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Environmental sustainable decision making- the need and obstacles for integration of LCA into decision analysis

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Highlights

- Extensive range of environmental impacts is rarely considered in decision analysis.
- LCA can provide sophisticated environmental profiles of decision alternatives.
- LCA and other decision analysis tools have different goals, principles and systems.
- Consistency of study system between LCA and other tools is the key for integration.

Abstract

Decision analysis is often used to help decision makers choose among alternatives, based on the expected utility associated to each alternative as function of its consequences and potential impacts. Environmental impacts are not always among the prioritized concerns of traditional decision making.

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This has fostered the development of several environmental problems and is nowadays a reason of concern. Life Cycle Assessment (LCA) can assess an extensive range of environmental impacts associated with a product or service system and supports a life cycle perspective on the alternative products or service systems, revealing potential problem shifting between life cycle stages. Through the integration with traditional risk based decision analysis, LCA may thus facilitate a better informed decision process. In this study we explore how environmental impacts are taken into account in different fields of interest for decision makers to identify the need, potential and obstacles for integrating LCA into conventional approaches to decision problems. Three application areas are used as examples: transportation, flood management, and food production and consumption. The analysis of these cases shows that environmental impacts are considered only to a limited extent in traditional evaluation of transport and food projects. They are rarely, if at all, addressed in flood risk management. Hence, in each of the three cases studied, there is a clear need for the inclusion of a better and systematic assessment of environmental impacts. Some LCA studies have been conducted in all three research areas, mainly on infrastructure and production systems. The three cases show the potential of integrating LCA into existing decision analysis by providing the environmental profiles of the alternatives. However, due to different goals and scopes of LCA and other decision analysis approaches, there is a general lack of consistency in study system scoping in terms of considered elements and boundaries, in uncertainty treatment, and in applied metrics. In the present paper, we discuss the obstacles arising when trying to integrate LCA with conventional evaluation tools and we propose a research agenda to eventually make such integration feasible and consistent.

Keywords

Decision analysis, Life cycle assessment, Cost benefit analysis, Risk assessment, Decision support

1. Introduction

Decision support systems are often used to guide decision makers towards the best decision. Decision theory as mathematical basis of decision making under uncertainty was formulated in the 20th century. Following a structured methodology, it aims at selecting one out of different available alternatives, based on the consequences associated to each alternative. Due to different context of application in several scientific disciplines, different approaches may be used in the specific field of application. Risk-based decision making, as one of the widely used approaches, has been used to address the concern for human, societal, economy and ecosystem health when exposed to unfavorable events, e.g. natural hazards, contamination etc. (Klüppelberg et al., 2014). Cost benefit analysis (CBA) is another approach, used to identify the alternative that can achieve a particular goal with lowest cost (Mishan and Quah, 2007). In parallel or in combination with CBA, Multi-criteria analysis (MCA) is often used to evaluate the alternatives based on a set of measurable criteria (Figueira et al., 2005). These approaches are broadly used in both public and private sectors with a particular aim: to help decision makers choose the most appropriate alternative to achieve their goals, according to a certain set of criteria.

Among these concerns and criteria, environmental problems are not well represented. This has caused tremendous problems in the past, examples being the London fog in last century and acid rains. The

67 former one is a result of burning soft coal for heating, while the later one comes from the excessive
68 emission of SO₂ and NO_x mainly from the burning of fossil fuels. These phenomenon happen because
69 there is not enough awareness of the potential damages, which leads to the lack of relevant control
70 measures. These environmental problems could have been avoided if the life cycle perspective of
71 environmental impacts associated with the energy product use are considered beforehand. To reduce
72 the occurrence of similar events, many regulations and proposals have appeared afterwards (Kahn,
73 2007), acting towards precautionary purposes, with different environmental focus and decision
74 analysis prospective in specific sectors. For example, noise problems have traditionally been
75 considered in transportation decision analysis (European Commission, 2014). Pathogen and chemical
76 impacts on human health have conventionally been taken into account in food safety decisions (FAO
77 and WHO, 2005). Impacts on climate change have recently been considered in decision analysis for
78 e.g. flood management, and transportation planning. Note that a wide range of environmental impacts
79 may arise as a consequence of man-made activities (e.g. climate change, eutrophication, acidification,
80 etc.). For the sectors mentioned above and the majority of other sectors, focus has been on a rather
81 limited selection of environmental impact categories following regulations and proposals. Some
82 attempts exist on accounting for a broader selection of environmental impact categories, via
83 approaches such as MCA (Halsnæs et al., 2015; Munda, 2005). However, it is not common to see a
84 decision analysis that covers an extensive set of environmental impacts for the alternatives, which
85 can sometimes lead to controversial results. For example, when facing several alternatives in a
86 transport project, the best alternative according to CBA may not have the best environmental
87 performance, due to e.g. neglecting environmental impacts from the life span of vehicles and
88 infrastructures (Chester and Horvath, 2009). Were these to be included through taking a life cycle
89 perspective, the preferred alternative may turn out to cause more damages on ecosystems, and the
90 cost for amending such damages may be more than the savings on the infrastructures. Without having
91 proper environmental impact assessment in decision analysis, such information cannot be revealed
92 and the decision making will be misguided. The lack of such practice may be ascribed to the lack of
93 a common understanding of the needs and of the possible ways to integrate the relevant environmental
94 impacts into existing decision analysis tools.

95 Many methods and tools were developed or adapted to assess environmental impacts, including e.g.
96 Environmental Impact Assessment, Life Cycle Assessment (LCA), input-output analysis, etc. Zijp et
97 al. (2017) summarized the methods available for assessing environmental sustainability and provides
98 a model for the selection of suitable method corresponds to the decision context. LCA standards out
99 for its inclusive of cradle to grave perspective, flexibly in spatial scale, and its feasibility of
100 application during product development and commercial stages. With its life cycle-based systems
101 perspective and broad coverage of environmental impacts, LCA is indeed a promising tool for
102 assessing environmental sustainability (Sala et al., 2013). It quantifies resource use and environmental
103 impacts that are associated with a product or service into an extensive set of impact categories (EC-
104 JRC, 2010). LCA is currently the most mature with its basic principles laid down in an international
105 set of standards (ISO 14040/14044) (Kloepffer, 2008). It has been adopted by public sectors for e.g.
106 prioritizing research in energy sector in USA (Bosso et al., 2012) and implementing sustainable
107 strategies in EU (European Commission, 2016). Private sectors also use LCA frequently for choosing

the environmentally friendly alternatives, materials and services and for communicating via environmental product declarations or ecolabels. However, LCA is not a legal requirement in any regulatory context (Bosso et al., 2012; European Commission, 2016). It has the potential to give a good overview of environmental impacts related to each decision alternative, to be taken into account in decisions.

