on generic statistical data at national level** and much more as the methodology is developed and suitable only for product LCAs **it is not suitable to evaluate the service related impacts of tourist related waste management activities**.

Suitability for assessment of changes on hotel level or on municipality level

As the methodology is based on generic statistical data on national level and much more as the methodology is developed and suitable only for product LCAs it cannot be used to assess changes on hotel or municipality level related to waste prevention issues.

3.19 Multi-Criteria Decision Making (MCDM)

Multiple-criteria decision making (MCDM) is one of the most well-known branches of decision making. It is **concerned with structuring and solving decision and planning problems involving multiple criteria** (Majumder, 2015).

The MCDA (multi-criteria decision analysis) **is a way of looking at complex problems** that are characterised by objectives, of **breaking the problem into a more manageable pieces**, to allow data and judgements to be brought to bear on the pieces, and **then of reassembling the pieces to present a coherent overall picture to decision makers** (Dodgson et al. 2009). The purpose of MCDA is to support decision-makers facing such problems. The MCDA is useful for:

- Dividing the decision into smaller and more understandable parts;
- Analysing each part;
- Integrating the parts to produce a meaningful solution.

Normally, there is **no unique optimal solution to an MCDM problem** and it is necessary to use decision maker's preferences to differentiate between solutions. According to Dodgson et al. (2009), the decision making process comprises a specific number of detailed steps which are illustrated in Figure 18.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

1. **Establish the decision context.**
   - 1.1 Establish aims of the MCDA, and identify decision makers and other key players.
   - 1.2 Design the socio-technical system for conducting the MCDA.
   - 1.3 Consider the context of the appraisal.
2. **Identify the options to be appraised.**
3. **Identify objectives and criteria.**
   - 3.1 Identify criteria for assessing the consequences of each option.
   - 3.2 Organise the criteria by clustering them under high-level and lower-level objectives in a hierarchy.
4. **'Scoring'. Assess the expected performance of each option against the criteria. Then assess the value associated with the consequences of each option for each criterion.**
   - 4.1 Describe the consequences of the options.
   - 4.2 Score the options on the criteria.
   - 4.3 Check the consistency of the scores on each criterion.
5. **'Weighting'. Assign weights for each of the criterion to reflect their relative importance to the decision.**
6. **Combine the weights and scores for each option to derive an overall value.**
   - 6.1 Calculate overall weighted scores at each level in the hierarchy.
   - 6.2 Calculate overall weighted scores.
7. **Examine the results.**
8. **Sensitivity analysis.**
   - 8.1 Conduct a sensitivity analysis: do other preferences or weights affect the overall ordering of the options?
   - 8.2 Look at the advantage and disadvantages of selected options, and compare pairs of options.
   - 8.3 Create possible new options that might be better than those originally considered.
   - 8.4 Repeat the above steps until a 'requisite' model is obtained.

Figure 18: : Detailed steps for application of MCDA (Source: Dodgson et al. 2009)

Among the weighing methods to represent importance required in Step 5, the use of the AHP (Analytical Hierarchy Process) is very frequent. The AHP is applied to relative critical weighing of indicators and evaluators (Kadafa et al. 2014).

Another very popular method is the Fuzzy Logic Decision Making (FLDM) (Majumder, 2015).

Underlying Concepts

MCDM and Life Cycle Assessment (LCA)

The analogy between MCDM and LCA has already been reported (Gaudreault et al. 2009). Nevertheless, **not many references have been found about the combined use of a general MCDM model with LCA**. Several MCDM methods (e.g. AHP, ELECTRE) have been applied in decision making related to reverse logistics and supply chain management problems, which cover downstream and upstream processes (Wang et al. 2004; Rezaei, 2015). This indicates, that some of the methods included in MDCM could potentially be combined with LCA studies so that MCDM would be a useful model to be further explored in that direction.

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

MCDM and Materials Flow analysis (MFA)

The use of a MCDM model to face a specific problem merely because of its compatibility with the MFA tool cannot be proved or confirmed. Therefore, it seems difficult to determine whether MCDM models could be of use when considering quantitative material flows.

Assessed Impacts

Social Assessment

As discussed previously, the MCDM is a flexible model which serves multiple purposes. It can be used not only to assess social aspects, but also **it can incorporate public participation into the decision-making process** (Hung, Ma & Yang, 2007). The evaluation of social aspects with MCDM has also been reported by Ramjeawon and Beerachee (2008), among other authors.

Economic Assessment

The MCDM is a **suitable methodology to evaluate economic impacts**. Several authors have analysed the use of MCDM as a socioeconomic assessment tool in solid waste management systems (Karagiannidis & Moussiopoulos, 1997; Rousis et al., 2008). Hung et al. (2007) have also proved the use of MDCM as an effective tool to accommodate environmental factors in combination with economic and social aspects.

Environmental Assessment

Many authors have informed about the successful use of MCDM in assessing environmental impacts (Hung et al. 2007; Ramjeawon & Beerachee, 2008; Garfi et al., 2009). Its use has been particularly relevant in municipal solid waste management (MSWM) decisions, as reported by several authors (Morrissey & Browne, 2004; Hung et al. 2007; Khan & Faisal, 2008).

Suitability for URBANWASTE

The MDCM is presented as **a model that would fulfil the characteristics of the URBANWASTE project**. Moreover, its frequent use in municipal waste management systems proves its suitability within the framework of the project.

For instance, Hung et al. (2007) reviewed a large number of models developed to support decision making in municipal solid waste management. They identified MCDM as a model which is often used to aid decision making in MSWM. Moreover, they developed a sustainable decision-making model for municipal solid waste management that also implemented public participation in the decision making process. The model provided an effective means for assisting decision making for real-world waste management problems.

Garfi et al. (2009) compared different waste management solutions, in this case by using a **combined model of MCDM and AHP**. This model considered environmental and social aspects. The integration of the AHP enabled decision making incorporating planning, setting priorities, selection of the best options among alternatives, and the allocation of resources.

Suitability for assessment of changes on hotel level or on municipality level

An article published by Allesch and Brunner (2014) – with results from 15 studies on MCDM models – reveals that the range of scale for application can vary significantly: from single waste streams to complete waste

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

management systems in cities, regions and countries. This indicates, that **the assessment of changes on both levels, hotel and municipality, would not imply major complications**.

Summary / conclusion

MCDM is a **decision-making tool that facilitates the selection of the best alternative among several alternatives.** It **evaluates a problem by comparing and ranking different options and by evaluating their consequences according to the criteria established**.

MCDM also proves to be a flexible model that could be combined with multiple different tools (e.g. LCA, AHP, CBA, etc.) (Ramjeawon & Beerachee, 2008; Khan & Fasial, 2008; Garfi et al. 2009, De Feo & De Gisi, 2010).

In addition, a study carried out by Allesch and Brunner (2014) concluded that MCDM appears to be one of the most complete methods for assessing environmental, social and economic impacts.

Its use in municipal waste management decisions, selection of waste treatment technologies, analysis of disposal sites, etc. (Hung et al., 2007; Manaf et al. 2008; Dodgson et al. 2009) confirms that this model could be of benefit for a project like URBANWASTE.

Nevertheless, the selection of the right tool to be integrated with the MCDM model seems to be a critical step, which might influence the outcome and alternative selected. The uncertainty about whether MCDM would be sufficiently MFA-based or not is another aspect to consider beforehand.

3.20 Multi-Scale Integrated Analysis of Societal and Ecosystem Metabolism (MuSIASEM)

Multi-Scale Integrated Analysis of Societal and Ecosystem Metabolism (MuSIASEM) is **a methodology developed to study "*biophysical and socioeconomic issues in an integrated manner, both for the level of the society and for the different compartments of that society*"** (Ginard-Bosch & Ramos-Martín, 2016, p. 26).

Underlying Concepts

MuSIASEM (Giampietro & Mayumi, 1997, 2000a, 2000b; Giampietro, Mayumi, & Ramos-Martin, 2009) is based on the integration of various underlying theoretical concepts, such as non-equilibrium thermodynamics applied to ecological analysis, complex systems theory or bioeconomics (Ramos-Martín et al. 2009). MuSIASEM aims to describe the metabolism of a human society by characterising "*the processes of energy and material transformation in a society that are necessary for its continued existence*" (Ramos-Martín et al., 2009).

Assessed Impacts

MuSIASEM is a **static analysis** that measures congruence between flows and funds over several scales (time, space etc.) and thereby allows observing the evolution of a system over time. But being a static analysis is at the same time one of the shortcomings of the methodology as it **provides a snapshot of a system but not of its dynamics** (Ginard-Bosch & Ramos-Martín, 2016).

MuSIASEM distinguishes between "endosomatic" and "exosomatic" metabolism that describes "*flows of energy and materials transformed under human control within socio-economic processes both inside*

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

(endosomatic) and outside (exosomatic) the physical body of the members of a given society" (Ramos-Martín et al., 2009, p. 4660). With reference to Giampietro et al. (2009) *flow coordinates* are elements that either enter but do not exit or exit but do not enter the production process. *Fund coordinates* are described as agents (e.g. capital, labour) that enter and exit the process and that transform input into output flows.

Examples for flow and fund indicators are: Total energy throughput (TET), total human activity (THA), gross domestic product (GDP), energy throughput in activity i (ET_i), human activity in activity i (HA_i), GDP per hour in the society (GDP_{hour}), exosomatic metabolic rate, average of the society (MJ/h) (EMR_{SA}), exosomatic metabolic rate (MJ/h) (EMR_i), economic labour productivity (h/h) (ELP_i), economic energy intensity (MJ/€) (EEI_i) (Ginard-Bosch & Ramos-Martín, 2016).

MuSIASEM does not distinguish different forms of energy, however the time dimension of energy transformations and the relation of the energy sector to other economic sectors is described (Ramos-Martín et al., 2009, p. 4660).

As MuSIASEM provides an **interdisciplinary approach** by combing several underlying systems it allows for the social, economic and environmental assessment of a system for a certain spatial scale and a certain point in time.

Suitability for URBANWASTE

MuSIASEM seems a **suitable and relevant method for URBANWASTE as it allows a comprehensive approach integrating social, economic and environmental assessment of a system** and **enables to compare various states of a system over time, e.g. after implementing certain waste reducing measures**. This is also supported by a study (D'Alisa, Di Nola, & Giampietro, 2012) that applies MuSIASEM (with an extended indicator set) as an approach to deal with the waste management crisis in Campania region (Italy).

Suitability for assessment of changes on hotel level or on municipality level

Applying the **method on a hotel is theoretically possible**, but does not fulfil the purpose of the method which seeks to gain a comprehensive analysis of a Societal and Ecosystem Metabolism. **Application on municipality/city level is suitable**.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

3.21 Social Life Cycle Assessment (SLCA)

SLCA has developed from environmental LCA, which addresses environmental impacts. O'Brien et al. (1996) first raised the idea of complementing LCA with social life cycle assessment. **Different indicators have been proposed**, such as **additional employment, Quality Adjusted Life Years and health impacts (positive and negative)** (Ekener Petersen, 2013). A document with guidelines of S-LCA has been developed by United Nations Environmental Programme (UNEP) and the Society of Environmental Toxicology and Chemistry (SETAC). These guidelines can be considered as a basis of this type of analysis, but according to Ekener Petersen (2013) they are also the result of what could be agreed on at that time and therefore do not completely cover all outstanding issues on S-LCA.

SLCA is based on (environmental) LCA, with some adaptations, and was developed in accordance with the ISO 14,040 and 14,044 standards for LCA (ISO, 2004). **LCA and SLCA share the life cycle perspective**, considering the full life cycle of products or services, including all down- and upstream processes. The main difference between SLCA and LCA is that LCA addresses environmental impacts, whereas S-LCA addresses social impacts, i.e. impacts on human beings and the society. Since **social impacts (both positive and negative) often are difficult to quantify** and to compare with other social impacts one major difference to the environmental LCA is the lack of quantitative and weighted results. Therefore, the **functional unit gets a slightly different use in SLCA in comparison with environmental LCA**. In **SLCA the (often) qualitative results cannot be expressed per functional unit** and therefore the FE only serve the purpose of specifying the scope of the study (Umair et al., 2015).

As described in Umair et al. (2015) the guidelines developed by UNEP & SETAC (2013) can be used to identify the different stakeholders and subcategories for social impacts to assess. The methodological sheets presented are grouped into five stakeholder groups: Workers, Consumers, Local Community, Society and Value Chain Actors. For each stake holder group UNEP & SETAC (2013) present of several assessment subcategories including for example freedom of Association is a subcategory of the Worker stakeholder group.

Underlying Concepts

SLCA was developed as a method to support the environmental LCA by adding social impacts to the environmental impact categories and SLCA was developed in accordance with the ISO 14040 and 14044 standards for LCA (ISO, 2004). It therefore shares many features with LCA and are supposed to include all upstream and downstream processes during the whole life cycle. However, SLCA differs from (environmental) LCA since social aspects are far from always possible to quantify and it is even less possible to aggregating negative and positive impacts as is normally done in (environmental) LCA.

Assessed Impacts

Social Life Cycle Assessment are for obvious reasons focused on social impact of any process or service. However, it can be seen as part of the combined method Life Cycle Sustainability Assessment (LCSA) that include the combination of life cycle assessment (LCA), life cycle costing (LCC), and social life cycle assessment (SLCA) (Vinyes et. al., 2013).

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Suitability for URBANWASTE

SLCA does only consider social aspects, but it is a method that can be used to asses all upstream and downstream impacts from a process or service. It is therefore **suitable to answer specific URBANWASTE questions with a social impact**. However even though the method could stand alone, there are possible synergies to perform SLCA as part of the broader concept Life Cycle Sustainability Assessment. An example of this can be found in Vinyes et al. (2013) where a LCSA of waste management options for used cooking is performed. Other concrete examples can be found in Sala et al. (2015).

Suitability for assessment of changes on hotel level or on municipality level

This is a method **suitable for assessing changes on both hotel and municipality level**, however since there are many possible social impacts to concern in large and complex systems there is probably a need for simplifications. For a small/simple system or an assessment based on observations and interviews with (all) relevant stakeholders would be possible to perform (e.g. Umair et al., 2015), but in order to asses more complex systems a simplified approach like the one used in Vinyes et al. (2013) could be more suitable.

3.22 Strategic Environmental Assessment (SEA)

Strategic Environmental Assessment (SEA) is a **systematic decision support process**, aiming to ensure that environmental and possibly other sustainability aspects are considered effectively in policy, plan and program making. It **was developed to supplement the Environmental Impact Assessment (EIA)** and due to this, these methods have several similarities, and the fundamentally difference is the level of decision making they are applied on where **EIA are applied on individual project and SEA are applied earlier on the plans and programs** that leads to concrete projects.

