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Francesco Castellani

Assessment of supply chain sustainability of bio-composite materials

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Assessment of supply chain sustainability of bio-composite materials

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submitted by
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È cosí che a forza di correr dietro a quelle immagini, io le raggiunsi.
Ora so di averle inventate.
Ma inventare è una creazione, non già una menzogna.
A forza di desiderio, io proiettai le immagini,
che non c'erano che nel mio cervello,
nello spazio in cui guardavo,
uno spazio in cui sentivo l'aria,
la luce ed anche gli angoli contundenti
che non mancarono in alcuno spazio per cui io sia passato.
Italo Svevo, *La coscienza di Zeno*

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List of Abbreviations and Acronyms

ABS	acrylonitrile butadiene styrene
BAT	Best Available Technique
C	carbon (chemical element)
CED	Cumulative Energy Demand (environmental impact indicator)
cf.	compare
CH_4	methane
CO_2	carbon dioxide
CO_2-eq	carbon dioxide equivalents
e.g.	*exempli gratia* (for example)
et al.	*et alii* (and others)
etc.	*et cetera* (and so on)
EPA	United States Environmental Protection Agency
EU	European Union
GHG	greenhouse gas
GWP	global warming potential
GWP_{100}	global warming potential (time horizon 100 years)
h	hour
ha	hectare = 0.01 km^2
i.e.	*id est* (that is)
K	potassium (chemical element)
km	kilometer
kWh	kilowatt hour
kWh_{el}	kilowatt hour (eletrical)
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LCSA	Life Cycle Sustainability Assessment

m	meter
m^3	cubic meter
MCDA	Multi-Criteria Decision Analysis
N	nitrogen (chemical element)
OR	Operations Research
P	phosphorous (chemical element)
PA	polyamide
PB	polybutylene
PBS	polybutyrate succinate
PBSA	polybutyrate succinate-co-adipate
PCL	polycaprolactone
PE	polyethylene
PEIT	polyethylene-co-isosorbite terephthlate
PET	polyethylene terephthlate
PHA	polyhydroxyalkanoate
PLA	polylactic acid
PP	polypropylene
ppm	part per million
PTT	polytrimethylene terephthalate
PUR	polyurethane
PVC	polyvinyl chloride
R&D	research and development
t	ton = 1000 kg
TPS	thermoplastic starch
UNFCC	United Nations Framework Convention on Climate Change
USA	United States of America

1 Introduction

Industrial processes are currently based on considerable consumption of fossil resources. Bringing down the level of greenhouse gases (GHG) emitted into the atmosphere when extracting and processing this kind of resources has been a main driver for the development of fossil resources substitutes (e.g., bio-based, biodegradable materials) and for the adoption of strategies to promote their efficient use. The European Commission, for instance, proposed a joint Statement on cascade use of wood (AEBIOM et al., 2013), in order to promote active forest management and improve resource efficiency (Geldermann et al., 2016a). However, the choice between conventional fossil-based materials and bio-based ones requires the consideration of many aspects along the supply chain. Supply chain optimization, increment of efficiency and product quality, reduction in GHG emissions, and reducing costs are only some of the goals that decision-makers must face to maintain and increase business competitiveness. This situation concerns any production system, both in industry and agriculture. The horticulture sector, a relevant sub-sector of agriculture, deals with the cultivation of edible products (e.g., fruit and vegetables) and ornamental products (e.g, decorative plants, trees, potted plants, and cut flowers). Recently, it has been realized that the supply chain of current production system in horticulture settings could be improved to reduce costs and emissions (Lazzerini et al., 2016). Horticultural market requests, indeed, a change, and it asks for the introduction of innovative, more sustainable products. However, crops cultivation is influenced by many factors, such as biological aspects (e.g., pathogens) and external factors (e.g., weather), which determine high vulnerability to risks. Growers, horticulturists, and plant nursery decision-makers face therefore the challenging task of choosing between conventional fossil-based resources and their substitutes.

Peat is a non-renewable fossil-based resource accumulated over millions of years in peatlands. The extensive exploitation of peatlands to extract peat as fuel or growth medium has remarkably reduced the global carbon storage (Gorham, 1991), and it has consequently raised environmental concern (Schmilewski, 2013). Peatland protection aims to maintain both biodiversity and such ecosystem services as climate protection and nutrient retention. However, peat is currently the main component of growth substrate for cultivating plants in horticulture. To preserve the environment from the massive use of this fossil-based resource against GHG emissions increase, limitation to peat utilization has been introduced in some European Countries (e.g., Niedersächsisches Ministerium für Ernährung, 2017). Consequently, horticultural decision-makers started to investigate and test peat substitutes for cultivating plants, and agro-waste compost is a promising suitable resource towards this end. Along with other agricultural media, compost can be mixed with sphagnum peat to form growth substrate for potted and field-grown plants. The substitution could lead to reduction in GHG emissions caused by the horticulture sector. In horticulture, another environmental concern is the plastic waste management of pots used for cultivating. Since they cannot be recycled easily due to soil and vegetable matter contamination, agrochemical residues and additives, they are generally landfilled after one usage (Schettini et al., 2013). Here, biodegradable pots made of renewable resources, which can be embedded in the soil with the plant or disposed of in composting facilities, represent a viable alternative to plastic pots. Both novel materials, compost and bio-based pots, can reduce the environmental emissions of a company and attract customers with willingness to pay for these more environmentally-friendly products. By doing so, plant nurseries might improve their image and become more competitive.