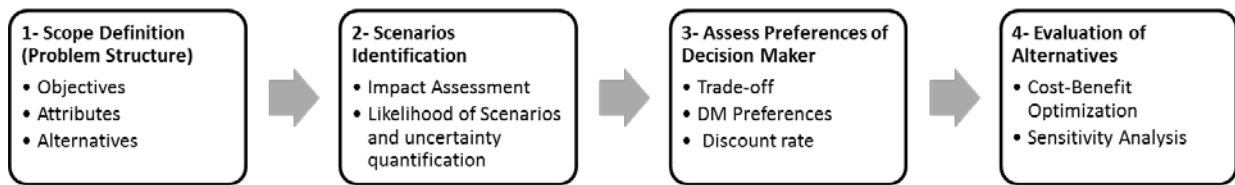
Taking some specific research disciplines as example, the aim of this study is to: 1) explore how environmental impacts are taken into account in the current decision analysis approaches; 2) review the application of LCA in those fields and 3) explore the need, obstacles and potential for integrating LCA into decision analysis. First, decision analysis theory and LCA will be introduced. Then we will look into three specific research disciplines, namely flood management, transport projects and food production and consumption to address the aim of the study. These three research fields have high societal relevance, and there is a strong need for considering environmental perspectives in their decision analysis. They have different conventional decision analysis approaches that represent a variety of challenges for the integration of LCA.

2. Decision analysis

The ultimate goal of decision making is to select one out of different available alternatives, which most likely leads to the most favorable outcome. Due to the uncertainty, we cannot identify the optimal choice by means of deterministic values (Faber, 2008). Therefore, decision analysis aims at evaluating alternatives based on the changes that they operate on expected utility associated to the performance of system, i.e. the benefits and the consequences. This facilitates objective and informed decisions by enclosing the decision process into a structured methodology, giving a mathematical representation to the evaluation process aiming at identifying the most favorable outcome with respect to possible alternatives.

According to Keeney (1982), any decision problem can be structured following four main steps as shown in Figure 1. The first step - scope definition of the decision problem - is of key importance, because choices and assumptions made in this phase will influence the entire decision process. Therefore, it is important to get a clear definition of the decision problem, the expected improvement (objectives) from the Decision Maker (DM) and the identification of the feasible and affordable alternatives out of all possible ones. The scenario identification phase (phase two) includes forecasting of the impact of each alternative on the performance of the system (through modelling and/or data collection and analysis) and the uncertainty characterization, quantification and propagation. The third phase refers to the quantification of DM's preferences. Preferences represent how attractive, valuable, convenient and favourable the DM judges the alternatives to be with respect to their impact on the system (Raiffa and Schlaifer, 1961). This value trade-off and risk attitude of the DM is translated into an objective function (utility function) representing a weighted average of the utility associated to all possible outcomes (French and Rios Insua, 2000; Raiffa and Schlaifer, 1961). The optimal alternative will be the one maximizing the expected value of the utility function. Phase four, deals with the optimization of the utility function and sensitivity analysis to assure robustness and consistency of the solution. The four steps in Keeney (1982) represent, however, a

147 very general decision problem paradigm. In real decision analyses, feedback loops and iterations are
 148 required as will also be shown in Figure 2.



149

150 **Figure 1. Steps of the Decision Making Process, adapted from Keeney (1982)**

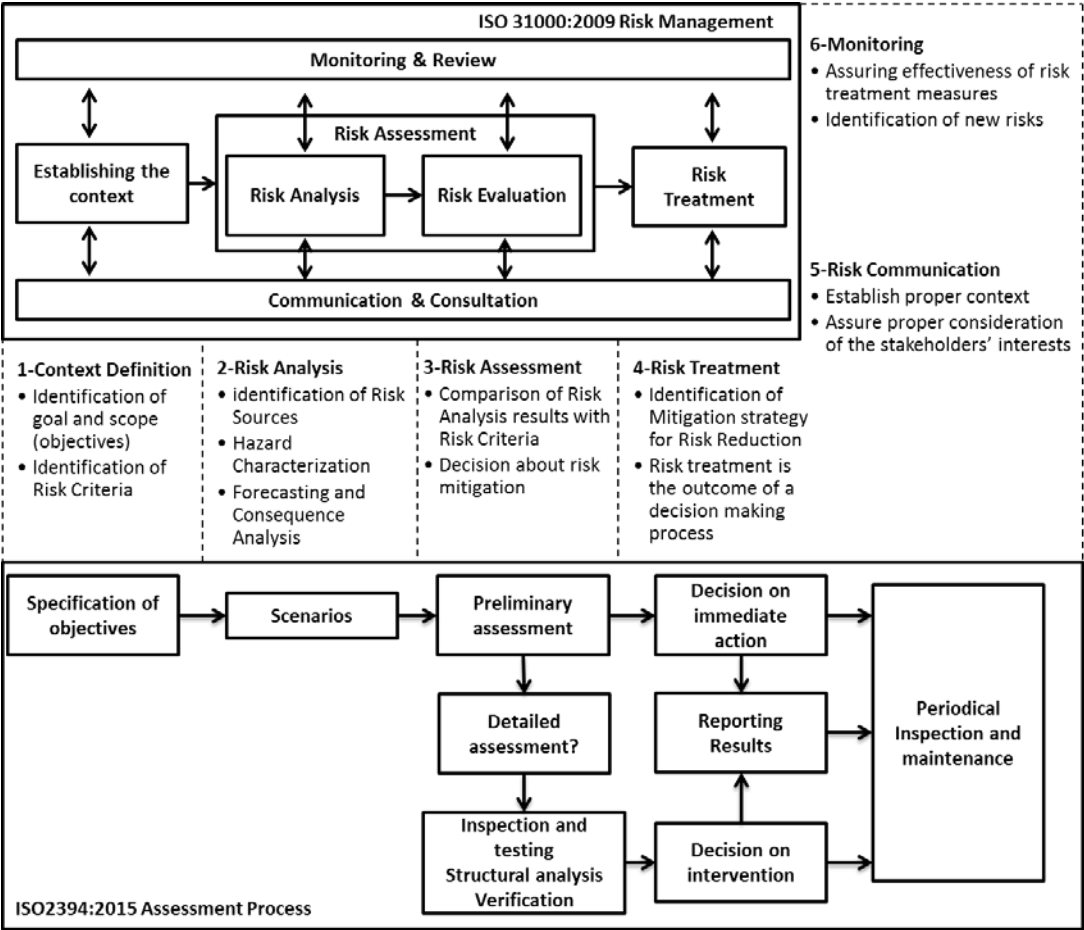
151 Uncertainty, which refers to the incompleteness of knowledge or the lack of understanding, affects
 152 largely the decision process. Variability, i.e. aleatory uncertainty, describes the inherent variations
 153 and randomness of the quantity, process or system of interest and it cannot be eliminated. Temporal,
 154 spatial or inter-object variations are some of the -not mutually exclusive- categories of variability.
 155 Epistemic uncertainty is caused by lack of knowledge and can be reduced e.g. by means of further
 156 measurement or study of the quantity, process or system. Where epistemic uncertainty and variability
 157 occur may vary, but two types are widely mentioned: model uncertainty and parameter uncertainty.
 158 Model uncertainty captures the imperfect representability of the true processes and systems.
 159 Parameter uncertainty refers to the lack of knowledge of the exact parameter value in a model. Other
 160 classifications and terminologies of uncertainty exist (e.g. Faber, 2012; Funtowicz and Ravertz, 1990;
 161 Kiureghian and Ditlevsen, 2009; Walker et al., 2003), that we do not further discuss in the paper.
 162 These include statistical uncertainty and scenario uncertainty, while errors, e.g. measurement and
 163 human errors, are often considered as uncertainty sources. Regardless of location and source,
 164 epistemic uncertainty and variability need to be properly treated throughout an analysis and
 165 communicated.