The SEA Directive (EU, 2001), which is in force since 2001 and should have been transposed to national law in all member states by July 2004, applies to a wide range of public plans and programmes. This means that in all EU member states a SEA have to be performed for public plans/programmes for agriculture, forestry, fisheries, energy, industry, transport, waste/ water management, telecommunications, tourism, town & country planning or land use and which set the framework for future development consent of projects listed in the EIA Directive. This means that for waste management facilities like incineration plants or landfills that are planned and constructed after 2004 a SEI and a EIA are already performed. Since the SEA Directive obligates all member states to monitor the significant environmental effects of the implementation of plans and programmes, but also to keep the process transparent, it is likely that there are environmental reports publically available. However, the SEA Directive does not include any requirements on how the monitoring should be performed and therefore it can hardly be described as a methodology for assessing sustainability.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Underlying Concepts

SEA is not based on any quantitative methodology like MFA or LCA. Instead it is a **methodology for a systematic decision support process**, aiming to ensure that environmental and possibly other sustainability aspects are considered effectively in plan and programme making.

SEA (under the SEA Directive) is based on the following phases:

- Screening, including investigation of whether the plan or programme falls under the SEA legislation.
- Scoping, defining the boundaries of investigation, assessment and assumptions required.
- Documentation of the state of the environment, effectively a baseline on which to base judgments.
- Determination of the likely (non-marginal) environmental impacts, usually in terms of Direction of Change rather than firm figures.
- Informing and consulting the public.
- Influencing "Decision taking" based on the assessment.
- Monitoring of the effects of plans and programmes after their implementation.

So it is not a method for assessing the impact of decision, but more a method of using the information from assessment methods to ensure that these are considered in public planning with potential impact on environment and health.

Assessed Impacts

A SEA are normally focuses on environmental issues (including public health) but it can include other aspects of sustainability like economic and social aspects, but also material assets and archaeological sites.

Suitability for URBANWASTE

SEA is not a suitable method answer specific URBANWASTE questions since it is used to ensure environmental concerns in public planning rather than monitor the effects of the plans. However, since the SEA Directive require monitoring there are a likeliness that this method will indirectly provide data to be used in other methods assessing waste strategies.

Suitability for assessment of changes on hotel level or on municipality level

SEA is mandatory to applied to public plans and programmes which means that many municipalities already should be using this method. However, since it does not include any requirements of how the monitoring of environmental impact should be performed, it is likely that the actual assessments are performed using other methodologies than SEA.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

3.23 Sustainability Assessment (SA)

Underlying Concepts

SA is more a group of approaches than one specific methodology. Dewan (2006) elaborated two key methodologies for Sustainability Assessment viz. monetary aggregation method and physical indicators and further explained that monetary aggregation method is primarily used by economists, whereas physical indicators are used by scientists and researchers (Spangenberg, 2005). The examples of economic approaches include natural resource accounting and modeling, sustainable growth modelling, and defining weak and strong sustainability conditions. Dewan (2006) also classified and discussed economic frameworks such as Lindahl and Solow–Hartwick framework in detail.

Assessed Impacts

Sustainability Assessment is more about system dynamics approach (factors-impact-qualitative-quantitative) and requires programming and these tools are to be elaborated during project while developing project deliverables implementation system. System Dynamics are focused on holistic impact and can be seen as concept combined in distinct computer models that analyse different variables and perform modelling on changing the qualitative and quantitative input of figures.

Suitability for URBANWASTE

Sustainability Assessment includes all other methodologies like CBA (Cost Benefit Analysis), NEB (Net Environmental Benefit), LCA (Life Cycle Assessment), EP&L (Environmental Profit and Loss) and other, this is suggested to be one of the finalizing method when all the data is gathered and put into system dynamics model. Then complete form of SA for whole recreation zone development project is possible, where known parameters of infrastructure and anthropogenic / economic loads are included. This model is a tool where modelling of parameters is possible with building scenarios opportunities. Relevance with URBANWASTE project is indirect as SA is the tool that requires resources comparable to large specific international project – it is holistic model requiring methodology.

Suitability for assessment of changes on hotel level or on municipality level

Project URBANWASTE performance **shall not use this methodology** in the frame of defined aims and tasks as other tools are more appropriate.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

3.24 Total Cost Assessment (TCA)

Total cost assessment (TCA) is a **methodology that was designed in the 1990s for internal managerial decision-making at the company level in the USA**. The **aim is to better select and justify waste management investment decisions that are environmentally sound and reduce long-term liabilities by capturing full range of costs and benefits from pollution prevention projects**. The TCA methodology provides the framework for that decision process, as well as the **framework for estimating baseline costs that have a much broader and potentially longer timeframe** (Centre for Waste Reduction Technology, 2000). The methodology is depicted in Figure 19. **TCA usually captures a longer time horizon using the net present value to discount future cash flows**, however internal rate of return, profitability index and payback period are used (Curkovic and Sroufe, 2007).

TCA belongs to a group of **environmental accounting methodologies**, which include full cost accounting, environmental cost accounting and total cost accounting which all **aim at including environmental cost in the decision process** (Beaver, 2000). The distinction between the various methodologies is now clear, but seems to depend on the scope of the cost allocation. The main cost allocation scopes are business activities that are to be included and the types of costs that should be included in the analysis. For the business activities a distinction can be made between the following activities: research and development, design, manufacturing, marketing and sales. TCA can encompass any combination of these activities (Curkovic and Sroufe, 2007). All authors agree that **both direct and indirect cost should be included in the analyses**. This means that **overhead costs should be included, but also liability costs that result from waste and materials management, and intangible costs such as cost associated to the corporate image**. There is no agreement about the allocation of the external (or societal) costs. In some cases, societal costs are included in a TCA in other cases not (see Social Assessment).

According to Angelakoglou and Gaidajis (2015: 744) the main pros of TCA are that it is useful for **internal managerial decision making and that it can identify hidden costs related to environmental issues.** The main cons are that there is no universally accepted approach available and that **environmental costs to monetary units can be a "challenging task".**

Underlying Concepts

The TCA methodology is **based on a life cycle approach** and is seen by Klöpffer (2003) as the **economic counterpart of environmental life cycle assessment** (in Figure 19 this is called LCI or life cycle inventory). TCA considers both upstream and downstream processes. It also considers quantitative material flows, but only if (direct or indirect) costs for the company are involved.

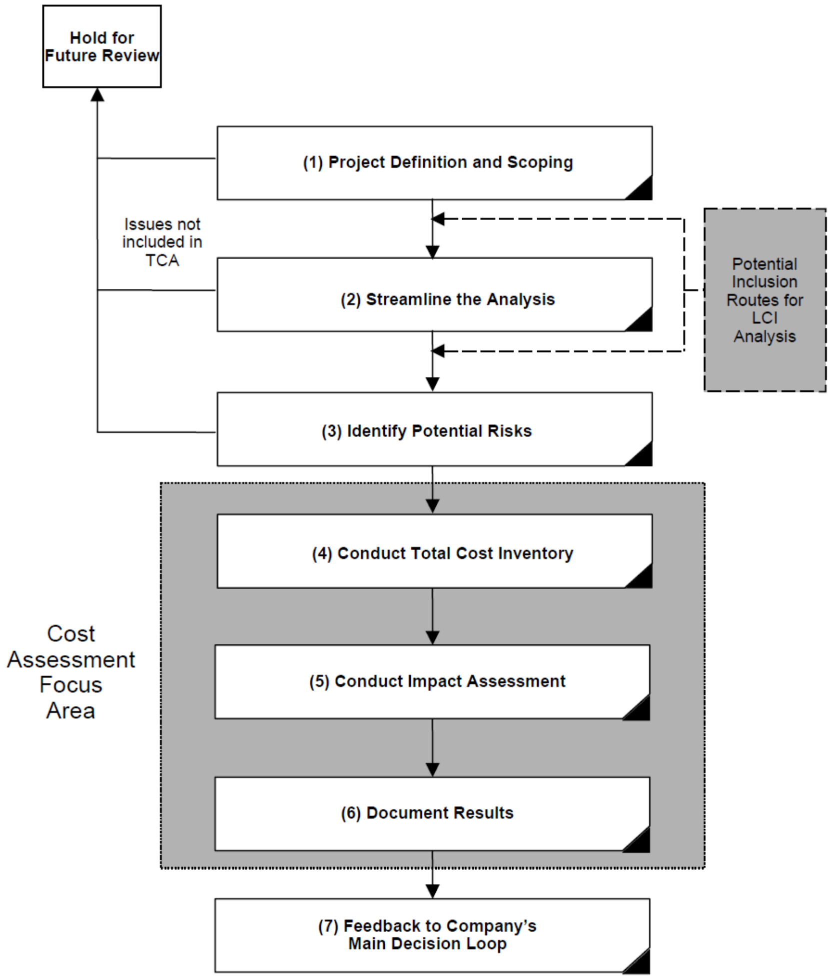


Figure 19: Overview of the TCA methodology. Source: Centre for Waste Reduction Technology, 2000: 3-4.

Assessed Impacts

Social Assessment

There seems to be some disagreement about whether TCA consider both internal and external (societal) costs. Beaver (2000) and Curkovic and Sroufe (2007) state, that TCA is concerned only with costs to the company itself. Social (or societal) costs are seen as externalities as far as they have no financial consequences for a company under current legal and regulatory conditions. However, it is likely that these conditions will change in the future, which implies that costs which are currently external are likely to be eventually internalized by companies (Beaver, 2000).

Other sources of information, such as the often cited TCA manual of the Centre for Waste Reduction Technology (2000), includes 'external costs' which are borne by society and not directly paid by the company in the TCA. In this manually these costs are called 'type V costs'.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Economic Assessment

TCA is a financial tool. It is used for assessing the cost and benefits derived from environmental initiatives and policies to support decision making within firms (Beaver, 2000). Other than financial issues are not covered by TCA.

Environmental Assessment

A live cycle inventory (LCI) is part of the TCA analysis. A LCI quantifies the raw materials used, energy use, and environmental releases associated with the system being evaluated. This includes the acquisition of raw materials, the acquisition of energy resources, processing of raw materials into usable components, manufacturing products and intermediates, transportation of materials to each processing step, use of the product and final disposition (which may include recycling, reuse, incineration or landfill) (Centre for Waste Reduction Technology, 2000: A-5).

From a practical point of view, according to the Centre for Waste Reduction Technology (2000: 1-17), a TCA analysis does not require a new and complete LCI. The use of "nearest neighbour" information can be sufficient.

Suitability for URBANWASTE

TCA is not suitable for URBANWASTE as its scope is too limited. It only assesses financial aspects and it is specially aimed at the company site and or company level. Also TCA seems to be adopted mainly in the manufacturing sector, and hardly the service sector (including tourism).

Suitability for assessment of changes on hotel level or on municipality level

TCA can be used as a 'business case' study for environmental investments on a company level and municipality level. However, the total cost assessment is only done in monetary terms, which do not include societal cost which are borne by society. Particularly this later aspect is relevant at the municipal level.

3.25 Urban and Industrial Symbiosis (UIS)

Urban and Industrial Symbiosis (UIS) could be referred to as a **sub-discipline of the industrial ecology** (Jacobsen, 2006) and a **combination of two equivalent concepts**: **Urban Symbiosis and Industrial Symbiosis** (Mulder, 2016). In the same way as two different biological species live and interact together in an ecosystem, UIS represents the close – and often long-term – interactions between urban and industrial areas. Considering the definition of urban symbiosis formulated by Vernay and Mulder (2015), it could be stated that **UIS is a strategy to create a more efficient metabolism of cities, including both urban and industrial systems**.

UIS **focuses on the built environment** and **refers to initiatives that aim at closing material and energy flows within a given urban and industrial area**. In this sense, households, neighbourhoods, firms, industries, etc. would share or exchange by-products, materials, energy or waste as a way to close specific cycles (Vernay & Mulder, 2015; Walls & Paquin, 2015).

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

By the proximity of urban and industrial infrastructures, a process of symbiosis might be created between urban and industrial systems, which might lead to a considerable reduction and optimization of resource consumption and/or carbon and other emissions of the systems involved. Moreover, it implies technological and social innovations that offer new ways to meet society's demand for products and services. Massard et al. (2014) pointed out, that cities can generally facilitate UIS when developing appropriate infrastructures in urban settings that would also meet the needs of certain industries.

For instance, a successful recycling system integrating urban and industrial sectors might change the energy content of municipal solid waste and affect the waste incinerator that produces district heating (Vernay and Mulder, 2015). Another example where urban and industrial systems are interlinked by means of materials and energy flows is illustrated in Figure 20.

Nevertheless, as Vernay and Mulder (2015) indicated, a successful integration of system requires a high level of consensus among all stakeholders; in terms of technological content as well as regarding the roles of each actor and their interrelation.

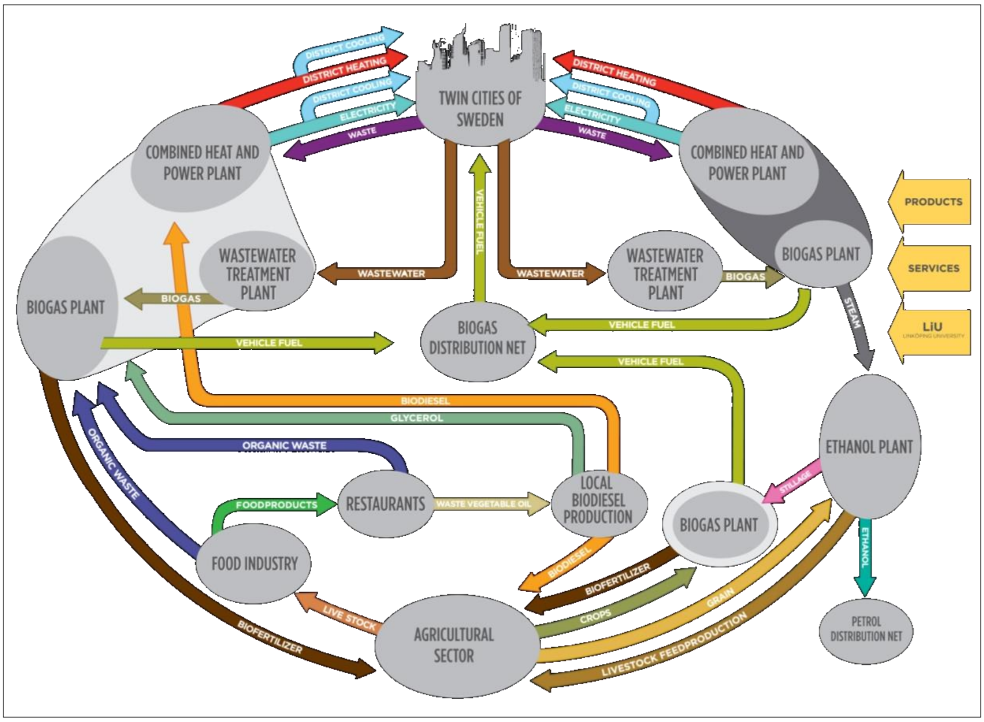


Figure 20: Example of urban and industrial symbiosis in Händelö area (Sweden) (Source: Sustainable Norrköping program, 2014)

Underlying Concepts

UIS and Life Cycle Assessment (LCA)

The application of **UIS initiatives based on LCA has already been reported**. It has been discussed that UIS encourages the interaction between urban and industrial systems so that synergy opportunities can be identified. Geng et al. (2010) simulated and evaluated the link between an innovative municipal solid waste management with local industries, in which materials from urban refuse were transferred directly to industrial applications. The implementation of such initiatives was assessed using a model based on the LCA approach.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

UIS and Material Flow Analysis (MFA)

The **application of UIS strategies is closely related to the understanding and measurement of material flows between specific urban and industrial systems**. Consequently, input-output schemes must be properly defined.