However, when mixing peat with compost beyond a certain replacement percentage, the agronomic quality of a potted plant becomes unacceptable, and substituting for peat may also result in higher costs for processing and handling. Moreover, biodegradable pots have higher costs than plastic pots. These aspects introduce conflicting goals for the decision-maker: reducing the environmental impacts and at the same time reducing additional costs that incur when substituting for peat and plastic pot.

This leads to the following research question: *Can decision-makers of plant nurseries substitute innovative, bio-based materials for conventional ones and improve the environmental sustainability of potted plants and at the same time minimizing the additional costs?*

The question of this thesis arises whether optimal mix of substrate (composed by peat and compost) and optimal material of planter container (e.g., fossil-based plastics or biodegradable, bio-based polymer) can be determined, such that additional costs when substituting and environmental impacts of a potted plant are simultaneously minimized—this is the *decision problem* of this dissertation. To answer to this question and find the trade-off between emissions and costs, approaches of Operations Research (OR) can be used. In general, emissions and costs relevant to the decision—which are different between the alternatives—should be taken into account. This thesis, which combines different disciplines, such as environmental engineering, agronomy, and business administration, is structured as follows.

Chapter 2 explores the value chains of peat, compost, and plastics in horticulture. Here, the following concepts are introduced: sustainability, peatland protection, waste management of plastics, cascade use of resources, resource efficiency, and substitutes for peat and plastic pots.

Peat, olive-mill waste compost, plastic pots made of polypropylene, and pots made of polylactic acid are the four case-specific technical solutions of the decision problem.

Chapter 3 describes methods to assess the environmental burdens of resources used in agriculture. Critical aspects and studies reported in literature are highlighted. A sustainability assessment of the supply chain of the technical solutions is preformed via Life Cycle Assessment (LCA). The analysis follows GHGs experimental detection of bio-based materials. These activities, which consist of hardware assembly, software programming, and laboratory analysis, have been conducted in cooperation with *Istituto per i sistemi Agricoli e Forestali del Mediterraneo* (ISAFOM), *Consiglio Nazionale delle Ricerche* (CNR - National Research Council), Perugia, Italy. Finally, LCAs of the four case-specific technical solutions are reported.

Chapter 4 presents studies that combine environmental and economic aspects. Moreover, the following aspects are described: decision-relevant additional costs when substituting, quality grade assessment in horticulture, correlation between selling price of potted plant and its agronomic quality. Finally, additional costs when substituting for peat and plastic pots are calculated and reported.

Chapter 5 presents existing approaches of OR for optimizing the use of resources in agriculture, with focus on blending problems. A blending model is then developed and applied to a case study of a plant nursery located in Pistoia (Italy) by using the four case-specific technical solutions. Objective functions, decision variables, constraints of the model are described. The bi-objective problem of minimizing decision-relevant environmental emissions and decision-relevant additional costs is formulated. Pareto optimal solutions are yielded for different scenarios within the case study.

Chapter 6 presents conclusions of this work, critical aspects, and it introduces robust optimization (for a survey, see Ide & Schöbel, 2016), which deals with uncertain data. The decision problem of this thesis with uncertainties is then investigated (Krüger et al., 2018) by using the approach described in Krüger (2018). The chapter describes at the end further paths for research in horticulture and other sectors.

2 Analysis of peat and plastic use in horticulture

In this chapter, the concept of sustainability is firstly described (Section 2.1), with insights on impacts of the agro-industrial sector, issues related to the use of peat as growth medium and petroleum-based planter containers, cascade utilization, and resource efficiency. Secondly, possible substitutes for peat and plastic containers are described (Section 2.2). Here, focus on compost and bio-based, biodegradable plastics are introduced. These preliminary aspects lay the basis for the evaluation of environmental and economic aspects, which will be analyzed in the next two chapters.