166 **2.1 Risk-based decision analysis**

167 Risk arises whenever there is uncertainty on potentially adverse events causing unfavourable
 168 consequences, within a specific time frame (JCSS, 2008). Risk-based decision making is a widely
 169 used tool to assess performance and evaluate policies for complex systems and services where
 170 potential risks exist. For instance, it is often applied to answer the decision problem such as choice of
 171 mitigation policies against natural disasters (earthquakes, floods etc.) and evaluation of food safety.

172 There are various definitions of risk, which may be defined as “combination of the consequences of
 173 an event (including changes in circumstances) and the associated likelihood of occurrence”, following
 174 ISO31000:2009. The evaluation of the risk can be formalized in different procedures according to the
 175 specific field of application. The ISO31000:2009 represents the general reference framework for risk
 176 management in industrial applications while ISO2394:2015 is the reference standard for both
 177 reliability and risk based decision making concerning design and assessment of structural systems.
 178 Figure 2 shows the two ISO standard frameworks, where ISO2394:2015 provides a more detailed
 179 description of the assessment procedure. The evaluation of risk analysis results, with respect to

180 acceptance risk criteria defined by current regulations - e.g. Seveso III (European Union, 2012),
 181 REACH (EU, 2006), EUROCODE0-to-8 (CEN 1990:2002) etc.- or in some cases by engineering
 182 judgment, is an important phase in identifying the mitigation strategy based on the possible
 183 alternatives. Risk assessment can be conducted in a qualitative way, semi-quantitative or quantitative
 184 way. Uncertainty is widely analyzed in quantitative risk-based decision analysis (Bedford and Cooke,
 185 2001; Klüppelberg et al., 2014) but less or not at all analyzed in qualitative and semi-quantitative risk
 186 assessment.



188 **Figure 2. Parallel between Risk Management Framework adapted from ISO 31000:2009 and**
 189 **Assessment Framework adapted from ISO2394:2015**

190 2.2 Cost-benefit analysis

191 Cost-benefit analysis (CBA) has been widely used to rank alternatives when the decision problem is
 192 structured in deterministic terms. CBA assumes that for each alternative, a specific consequence can
 193 be assigned in terms of cost and different benefits. In addition, we can assume that the measures of
 194 the different benefits are of such nature that they cannot be simply summed up and that the DM puts
 195 a limitation to the available budget to achieve a certain improvement. The best alternative will be the
 196 one whose cost does not exceed the available budget and whose benefits fulfil minimum “*aspiration*
 197 *levels*” (Keeney and Raiffa, 1993). By applying CBA, the benefits are converted into one metric using
 198 conversion factors, so that they can be summed up in one equivalent benefit. The alternatives are then

ranked according to e.g. benefit-cost ratio and net present value. It is intuitive that CBA simplifies the evaluation of alternatives, especially considering that sometimes the conversion of the benefits into one measure can be very difficult to perform in rigorous terms. Additional limitations arise since both costs and benefits are often monetarized in CBA by deterministic values, where uncertainty of those values are not always available. CBA has been used in combination with risk management especially in flood management, where the probability distribution of the consequences associated with each alternative are identified by risk-based methods and the ranking of alternatives is conducted via CBA. However, CBA has also been applied in decision problems where no risk assessment is necessary such as improvement of the mobility of a region. Here the consequences are identified via a set of criteria assessed by mobility performance indicators, where CBA is applied for ranking. Discounting of costs and benefits over time is always applied in CBA, to actualize the future monetarized value of costs and benefits.

2.3 Life Cycle Assessment

LCA is applied in various cases to: 1) identify the environmental hotspots in the studied system or/and 2) compare the environmental impacts of different alternatives that can be applied in the studied system to achieve the same function. The system is analyzed over the whole life cycle of “goods or services (“products”)” (EC-JRC, 2010). According to ISO 14040 and 14044, there are four phases to conduct an LCA (Figure 3). The first phase formulates the question to be answered and defines the studied system. It first identifies the function to be provided by the product or service and describes it quantitatively and qualitatively in the form of functional unit, e.g. transport a certain number of people from one location to another over a certain number of years. Afterwards relevant elements that are needed to fulfill the functional unit is defined, e.g. two buses with capacity of 50 people. In the next phase, all relevant input and output in the form of resource consumption and emissions associated with the system’s delivery of the functional unit are quantified in an inventory. Here the data is collected for all life stages of the product or service, i.e. raw material, manufacturing, use and end-of-life stages. In the third phase, the environmental impacts caused by the flows listed in the inventory are quantified. For each flow in the inventory, there is a cause-effect chain that describes the relationship between the flow and the damages on an area of protection (natural environment, human health or resources). Depending on the location of indicators in the cause-effect chain, the impacts can be characterized either at midpoint level with relevant indicators and metrics (e.g. kg CO₂ equivalent for climate change, kg SO₂ equivalent for acidification, kg CFC-11 equivalent for ozone depletion, kg P or N equivalent for eutrophication, kg NMVOC for photochemical oxidant formation, etc.), or at endpoint level (i.e. human health damages described in disability-adjusted life years (DALY), ecotoxicity damage described in Potential Disappeared Fraction of natural species in the ecosystem (PDF) or resource depletion described in monetary terms) (Hauschild et al., 2013). It’s also possible to integrate the result into one single score, using weighting factors. In the fourth and final phase the outcomes are interpreted to answer the question that was posed in the goal definition, i.e. which product performs better or where is the hotspot? The interpretation can be performed either on the midpoint scores, endpoint scores, or on the single score, depending on the goal and stakeholder’s preference.

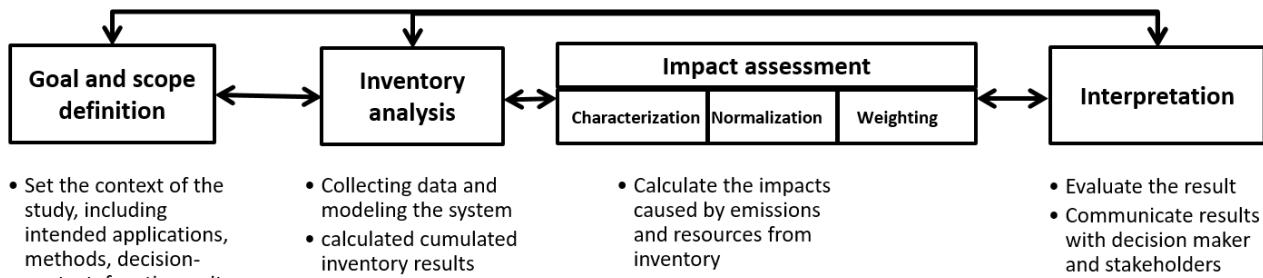


Figure 3. Steps of LCA, adapted from ISO 14040:2006.