A successful and structured UIS should therefore make use of reliable data provided by MFA and consider different material streams. The compatibility between UIS and MFA is essential.

Assessed Impacts

Social Assessment

UIS requires rearranging pre-existing socio-technical configurations (Vernay and Mulder, 2015). It means that the majority of actions taken towards the achievement of UIS will have an impact on social aspects. Consequently, UIS offers the opportunity to measure social impacts before and after any configuration has been modified. It can be concluded that UIS is suitable for social assessment as it must integrate social aspects since the beginning.

Economic Assessment

Although the UIS is not a methodology to measure economic aspects as such, the exchange and sharing of resources between urban and industrial systems implies an economic benefit (van Berkel et al. 2009). For this reason, the economic assessment of every activity aimed at closing cycles must be fully integrated in the UIS.

Environmental Assessment

The main objective of UIS is to create more efficient metabolic cities. This translates into reduction and optimization of resource consumption and/or carbon and other emissions of the systems involved, among other aspects. Therefore, the environmental benefits of a well-established urban and industrial symbiosis are clear.

It can be concluded that UIS represents a suitable methodology for consideration and assessment of different environmental impacts resulting from the actions implemented.

Suitability for URBANWASTE

The URBANWASTE project would be a great opportunity to further explore the concept of UIS, as it offers the right scenario to work on the general target proposed by UIS: create a more efficient metabolism in cities. The municipal waste stream is one of the key material flows considered in urban metabolism, which can integrate LCA and MFA during the analysis of municipal waste generation, treatment and disposal. Therefore, municipal waste management offers a great opportunity to create symbiosis between different systems and stakeholders involved.

However, as Vernay and Mulder (2015) indicated after studying several urban symbiosis projects, the creation of appropriate and fitting bridges between urban and industrial systems is crucial to the success of their integration, and it needs to be carefully assessed. These bridges could be of technical but also of social nature.

Another important aspect for the successful integration of UIS within the URBANWASTE project is the long term visioning. A future vision supported by the main stakeholders involved will create a framework that facilitates the future presence of implemented strategies and innovations.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Suitability for assessment of changes on hotel level or on municipality level

Urban and Industrial Symbiosis implies the interaction of both urban and industrial systems. In order to achieve this prerequisite, hotels could be considered as part of the urban fabric and it would require interaction with industrial activity. It would imply, for instance, the connection of certain material flow between hotel and industry to evaluate potential changes in the hotel.

Regarding changes on municipal level, UIS is a suitable strategy to be applied at that level. UIS considers districts, municipalities and industries associated as components of a living organism (i.e. city) and this allows the measurement of its metabolic interactions.

Urban and Industrial Symbiosis (UIS) has been introduced as a methodology to break linear relationships between consumption and waste, and by returning outputs as inputs between urban and industrial systems. As Berkel et al. (2009) observed, the geographic proximity of urban and industrial areas allows the use of physical resources discarded in urban areas (i.e. waste) as alternative raw materials or energy source for industrial operations.

In this sense, the URBANWASTE project appears to be the right scenario to apply UIS initiatives and explore resource-waste cycles in tourist cities. The scope of the project offers the possibility of creating a large number of symbioses between houses, buildings, services and industries.

3.26 Water Footprint

The water footprint (WF) is a consumption-based indicator of water use defined as the total volume of water that is used to produce the goods and services consumed by an individual or community (Hoekstra and Chapagain, 2011).

Water footprint assessment (WFA) has raised as a methodology for assessing water use and its environmental impacts on a life cycle basis. Conceptually it can be considered as similar to the ecological and carbon footprint used in the LCA (Jeswanu and Azapagic, 2011).

In 1993, Professor Tony Allan introduced the concept of virtual, or embedded, water to understand how arid countries can feed their people. Building on this concept of virtual water, in 2002, Professor Arjen Hoekstra, whilst working at UNESCO-IHE, created the water footprint as a metric to measure the amount of water consumed to produce goods and services along the full supply chain in response to the need for a consumption based indicator of freshwater use (Hoekstra, 2003). This provided a metric for measuring the virtual water of goods that are produced in one location, and consumed elsewhere.

Interest in the water footprint grew rapidly after its introduction in academic literature. In the mid-2000's, companies, in particular food and beverage companies such as Unilever, SABMiller, Heineken, Coca-Cola, Nestle and Pepsico, became increasingly aware of their water dependence and the water-related risk facing their companies.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

The WF looks at both direct and indirect water use of a consumer and a producer and three key water components are tracked in its calculation:

- the Blue Water Footprint: refers to the consumption of surface and groundwater
- the Green Water Footprint refers to the consumption of rainwater stored in the soil as soil moisture
- the Grey Water Footprint refers to pollution and it is defined as the volume of freshwater required to assimilate the load of pollutants based on existing ambient water quality standards (Galli et al. 2012).

The water footprint thus offers a better and wider perspective on how a consumer or producer relates to the use of freshwater systems. It is a volumetric measure of water consumption and pollution. It is not a measure of the severity of the local environmental impact of water consumption and pollution (Hoekstra et al., 2011).

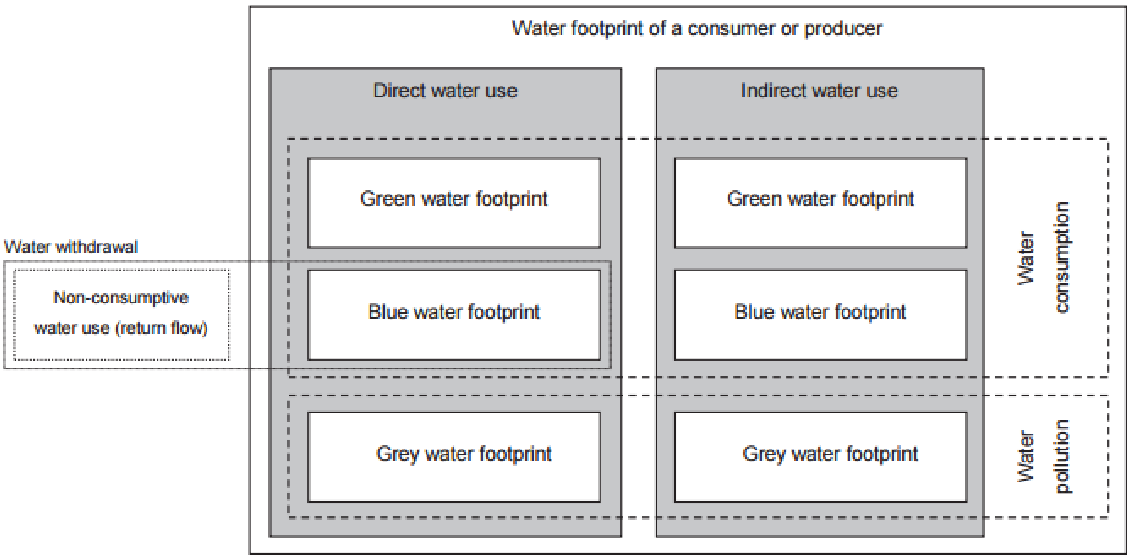


Figure 21: Schematic representation of the components of a water footprint (Source: Hoekstra et al.,2011)

Underlying Concepts

The Water Footprint assessment can be divided in four different phases:

- Setting the goal and the scope
- Water footprint accounting
- Water footprint sustainability assessment
- Water footprint response formulation

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

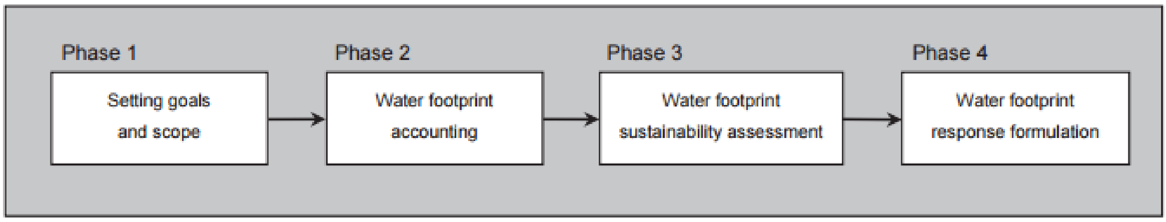


Figure 22: The four phases of the water footprint assessment (Source Hoekstra et al.,2011)

Water Footprint and Life-Cycle Assessment (LCA)

The water footprint of a product or a process can be an indicator in the life cycle assessment (LCA) of the product or the process assessed. Being applied in an LCA is one of the many applications of the water footprint. The debate regarding the possible integration of the WF methodology and the LCA has been deeply analysed in recent and relevant literature. Jeswani et Azapagic, 2011, Jefferies at al, 2012, Pfizer et al. 2009 and Hoekstra et al., 2009 argued in several papers which could be the approach for evaluating the impacts of water use.

Methodologies, approaches and indicators for assessing the impact of freshwater usage are still evolving. Most of the methodologies and approaches assess the quantity of water used rather than the related impacts.

Although there is a recognised need to consider the related impacts, particularly on a life cycle basis, the difficulty is that there are little or no reliable data on water usage in life cycle databases; furthermore, there is no agreed life cycle impact assessment method for estimating impacts related to freshwater use. However, several methodologies for inventory modelling and impact assessment for water use in LCA have been proposed and analysed. Most LCA studies that report water use simply the total input without differentiating between the in-stream and off-stream uses or between freshwater and seawater. Most of the databases do not consider systematically wastewater discharges and hence do not report consistently water outputs. The distinction between the input and output water flows is essential for an adequate assessment of the potential impacts of water use. Because of these inadequacies at the inventory level, LCA is not generally considered suitable for quantifying water use. In an attempt to address these issues, several approaches have been proposed for water use modelling in LCI (Milà i Canals et al., 2009; Pfister et al., 2009). In addition to the LCA-related methodologies, the water footprint approach, developed in the context of water resource management, also provides a potentially useful methodology for quantifying water use for LCI.

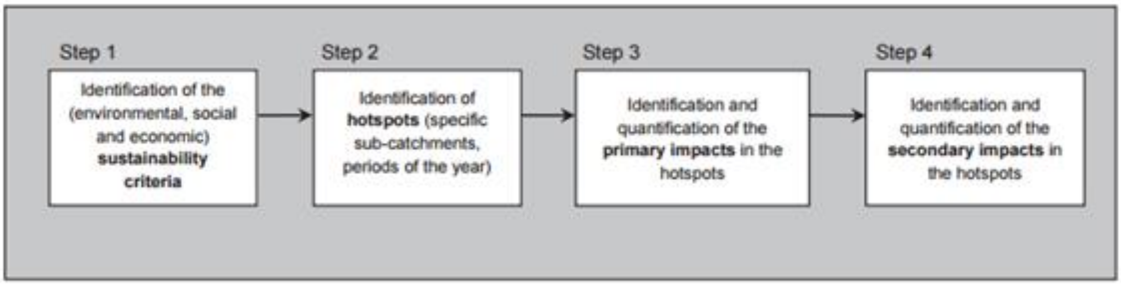
Water Footprint and Materials Flow Analysis (MFA)

MFA is descriptive, systematic approach to assess the metabolism of a defined system and is based on the principle of mass conservation (Brunner and Rechberger, 2004). In this sense the water footprint can be considered as particular fragment of the MFA that take in consideration just the volume of water used in one process.

Assessed Impacts

The water footprint within a catchment needs to meet certain criteria in order to be sustainable. Sustainability has an environmental dimension as well as a social and economic dimension.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452



Economic Assessment

A practical example describing of an economic assessment of the water footprint can be found in Aldaya et LLams, 2009. In this sense the paper analysed a case study of the Guadiana river in Spain, assessing firtstly the hydrological data and secondly the economic data. In this sense the paper analysed the water footprint, virtual water and economic relevance of each economic sector in different rainfall years (evaluating an average - 2001, dry -2005, and humid year -1997). Special emphasis has been given to the agricultural sector, which consumes about 95% of total green and blue water resources. Based on this consideration the estimated economic values of such water could be used to develop on the one side recommendations to decision makers that will be responsible to design water management strategies, and on the other side to assess the economic sustainability of the processes.

Environmental Assessment

To be environmentally sustainable, water use must not exceed the maximum sustainable limits of a freshwater resource. Blue water scarcity is used to measure the environmental sustainability of the blue water footprint. It's a measure of the blue water footprint compared to the water available after considering environmental flow requirements. When the blue water footprint is larger than the available water, environmental flows are not met and over time, freshwater ecosystems degrade.

When we consider the environmental sustainability of water use from the perspective of water quality, we compare the grey water footprint with the available assimilation capacity to measure the water pollution level. If the grey water footprint exceeds the assimilation capacity water quality standards are violated and the quality of the water will not meet socially agreed upon purposes.

Both, blue water scarcity and water pollution levels, are assessing the cumulative impact of all water uses of the freshwater resource. This can be done for sub-catchment or a local aquifer all the way up to large river basins and regional groundwater reserves (Hoekstra et al., 2011)

Social Assessment

A minimum amount of the freshwater available on Earth needs to be allocated to 'basic human needs', most notably a minimum domestic water supply for drinking, washing and cooking and a minimum allocation of water to food production to secure a sufficient level of food supply to all. A minimum domestic water supply for drinking, washing and cooking needs to be guaranteed at the catchment or river basin level. Moreover, a minimum allocation of water to food production is to be secured at global level, since river basin communities are not necessarily self-sufficient in food, if food security is ensured through food imports (Hoekstra et al. 2011). In this sense, basic human needs and rules of fairness are criteria that are difficult to quantify in the form of sharp boundaries. Whether water-related basic human needs or rules of fairness in a certain catchment are violated will depend on expert judgement. The existence of conflicts over water can be a

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

practical indication (Hoekstra et al., 2011). In practice, social conflicts over water will often arise at the same time as when environmental conflicts occur. Therefore, no set of indicators exists to clearly define social impacts, but it is based on a subjective judgement.

Suitability for URBANWASTE

URBANWASTE may adapt the water footprint assessment in some particular cases where a specific research on water will be requested, but not to assess the overall urban metabolism. Water cycle was also not mentioned in the project's objectives, but some partners demonstrated their interest in water related issues and they can benefit from the implementation of this methodology.

Suitability for assessment of changes on hotel level or on municipality level

The water footprint assessment methodology can be interesting for hotels based in areas where water scarcity is an increasing issue (Syracuse, Florence, Tenerife, Cyprus, Kavala, Nice, etc.). Hotels could in fact assess their water consumption and redesigning their water management in a more sustainable way, reducing wasted water and improving overall performances.