2.1 Environmental sustainability

The concept of *sustainability* follows the principle of the 'triple bottom line' (formalized by Elkington, 1997), where three dimensions of sustainability, i.e., environmental, economic, and social, are taken into account—more aspects can be also included for assessing sustainability (see, e.g., Gibson, 2006; Sianipar et al., 2013). Environmental sustainability, in particular, deals with the global climate change, which is directly correlated to the increment of population and therefore needs of food (and field for cultivating), increment of consumption of natural non-renewable resources, and rising of outdoor air pollution. These aspects were early introduced by Meadows et al. (1974) and, with the passing of the time, has driven the current research in many fields. The Conference hold in Kyoto, Japan (UNFCCC, 1997) presented the most relevant international agreement among State Parties with the dissemination of the Kyoto Protocol, an international treaty that currently include 192 parties worldwide (UNTC, 2005). Shared intents were written towards a systematic way of limiting human activities that are responsible for the release into the atmosphere of GHG emissions (e.g., carbon dioxide, methane, and nitrous oxide). Ten years later, the fourth report of the Intergovernmental Panel on Climate Change (IPCC, 2007) stated that warming of the climate change is unequivocal, due to "observations of increases in global average air and ocean temperatures, widespread melting of snow and ice, and rising global average sea level". This *global warming* is also driver of other extreme natural reactions, such as hurricanes, floods, droughts, desertification, and alteration of natural habitats that would otherwise be preserved (Solomon et al., 2009). IPCC (2007) stated also that increment in carbon dioxide (CO_2) concentration are mainly due to fossil fuel use, while increment of concentrations of methane (CH_4) and nitrous oxide (N_2O) are primarily due to agricultural activities. Human activities are therefore the most relevant drivers of these environmental changes in many sectors (Rosenzweig et al., 2008; Martin & Saikawa, 2017). Reducing GHG emissions and modifying the demand of fossil resources are drivers for seeking alternative renewable resources (Geldermann et al., 2016a). In horticulture, to determine which resources can be substituted in the supply chain, an overall picture of the impacts should be outlined. This topic is further discussed in the following section.

2.1.1 Impacts of the agro-industrial sector

The agro-industrial and forestry sector contributes 20% of the total anthropogenic GHG emissions (see Figure 2.1), which are increasing at around 1% per year (IPCC, 2014). Changing agricultural practices to reduce GHG emissions in agriculture is nevertheless a challenge, due to economic and implementation limits (Duxbury & Mosier, 1993; Lamb et al., 2016).

An agricultural value chain is a linear concatenation of activities, from livestock/crop production to waste management (Figure 2.2). Other intermediate activities are related to livestock processing (e.g., harvesting, primary and secondary processing), transporting, retailing, and consuming. Here, fundamental drivers for an agricultural value chain are global trends, consumer preferences, costs, sustainability, and product quality. This value chain can be also referred to the horticulture sector, which is a subset of the agricultural ones.

The Italian horticulture sector is one of the most relevant among the agricultural area in Europe (Beccaro et al., 2014). The Pistoia district has been selected due to its relevance in the sector, with about 30% of the national nursery production (Pardossi et al., 2009). Here, about 5200 ha are covered by plant cultivation in nurseries, and over 5500 workers are employed. Horticultural products comprise field-grown plants, potted plants, garden shrubs, broad-leafed and coniferous trees. The potted-plant cultivation of ornamental species, e.g., *photinia*, *osmanthus*, *acer platanoides*, and *cupressus*, exploits a huge quantity of resources (e.g., consumption of diesel for transportation, electrical energy for pumping irrigation water, use of peat as main component of substrate, and use of fertilizers), and it is an important contributor to the GHGs emissions at the local level (Recchia et al., 2013).

A plant nursery has a specific value chain, shown in Figure 2.3. The figure identifies inputs (upper side of the figure), outputs (lower side), and phases that characterize the cultivation activities inside a plant nursery:

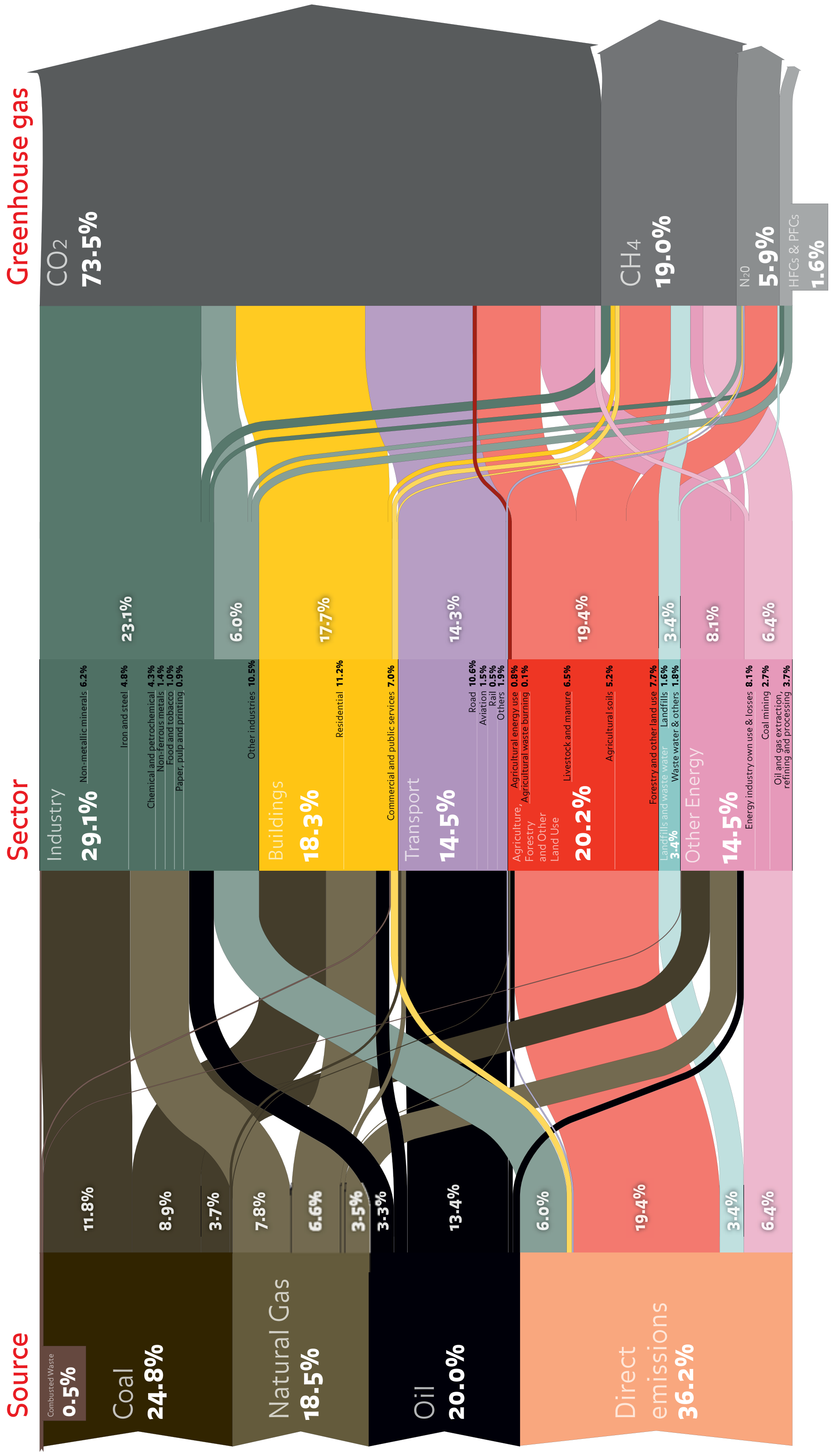

Figure 2.1: World GHG emissions flow chart (2012). Source: ASN Bank & Ecofys 2016 (`www.ecofys.com`), based on International Energy Agency 2014 "CO_2 emissions database" (`www.iea.org`), and Joint Research Centre, European Commission, 2013, Global Emissions EDGAR v4.2 FT2010 (October 2013)