Note that in addition to the conventional environmental LCA, social LCA (sLCA) and life cycle costing (LCC) also exist. They share the same principle as the conventional LCA, but looking at social impacts and cost flows respectively. Norris et al. (2001) proposed to use LCC for accounting economical cost and discussed two approaches for combining LCA and LCC for providing sustainability decision support. Hoogmartens et al. (2014) argue that LCA, sLCA and LCC together, the so-called life cycle sustainability assessment (LCSA), can well deliver a sustainability decision support. So far, sLCA, LCC and LCSA are used to a lesser extent due to the less mature methodology.

Uncertainty in LCA is addressed primarily on the input parameters by the practitioners, whereas the uncertainty on the impact assessment and model itself is rarely considered in current LCA practice, but mainly confined to academic applications (e.g. van Zelm and Huijbregts, 2013).

2.4 Summary

Decision analysis allows to rank available alternatives based on their consequences (e.g. in terms of human health impacts and economic benefit/loss), where environmental impacts are barely considered. LCA has the potential to fill this gap. It can be applied to analyze the environmental impacts associated with each alternative allowing their consideration together with the other conventional consequences for optimization. To identify the status and challenges involved in such combination, we look at application in three different research areas in the following sections.

3. Decision analysis in three research fields

In this section, we look into three research domains, namely flood management, transport and food production and consumption. We discuss and propose possible directions for integrating LCA-based environmental impacts into existing approaches used for decision support.

3.1 Decision analysis in flood management

Flood management is a fundamental societal service as it greatly increases the space where society can evolve and flourish. Protecting society is really protecting people and their assets from natural extremes, and since natural processes cannot be prevented, what is done is flood risk management rather than managing the floods; i.e. making sure that the risk of flood is acceptable from a societal point of view.

269 The frequently asked questions by the decision makers are “which technical solutions” will provide
270 “which degree of protection” and “at what cost”. To answer these questions, traditionally risk
271 assessment in combination with CBA is used as the decision analysis tool. In practice, risk assessment
272 is translated into risk (cost) curves as illustrated by Zhou et al. (2012) and Halsnæs et al. (2015). Here
273 the probability of scenarios are equal to the rarity of the natural phenomena causing the floods. The
274 consequence is a complex product based on GIS analysis of which assets are actually affected by the
275 flood and to what degree. Besides this, potential health issues stemming from the investigated flood
276 problems are also considered in the consequences (as in e.g. Halsnæs et al., 2015). After an evaluation
277 of the consequences in terms of monetary cost/benefit, CBA can be made and an optimal state where
278 the risk (cost) of flooding is balanced against the cost of the technical solutions set in place to protect
279 assets against flooding. Flood risk management is normally done on water catchment scale, either
280 locally or regionally, depending on the nature of the risk in a given catchment. Uncertainty exists on
281 input parameters of the consequence modelling, e.g. the variability of flood hazard occurrence rate
282 and the “range of climate change risk estimates”, and costing (Halsnæs et al., 2015). Those parameter
283 uncertainties are often quantified, but the underlying model uncertainty is rarely treated explicitly.

284 Even though risk assessment in combination with CBA is a mainstream decision analysis tool in flood
285 management, it is generally agreed that in its most widely used form, it excludes important relevant
286 aspects in decision making if they are difficult to monetarize (Merz et al., 2014). Or the conclusions
287 are made mainly dependent on the chosen pricing of the non-structural values (e.g. amenity and health)
288 (Halsnæs et al., 2015; Zhou et al., 2012). Thus, MCA is sometimes used instead of CBA to include
289 consequences at different levels, including e.g. technical, hydraulic, environmental, sociological,
290 economic, planning, operation and maintenance aspects (Martin et al., 2007), sustainability (Lai et
291 al., 2008), and non-stationarity, i.e. and when in time to optimally invest in protection for systems
292 undergoing climate change (Åström et al., 2014).

293 **How are environmental aspects considered?**

294 As stated in the EU flood risk management directive, “It is feasible and desirable to reduce the risks
295 of adverse consequences, especially for human health and life, the environment, cultural heritage,
296 economic activity and infrastructure associated with floods” (European Commission, 2007). This
297 clearly indicates that environmental impacts are desirable, but not mandatory to be included in flood
298 risk management. As a consequence, it has “rarely been included in cost assessments up to now”
299 (Meyer et al., 2013). Some studies have made the effort. For example, health costs were accounted in
300 the case of Odense urban flooding management (Halsnæs et al., 2015). Another example is the
301 inclusion of “greening” of cities in urban water management, such as green roof and green areas, is
302 considered as the main sustainable measures (Belmeziti et al., 2015; Zhou et al., 2012). Nevertheless,
303 methods have been tested to assess environmental impacts of infrastructure systems, mainly using
304 LCA-based methods. But unfortunately only on cases that have come out of existing flood risk
305 practice for further qualification of decision (e.g. Brudler et al., 2016), and not as input to decision
306 regarding acceptable flood risk.

307 When facing several alternatives for urban water management strategies (e.g. “green” or “grey”
308 infrastructure), LCA is a good tool for accounting resource consumption and environmental impacts
309 for the whole life cycle of the strategy as demonstrated by Brudler et al. (2016), De Sousa et al. (2012)
310 and Spatari et al. (2011). LCA has also been conducted on smaller scales of infrastructure, e.g.
311 stormwater treatment devices (Andrew and Vesely, 2008) and bio-infiltration rain gardens (Flynn and
312 Traver, 2013). The results indicate that sometimes the apparent “green” strategy does not necessarily
313 perform better in environmental impacts (De Sousa et al., 2012). Hence, it introduced important
314 insight into the environmental impacts and it is worthwhile to integrate LCA into the current decision
315 analysis for flood management. The provided environmental profile of the alternatives may have an
316 impact on the stakeholder’s decision. To achieve this, a possible approach is to assess the
317 environmental impacts of each alternative and monetarize it as one cost/benefit in the traditional CBA,
318 which will be further discussed in section 4.

319 **3.2 Decision analysis in transport projects**

320 The primary aim of transport projects is to improve the mobility of persons and goods, often on local
321 or regional level. This aim can be achieved in different ways, such as building a new infrastructure to
322 increase the access to a specific location or providing a new public transport service. However, often
323 several alternatives are at hand and a decision has to be taken in order to decide which one to
324 implement. Traditionally, the decision analysis for transport projects is based on CBA. It facilitates
325 the decision makers to choose the preferred alternatives based on a number of socio-economic
326 budgets. In transport CBA some of the key variables are the output of transport demand models, such
327 as travel time savings and vehicle kilometers travelled, while others are derived from such output,
328 like number of accidents, noise and emissions, the so called “external costs”. A challenge is to address
329 the uncertainty inherent in the variables included in CBA and how it propagates to the final results.
330 This issue has not been included in the standard CBA for transport until recently, although literature
331 reports on investigations of how to quantify parameter uncertainty in both transport models (de Jong
332 et al., 2007; Rasouli and Timmermans, 2012) and CBA (Fagnant and Kockelman, 2012; Salling and
333 Leleur, 2015). In this respect, uncertainty in transport projects is commonly treated through stochastic
334 simulations techniques such as Monte Carlo Simulation, and scenario analysis, in both scientific
335 literature (de Jong et al., 2007) and practice (European Commission, 2014).