Several hotels all around Europe and not only are trying to redesign their water management to reduce their water footprint. In this sense the International Tourism Partnership has recently launched the Hotel Water Measurement Initiative (HWMI) that aims at allowing any hotel anywhere in the world to measure and report on the water footprint of a hotel stay or meeting / event (http://www.greenhotelier.org/our-themes/water/global-hotel-groups-collaborate-for-industry-first-on-water-measurement/).

Furthermore, the EMAS scheme promotes Best Environmental Management Practice in the Tourism Sector, including an entire chapter on water assessment and management (JRC, 2013).

Summary / Conclusions

Freshwater availability is an increasing issue and it is crucial both for businesses and local authorities to pay more attention to water consumption and management. On the one hand, businesses could benefit from WFA to assess their own consumption while local authorities could take advantage of the implementation of WFA to understand and better manage water use at a river/catchment level.

The water footprint measures the amount of water used to produce each of the goods and services we use. It can be measured for a single process, such as growing rice, for a product, such as a pair of jeans, for the fuel we put in our car, or for an entire multi-national company. The water footprint can also tell us how much water is being consumed by a particular country – or globally – in a specific river basin or from an aquifer.

Water Footprint Assessment is a comprehensive methodology aiming at understanding and quantifying water uses dividing them in green, blue and grey water footprints, for better managing those.

The WFA could be useful for government, river basin authorities and companies- hotel in this case- to monitor their dependence on scarce water resources alongside their supply-chain.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

4. Conclusions: A set of methodologies suitable for URBANWASTE

In order to be suitable for the URBANWASTE project, a method has to allow an accompanying assessment of environmental, economic and social impacts of the current situation ("baseline") in the URBANWASTE pilot cases and support the development of selected strategies aiming at reducing the amount of municipal waste production and at further support the re-use, recycle, collection and disposal of waste in tourist cities.

4.1 Grouping of methodologies

The detailed analysis of the **26 methodologies** allowed grouping of them into the following three categories:

1. **Methods directly covering assessments**: these methods include the direct assessment of certain impacts in a sustainability assessment (could be environmental, economic or social impacts). The authors consider three sub-sets of methodologies within this group:
    - Environmental assessment: LCA, subset methodologies of LCA such as EF, CF (corporate and product carbon footprint), WF
    - Cost-related assessment: CBA, EIOLCA, ABC, LCC, EP&L, TCA and EE
    - Social assessment: LCWE, SLCA
2. **Methods used for visualisation purposes**: these methods are not directly covering an assessment of impacts, but are more related to display or visualise. MFA / EFA, for example, are not providing information on impacts, but are displaying material flows in a system which is requirement for a subsequent impact assessment. Other methods considered as being part of this group are IS and UIS.
3. **Methods used to structure / prioritize**: this group of methods helps to prioritize the results of certain impacts, e.g. within a group of impact categories (e.g. impact categories related to environmental assessment) or within certain categories of impact assessment. These methods aim at making individual results comparable and deal with issues of weighting and normalizing individual results. For example, sometimes it is important to find a way to provide overall results of assessments in order to compare certain scenarios. This group also contains methods aiming at structuring the process of impact assessments, such as SEA and EIA. MuSIASEM, SA, CSR, AHP, CRA, MCDM, BSC and DPSIR are further methods belonging to this group.

4.2 Selection of methodologies

By applying the criteria described in Chapter 2 to judge on the suitability of a reviewed method to meet the URBANWASTE project's objectives, out of the total 26 methodologies a set of 6 methods has been identified as being suitable. The detailed results are displayed in Table 10.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Table 10: Results of the assessment of methodologies based on a set of five criteria

		Criterion I	Criterion II	Criterion III			Criterion IV	Criterion V	
		Methodology…		Suitability for social, economic and environmental assessment			Suitable for URBANWASTE	suitability for assessment of changes based on implementing measures…	
	Name of Methodology	…based on a life cycle perspective	…considers or allows the consideration of MFA	**III a.** Social assessment	**III b.** Economic assessment	**III c.** Environmental assessment		… on hotel level	… on municipal level
1	Activity-Based Costing (ABC)	Yes	n/a	partly	Yes	Yes	No	Yes	Yes
2	Analytical Hierarchy process (AHP)	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
3	Balanced Scorecard Approach (BSC)	No	No	Yes	Yes	Yes	No	Yes	No
4	Carbon Footprint (CF, Corporate Carbon Footprint, Product Carbon Footprint)	Yes	Yes	No	No	Yes	partly	Yes	Yes
5	Comparative Risk Assessment (CRA)	No	No	Yes	Yes	Yes	No	No	No
6	Corporate Social Responsibility (CSR)	No	No	Yes	Yes	Yes	No	Yes	Yes
7	Cost-Benefit Analysis (CBA)	Yes	n/a	Yes	Yes	Yes	partly	Yes	Yes
8	Drives-Pressures-State-Impact-Response (DPSIR)	Yes	Yes	Yes	Yes	Yes	partly	No	Yes
9	Eco-Efficiency (EE)	Yes	Yes	partly	Yes	Yes	Yes	Yes	Yes
10	Ecological Footprint (EF)	partly	no	partly	No	Yes	partly	Yes	Yes
11	Economic Input-Output (EIO)	Yes	n/a	No	Yes	partly	No	No	No
12	Material Flow Analysis (MFA) and Energy Flow Analysis	No	Yes	No	No	No	Yes	partly	Yes
13	Environmental Impact Assessment (EIA)	Yes	Yes	No	No	Yes	No	No	partly
14	Environmental Profit and Loss (EP&L)	Yes	Yes	No	Yes	Yes	No	Yes	No
15	Industrial Symbiosis (IS)	Yes	Yes	No	Yes	Yes	partly	No	No
16	Life Cycle Assessment (LCA)	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
17	Life Cycle Costing (LCC)	Yes	No	Yes	Yes	Yes	partly	partly	partly
18	Life Cycle Working Environment (LCWE)	Yes	Yes	Yes	Yes	No	No	No	No
19	Multi-Criteria Decision Making (MCDM)	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
20	Multi-Scale Integrated Analysis of Societal and	partly	Yes	Yes	Yes	Yes	partly	partly	Yes
21	Social Life Cycle Assessment (SLCA)	Yes	No	Yes	No	No	partly	Yes	Yes
22	Strategic Environmental Assessment (SEA)	No	No	No	No	Yes	No	No	No
23	Sustainability Assessment (SA)	Yes	n/a	Yes	Yes	Yes	No	No	No
24	Total Cost Assessment (TCA)	Yes	Yes	No	Yes	No	No	Yes	no
25	Urban and Industrial Symbiosis (UIS)	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes

It can be seen, that almost three quarters of the assessed methods are based on a life cycle perspective, which is an important consideration in assessing sustainability. A little bit more than 50% of the methods consider or at least allow the consideration of material flow based data as input for assessments. Similarly, 54% of the methods contain elements allowing for considering social aspects in the assessment. A higher share of the methods is working with economic data and allows economic assessments (76%). 81% of all reviewed methods are dedicated to be suitable for environmental assessments.

The overall suitability ("yes") for URBANWASTE was assessed with 23% (6 methods). 8 methods (31%) will be considered only "partly" in the future project activities, meaning some indicators based on these concepts will be used or methods related to structuring / prioritizing and visualisation are considered. 12 methods (46%) are not considered in URBANWASTE. The results of the methodology selection based on criterion IV are displayed in Table 11.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Table 11: Results of methodology selection based on Criterion IV

	Name of Methodology	Criterion IV suitability for URBANWASTE	Reason
1	Activity-Based Costing (ABC)	No	Too complex in the implementation and therefore time and resources consuming. It is recommended to include not the methodology itself, but to consider the fact of including indirect costs as good as possible in order to have real costs. Of course the extent of including indirect costs depends on the scenarios and strategies for implementation.
2	Analytical Hierarchy process (AHP)	Yes	AHP is an effective tool to deal with complex decision making processes, helping the decision maker to set priorities and make the best decision. By reducing complex decisions to a series of pairwise comparisons, and then synthesizing the results, the AHP helps to capture both subjective and objective aspects of a decision. Its use is therefore recommended in combination with other methodologies such as LCA and MFA.
3	Balanced Scorecard Approach (BSC)	No	BSC is a concept to measure, document and control the activities of a company / organisation related to its vision and strategy. The method offers a tool for translating an organization's strategy into specific objectives. Although the BSC approach has been introduced as a very useful tool to achieve a balance between different perspectives on the basis on targets, key performance indicators (KPIs) and measures, and to evaluate the economic, social and environmental aspects of an organisation, this tool is not the right method to be used in URBANWASTE.
4	Carbon Footprint (CF, Corporate Carbon Footprint, Product Carbon Footprint)	Partly	Carbon footprinting is just partly considered as suitable for the project, as CF is only a smaller part of the more standardised method of LCA. Corporate carbon footprints are used as background information in the assessment procedure, for example information available under: www.hotelfootprints.org; www.bookdifferent.com
5	Comparative Risk Assessment (CRA)	No	CRA is not suitable to answer specific URBANWASTE questions related to the impact of changes resulting from implementing certain waste management measures in the URBANWASTE pilot cases as this tool usually is applied to evaluate and rank the risks a given set of problems pose to the natural environment, to human health or to the quality of life and not for assessing the impacts of certain measures such as waste prevention activities.
6	Corporate Social Responsibility (CSR)	No	This methodology does not include assessments of sustainability and is therefore not suitable to answer specific URBANWASTE questions. However, the implementation of CSR in organisations could be used as a strategy to improve sustainability, but these improvements should then be assessed with a supplementary methodology.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

	Name of Methodology	Criterion IV suitability for URBANWASTE	Reason
7	Cost-Benefit Analysis (CBA)	Partly	CBA is an analytical tool for judging the economic advantages or disadvantages of an investment decision by assessing its costs and benefits in order to measure the welfare change attributable to it. A value is ascribed to all effects, converting them into monetary units. The various costs and benefits may occur at different times in the future, which is why a so-called net present value is calculated to illustrate the aggregated value in today's prices. For this calculation, a discount rate is used. Problem in the application for URBANWASTE is to translate all effects into monetary units and to assign respective discount rates. Yet, for economic assessment the concept is considered.
8	Drives-Pressures-State-Impact-Response (DPSIR)	Partly	This conceptual framework helps to organize and structure indicators in the context of a so-called causal chain that links indicators of the environmental driving forces, to pressure indicators, to environmental state indicators, to impact indicators and finally to indicators of societal response. URBANWASTE could use the DPISR framework together with an LCA approach to assess and evaluate the municipal solid waste management system of the pilot cases involved in the project. But overall it has to be concluded that DPSIR is more a causal framework for describing the interactions between society and the environment than an assessment methodology. But it could support the understanding of the driving forces and the pressures that the tourism industry has on the entire waste management system, thus helping in the development of tailored strategies.
9	Eco-Efficiency (EE)	Yes	Generally, EE is defined as the ratio between economic performance and environmental influence, EE can be used in relation with LCA to make a linking of economic costs and environmental impacts in a systematic manner. Though EE is mostly attributed to LCA, EE indicators can also be derived from a combination of MFA factors and economic indexes. Economic data from the defined system is used to derive the EE indicators for material and energy inputs, consumption and emission outputs. The EE method is highly flexible and adaptable, and relatively simple to communicate which makes it an appropriate tool to meet the objectives of URBANWASTE. This means, the concept of EE is used, when cost-related aspects are assessed. The selection of indicators depends on data availability (see Task 2.3 to 2.5) and the indicators shall display the scenarios and implementation measures adequately.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Name of Methodology		Criterion IV suitability for URBANWASTE	Reason
10	Ecological Footprint (EF)	Partly	EF can be an indicator in LCA, evaluating the impact on land of a certain product or process. Main purpose of the EF is the assessment of human consumption's impact on the environment, i.e. the use of natural capital and more specific the use of global hectares land (gha). EF could be appropriate for dissemination and communication campaigns. If in URBANWASTE a use of LCA is carried out for environmental assessment, then EF might be considered, but this depends on data availability.
11	Economic Input-Output (EIO)	No	The method has several problems for implementation in URBANWASTE. The method uses the calculation of whole sectors, this means on the one hand aggregates are used, that are presenting "averages" within the sector. The more diverse a sector is, the less representative the results might be for specific parts of the sector. Tourism on the one hand might be considered uniform in the products and services delivered, but results cannot be derived at local level (cities) or at lower levels (accommodation types). Secondly, it is difficult to allocate the environmental impacts to the EIO models. Therefore, the method is considered not suitable for the project.
12	Energy Flow Analysis (EFA) and Material Flow Analysis (MFA)	EFA (no) MFA (yes)	MFA traces the input, storage, transformation, and output processes and it allows following the material flows throughout the life cycle within an urban system, based on the physical principle that matter can neither be created nor destroyed. For URBANWASTE only the MFA is chosen, as quantities of wastes and materials (depending on data availability) are prerequisite for sustainability assessments.
13	Environmental Impact Assessment (EIA)	No	EIA sets a methodological framework for assessing environmental impacts of a project. The approach seems too broad in the sense of URBANWASTE. EIA is a methodological framework and foreseen for much bigger projects than the hotel level. On municipality level SEA would be a more appropriate methodology. Therefore, it can be concluded that EIA is not suitable for URBANWASTE.
14	Environmental Profit and Loss (EP&L)	No	In EP&L, the municipal level is hardly to be approached and in general the EP&L is better for products and not for services nor a group of services and interactions like the tourism is. The EP&L analysis provides monitoring the footprint of the company's operations and identifying new opportunities to enhance the sustainability of a company's products - product/impact in metric values. Therefore, project URBANWASTE performance shall not use this methodology in full manner, however some scientific idioms are useful for defining monetary environmental expenditures in specific cases.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