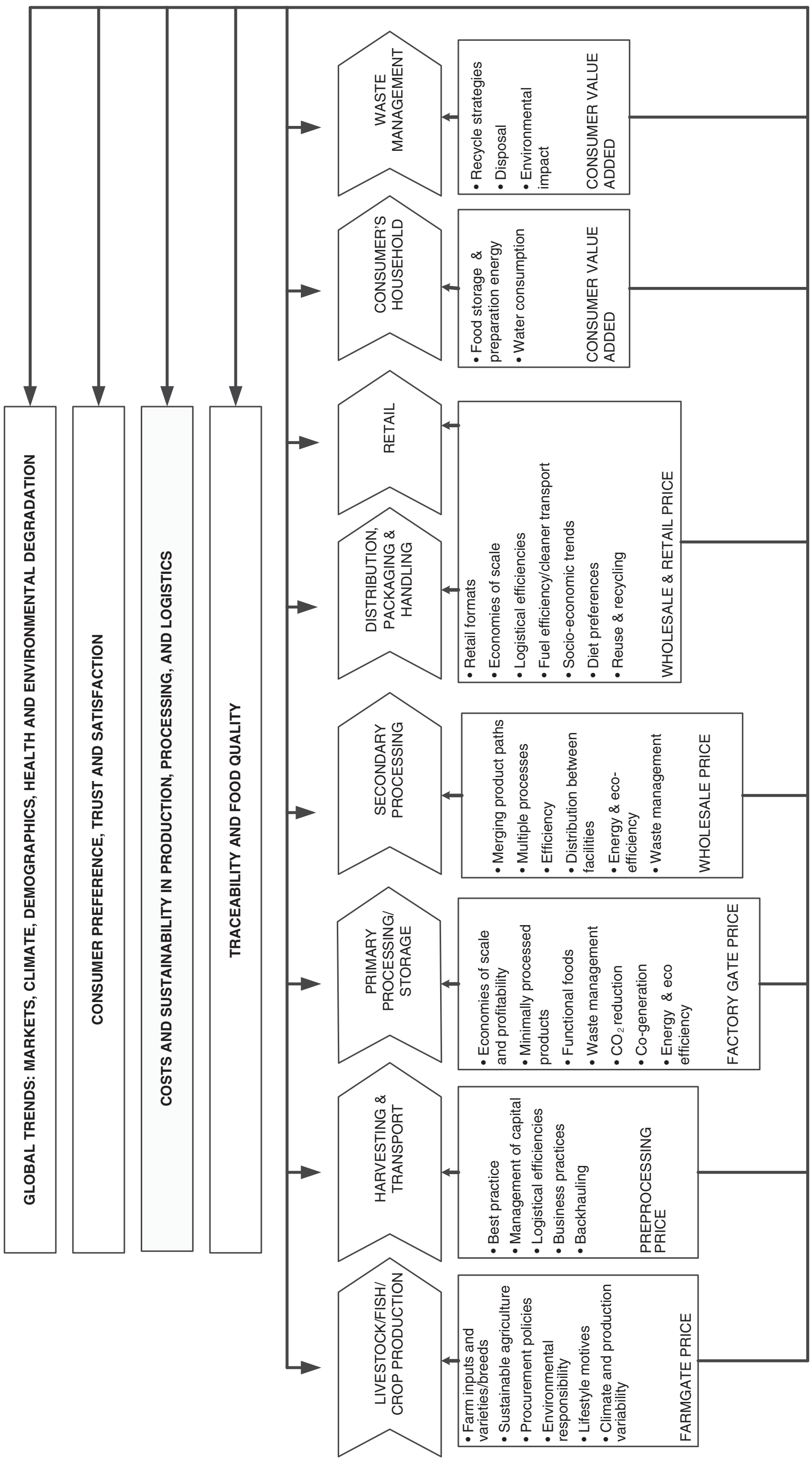

Figure 2.2: General agricultural value chain (Higgins et al., 2010)

- acquiring raw materials required to prepare the substrate for cultivating, mixing the suitable input materials for the specific *species* to prepare the growth substrate—they can be prepared inside the plant nursery or, alternatively, premixed by suppliers and customized for specific applications;
- potting and placing the plant on the ground;
- packaging the potted plant in order to be transported and sold.

Final product of the value chain is an ornamental potted plant cultivated for a season (e.g., one year). This final product is the object of the decision problem of this thesis, described in Chapter 1.

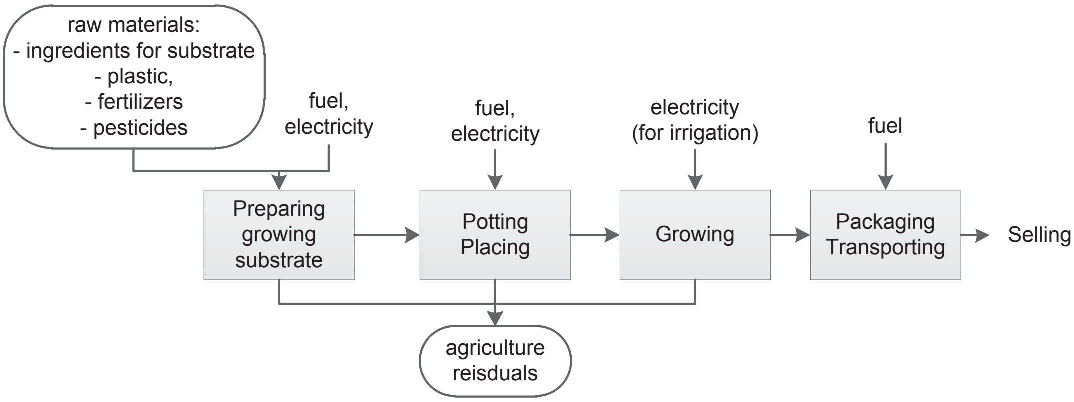

Figure 2.3: Flowchart of a horticultural value chain (for an exemplar plant nursery)

To understand better the implications of substitution of resources used in horticulture, a relevant study (Lazzerini et al., 2014) conducted on the inputs of this value chain is reported here. The study investigated 11 plant nursery located in the Pistoia district (Tuscany region), one of the most important agricultural activity of the district. The study, conducted on potted plant cultivation (pots between 20 and 30 cm of diameter) and field-grown cultivation (plants which are directly grown inside the soil), takes into account six technical parameters for the cultivation of plants: fertilization production, pesticide production, diesel fuel use, electricity use, plastic production, and peat production. The results of the study (Figure 2.4) show that the largest sources of the environmental emissions for potted plant cultivation are related to the use of plastics (e.g., containers, mulching films, packaging plastics) and the use of peat as horticultural medium.