336 Transport CBA is usually complemented by other evaluation methods to cover more impacts.
337 Particularly relevant is the assessment of the so called wider economic impacts of the (transport)
338 project, such as the agglomeration impacts and the effects on the labor market (Eddington, 2006),
339 although some criticism has been raised with respect to the possibility of including them as part of
340 standard transport evaluation frameworks (Gibbons and Overman, 2009). Besides, traditional CBA
341 is sometimes replaced, combined or conducted in parallel with MCA. Nevertheless, some impacts
342 remain difficult, if at all possible, to quantify through standard transport decision support methods,
343 such as long term environmental impacts (Engelbrecht, 2009).

344 **How are environmental aspects considered?**

345 With respect to the appraisal of the environmental impacts deriving from transport projects, as
346 mentioned in the above paragraph, the assessment of some of the environmental costs, such as air
347 pollution and noise from vehicle operation, are normally included in standard CBA frameworks,
348 following the EU guideline on CBA (European Commission, 2014). To calculate the costs related to
349 noise and air pollution a bottom-up approach is commonly used. First, the amount of noise and
350 pollution is quantified based on the estimated volumes of traffic, expressed in terms of vehicle
351 kilometers (passenger vehicles) or ton kilometers (freight vehicles) travelled. Then, the estimated
352 quantities are translated into monetary terms, based on available values from national and
353 international guidelines.

354 However, some environmental costs related to the entire life-span of the project, such as the resource
355 use, and some impacts on ecosystem and human health, are not covered. In addition, impacts
356 associated with vehicles and infrastructures manufacturing and maintenance are usually not
357 considered either. For instance, the Danish CBA guidelines (Danish Ministry of Transport, 2015)
358 requires the inclusion in transport CBA of the monetary impacts of CO₂, NO_x, HC, CO, PM_{2.5} and
359 SO₂ deriving from vehicles emissions but not from e.g. the construction of the infrastructure or the
360 maintenance of the vehicles. Consistently, transport CBA projects only addresses vehicles emissions
361 as reviewed by Annema et al. (2017).

362 In order to include the project life-span environmental costs into quantitative decision analysis for
363 transport projects, some studies have applied LCA in evaluations. The existing literature can broadly
364 be divided in two topic areas. The first, identifies the missing elements in the environmental impacts
365 embedded in current decision analysis (Chester and Horvath, 2009) and points to hotspots where
366 environmental improvement can be made, e.g. passengers and household behavior (Chester et al.,
367 2010; Kimball et al., 2013). The second research area focuses instead on using LCA to quantify the
368 environmental impacts of single elements in transport system, such as railway infrastructure
369 (Linneberg et al., 2014), bridges (Hammervold et al., 2013), vehicles driven by different fuels or
370 electricity (Bohnes et al., 2017; Garcia and Freire, 2017; Lombardi et al., 2017) and different mobility
371 modes such as public bus and trucks (Ercan et al., 2015; Sen et al., 2017).

372 Despite the proven importance and possibility of assessing a broader range of environmental impacts,
373 LCA is commonly not included within standard transport projects assessment frameworks. A feasible
374 way for integrating LCA into current decision analysis for transport might be to monetarize
375 environmental impacts assessed in the LCA, catering to their inclusion into standard CBA. Manzo
376 and Salling (2016) practiced the proposal. The result shows that the inclusion of environmental
377 impacts assessed by LCA indeed affecting the final project evaluation, while modifying the
378 contribution from different components in the system. However, a better guideline for practice is
379 needed for a better integration, which will be further discussed in section 4.

380 **3.3 Decision support in food production and consumption**

381 The decisions made within the food production and consumption are largely aimed at assuring food
382 safety and food security: prevent foodborne illness and guarantee its availability and good quality for
383 the whole population. For this purpose, risk management (also called food safety risk analysis) is the

384 most common used decision analysis tool in the field as illustrated in Section 2. A hazard in this
385 context is defined as “a biological, chemical, or physical agent in or property of food that may have
386 an adverse health effect” (WHO, 1995). After identifying the potential hazards, a full profile of the
387 associated adverse effects on health is characterized, quantitatively or qualitatively. Exposure
388 assessment is applied to find the amount and likelihood of intake. Which is then applied in a dose-
389 response relationship to estimate the risk of disease, i.e. the probability and severity of health effects
390 that is caused by the hazard investigated (FAO and WHO, 2005). Food risk management can be
391 operated on all geographical scales, including local, regional and global.

392 There are two major types of food risk assessment, according to the hazard property: microbial and
393 chemical risk assessments. The major challenge for microbial risk assessment is to estimate the
394 ingested dose, which is often done using stochastic modelling. Uncertainty on our knowledge of e.g.
395 foods items contamination, pathogen survival and growth in the food product, and the probability of
396 disease given a certain dose are often taken into account. The focus of chemical risk assessment is on
397 the presence of potential harmful chemicals in the food (FAO and WHO, 2005). The allowed dose
398 for a certain substance is often derived from animal testing, or calculated using models. The dose
399 level where no adverse effects are observed is then divided by uncertainty factors (or safety factors)
400 to ensure their safe application on human beings. These uncertainty factors are applied to account for
401 interspecies and intraspecies variability. They are intended to assure adequate safety of the final
402 toxicological value but may actually result in overly conservative safe dose estimates. The uncertainty
403 mentioned above focus on parameter uncertainty, which is usually treated in studies. In contrast,
404 model uncertainty is rarely quantified.

405 In addition to the traditional risk assessment mentioned above, CBA is sometimes also used for food
406 decision making. Some food such as fish and nuts has positive benefits on human health, but they can
407 also contain harmful substances, e.g. heavy metals and carcinogenic toxins. The negative risks are
408 sometimes compared with the positive benefits of food to help determine whether the food has an
409 overall health benefit (EFSA, 2006).

410 **How are environmental impacts considered?**

411 In terms of food security, there is a strong need for a more sustainable food production and
412 consumption to be able to feed the predicted global population of 9.5 billion people by 2050, with
413 respect of less environmental impacts and resource depletion. UNEP presented several long term
414 targets and indicators for “sustainable agriculture and food security” (UNEP, 2014), e.g. reducing
415 food and nutrient loss along production and consumption. Van der Goot et al. (2016) concluded that
416 current food production manners are not very efficient, where the losses of food are significant along
417 the production chain. Research and application of technology development and new farming systems
418 are promoted by European Commission to enhance sustainability in food production (Freibauer et al.,
419 2011). These sources point to the fact that currently there is an increasing attention to reduce
420 environmental impacts in food production and consumption. This is also reflected in the EU
421 regulations such as food law (European Commission, 2002) that “the protection of animal health and
422 welfare, plant health and the environment” should be pursued in food regulations. But still a

423 harmonized system and operation procedure is missing to implement environmental considerations
424 in decision making along the whole food production chain from primary production to consumption.
425 LCA has been extensively conducted for the production and processing of industrial food products,
426 dairy and meat production, fruits, and agricultural products. As summarized in Arvanitoyannis et al.
427 (2014) and Roy et al. (2009), these studies mainly aim at 1) identifying hotspots in the system for
428 future improvement, and 2) comparing different food and their related products (e.g. packaging), to
429 identify the best choice regarding environmental impacts. These LCA studies shows a strong potential
430 of solving food security problem in the cause of less environmental impacts.