	Name of Methodology	Criterion IV suitability for URBANWASTE	Reason
15	Industrial Symbiosis (IS)	Partly	IS is not by itself a methodology to measure quantity, qualitative composition or treatment of waste materials. Yet the idea of closing loops and circular economy considerations are part of the assessment procedures in URBANWASTE.
16	Life Cycle Assessment (LCA)	Yes	LCA is highly relevant to URBANWASTE as it is a necessary method for evaluating the targets of the project. Only with LCA environmentally-related targets such as cutting GHG and fresh water use can be measured. The methodology of LCA is convertible to a high extent as an international standard method exists. Technical and operational feasibility can be achieved by using standard LCA methods and social and cost-related targets can also be evaluated by this approach. The potential to display material flows and to estimate their impacts by using LCA is without controversy. LCA is compatible with EU policy and therefore is suitable for the purpose of URBANWASTE.
17	Life Cycle Costing (LCC)	Partly	Regarding the project URBANWASTE it might be possible to use a very reduced form of LCC, that cannot be called LCC anymore, as it only covers specific parts of the waste management system. One option could be to focus on a clear set of defined costs / revenues in a system, that are available, in good quality and that are representable (e.g. for future scenarios in an assessment). This is important to reduce the time efforts to a minimum, the drawback is that only parts of systems and parts of life cycles can be considered, this may lead to the effect, that important cost / revenue aspects are not included or important social and environmental impacts are not considered.
18	Life Cycle Working Environment (LCWE)	No	As the methodology is based on generic statistical data at national level and much more as the methodology is developed and suitable only for product LCAs it is not suitable to evaluate the service related impacts of tourist related waste management activities.
19	Multi-Criteria Decision Making (MCDM)	Yes	MCDM is a decision-making tool that facilitates the selection of the best alternative among several alternatives. It evaluates a problem by comparing and ranking different options and by evaluating their consequences according to the criteria established. Not many references have been found about the combined use of a general MCDM model with LCA. But the method is considered in URBANWASTE by using a combined model of MCDM and AHP.
20	Multi-Scale Integrated Analysis of Societal and Ecosystem Metabolism (MuSIASEM)	Partly	The method can only be considered partly, as indicators will be used, that fit to the concept.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Name of Methodology		Criterion IV suitability for URBANWASTE	Reason
21	Social Life Cycle Assessment (SLCA)	Partly	The method is suitable to answer specific URBANWASTE questions with a social impact. This is a method suitable for assessing changes on both hotel and municipality level, however, since there are many possible social impacts to concern in large and complex systems there is probably a need for simplifications. Since social impacts (both positive and negative) often are difficult to quantify and to compare with other social impacts one major difference to the environmental LCA is the lack of quantitative and weighted results. This means, the life cycle perspective is included related to data availability. Based on this, social indicators will be developed, and the LCC concept is considered in this process.
22	Strategic Environmental Assessment (SEA)	No	SEA is not a method for assessing the impact of decisions, but more a method of using the information from assessment methods to ensure that these are considered in public planning with potential impact on environment and health (similar is EIA). SEA is not a suitable method answer specific URBANWASTE questions since it is used to ensure environmental concerns in public planning rather than monitor the effects of the plans. However, since the SEA Directive require monitoring there are a likeliness that this method will indirectly provide data to be used in other methods assessing waste strategies.
23	Sustainability Assessment (SA)	No	Sustainability Assessment includes all other methodologies like CBA (Cost Benefit Analysis), NEB (Net Environmental Benefit), LCA (Life Cycle Assessment), EP&L (Environmental Profit and Loss) and other, this is suggested to be one of the finalizing method when all the data is gathered and put into system dynamics model. Then complete form of SA for whole recreation zone development project is possible, where known parameters of infrastructure and anthropogenic / economic loads are included. This model is a tool where modelling of parameters is possible with building scenarios opportunities. Relevance with URBANWASTE project is indirect as SA is the tool that requires resources comparable to large specific international project.
24	Total Cost Assessment (TCA)	No	Despite its life cycle approach, the method TCA is not considered in the URBANWASTE context, as the cost-related aspects are partly covered in the well-known CBA and in the more standardised LCC.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

	Name of Methodology	Criterion IV suitability for URBANWASTE	Reason
25	Urban and Industrial Symbiosis (UIS)	Yes	IS was considered partly as suitable for URBANWASTE above in this Table. In this sense, the URBANWASTE project appears to be the right scenario to apply UIS initiatives and explore resource-waste cycles in tourist cities. The scope of the project offers the possibility of creating a large number of symbioses between houses, buildings, services and industries. This method will be considered more in the scenario building process for future waste management / prevention options.
26	Water Footprint (WF)	No	URBANWASTE resources are to be exhausted when the total urban metabolism is balanced and assessed. Despite the fact, that touristic water consumption and wastewater treatment is important, the water cycle was not mentioned in the project's objective. Therefore the focus is on solid waste and not related to wastewater.

As Table 11 shows, several of the reviewed methodologies can be considered suitable to answer specific URBANWASTE questions. Some of these methods are "real assessment" methods, others are subsets of other methods (e.g. the whole "footprinting methods"), others are used for structuring results of prior assessments, others are important to visualize or structure e.g. material and waste flows in a system. Some methods can be used "standalone", some consider integrated assessment options, some only consider one specific (environmental) aspect in their assessment. However, especially for MFA and LCA it is true that the original methodologies are more suitable for comprehensive sustainability assessments than subset methods that only focus at certain aspects. Furthermore, the results of the literature review carried out within task 2.1 of this project show that MFA and LCA are commonly used for assessments with goals similar to the ones of URBANWASTE. Thus, **the decision to apply MFA and LCA for environmental assessments within URBANWASTE would ensure comparability of the results with a wide range of other studies**.

4.3 Assessment approach

In order to meet the objectives of the URBANWASTE project, it seems to be reasonable to **choose a modular design for the assessments to be performed within URBANWASTE**:

1. Based on an **MFA,** which will provide an **inventory of material / waste flows** and thus will lay the **basis for the subsequent impact assessment**,
2. an **LCA** will be carried out as especially **for assessing environmental impacts** the use of a life cycle based approach is important.

   The LCA **impact categories** that the authors consider relevant for URBANWASTE are **Global Warming Potential (GWP), Acidification Potential (AP) Eutrophication Potential (EP) and Resource Depletion**. GWP is seen as a relevant impact category as all touristic activities as well as products consumed by tourists and, thus, are becoming or producing waste, are somehow related to energy consumption.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Thus, their environmental impact can be expressed through CO_2-equivalents. The GWP also considers the issues raised in the carbon footprinting method. As **food waste** was identified in Deliverable D2.1 as being one of the most important waste streams related to tourism AP and EP also should be considered for assessing the environmental impacts of touristic activities.

3. As this project also aims at implementing strategies and measures aiming at reducing waste production from tourism / touristic activities on municipal or hotel level respectively, an **assessment of costs** is important as well. Therefore, in a third step, the method of **Ecological Efficiency (EE)** will be applied to assess the **economic impacts of the strategies and measures** that will be developed within this project. Accompanying this, concepts of other cost-related methods are considered, e.g. from **CBA** and **LCC**.

4. To cover all three main aspects of sustainability, also **social impacts** have to be assessed. In Europe, social sustainability is rarely discussed, probably because the prevailing standards are very high. Low-wages and child labour for example, which are typically applied social indicators for assessments on global scale, are not an issue in Europe. As URBANWASTE focusses on waste generation from tourists and touristic activities and, thus, on services related to tourism with local impacts mainly, the global view seems negligible. Nevertheless, despite the high standards in Central Europe, it is still worth to take a look at the definition and the compliance with these standards. As existing methods for assessing social sustainability tend to have a global perspective and to focus on processes or producers, **only individual parameters** will be selected and analysed within URBANWASTE, but under consideration of general aspects of one methodology (**SLCA**).

5. The group of methods related to **structure or rank results of sustainability assessments** or to build / structure / prioritize scenarios is represented with the methods **AHP** (similar to MCDM) and partly DPSIR.

Table 12 displays an overview of the selected methods ("yes") and the methods and approaches that will be additionally considered in the assessment ("partly").

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Table 12: Overview of selected methods and additionally considered methods / concepts

Assessment part	Selected method	Additional considered method
Structuring data and visualization of waste and material flows	Material Flow Analysis (MFA)	----
Environmental assessment	Life Cycle Assessment (LCA)	----
Economic assessment	Ecological Efficiency (EE)	Cost Benefit Analysis (CBA) and Life Cycle costing (LCC)
Social assessment	Individual indicators	Social Life Cycle Assessment (SLCA)
Structuring / ranking of results of sustainability assessment	Analytical Hierarchy Process (AHP)	Driving forces – Pressures – States – Impacts – Responses Framework (DPSIR)
Scenario building	Urban and Industrial Symbiosis (UIS) approaches	----

4.4 Impact categories and indicators

In the following section a first proposal of possible impact categories and allocated indicators to be used within URBANWASTE is given. Indicator in this work is understood as "Quantifiable representation of an issue of concern (impact category) to which life cycle inventory results may be assigned". It has to be taken into account that depending on the availability of data and especially depending on the planned waste prevention activities to be assessed the impact categories might be changed.

Environmental assessment criteria

Climate change

The indicator used for the category climate change is the Global Warming Potential GWP. This category is based on the emissions of greenhouse gases and the endpoint is the increase of the global average temperature. The Intergovernmental Panel on Climate Change (IPCC) defines the GWP as an indicator that reflects the potential relative climate change effect per kg of a greenhouse gas over a fixed time period, such as e.g. 100 years (GWP100). According to EC-JRC (2009), the carbon footprint is quantified using the indicator Global Warming Potential (GWP).

The substances normally considered as contributors to global warming are:

- Carbon dioxide (CO_2)
- Methane (CH_4)

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

- Nitrous oxides (N_2O)
- CFC's
- HCFC's
- HFC's
- Halons
- Tetrachloromethane
- 1,1,1-Trichloroethane (CCl_3CH_3)
- Carbon monoxide (CO)

Indirect measurable indicators:

- Use of Electricity
- Use of Fuel

Acidification

The indicator for the category acidification is e.g. the Acidification Potential AP which is based on acidifying emissions like NH3, NOX or SO2. These emissions have impacts on vegetation and on the aquatic biodiversity.

Acidification is also a category with transparent calculation. Differences in the case studies may occur because of combustion processes as NOX source. This category is related to energy production, transportation and some industrial processes, so it may be relevant for the URBANWASTE case studies.

Proper characterisation models do exist for this category. Acidification can be modelled by various methodologies, e.g. by applying the methodology proposed by the European Commission (2010b).

Substances normally considered as contributors to acidification are:

- Sulfur dioxide (SO_2)
- Sulfur trioxide (SO_3)
- Nitrogen oxides (NO_x)
- Hydrogen chloride (HCl)
- Nitric acid (HNO_3)
- Sulfuric acid (H_2SO_4)
- Phosphoric acid (H_3PO_4)
- Hydrogen fluoride (HF)
- Hydrogen sulfide (H_2S)
- Ammonia (NH_3)

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Eutrophication

There is terrestrial and aquatic eutrophication which is based on emissions of nutrients like phosphor and nitrogen. Eutrophication has impacts on the vegetation and on the aquatic biodiversity because it leads to a lack of O_2. Eutrophication is mainly caused by agriculture and fertilising and was considered a relevant impact category because of the high food waste amounts caused by touristic activities. Because of this connection eutrophication will be used as an assessment category for the assessments carried out within URBANWASTE.

This category counts also to standard categories with transparent calculations.

Substances normally considered as contributors to eutrophication are:

- Nitrogen oxides (NO_x)
- Ammonia (NH_3)
- Nitrous oxides (N_2O)
- Ammonium (NH_4)
- Phosphate (PO_4)

Resource depletion

An indicator for resource depletion is e.g. shortage of mineral resources. It is therefore due to inputs of non-renewable energy like coal, oil, bauxite etc. Because of the decrease of non-renewable resources the future generation will suffer. The relevance for URBANWASTE case studies was considered as high, as it is one of the targets to reduce waste and resources consumption.

The category is easily to measure in case of fossil energy use on a production site. It is modelled by various methodologies, although they are not considered to include all aspects they should. The method CML 2002 includes non-renewable resources (fossil fuels and minerals). In Guinée et al. (2002) only the ultimate stock reserves are included, which refers to the quantity of resources that is ultimately available, estimated by multiplying the average natural concentration of the resources in the earth's crust by the mass of the crust (Guinée, 1995). In Oers et al. (2002), additional characterisation factors have been listed on the basis of USGS economic reserve and reserve base figures in addition to the ultimate reserve. The characterisation factors are named 'abiotic depletion potentials' (ADP) and expressed in kg of antimony equivalent, which is the adopted reference element. The abiotic depletion potential is calculated for elements and, in the case of economic reserves and reserve base, several mineral compounds. The CML method is recommended in the ILCD framework since it captures scarcity by including extraction as well as reserves of a given resource. Characterization factors are given for metals, fossil fuels and, in the case of reserve base and economic reserves, mineral compounds (European Commission 2010a).

As another indicator that could be used in this connection is the Cumulative Energy Demand (CED), which is broadly used in LCA studies. This indicator is based on non-renewable and renewable energy demand and therefore linked to the category climate change. It should be included in the assessment next to the Global Warming Potential GWP as CED is well known beyond stakeholders.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Economic criteria

As stated in chapter 4.3 parts of the assessment of economic impacts will be carried out by applying LCC together with other methodologies. The main differences between existing LCC methodologies concern the selected cost categories that are considered as relevant. As a starting point, a list of general cost categories was adopted from Woodward (1997). Additional costs which could be relevant for URBANWASTE purposes were added. Advantages and disadvantages need to be taken into account according to partner feedback on data availability.

Material costs

Material costs cover raw materials and operating materials. Material costs are a basic item of any cost accounting.

Data are relatively easy available to any company management. Raw materials may be generated from the documentation of materials of the components from BOM (Bill of materials). A documentation of hazardous materials may also be available. Documentation is easily carried out for simple, mass-relevant products. Operating materials have assumingly lower mass-intensity than raw materials.

Operating costs

Operating costs consist of electricity, water, air, etc., operating manpower and costs for environmental compliance. Costs for electricity, water, air etc. and operating manpower are also included in any financial accounting. Data shall be easily available for company management.

Higher costs for operating manpower indicate increased employment which is a positive social impact. These costs are easy to obtain, if cost drivers such as x hours working time per unit are known. Allocation problems may also occur if cost accounting or process controlling is not done in the company.

Costs for environmental compliance allow demonstrating savings due to improved performance – e.g. less waste and lower toxicity of waste means lower disposal fees. These costs may be difficult to obtain from companies.

Maintenance costs

Maintenance costs contain corrective maintenance labour, preventative maintenance labour and parts. These costs are only important on the manufacturer level and seem less important for the URBANWASTE related touristic activities.

Transport costs

Data on transport costs are mainly relevant on the individual tourist level and therefore seem be difficult to obtain within URBANWASTE. But as transport costs are an important issue in tourism it might be necessary to include them, depending on the individual pilot cases. The final decision on whether to include this cost category or not has to be made at a later stage of the project.

Disposal costs/resale value

Disposal costs are important to include, as the waste prevention potential of different scenarios will be investigated within URBANWASTE. Additionally, disposal costs may be a very important reason for initialising and industrial symbiosis. Data availability is also given. Data can also be calculated.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

End of Life costs for postconsumer waste

End of Life (EOL) costs for postconsumer waste cover costs outside of production phase or out-plant costs, but are within the scope of a life cycle approach. The following costs have been discussed:

- separate collection costs, born by commune
- disassembly costs, born by the recycler
- costs of reprocessing of secondary materials
- revenues from secondary materials

Separate collection costs born by the municipality can be important especially for IT products or refrigerators for the purpose of providing a full scope of the costs, not only those born by the hotels. The difficulty is the data availability. Sources could be municipalities, private collection companies, associations of producers. These costs can be especially relevant for potential ReUse Pilot Actions as they, for example, can show the difference between the current mixed WEEE collection (commingled collection) and the value preserving collection with a purpose of reuse.

Costs for reprocessing of secondary materials can be compared with the costs of raw/operating materials and show whether industrial symbiosis is economically feasible. Data could be obtained from recyclers and from literature. It depends on the case study where these costs should be accounted for.