Among the aspects presented so far which raised environmental concern, two are considered in this dissertation due to their relevant contribution to the total environmental emissions of a plant nursery: peat as main constituent of the growth substrate and plastics usage for potting plants.

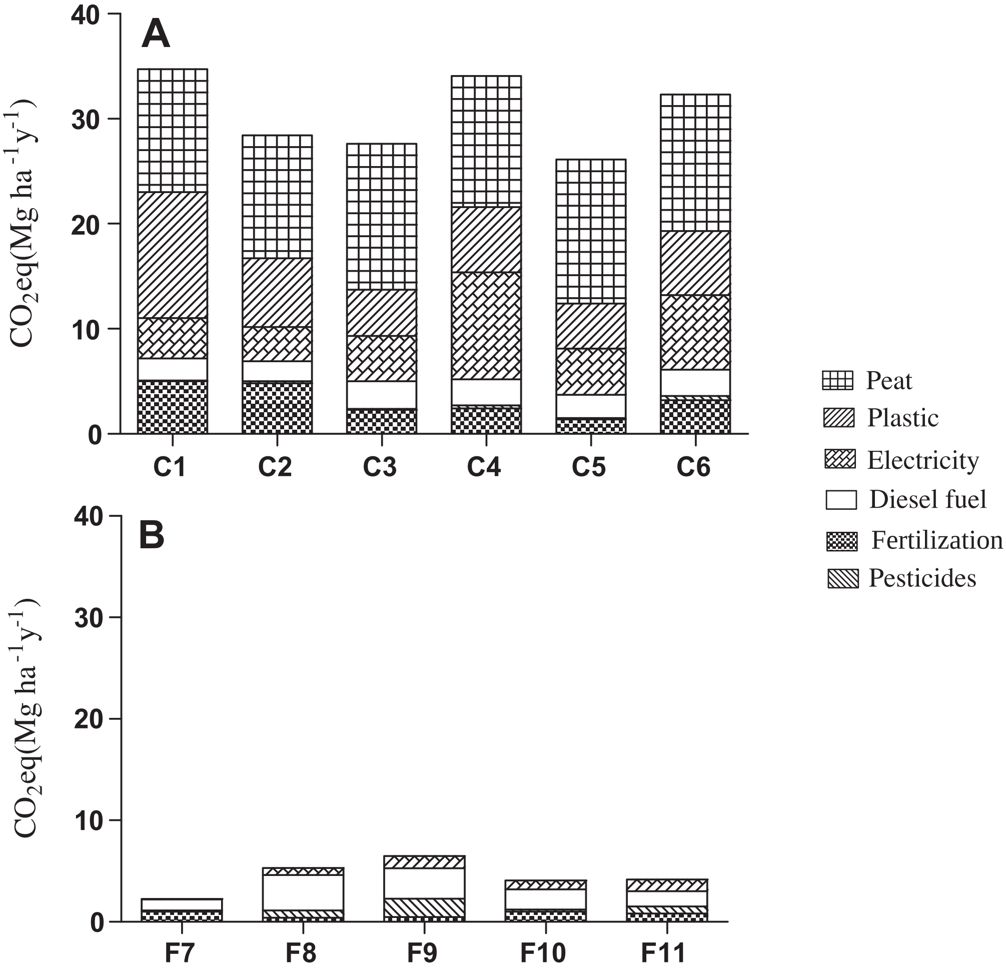

Figure 2.4: Emission factors (t CO_2-eq per hectare per year) of a plant nursery in central Italy.
Notes: The analysis was conducted using the hectare (1 ha) as functional unit of the life cycle assessment for potted plant cultivations in 6 plant nurseries (indicated with "A") and in-field cultivation in 5 plant nurseries (indicated with "B") (Lazzerini et al., 2014)

2.1.2 Peatland and climate protection

Peat is a "sedentarily accumulated material consisting of at least 30% (dry mass) of dead organic material" (Joosten & Clarke, 2002), and it accumulates in mires and peatlands. The latter are areas with or without vegetation with a naturally formed peat layer of 30 cm or more on the surface (Strack, 2008, p.17). Peatlands are the most widespread wetland type, cover 4,000,000 km^2 (about 3% of the Earth's land surface), contain 10% of the global freshwater resources, and store one-third of the worldwide soil carbon storage (Gorham, 1991). Over 90% of all peatlands are in temperate and cold belt in the Northern Hemisphere (Maltby & Proctor, 1996). In Europe, peatlands cover more than 282,000 km^2 (about 7% of the worldwide peatland) and it is estimated that around 42% of the total peat usage is used as nursery growth medium for edible plants, ornamental plants, and landscape horticulture (Altmann, 2008). Figure 2.5 presents the share between different horticultural sectors in Europe, with the largest share by floriculture sector (48%), followed by vegetable growing (27%), nursery stock (17%, for potting and in soil use), and other uses.