431 Food production is an important source of many environmental impacts and there are potential trade-
432 offs between food risk minimization and sustainability of the alternatives in food production system.
433 However, environmental sustainability aspects are rarely taken into account. It will thus be beneficial
434 to integrate LCA into the current food risk management practice to quantitatively assess
435 environmental impacts associated with the alternatives that minimizes the risks. There are some
436 common metrics (e.g. DALY) that are used to present results both in LCA and food safety risk
437 assessment, which may potentially serve as the basis for the integration of LCA and risk assessment
438 for food safety as shown in Stylianou et al., (2016). Note that DALY only describes human health
439 consequences, whereas animal health and welfare, and impacts on ecosystems cannot be expressed.
440 For those aspects, research is needed to convert LCA output in a valuable metric for food safety risk
441 assessment.

442 **4. Discussion on the need, obstacles and research agenda for integrating LCA into decision** 443 **analysis**

444 We have screened the major criteria considered in the current decision analysis of three application
445 areas. It turns out that economical costs and benefits are the major concerns in transport and flood
446 management, where CBA is often used to prioritize alternatives in decision analysis. Human health
447 is the focus in food safety related decisions, where traditional risk management is often used to
448 prioritize alternatives. Environmental benefit/cost is rarely considered in flood management, and to a
449 very limited extent in transport projects, focusing on few pollutants in few life stages. In food safety
450 related decisions, though human health caused by the food itself is well taken into account, the
451 environmental impacts arising from the rest of the food system, are not considered and these also
452 have the potential to impact negatively on human health e.g. through climate change or release of
453 toxicants that expose humans through the environment. Comprehensive environmental considerations
454 are not well addressed in decision analysis tools, making them inapt to support decisions towards
455 sustainability.

456 LCA offers a solution to the problem. It has been applied in transport and flood management to assess
457 the environmental impacts arising from infrastructures and resource consumptions. LCA also has the
458 potential to quantify the environmental performance of food risk mitigation actions. Such results can
459 provide valuable information to support decisions if combined with the main decision analysis tool
460 such as CBA or risk assessment. A summary of the main elements in decision analysis for the three
461 research areas covered in this study and LCA is presented in Table 1. It can be clearly seen that there

are many discrepancies between the conventional decision analysis and LCA. First of all the goal is different. While LCA aims at assessing environmental impacts, the conventional decision analysis tries to solve decision problem such as how to protect human beings and properties from certain risks, and how to provide a service to satisfy human being's basic needs. This results in the different choice of principles, i.e. non-precautionary, precautionary and cautionary principle. The covered impacts, study system and uncertainty treatments also vary between different methods.

These discrepancies are confirmed by other references. Cowell et al. (2002) discussed the application of risk assessment and LCA in regulatory context. They found that, similar to the situation in flood management and food risk assessment in this study, risk assessment works with a precautionary principle, where both absolute and comparative results can be delivered. LCA, in contrast, aims at quantifying the average or marginal consequences, and only comparative results are expected. In analogy to the transport project, Hoogmartens et al. (2014) highlights that CBA has been mainly applied for policy or strategic decision making, where the project is the main focus, meaning that the time span and system boundaries are defined by the project. CBA puts emphasis on the socio-economic impacts rather than external environmental impacts. In contrast, conventional LCA is product oriented, which results in the different system boundaries, and it focuses on environmental impacts rather than social and economic impacts. In recent years, LCA has been used in a broader scope, such as assessing impacts for services (Barjoveanu et al., 2014), urban metabolism (Goldstein et al., 2013), and larger scale applications (Lotteau et al., 2015) and territorial planning (Loiseau et al., 2018). Correspondingly, the term of "product" has been extended from a single product to services, sectors, cities, etc. Still, the focus is on the function provided by the service, product or system, which is different from CBA and risk assessment. By integrating two different methods together to solve the same decision problem, those discrepancies bring us the opportunities to obtain a more comprehensive picture of the potential consequences of the decision. But the methods and results also need to be integrated in a transparent and coherent way to avoid inconsistency among options under comparison. Main discrepancies from our three application areas are discussed in the following section, concluding with a proposed future research agenda.

4.1 Compatible study system

A clear identification of the boundaries and temporal scope of the system under assessment is necessary, such that the system is identical for both LCA and the other traditional decision analysis approaches. System boundaries are not always easy to identify. In CBA and risk assessment, the system is defined by the project or hazard prone area, meaning that only elements and associated life stages relevant for the life cycle of the system are included (i.e. both spatial and temporal boundaries). In contrast, the system boundary defined in LCA as function to be provided (function unit), within all life stages of the system (i.e. both temporal and spatial boundaries). This leads to difficulties in identifying a consistent system boundary when the results delivered by two methods need to be combined into a common decision pathway. Whereas incompatibility of boundaries is not avoidable, it is on a case-by-case discussion whether that incompatibility has a large influence on the results. And it needs to be transparently presented to the audience so that they are aware of the differences. For example, in flood management and transport projects, an infrastructure is a mean to improve mobility or to prevent floods. According to the specific goal and scope of the analysis, the

503 infrastructure may provide more than one function from the perspective of CBA, risk assessment and
504 LCA. For instance, a dike may be built with a road on the top. This provides extra mobility function
505 in addition to its primary function, i.e. storm water management. If those two alternatives are for
506 comparison, one dike with road and one without, and the scope of the project is only storm water
507 protection with system boundaries limited to the dike itself, then the two alternatives are equivalent
508 in the prospective of conventional flood management. However, they are still different in LCA, since
509 one alternative provides an extra mobility function, which needs to be accounted for and compared
510 to a matching infrastructure providing the same mobility function within the system boundary.
511 Allocating the proper share of environmental impacts to the main function can potentially align the
512 system boundary with CBA, where caution is needed for the alignment. Another concern is which
513 life stages to include in the system when considering that some of the environmental impacts
514 occurring along the life cycle do not affect the project location, e.g. the emissions for construction
515 material production may not happen in the same place where the infrastructure is built and used. From
516 the perspective of LCA, all emissions regardless of location should be included. In contrast, CBA of
517 an infrastructure may only take into account the emissions that happens within its concerned local
518 geographical scope as a valid environmental cost (EIB, 2013; European Commission, 2014). This
519 emphasizes that the geographical coverage needs to be clearly identified during the project assessment
520 and communicated to the decision process.

521 Temporal scope is another issue that needs to be addressed. Decision analysis is always dealing with
522 time. Benefits and costs are discounted over years in e.g. transport projects and flood management in
523 standard methods. But when it comes to impacts on human health, discounting is not always
524 conducted due to ethical issues (Motarjemi et al., 2014). LCA calculates impacts from Life Cycle
525 Inventory (LCI) flows representing the aggregated load of emissions over the life cycle of a product
526 or service. The traditional LCA thus only provides time-integrated results over the product. Recently,
527 studies have explored dynamic LCA, considering that emissions in reality often happen over a period
528 of time (Levasseur et al., 2010). However, applying discounting across generations on environmental
529 impacts in LCA is not encouraged, due to ethical concerns similar to the ones applying to the human
530 health impacts (Hellweg, 2003). There is no single answer to whether LCA results should be
531 discounted when they are being integrated with other decision analysis tools. It depends on the impact
532 category and the decision context. However, as a rule of thumb, it is essential to have consistency in
533 discounting when aggregating similar impacts in LCA and other decision analysis tools.