Revenues from secondary materials can be included in combination with the costs of reprocessing and can also be compared to the disposal costs.

Other costs

Some additional costs might be relevant in the context of URBANWASTE:

Taxes: costs for all taxes occurring within URBANWASTE project's framework

Tooling cost: costs including the depreciation of the App (WP5), the maintenance and the cost for any consumables (water, paper) used while the tool is in operation

Other non-obligatory costs are assurance costs, infrastructure costs, building costs, settlement costs, control costs, financing costs, appliance costs, scrap costs and service costs.

Social criteria

The list of impact categories and indicators for a social LCA (SLCA) was selected following a survey from JØrgensen et al. (2008). The categories and the corresponding indicators summarised in JØrgensen et al. (2008) follow the international approach on SLCA. As mentioned above most of them are not suitable for European cases. The selection of suitable indicators was therefore performed according to the data availability and relevance.

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 690452

Labour practices and decent work conditions

Labour practices and decent work conditions can be evaluated by various indicators, e.g.

- Wages, including equal remuneration on diverse groups, regular payment, length and seasonality of work and minimum wages
- Benefits, including family support for basic commodities and workforce facilities
- Physical working conditions, including rates of injury and fatalities, nuisances, basal facilities and distance to workplace
- Psychological and organisational working conditions, such as maximum work hours, harassments, vertical, two-way communication channels, health and safety committee, job satisfaction, and worker contracts
- Training and education of employees

Society

Concerning the category 'society' the following criteria can be mentioned:

- Development support and positive actions towards society, including job creation, support of local suppliers, investments in research and development, infrastructure, and local community education programmes
- Local community acceptance, such as complaints from society, and presence of communication channels
- Ensuring of commitment to sustainability issues from and towards business partners

The above mentioned assessment criteria and indicators can generally be used for the assessment of status quo within WP 2 task 2.6 as well as the assessment of the Pilot Activities (WP 7). Depending on the outcomes of the investigation on data availability as well as the selection of specific case studies in future there might be changes of criteria as well as indicators. A first finalised overview on planned criteria and indicators for URBANWASTE will be presented in Deliverable 2.3.

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

5. References

Aldaya, M.M., Llamas, M.R. 'Water Footprint Analysis (Hydrologic and Economic) of the Guadiana River Basin'. World Water Assessment Programme, 2009 Side Publication Series.

Allesch, A. and Brunner, P.H., 2014. Assessment methods for solid waste management: A literature review. Waste Management & Research, p.0734242X14535653.

Andrews, C.J., Apul, D.S., Linkov, I., 2005. Comparative Risk Assessment: Past Experience, Current Trends and Future Directions, in: Linkov, I., Ramadan, A.B. (Eds.), Comparative Risk Assessment and Environmental Decision Making. Springer Netherlands, Dordrecht, pp. 3-14.

Angelakoglou, K. & G. and Gaidajis (2015) A review of methods contributing to the assessment of the environmental sustainability of industrial systems. Journal of Cleaner Production, 108, 725-747.

Arjen, Y., Hoekstra, A., Ashok, K., Chapagain, A., Maite M., Aldaya,M., Mesfin, M. et Mekonnen, M. 'The Water Footprint Assessment Manual Setting the Global Standard' Earthscan 2011

Baas, L.lW. & Boons, F.A., 2004. An industrial ecology project in practice: exploring the boundaries of decision-making levels in regional industrial systems. Journal of Cleaner Production 112, 3376-3387.

Beaver, E. (2000) LCA and Total Cost Assessment. Environmental Progress, 19(2), 130-139.

Beck, T., Bos, U., Makishi, C., Lindner, J.: Quantifying and assessing working environment related social aspects along product lifecycles -the LCWE approach. http://www.lcacenter.org/LCA9/presentations/295.pdf (available at 16.11.2016)

Berkel, R.V., Fujita, T., Hashimoto, S. and Fujii, M., 2009. Quantitative assessment of urban and industrial symbiosis in Kawasaki, Japan.Environmental science & technology, 43(5), pp.1271-1281.

Bhushan N. and Rai K. (2004). Strategic Decision Making, Applying the Analytic Hierarchy Process. Springer-Verlag London. 978-1-85233-864-0

Bhushan, N. and Rai, K., 2004. Strategic decision making: applying the analytic hierarchy process. Springer Science & Business Media.

Blanchard B, Fabrycky WJ (1998): Systems Engineering and Analysis. Prentice Hall, Upper Saddle River, New Jersey, USA, p 560.

BookDifferent Foundation, 2016. Carbon footprint per hotel. BookDifferent Foundation. [Online]. Available from http://www.bookdifferent.com. [Last accessed 25 October 2016].

Brown, I., 2015. Comparative Risk Assessment to Inform Adaptation Priorities for the Natural Environment: Observations from the First UK Climate Change Risk Assessment. Climate 3, 937.

Brunner, P. H. et Rechberger, H. 'Practical Handbook of Material Flow Analysis' 2004 CRC Press LLC, Boca Raton, Florida

Brunner, P.-H. & Rechberger, H. 2004. "Practical Handbook of Material Flow Analysis. Advanced Methods in Resource and Waste Management". Lewis Publishers, CRC Press LLC.

BSI, 2011. PAS 2015:2011 Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. The British Standards Institution, London.

Bukh, P. N. & P. Israelsen. 2004. Activity Based Costing. Dansk økonomistyring under forvandling. 1st edition. Jurist- og Økonomforbundets Forlag.

Buxmann, K.; Chudacoff, M.; Frischknecht, R.; Hellweg, S.; Müller-Fürstenberger, G.; Schmidt, M. (1998): „Ökobilanz-Allokationsmethoden. Modelle aus der Kosten- und Produktionstheorie sowie praktische Probleme in der Abfallwirtschaft", Eidgenössische Technische Hochschule Zürich.

Byström, J.. 2012. "Eco efficiency, a path towards integrated resource management". Waste Management, Vol. 32, pp. 797-798. DOI:10.1016/j.wasman.2012.01.0022

Carbon Trust, 2012. Carbon Footprinting - The next step to reducing your emissions. Management guide. The Carbon Trust, UK.

Centre for Waste Reduction Technology (2000) Total Cost Assessment Methodology. Internal Managerial Decision making Tool. American Institute of Chemical Engineers.

Chang, N.-B. & Pires, A. 2015. "Sustainable Solid Waste Management : A Systems Engineering Apporach". IEEE Press Series on Systems Science and Engineering. Wiley, MengChu Zhou, Series Editor.

Chapagain, A., Orr, S, 'An improved water footprint methodology linking global consumption to local water resources: A case of Spanish tomatoes' Journal of Environmental Management 2009 vol: 90 (2) pp: 1219-1228

Curkovic, S. & R. Sroufe (2007) Total Quality Environmental Management and Total Cost Assessment: An exploratory study. International Journal of Production Economics, 105, 560-579.

D'Alisa, G., Di Nola, M. F., & Giampietro, M. (2012). A multi-scale analysis of urban waste metabolism: density of waste disposed in Campania. Journal of Cleaner Production, 35, 59–70. https://doi.org/10.1016/j.jclepro.2012.05.017

De Feo, G. and De Gisi, S., 2010. Using an innovative criteria weighting tool for stakeholders involvement to rank MSW facility sites with the AHP. Waste Management, 30(11), pp.2370-2382.

de Sousa, T.B., Camparotti, C.E.S., Esposto, K.F. and Guerrini, F.M., 2014. Alignment of Balanced Scorecard Perspectives with Supply Chain Management Objectives: a Literature Review. Independent Journal of Management & Production, 5(4), pp.1050-1070.

Denton, G.A. and White, B., 2000. Implementing a Balanced-scorecard Approach to Managing Hotel Operations The Case of White Lodging Services. Cornell Hotel and Restaurant Administration Quarterly, 41(1), pp.94-107.

Deo, B. S. 2005. "Materials Flow Analysis (MFA) - a Translation Problem for Management". IIE Annual Conference 2005: 1-6

Dewan, H. (2006): Sustainability Index: An Economics Perspective. Link: http://economics.ca/2006/papers/0409.pdf

Dias-Sardinha, I. and Reijnders, L., 2005. Evaluating environmental and social performance of large Portuguese companies: a balanced scorecard approach. Business Strategy and the Environment, 14(2), pp.73-91.

DIRECTIVE 2011/92/EU OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 13 December 2011on the assessment of the effects of certain public and private projects on the environment (codification)

DIRECTIVE 2014/52/EU OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 16 April 2014 amending Directive 2011/92/EU on the assessment of the effects of certain public and private projects on the environment

Dodgson, J.S., Spackman, M., Pearman, A. and Phillips, L.D., 2009. Multi-criteria analysis: a manual. Department for Communities and Local Government: London. ISBN 9781409810230

Dreyer, L.C., Hauschild, M.Z., Schierbeck, J. (2006): "A framework for social life cycle impact assessment" International Journal of Life Cycle Assessment, 11 (2), pp. 88-97.

Eagan, P. and Weinberg, L., 1999. Application of analytic hierarchy process techniques to streamlined life-cycle analysis of two anodizing processes. Environmental science & technology, 33(9), pp.1495-1500.

Eckelman, M. J., & Chertow, M. R. (2009). Using Material Flow Analysis to Illuminate Long-Term Waste Management Solutions in Oahu, Hawaii. Journal of Industrial Ecology, 13(5), 758–774. https://doi.org/10.1111/j.1530-9290.2009.00159.x

Eckelman, M.J. & Chertow M.R., 2009. Using material flow analysis to illuminate long-term waste management solutions in Oahu, Hawaii. Journal of Industrial Ecology 13, 758–774.

Eckelman, M.J., Ashton, W., Arakaki, Y., Hamaki, K., Nagashima, S. & Malone-Lee, L.C., 2014. Island waste management systems. Journal of Industrial Ecology 18(2), 306-317.

Ekener Petersen, E., 2013, Tracking down Social Impacts of Products with Social Life Cycle Assessment, Doctoral thesis, KTH Royal Institute of Technology, Stockholm.

Environmental Profit and Loss. http://www.kering.com/en/sustainability/epl

EU, 2001, Directive 2001/42/EC of the European Parliament and of the Council of 27 June 2001 on the assessment of the effects of certain plans and programmes on the environment, Official Journal of the European Communities, Brussels.

European Commission – Joint Research Centre, Institute for Environment and Sustainability (EC-JRC) (2009): CARBON FOOTPRINT - what it is and how to measure it. Available at lca.jrc.ec.europa.eu/Carbon_footprint.pdf , accessed at November 2010.

European Commission (2010a): International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. First edition March 2010. EUR 24708 EN. Luxembourg. Publications Office of the European Union; Joint Research Centre - Institute for Environment and Sustainability 2010

European Commission (2010b): Framework and requirements for Life Cycle Impact Assessment (LCIA) models and indicators", First edition March 2010. EUR 24708 EN. Luxembourg. Publications Office of the European Union; Joint Research Centre - Institute for Environment and Sustainability 2010

European Commission -Joint Research Centre -Institute for Environment and Sustainability (2010): International Reference Life Cycle Data System (ILCD) Handbook -General guide for Life Cycle Assessment -Detailed guidance. First edition March 2010. EUR 24708 EN. Luxembourg. Publications Office of the European Union; 2010

European Environment Agency (EEA). 1998. "Making sustainability accountable: Eco-efficiency, resource productivity and innovation". Topic report No 11/1999.

Fathi, Z. & E. S. M. Dozdahiri. 2015. "A survey of activity-based costing in hotel industry". Manage Science Letters. DOI: 10.5267/j.msl.2015.6.012.

Ferrer, G., Cortezia, S. & Morbach Neumann, J., 2012. Green City. Environmental and social responsibility in an industrial cluster. Journal of Industrial Ecology 16(1), 142-152.

Filimonau, V., Dickinson, J., Robbins, D., Huijbregts, M.A.J., 2011. Reviewing the carbon footprint analysis of hotels: Life Cycle Energy Analysis (LCEA) as a holistic method for carbon impact appraisal of tourist accommodation. Journal of Cleaner Production 19, 1917-1930.

Fleischer, Günther and Riebe, Olaf (2002): "Ökobilanz/Life Cycle Assessment", Technical University TU Berlin.

Frischknecht, R.. 2010. "LCI modelling approaches applied on recycling of materials in view of environmental sustainability, risk perception and eco-efficiency". International Journal of Life Cycle Assessment, 15:7, pp. 666–671. DOI: 10.1007/s11367-010-0201-6

Galli, A., Wiedmann, T., Ercin, E., Knoblauch, D., Ewing, B. et Giljum, S. 'Integrating Ecological, Carbon and Water footprint into a "Footprint Family" of indicators: Definition and role in tracking human pressure on the planet' Ecological Indicators 2012 vol: 16 pp: 100-112

Garfi, M., Tondelli, S. and Bonoli, A., 2009. Multi-criteria decision analysis for waste management in Saharawi refugee camps. Waste management, 29(10), pp.2729-2739.

Gaudreault, C., Samson, R. and Stuart, P., 2009. Implications of choices and interpretation in LCA for multi-criteria process design: de-inked pulp capacity and cogeneration at a paper mill case study. Journal of cleaner production,17(17), pp.1535-1546.

Geng, Y., Tsuyoshi, F. and Chen, X., 2010. Evaluation of innovative municipal solid waste management through urban symbiosis: a case study of Kawasaki. Journal of Cleaner Production, 18(10), pp.993-1000.

Giampietro, M., & Mayumi, K. (1997). A dynamic model of socioeconomic systems based on hierarchy theory and its application to sustainability. Structural Change and Economic Dynamics, 8(4), 453–469. https://doi.org/10.1016/S0954-349X(97)00017-9

Giampietro, M., & Mayumi, K. (2000a). Multiple-Scale Integrated Assessment of Societal Metabolism: Introducing the Approach. Population and Environment, 22(2), 109–153. https://doi.org/10.1023/A:1026691623300

Giampietro, M., & Mayumi, K. (2000b). Multiple-Scale Integrated Assessments of Societal Metabolism: Integrating Biophysical and Economic Representations Across Scales. Population and Environment, 22(2), 155–210. https://doi.org/10.1023/A:1026643707370

Giampietro, M., Mayumi, K., & Ramos-Martin, J. (2009). Multi-scale integrated analysis of societal and ecosystem metabolism (MuSIASEM): Theoretical concepts and basic rationale. Energy, 34(3), 313–322. https://doi.org/10.1016/j.energy.2008.07.020

Ginard-Bosch, F. J., & Ramos-Martín, J. (2016). Energy metabolism of the Balearic Islands (1986–2012). Ecological Economics, 124, 25–35. https://doi.org/10.1016/j.ecolecon.2015.12.012

Guinée, J.B. (1995). Development of a methodology for the environment life-cycle assessment of products; with a case study on margarines. Thesis, Leiden University.

Guinée, J.B. (Ed.), Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., Van Oers, L., Wegener Sleeswijk, A., Suh, S.,. Udo de Haes, H.A, De Bruijn, J.A., Van Duin R., Huijbregts, M.A.J. (2002). Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards. Series: Eco-efficiency in industry and science. Kluwer Academic Publishers. Dordrecht (Hardbound, ISBN 1-4020-0228-9; Paperback, ISBN 1-4020-0557-1).