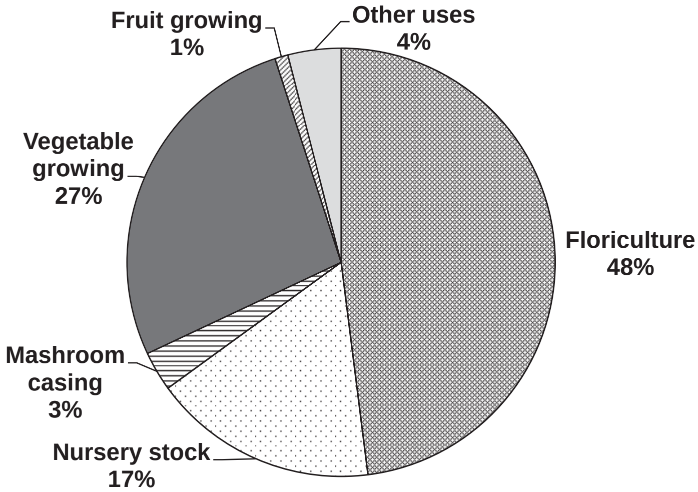

Figure 2.5: Share of peat usage in the growth substrate industry (adapted from Altmann, 2008)

Peat is a very flexible material that can be adapted for most plants, since it is generally low in nutrients, pH, and bulk density. Moreover, it exhibits favorable cation exchange capacity and air-filled porosity characteristics, as well as high structural stability, long-term availability, and uniform properties (Robbins & Evans, 2001). For these reasons, peat is the main growth medium constituent in Europe (Schmilewski, 2013; Reinikainen, 2001).

However, preparing the surface of peatland for harvesting (i.e., removing vegetation and digging ditches), extracting, storing, transporting the peat, and treating the cutaway area have substantial negative impacts on the environment: intensive drainage, water contamination and removal, biodiversity alteration, increment of air pollution, reduction of carbon storage, and increment of atmospheric carbon concentration (Raeymaekers, 1999). Since peat has formed over thousand of years (1 mm per year formation rate, Joosten & Clarke, 2002), and although minimal methane emissions could escape from peatlands, its carbon cycle is assumed isolated from the biological ones. Consequently, peat is considered a non-renewable resource. In Italy, Camporese et al. (2006) described peatland subsidence in the Venice watershed as "irreversible long-term critical issue": drained peat soils, which are mainly related to biochemical aerobic oxidation of the organic matter, cannot be recovered, and plowing has enhanced fossil carbon dioxide release and, subsequently, the sinking rate. In this way, alteration of the soil and mechanical destruction of the peat structure has been performed during the past decades (Camporese et al., 2006).

In general, CO_2, CH_4, and N_2O emissions are accounted when extracting, processing at factory, and transporting peat (due to fuel combustion). Emissions can be also accounted after use, due to peat decomposition in fossil CO_2 (Robertson, 1993). The extraction of peat and the role of peatland in climate change has been widely studied (e.g., Flessa et al., 2002; Cleary et al., 2005; Grönroos et al., 2013; Helin et al., 2014), and the debate on peat exploitation is still ongoing. For example, the socio-economic report by Altmann (2008) described strong divergent positions of the European growth media industry and environmental organizations about the sustainability of extracting, processing, and using peat for horticultural purposes. Raeymaekers (1999) indicated peat extraction as serious threat that brings to the increment of GHG emissions into the atmosphere. Regina et al. (2016) estimated 98 t CO_2-eq/ha per year as maximum emission rate from cultivating peatland (value adopted from IPCC, 2013). Undisturbed peatlands represent also an important part of the global carbon sink, with annual uptake peak value of -0.5 t CO_2-eq/ha (Waddington et al., 2009). Many studies have investigated the reduction of the carbon pool (see e.g., Minkkinen & Laine, 1998; Tuittila et al., 1999; Minkkinen et al., 2002; Turetsky, 2002), and the anthropogenic disturbance on peatland ecosystem has been mainly attributed to agriculture and silviculture (Laine et al., 2009), which is the practice of control growth, health, and quality of forests. On the one hand, with agricultural use the aeration of the cultivated area is increased, with consequent increment of microbial activity and therefore CO_2 emissions. On the other hand, silviculture is assumed similar to agriculture but with reduced impacts due to the different soil conditions: "CO_2 sink function of the peatland usually remains as long as the tree stand grows" (Laine et al., 2009). For this reason, re-covering exploited peatland by planting trees is an option to preserve its carbon sink effect.