534 **4.2 Cautionary principle vs. non-cautionary principle**

535 "Cautionary principle means that caution, for example by not starting an activity or by implementing
536 measures to reduce risks and uncertainties, shall be the overriding principle when there is uncertainty
537 linked to the consequences" (Aven, 2008). In the extreme case where scientific uncertainty of the
538 consequence is lacking, e.g. flooding due to climate change, it can be referred as precautionary
539 principle (Aven, 2008). The cautionary principle is applied in many contexts, especially where risk-
540 based decision approaches are applied. Thresholds are one way to apply a cautionary principle, e.g.
541 there are thresholds for many chemicals that cannot be exceeded in food products. In EU food policies,
542 those thresholds are even set up according to precautionary principle that supports taking protective
543 action before a complete scientific proof of a risk, e.g. prohibition of the use of growth hormones in

544 beef production. Similarly, cautionary requirements exist for infrastructures, where a certain level of
545 safety against natural disaster, such as flood and earthquakes, needs to be achieved. In those cases,
546 the cautionary and precautionary thresholds and requirements will limit the number of alternatives
547 available for optimization in decision analysis. In contrast, LCA aims at comparing environmental
548 burdens with “best estimate of risk on the basis of scarce knowledge” (Hauschild, 2005). The
549 cautionary constraints are not applied in LCA, which does not limit the applicability of alternatives
550 via thresholds and requirements as in the cautionary principle based tools. Recently developed
551 methods aiming at translating LCA results into limited carrying capacity may help harmonize this
552 discrepancy (Bjørn and Hauschild, 2015).

553

554 Table 1. Summary of main elements in decision analysis and LCA for the three research areas in this study

		Conventional decision analysis			Life Cycle Assessment
		Flood management decision analysis	Transport project decision analysis	Food production and consumption decision analysis	
Main method		Risk assessment in combination with CBA	CBA (often combined with MCA)	Risk assessment	Life cycle inventory quantification and life cycle impact assessment
Studied system	System components	People and assets within the urban environment (e.g. buildings, roads, water infrastructure).	Individuals and households transport choices, transport services, policy and infrastructures	Agriculture farming, food production system, food consumption and waste treatment	<ul style="list-style-type: none"> - For flood: Infrastructure of urban water management system - For transport projects: Infrastructure of transportation system, vehicles driven by different fuels and mobility modes - For food production and consumption: Farming process, food production system, consumption and waste management
	System boundary	All relevant elements within the utility function; including: identified risks, actions to be taken, investment, costs of consequences, etc.	Costs and benefits foreseen from the project expressed in monetary terms	All relevant elements within the utility function, including identified risks, actions to be taken, investment and costs, etc.	All flows related to delivering the functional unit, including raw materials, production, consumption/ maintenance and waste management
Goal and Scope	General goal	Protect people and assets from flood. Control the risk of flood to an acceptable level	To guarantee and enhance the mobility of persons and goods	Guarantee food safety and availability	Assess impacts on ecosystem and human health on comparative bases
	Scale	Water catchment (Local/Regional,	Local or regional	Local, regional and global	Impacts caused by the studied system are assess in a comprehensive way,

		usually following natural water divides)			Depends on the impact category, the scale of consequences can be global (such as climate change) or local (such as eutrophication)
	Cautionary or non-cautionary	Precautionary	Cautionary in most cases	Precautionary and cautionary	Non-cautionary
Covered impacts		Economic benefit/loss	Socio-economic benefits/cost; environmental impacts to a much less degree	Human health; impacts on ecosystem to a less degree	Environmental impacts, human health impacts and resource impacts
Location and time		The consequences are time and location dependent			The consequences are integrated over time and location is unspecific
Uncertainty	Parameter uncertainty	Usually treated	Usually treated	Usually treated	Occasionally treated
	Model uncertainty	Rarely treated	Rarely treated, but recommended by guidelines	Rarely treated	Rarely treated, but recommended by guidelines

556 4.3 Uncertainties

557 The reliability of the decision analysis results can only be assessed through an evaluation of the
558 uncertainty. As mentioned in section 2, parameter and model uncertainty are the two essential forms
559 of epistemic uncertainty and variability in an analysis. The usual approach for dealing with the
560 parameter uncertainty is to assign a probability distribution to each input parameter, fitting the best
561 to data. In LCA, only four types of distributions are normally used due to software limitations and
562 the complexity of the modelled product life cycle: normal, lognormal, uniform and triangular
563 distributions (Heijungs and Frischknecht, 2005). In the decision analysis approaches applied in flood,
564 transport and food, more distribution patterns are applied, e.g. Gamma, Beta, Weibull and
565 Generalized Extreme Value distributions (Faber, 2012). Generally, distribution patterns are fitted to
566 data in the other decision analysis approaches, but not often in LCA. Compared to parameter
567 uncertainty, model uncertainty is given much less attention by practitioners, both in LCA and decision
568 analysis within the three discussed research areas, and often it is completely neglected.

569 Monte Carlo and other simulation methods are the primary approach to propagate the uncertainty to
570 the output in the decision analysis for all three considered research areas and LCA, assuming different
571 distribution patterns of chosen input parameter (de Jong et al., 2007; Halsnæs et al., 2015; Lloyd and
572 Ries, 2007; Vose, 1998). An important feature of LCA is that it provides impacts on a site-generic
573 and time-integrated scale. Thus, spatial and temporal variations are not often represented in the result.
574 In the recent years efforts have been made to develop spatial differentiated LCA methods (Huijbregts
575 et al., 2015; Wernet et al., 2016). However, due to the diverse location of elements included in the
576 system, data availability, and the diverse location of impacts, it is not easy to reach a systematically
577 regionalized LCA result. On the contrary, risk-based methods and CBA provide full uncertainty
578 quantification in time and space. These aspects need to be harmonized in the problem definition when
579 integrating LCA with other decision analysis approaches.