Haladu, A. 2016. "The Practablilty of Activity-Based Costing in Service Firms". International Journal of Management Research & Review, 6:7 pp. 876-886.

Hendrickson, C. T., Lave, L. B., and Matthews, H. S. (2006): Environmental Life Cycle Assessment of Goods and Services: An Input-Output Approach. Resources for the Future Press. ISBN 978-1-933115-24-5.

Hermann, B.G., Kroeze, C. and Jawjit, W., 2007. Assessing environmental performance by combining life cycle assessment, multi-criteria analysis and environmental performance indicators. Journal of Cleaner Production, 15(18), pp.1787-1796.

Hinterberger, F., Giljum, S. et Hammer, M. 'Material Flow Accounting and Analysis (MFA) A Valuable Tool for Analyses of Society-Nature Interrelationships, Sustainable Europe Research Institute(SERI) Nr. 2, August 2003

Hoekstra, A., Ercin A. 'United Nations World Water Assessment Programme: Carbon and Water Footprints Concepts, Methodologies and Policy Responses', Published in 2012 by the United Nations Educational, Scientific and Cultural Organization

Holman, P., Rounsvell, M. ,Shackley, S., Harrison, P.A., Nicholls, R.J., Berry, P.M. et Audsley, E. 'A regional, multi-sectoral and integrated assessment of the impacts of climate and socio economic change in the UK PartI Methodology I'. Climatic Change (2005) 71: 9–41

Holmes, T., & Pincetl, S. (2012). Urban Metabolism Literature Review. Center for Sustainable Urban Systems. UCLA Institute for the Environment. Director Center for Sustainable Urban Systems.

Horngren, C. T., A. Bhimani, S. M. Datar, G. Foster. 2002. "Activity-based costing." Management and Cost Accounting. 2nd edition, pp. 331-357. Prentice Hall Europe. Pearson Education Limited.

Hu, A.H., Huang, C.Y., Chen, C.F., Kuo, C.H., Hsu, C.W., 2015. Assessing carbon footprint in the life cycle of accommodation services: the case of an international tourist hotel. Int. J. Sustain. Dev. World Ecol. 22, 313-323.

Hudekova, Z., Krajcsovics, L., Martin, P., Pauditsova, E., & Rehackova, T. (2007). Ecological footprint, climate change and cities. Bratislava: REC Slovakia.

Hung, M.L., Ma, H.W. and Yang, W.F., 2007. A novel sustainable decision making model for municipal solid waste management. Waste management, 27(2), pp.209-219.

Hunkeler, D. and Rebitzer, G. (2003): "Life Cycle Costing – Paving the Road to Sustainable Development?" International Journal of Life Cycle Assessment, 8 (2) pp. 109-110.

Huppes, G. & M. Ishikawa. 2005. "Eco-efficiency and Its Terminology". Journal of Industrial Ecology, 9:4, pp. 43-46. DOI: 10.1162/108819805775247891

ISO 14040: "Umweltmanagement – Ökobilanz – Grundsätze und Rahmenbedingungen/Environmental management – Life cycle assessment – Principles and framework", DIN EN ISO 14040:2006.

ISO 14044: "Umweltmanagement – Ökobilanz – Anforderungen und Anleitungen/Environmental management – Life cycle assessment – Requirements and guidelines", DIN EN ISO 14044:2006.

ISO, 2004, ISO 14000 - Environmental management, International Organization for Standardization, Geneva, Switzerland.

ISO, 2006. New ISO 14064 standards provide tools for assessing and supporting greenhouse gas reduction and emissions trading. International Organization for Standardization. Available at: http://www.iso.org/iso/home.htm [Last accessed 24 October 2016].

ISO, 2010, ISO 26000:2010 - Guidance on social responsibility, International Organization for Standardization, Genève, Switzerland.

ISO, 2013. ISO 14067:2013 Greenhouse gases - Carbon footprint of products - Requirements and guidelines for quantification and communication. International Organization for Standardization.

ISO, 2014, Discovering ISO 26000, International Organization for Standardization, Genève, Switzerland.

ITP and GreenView, 2016. Hotel Footprinting Tool. International tourism Partnership and GreenView. [Online]. Available from http://www.greenhotelier.org/our-themes/energy/hotels-can-now-benchmark-and-compare-carbon-footprints/. [Last accessed 25 October 2016].

Jacobsen, N.B., 2006. Industrial symbiosis in Kalundborg, Denmark. A quantitative assessment of ewconomic and environmental aspects. Journal of Industrial Ecology 10(1-2), 239-255.

Jacobsen, N.B., 2006. Industrial symbiosis in Kalundborg, Denmark: a quantitative assessment of economic and environmental aspects. Journal of industrial ecology, 10(1-2), pp.239-255.

Jankovic, S., V. Vejzagic & D. Vlasic. 2011. "Tourist Destination Integral Product Eco-Efficiency". International Business & Economics Research Journal, 10:1, pp. 63-72. DOI: 10.19030/iber.v10i1.928

Jefferies, D., Muñoz, I., Hodges, J., King, V., Aldaya, M., Ercin, A., Milà i Canals, L. et Hoekstra, A. 'Water Footprint and Life Cycle Assessment as approaches to assess potential impacts of products on water consumption. Key learning points from pilot studies on tea and margarine' Journal of Cleaner Production 2012 vol: 33 pp: 155-166

Jeswani, H. et Azapagic, A. 'Water footprint: methodologies and a case study for assessing the impacts of water use' Journal of Cleaner Production 2011 vol: 19 (12) pp: 1288-1299

JØrgensen, Andreas; Le Bocq, Agathe, Nazarkina, Liudmila; Hauschild, Michael (2008):

Methodologies for Social Life Cycle Assessment, International Journal of Life Cycle

Assessment 13 (2) 96 – 103 (2008), Springer-Verlag

JRC, 2015, Social Life Cycle Assessment - State of the art and challenges for supporting product policies,

Kadafa, A.A., Manaf, L.A., Nur, W. and Sulaiman, A., 2014. Applications of system analysis techniques in solid waste management assessment. Pol. J. Environ. Stud, 23(4), pp.1061-1070.

Kaplan, R. S. and Norton, D.P., 1992. The Balanced Scorecard: Measures that Drive Performance, Harvard Business Review, (January-February): 71-79.

Kaplan, R. S., A. A. Atkinson. 1998. "Activity-Based Cost Systems". Advanced Management Accounting. 3rd edition, pp. 97-148. Prentice Hall.

Kaplan, R.S., 2010. Conceptual Foundations of the Balanced Scorecard. Working Paper 10-074. Harvard Business School, Harvard University.

Karagiannidis, A. and Moussiopoulos, N., 1997. Application of ELECTRE III for the integrated management of municipal solid wastes in the Greater Athens Area. In Multiple Criteria Decision Making (pp. 568-578). Springer Berlin Heidelberg.

Kaza, N. & Curtis, M.P., 2014. The land use energy connection. Journal of Planning Literature 29(4), 355-369

Kelly, J., W.Haider, P. W. Williams & K. Englund. 2007. "Stated preferences of tourists for eco-efficient destination planning options". Tourism Management, 28:2, pp. 377 – 390. DOI: 10.1016/j.tourman.2006.04.015

Khajuria, A., Matsui, T. et Machimura, T.'Life cycle assessment of municipal solid waste with regard to DPSIR framework: Evidence from India' Environmental Science An Indian Journal Trade Science Inc. Volume 7 Issue 2 ESAIJ, 7(2), 2012 [41-46]

Khan, S. and Faisal, M.N., 2008. An analytic network process model for municipal solid waste disposal options. Waste management, 28(9), pp.1500-1508.

Klöpffer, W. (2003) Life-Cycle Based Methods for Sustainable Product Development. The International Journal of Life Cycle Assessment, 8(3), 157-159.

Korhonen, J., 2001. Regional industrial ecology: examples from regonial economic systems of forest industry and energy supply in Finland. Journal of Environmental Management 63, 367-375.

Koskela, M. & J. Vehmas. 2012. "Defining Eco-efficiency: A Case Study on the Finnish Forest Industry". Business Strategy and the Environment, 21:8, pp. 546–566. DOI: 10.1002/bse.741

Kristensen, P. 'The DPSIR Framework' National Environmental Research Institute, Denmark Department of Policy Analysis European Topic Centre on Water, European Environment Agency presented at the 27-29 September 2004 workshop on a comprehensive / detailed assessment of the vulnerability of water resources to environmental change in Africa using river basin approach. UNEP Headquarters, Nairobi, Kenya

Kuzyk, L., & Rockley, M. (2014). Measuring Environmental Sustainability in Plans. Plan Canada, 2014(Spring), 30–35.

Kytziaa, S., A. Walzb & M. Wegmanna. 2011. "How can tourism use land more efficiently? A model-based approach to land-use efficiency for tourist destinations". Tourism Management, 32:3, pp. 629–640. DOI: 10.1016/j.tourman.2010.05.014

Larsen, H.N., Hertwich, E.G., 2010. Identifying important characteristics of municipal carbon footprints. Ecological Economics 70, 60-66.

Leiman, T. & Tuomi, K. 2004. "Cost Benefit Analysis, Integrated Environmental Management, Information Series 8, Department of Environmental Affairs". Department of Enfironmental Affairs and Tourism (DEAT)", Pretoria. ISBN 0-9584728-9-0.

Luciano, A., Barberio, G., Mancuso, E., Sbaffoni, S., La Monica, M., Scagliarino, C. & Cutaia, L., 2016. Potential improvement of the methodology for industrial symbiosis implementation at regionalscale. Waste Biomass Valor 7, 1007-1015.

Mackay, A., 2005. A Practitioner's Guide to the Balanced Scorecard. Chartered Institute of Management Accountants.

Majumder, M., 2015. Impact of Urbanization on Water Shortage in Face of Climatic Aberrations. Chapter 2: Multi Criteria Decision Making. Springer Verlag, Singapor.

Manaf, L.A., Basri, H. and Basri, N.E.A., 2009. UrusSisa: An intelligent system for integrated solid waste management. Journal of Sustainable Development, 1(2), p.39.

Massard G., Jacquat O., Zürcher D. 2014: International survey on ecoinnovation parks. Learning from experiences on the spatial dimension of eco-innovation. Federal Office for the Environment and the ERANET ECO-INNOVERA, Bern. Environmental studies no. 1402: 310 pp.

Maxim, L., Spangenberg, J.,H. et O'Connor, M. 'An analysis of risks for biodiversity under the DPSIR framework' Ecological Economics 2009 vol: 69 (1) pp: 12-23

Mickwitz, P., M. Melanen, U. Rosenström & J. Seppälä. 2006. "Regional eco-efficiency indicators – a participatory approach". Journal of Cleaner Production, 14:18, pp. 1603–1611. DOI: 10.1016/j.jclepro.2005.05.025

Milà i Canals, L., Chenoweth, J., Chapagain, A., Orr S., Antón, A. et Clift, R. 'Assessing freshwater use impacts in LCA: part I – inventory modelling and characterisation factors for the main impact pathways' International Journal of Life Cycle Assessment, 14 (1) 2009, pp. 28–42

Mohammed, F., Biswas, W.K., Yao, H. & Tadé, M., 2016. Identification of an environmentally friendly symbiotic process for the reuse of industrial byproduct. – and LCA perspective. Journal of Cleaner Production 112, 3376-3387.

Morgenstern, R.D., Shih, J.-S., Sessions, S.L., 2000. Comparative risk assessment: an international comparison of methodologies and results. Journal of Hazardous Materials 78, 19-39.

Morrissey, A.J. and Browne, J., 2004. Waste management models and their application to sustainable waste management. Waste management, 24(3), pp.297-308.

Mulder, K.F., 2016. Urban Symbiosis, a New Paradigm in the Shift Towards Post-Carbon Cities. TU Delft/The Hague University of Applied Science

National Institute of Standards and Technology, 2000. BEES 2.0: Building for environmental and economic sustainability – technical manual and user guide. National Institute of Standards and Technology (NIST), Caithersburg, MD, USA, NISTIR 6520.

Niemeijer, D. et de Groot, R.S. 'Framing environmental indicators: moving from causal chains to causal networks' Environ Dev Sustain (2008) 10:89–106

Norris, G.A. (2001): "Integrating life cycle cost analysis and LCA", International Journal of Life Cycle Assessment, 6 (2), pp. 118-120.

Norris, G.A. (2001): "Integrating life cycle cost analysis and LCA", International Journal of Life Cycle Assessment, 6 (2), pp. 118-120.

O'Brien, M., Doig, A., Clift, R., 1996, Social and environmental life cycle assessment (SELCA), Internstionsl Journal of Life Cycle Assessment 1, 231-237.

OECD. 1998. "Eco-Efficiency". OECD. pp. 7-21. DOI:10.1787/9789264040304-en

Oers, L. van, A. de Koning, J.B. Guinée & G. Huppes, 2002. Abiotic resource depletion in LCA - Improving characterisation factors for abiotic resource depletion as recommended in the new Dutch LCA Handbook. DWW report, Delft; see http://www.cml.leiden.edu/research/industrialecology/researchprojects/finished/abiotic -depletion-lcia.html

Paquin, R.L., Busch, T. & Tilleman, S.G., 2015. Creating ecconomic and envionmental value through industrial symbiosis. Long Range Planning 48, 95-107.

Pearce, D., Atkinson, G. & Mourato, S. 2006. "Cost-Benefit Analysis and the Environment. Recent Developments". Secretary-General of the OECD, Paris, France. OECD Publishing. ISBN 92-64-01004-1.

Petcharat, N. N. & J. M. Mula. 2012."Towards a conceptual design for environmental and social cost identification and measurement system". Journal of Financial Reporting and Accounting, 10:1, pp. 34-54. DOI: 10.1108/19852511211237435.

Pincetl, S. (2012). 1 - A living city: using urban metabolism analysis to view cities as life forms A2 - Zeman, Frank. In Metropolitan Sustainability (pp. 3–25). Woodhead Publishing. Retrieved from http://www.sciencedirect.com/science/article/pii/B9780857090461500011

Pirani, S.I., Arafat, H.A., 2014. Solid waste management in the hospitality industry: a review. Journal of Environmental Management 146, 320-336.

Psidter, S., Koehler, A. et Hellweg, S. 'Assessing the Environmental Impacts of Freshwater Consumption in LCA' ETH Zurich, Institute of Environmental Engineering, 8093 Zurich, Switzerland Received August 28, 2008. Revised manuscript received March 20, 2009. Accepted March 20, 2009

PUMA – First environmental profit and loss account, http://about.puma.com/pumacompletes-first-environmental-profit-and-loss-account-which-values-impacts-at-e-145- million/

Qian, Huang and Yan, 2007. Integrated assessment of environmental and economic performance of chemical products using analytic hierarchy process approach. Chinese Journal of Chemical Engineering, 15(1), pp.81-87.

Ramjeawon, T. and Beerachee, B., 2008. Site selection of sanitary landfills on the small island of Mauritius using the analytical hierarchy process multi-criteria method. Waste Management & Research, 26(5), pp.439-447.

Ramos-Martín, J., Cañellas-Boltà, S., Giampietro, M., & Gamboa, G. (2009). Catalonia's energy metabolism: Using the MuSIASEM approach at different scales. Energy Policy, 37(11), 4658–4671. https://doi.org/10.1016/j.enpol.2009.06.028

Rebitzer G (2002): Integrating Life Cycle Costing and Life Cycle Assessment for Managing Costs and Environmental Impacts in Supply Chains. In: Seuring S, Goldbach M (eds): Cost Management in Supply Chains. Heidelberg: Physica-Verlag, p 128-146.

Rebitzer, G. and Hunkeler, D. (2003): "Life cycle costing in LCM: Ambitions, opportunities, and limitations - Discussing a framework", International Journal of Life Cycle Assessment, 8 (5), pp. 253-256.

Ren, J, Liang, H., Dong, L. & Gao, Z., 2016. Design for sustainability of industrial sysmbiosis based on emergy and multi-objective particle swarm optimization. Science of the Total Environment 526, 789-801.

Rex, E. and Baumann, H. (2008): "Implications of an interpretive understanding of LCA practice", Business Strategy and the Environment, 17 (7), pp. 420-430.

Rezaei, J., 2015. A systematic review of multi-criteria decision-making applications in reverse logistics. Transportation Research Procedia, 10, pp.766-776.

Rosselló-Batle, B., Moià, A., Cladera, A., Martínez, V., 2010. Energy use, CO2 emissions and waste throughout the life cycle of a sample of hotels in the Balearic Islands. Energy and Buildings 42, 547-558.

Rousis, K., Moustakas, K., Malamis, S., Papadopoulos, A. and Loizidou, M., 2008. Multi-criteria analysis for the determination of the best WEEE management scenario in Cyprus. Waste Management, 28(10), pp.1941-1954.

Saaty, R.W., 1987. The analytic hierarchy process—what it is and how it is used. Mathematical modelling, 9(3), pp.161-176.

Saaty, T.L., 2008. Decision making with the analytic hierarchy process. International journal of services sciences, 1(1), pp.83-98.

Sala, S., Vasta, A., Mancini, L., Dewulf, J., 2015, Rosenbaum, E; Social Life Cycle Assessment - State of the art and challenges for supporting product policies, Joint Research Centre, Ispra.

Sartori, D., Catalano, G., Genco, M., Pancotti, C., Sirtori, E., Vignetti, S. & Del Bo, C. 2015. " Guide to Cost-Benefit Analysis of Investment Projects". European Commission, Directorate-General for Regional and Urban policy, pp. 25-76.

Schmoldt, D., Kangas, J., Mendoza, G.A. and Pesonen, M. eds., 2013. The analytic hierarchy process in natural resource and environmental decision making (Vol. 3). Springer Science & Business Media.

Schütz, H., Wiedemann, P.M., Hennings, W., Mertens, J., Clauberg, M., 2006. Introduction, Comparative Risk Assessment. Wiley-VCH Verlag GmbH & Co. KGaA, pp. 7-12.

Seppälä, J., 2003. Life cycle impact assessment based on decision analysis. Helsinki University of Technology.

Smeets, E. and Weterings, R. 'Environmental indicators:Typology and overview' Technical report No 25 EEA, Copenhagen, 1999

Smith, R.L., Sengupta, D., Takkellapti, S. & Lee C.C., 2015. An industrial ecology approach to munuicipal solid waste managenemn: II. Cse studoes for recovering energy from the organic fraction of MSW. Resources, Conservation and Recycling 104, 317-326.

Sokka, L., Pakarinen, S. & Melanen, M., 2011. Industrial symbiosis contributing to more sustainable energy use – An example from the forest industry in Kymenlaakso, Finland. Journal of Cleaner Production 19, 285-293.

Sovacool, B. & A. Gross. 2015. "The social acceptance of artificial photosynthesis: towards a conceptual framework". Interface focus, 5:3, pp. 20140089–20140089, The Royal Society Publishing. DOI: 10.1098/rsfs.2014.0089

Spangenberg, J.H. (2005). Economic sustainability of the economy: concepts and indicators. International Journal of Sustainable Development.

Stefano, N. M. & M. C. D. Freitas. 2014. "Framework: Activity-Based Costing in Services". Revista de Administração da Universidade Federal de Santa Maria. 7:1, pp. 153-169. DOI: 10.5902/1983465910079.

Strazza, C., Magrassi, F., Gallo, M. & Del Borghi, A., 2015. Life cycle assessment from food to food: a case study of circular economy from cruise ships to aquaculture. Sustainable Production and Consumption 2, 40-51.

Styles, D. Schönberger, H. et Galvez Martos, J.L. 'Best Environmental Management Practice in the Tourism Sector' JRC Scientific and Policy Report, 2013

Sun, Y.-Y., S. Pratt, "The Economic, Carbon Emission, and Water Impacts of Chinese Visitors to Taiwan: Eco-efficiency and Impact Evaluation". Journal of Travel Research, 53:6, pp. 733-746. DOI: 10.1177/0047287513517420

Sustainable Norrköping program, 2014. Urban-industrial symbiosis. Website: https://www.iei.liu.se/envtech/forskning/forskningsprojekt/hallbara-nkpg/sustainable-nkpg4?l=en. Last accessed: 16-11-2016

Svarstad, H., Petersen, L., Rothman, D., Siepe,l H. et Wätzold, F. 'Discursive biases of the environmental research framework DPSIR' Land Use Policy 2008 vol: 25 (1) pp: 116-125

Thomas E. Swarr, David Hunkeler, Walter Klöpffer, Hanna-Leena Pesonen, Andreas Ciroth, Alan C. Brent and Robert Pagan (2011): Environmental life-cycle costing: a code of practice. Int J Life Cycle Assess (2011) 16:389, p.391. DOI 10.1007/s11367-011-0287-5.

Tolle, D., Vigon, B. and Evers, D., 1998. Life-cycle impact assessment demonstration for the GBU-24. BATTELLE COLUMBUS OPERATIONS OH.

Tsai, W.-H. & Shih-Jieh Hung. 2009. "Treatment and recycling system optimisation with activity-based costing in WEEE reverse logistics management: an environmental supply chain perspective". International Journal of Production Research, 47:19, pp. 5391-5420. DOI: 10.1080/00207540801927183.

Tsai, W.-H., T.-S. Tsaur, Y.-W. Chou, J.-Y. Liu, J.-L. Hsu & C.-L. Hsieh. 2015. "Integrating the Activity-Based Costing System and Life-Cycle Assessment into Green Decision-Making." Journal of Production Research, 53:2, pp. 451-465. DOI: 10.1080/00207543.2014.951089.

Tsai, W.-H., Y.-S. Shen, P.-L. Lee, H.-C. Chen, L. Kuo, C.-C. Huang. 2012. "Integrating information about the cost of carbon through activity-based costing." Journal of Cleaner Production, Vol. 36, pp.102–111. DOI: 10.1016/j.jclepro.2012.02.024.

Tscherning, K., Helming, K., Krippner, B., Sieber, S. et Paloma, S. 'Does research applying the DPSIR framework support decision making?' Land Use Policy 2012 vol: 29 (1) pp: 102-110

Turner, D., Williams, I. et Kemp, S. 'Combined material flow analysis and life cycle assessment as a support tool for solid waste management decision making'. Journal of Cleaner Production 2016 vol: 129 pp: 234-248

Turner, D.A., Williams, I.D. & Kemp, S., 2016. Combined material flow analsysi and life cycle assessment as a support tool fro waste management decision making. Journal of Cleaner Production 129, 234-248.

Umair S., Björklund A., Ekener-Petersen E., 2015, Assessing Social Impacts of Informal Recycling of Electronic ICT Waste in Pakistan with a Life Cycle approach. Resources, Conservation and Recycling, 95, 46-57.

UNEP & SETAC, 2009, Guidelines for social life cycle assessment of products: a social and socioeconomic LCA code of practice, United Nations Environment Programme.

Unger, N., Schneider, F., Salhofer, S. (2008). A review of ecodesign and environmental assessment tools and their appropriateness for electrical and electronic equipment. Progress in Industrial Ecology, an International Journal 5, 13-29.

Van Berkel R., Fujita T., Hashimoto S. & Fujii M., 2009. Quantitative assessment of Urban and Industrial Symbiosis in Kawasaki, Japan. Environmental Science & Technology 43, 1271–1281.

Vazakidis, A. & I. Karagiannis. 2008. "Activity-based management and traditional costing in tourist enterprises (a hotel implementation model)". International Journal of Operational Research, 11, pp. 123-147. DOI: 10.1007/s12351-009-0049-3

Vercalsteren, A., C. Spirinckx & T. Geerken. 2010. "Life cycle assessment and eco-efficiency analysis of drinking cups used at public events". The International Journal of Life Cycle Assessment, 15:2, pp. 221-230. DOI: 10.1007/s11367-009-0143-z

Vernay, A.L. and Mulder, K.F., 2015, September. Organising urban symbiosis projects. In Proceedings of the Institution of Civil Engineers-Engineering Sustainability. Thomas Telford Ltd.

Villanueva, A., Kristensen, K.-B. & Hedal, N. 2006. "A quick guide to LCA and CBA in waste management". 1st edition. Danish Topic Centre on Waste and Resources.

Vinyes, E., Oliver-Solà, J., Ugaya, C., 2013, Application of LCSA to used cooking oil waste management, The International Journal of Life Cycle Assessment, 18, 445-455.

Wackernagel, M., & Rees, W. (1995). Our Ecological Footprint: Reducing Human Impact on the Earth. Gabriola Island, BC, and Philadelphia, PA: New Society Publishers.

Walker, R. V. , F. Jiang, O. O. Osidele & M. B. Beck. 2009. "Eco-Effectiveness, Eco-efficiency, and the Metabolism of a City: a Multi-sectoral Analysis". IEEE International Conference 2009 on Systems, Man and Cybernetics. DOI: 10.1109/ICSMC.2009.5346300

Walls, J.L. and Paquin, R.L., 2015. Organizational Perspectives of Industrial Symbiosis A Review and Synthesis. Organization & Environment, 28(1), pp.32-53.

Wang, G., Huang, S.H. and Dismukes, J.P., 2004. Product-driven supply chain selection using integrated multi-criteria decision-making methodology.International journal of production economics, 91(1), pp.1-15.

Wang, W., D. Jiang, D. Chen, Z. Chen, W. Zhou & B. Zhu. 2015. "A Material Flow Analysis (MFA)-based potential analysis of eco-efficiency indicators of China's cement and cement-based materials industry". Journal of Cleaner Production, Vol. 112, pp. 787-796. DOI: 10.1016/j.jclepro.2015.06.103

Wati, Y. and Koo, C., 2011, January. An Introduction to the Green IT balanced scorecard as a strategic IT management system. In System Sciences (HICSS), 2011 44th Hawaii International Conference on (pp. 1-10). IEEE.

WBCSD (2011). Guide to Corporate Ecosystem Valuation – A framework for improving corporate decision-making. Link: https://www.google.at/url?sa=t&rct=j&q=&esrc=s&source=web&cd=2&ved=0ahUKEwjR28ai2L7QAhXFCcAKHQjCAUQQFggkMAE&url=http%3A%2F%2Fwww.wbcsd.org%2Fcontentwbc%2Fdownload%2F573%2F6341&usg=AFQjCNEdbHllqrIPXWmhtIVYVRLvZ8jAMw&cad=rja

Weidema, B.P., Thrane, M., Christensen, P., Schmidt, J., Løkke, S., 2008. Carbon Footprint. Journal of Industrial Ecology 12, 3-6.

Wiedmann, T., & Barrett, J. (2010). A Review of the Ecological Footprint Indicator—Perceptions and Methods. Sustainability, 2(6), 1645–1693. https://doi.org/10.3390/su2061645

Wiedmann, T., Minx, J., 2007. A Definition of 'Carbon Footprint', in: Pertsova, C.C. (Ed.), Ecological Economics Research Trends: Chapter 1, pp. 1-11. Nova Science Publishers, Hauppauge NY, USA.

Wittstruck, D. and Teuteberg, F., 2011. Development and simulation of a balanced scorecard for sustainable supply chain management–a system dynamics approach.

Woodward, D. G. (1997): Life cycle costing - theory, information acquisition and application, International Journal of Project Management, Vol. 15, No. 6, pp. 335-344, 1997

WRI, WBCSD, 2016. Greenhouse Gas Protocol. World Resources Institute (WRI) and World Business Council on Sustainable Development (WBCSD). [Online]. Available from: http://www.ghgprotocol.org/ [Last access 25 October 2016].

WWF. (2016). Living Planet Report 2016. Risk and resilience in a new era.

Yang, Z., X. Zhou & L. Xu. 2015. "Eco-efficiency optimization for municipal solid waste management". Journal of Cleaner Production, 104:1 pp. 242–249. DOI: 10.1016/j.jclepro.2014.09.091

Yazan, D.M., Romano, V.A. & Albino, V., 2016. The design of industrial symbiosis: an input-output approach. Journal of Cleaner Production 129, 537-547.

This project has received funding from the European Union's Horizon 2020 research
and innovation programme under grant agreement No 690452

Yuan, H., Z. Huang & P. Xu. 2016. "A framework for eco-efficiency of C&D waste management". Procedia Environmental Sciences Vol. 31, pp. 855–859. DOI: 10.1016/j.proenv.2016.02.097

Zhang, J.J, N. Joglekar, J. Heineke & R. Verma. 2014. "Eco-efficiency of Service Co-production: Connecting Eco-certifications and Resource Efficiency in U.S. Hotels". Cornell Hospitality Quarterly, 55:3, pp. 252-264. DOI: 10.1177/1938965514533988

Zhang, Y. (2013). Urban metabolism: A review of research methodologies. Environmental Pollution, 178, 463–473. https://doi.org/10.1016/j.envpol.2013.03.052

Zhang, Y., Yang, Z., & Yu, X. (2015). Urban Metabolism: A Review of Current Knowledge and Directions for Future Study. Environmental Science & Technology, 49(19), 11247–11263. https://doi.org/10.1021/acs.est.5b03060

Zhao, Y. & C. Zhao. 2011. "Eco-Efficiency Evaluation Indicator of Plantation Harvesting System and Its Improvement Based on Material Flow Analysis". IEEE Asia-Pacific Power and Energy Engineering Conference 2011. DOI: 10.1109/APPEEC.2011.5748784

Zhou, Z. and Zheng, L., 2010. The Evaluation of Recycling Economy Development in Flow Manufacturing Enterprise-An Case Study of Electrolytic Aluminum Enterprise in China. In Management and Service Science (MASS), 2010 International Conference on (pp. 1-4). IEEE.

URBAN WASTE

URBAN STRATEGIES FOR WASTE MANAGEMENT IN TOURIST CITIES