580 4.4 Combining LCA and other decision analysis tools – current status and recommendations

581 There are some studies discussing the similarity and difference between LCA and risk assessment.
582 Olsen et al. (2001) concluded that LCA and risk assessment are not substitutable due to different aims,
583 scopes, etc. Bare (2006), Flemström et al. (2004) and Cowell and Clift (2000) reached similar
584 conclusions after comparing LCA and risk assessment within the context of human health impact,
585 chemical toxicity and public decision making respectively. Despite the difficulties, a few studies have
586 still attempted to combine LCA with risk assessment for a better decision support. Linkov et al. (2017)
587 summarized that two mainstream methods exist for such integration. One is to apply risk assessment
588 on different life stages where risky materials appear. The other one is to apply risk exposure pathway
589 and impact in LCA methodologies in some impact categories, such as the method proposed by OECD
590 (2015) for nano-enabled applications. Harder et al. (2015) reviewed case studies blending risk
591 assessment and LCA. They conclude that in addition to the two methods mentioned above, LCA and
592 risk assessment in many cases studies were conducted in parallel to complement each other. However,
593 none of them can really deliver results integrating the environmental impacts and risk consequences
594 (e.g. Barberio et al., 2014; Dhingra et al., 2010; Ribera et al., 2014). Guinée et al. (2017) points out
595 that tools such as multi-criteria decision-making is emerging in the past decade to deliver a full
596 combination of LCA and risk assessment result (e.g. Benetto et al., 2007; Linkov and Seager, 2011;

597 Tsang et al., 2014). Normalization or weighting may be performed to allow the harmonization of
598 assessment results from LCA and risk assessment depending on the stakeholders' preference, where
599 the results can be combined together. These studies show that though obstacles exist, efforts has been
600 made to integrating LCA with risk assessment, that may serve as inspiration e.g. for flood and food
601 decision analysis when risk assessment is the main tool.

602 The combination of LCA and CBA has also been conducted in few studies. Møller et al. (2013) used
603 LCA to quantify energy consumption and CO₂ of biofuel production. There "welfare economic
604 accounting prices" were assigned to those results and integrated into traditional CBA for comparing
605 the consequences of using three different biofuels in Denmark. In Jones et al. (2017), CO₂, SO₂,
606 PM₁₀, NMVOC and N emissions were quantified over the life cycle of a transport service provided
607 by train, which are further monetized and integrated into CBA to calculate NPV. Huang et al. (2017)
608 conducted LCA and CBA in parallel for assessing the "environmental and cost impacts of reusing fly
609 ash", where a normalization factors were given to LCA and CBA results respectively for combination.
610 On the other hand, Hoogmartens et al. (2014) identified the obstacles for combining LCA and CBA
611 in terms of discrepancies in key focus (product vs. strategies), life span, life stages covered, metrics,
612 and system boundaries such as whether to include impacts on broader society. These examples show
613 that multiple ways of combining LCA with CBA exist, that can serve as the basis when integrating
614 LCA into decision analysis in transport and flood management. But caution is needed as also
615 discussed in above sections.

616 Note that in addition to the conventional environmental LCA (eLCA) as we mentioned in this study,
617 social LCA (sLCA) and life cycle costing (LCC) also exist, though used to a less extent due to the
618 less mature methodology. They share the same principle as the conventional LCA, but looking at
619 social impacts and cost flows respectively. Norris et al. (2001) proposed to use LCC for accounting
620 economical cost and discussed two approaches for combining LCA and LCC for providing
621 sustainability decision support. Hoogmartens et al. (2014) argue that eLCA, sLCA and LCC together,
622 the so-called life cycle sustainability assessment (LCSA), can well deliver a sustainability decision
623 support. They imply that instead of integrating LCA into CBA, effort should be put on translating
624 CBA into LCC for such integration. However, obstacles mentioned in the previous sections still need
625 to be conquered, and it may face even more challenges to convince the decision makers to switch
626 from CBA to another method as the main decision analysis tool.

627 An increasing trend of using LCA for sustainability assessment is observed, in addition to the
628 combination with risk assessment, CBA and LCC as mentioned above. LCA used often for assisting
629 eco-design of products (Bovea and Pérez-Belis, 2012). Arena et al. (2013) propose a streamlined
630 LCA framework, where important impacts in each life stages of a car life cycle was extracted from
631 LCA and other guidelines or standards. They are developed into a qualitative performance evaluation
632 system to be used in the early stage decision making. Welsh-huggins and Liel (2017) did parallel risk
633 assessment and LCA study on buildings with green roofs, where trade-offs was demonstrated between
634 cost, material, hazard resistance and environmental impacts. As pointed out by Jeswani et al. (2010),
635 many possibilities exist for broadening LCA in its use for a better sustainable decision support,
636 especially in combination with other decision analysis tools such as strategic environmental

assessment, environmental impact assessment, multi-criteria decision analysis, LCC, CBA and sLCA. However, whether a better and more systematic LCA is needed in relevant decisions largely depends on the context of study and more importantly, the stakeholders' preference. Indeed, one reason for not including comprehensive environmental assessment results in the decision analysis is stakeholders' perceptions and priorities. Although integrating LCA into decision analysis may give a better overview of potential environmental consequences that may eventually cause damages to society, these consequences are considered external in the conventional projects. As no mandatory requirements exist, most stakeholders prefer not to internalize such external costs. Another concern is the limited social acceptance when environmental impacts and their associated damages to humans and ecosystems need to be monetized or normalized to another metrics, to be able to integrate it with results from other decision analysis tools. Monetizing of non-market things using "willingness to pay" assessments assumes that individuals have the same preference in giving up or obtaining the same thing and that this willingness is not changed when facing public decision instead of private transactions (Kelman, 1981). Both assumptions are not true in reality, which may result in far-off monetary values. Putting values on human life is another issue that will always be argued on fairness and human rights (Bayles, 1978). These methodological and ethical issues may further hinder the stakeholders' willingness to include external environmental impacts into decision analysis.

Even though obstacles exist, both from methodology and stakeholders' willingness, to integrate LCA into risk assessment and CBA, the examples given above show that it may make an important difference to the decision when LCA is taken into account. The three application areas, transportation, flood management and food production and consumption, all target at decision making at societal level. Their environmental consequences will cause damage to nature and society, and disregarding them in decision analysis will eventually cause more problems to fix afterwards. Climate change is a good example of paying such prices after ignoring GHG emissions in the past. Therefore, it is highly relevant to integrate sustainability considerations into decision analysis now, e.g. using the LCA-based approaches described above to support robust decisions that avoid shifting burdens to the future.

5. Conclusions and perspectives

It is clear that economic benefit and cost, and impacts on human health are major concerns in decision analysis within the three research areas presented in this study. Few attempts exist to assess environmental impacts, e.g. noise and air quality assessment in transportation CBA, classification of green or grey facilities in flood management and reducing waste and harmful elements in food production and consumption. However, those methods either cover only few environmental impact categories, or act as a guideline without actually assessing the impacts. There is thus a clear need for a better assessment of environmental impacts to be incorporated into decision analysis for these research areas as well as in general in order to support sustainable system choices. As a promising tool, LCA provides a mature and ISO standardized methodology to assess a full set of environmental impacts. Previous applications in the three studied research areas have demonstrated its ability to support an informed judgement on the environmental profile of the compared alternatives. However, there are still many challenges ahead. Due to lack of common scopes and purposes, and methodological differences, the aim of study and system boundaries need to be aligned as much as

677 practically possible between LCA and the traditional decision analysis tools, assuring the
678 compatibility in the comparison and/or aggregation of the results when possible. Similarly,
679 uncertainty and discounting are treated differently, where an alignment is needed. Moreover, metrics
680 in LCA (e.g. DALY and PDF) are different from the ones used in other decision analysis tools.
681 Although recent studies show that there are multiple ways of integrating LCA and other decision
682 support tools, further research is needed to overcome these challenges.

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686

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