Since GHGs are well known drivers of the climate change, the Intergovernmental Panel for Climate Change (IPPC) has provide guidance on how to account for peatland emissions. However, this topic has been controversially debated. Many authors debated over the legitimacy of that guidance (see, e.g., Altmann, 2008), and others indicated the potential in peatland management for reducing the emissions with different scenarios (see, e.g., Christensen et al., 2004). These studies have highlighted that human activities have started to threat peatlands, whose massive exploitation for energy and agricultural purposes can be reduced to preserve their precious function as world's carbon storage (Regina et al., 2016). Towards this end, many Countries have started to prescribe limits to peatland excavation, and the scientific community has intensified the research for alternative, organic sources to create suitable growth substrates. For example in Lower Saxony (Germany), the Revision of the Landesraumordnungsprogramm (LROP) (Regional planning program) has regulated peatland area (around 25,000 km^2) with specific indication of land use: e.g., only 12% of the total area can be exploited for peat extraction, and maximum thickness of a peatland layer that can be exploited by peat's industry is regulated (Niedersächsisches Ministerium für Ernährung, 2017).

2.1.3 Petroleum based plastic and waste management

Plastic materials are universally used for diverse applications, from agriculture to medical devices or electronics. Figure 2.6 shows a general plastic's supply chain with input/output materials and processes.

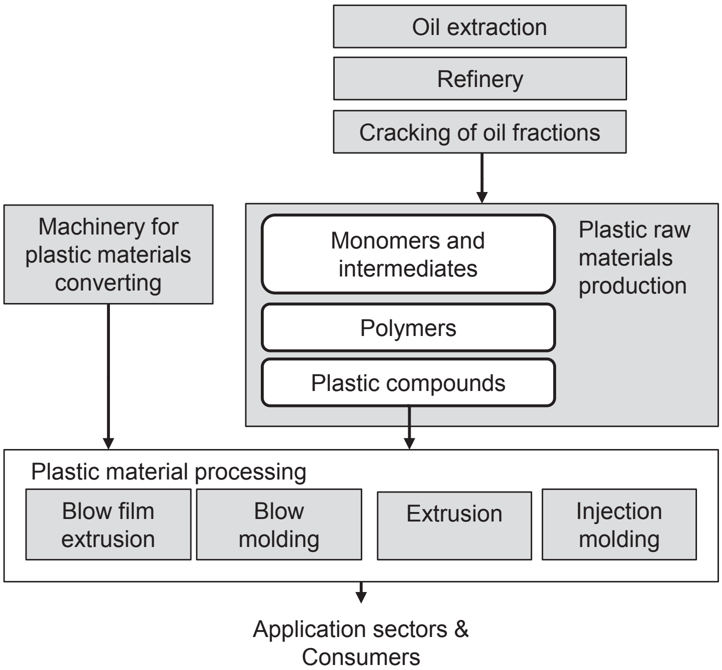

Figure 2.6: Plastics' supply chain (adapted from Azzone et al., 2014, p.49)

Plastic usage is expanding and expected to grow at 3–4% rate per year. Approximately 6,300 million tons of plastic waste had been globally generated (Geyer et al., 2017), and the consumption in agriculture is estimated around 6.5 million tons (Scarascia-Mugnozza et al., 2011). It is estimated that the agro-food processing industry produces about 250 million tons per year of plastic wastes in Europe (AWARENET, 2004); in the United States (USA), an estimated 750 thousand tons of plastic pots are disposed of annually (Nambuthiri et al., 2015); and in Italy, 91 thousand tons of plastic pots, sheets, and twines used in agriculture are disposed of annually (Scarascia-Mugnozza et al., 2011). The European Commission Directive on Packaging and Packaging Waste (EU Council, 1994) requires recycling, in order to prevent or minimize environmental impacts. The landfill directive (European Landfill Directive EC/31/1999) set the targets for a "major shift in waste management practice" (Platt, 2006, chapter 4). Also in 2014, in EU 30.8% of post-consumer plastics waste went to landfill (PlasticEurope, 2016, p.24), and in Italy it is estimated that plastic waste represents 12.7% (by weight) of the national urban waste treated by municipal facilities (ISPRA, 2015, p.72). Among the disposal options for plastic materials after usage, the following are currently performed: landfill, off-site incineration, recycling, and re-use.

In horticulture, petroleum based containers are employed due to suitable chemical and mechanical characteristics for cultivating the plants. However, recycling plastic at the end of the supply chain in agriculture remains a challenge due to the following factors (Briassoulis et al., 2012; Hall et al., 2010; Garthe & Kowal, 1993): the presence of unstable markets that can acquire the recycled materials, the lack of good housekeeping practices, the dispersed sources of plastic wastes (i.e., horticulture industries, households, farms, and therefore a problem of coordination in collection from different usage patterns), the presence of contaminants (residual soil, vegetable matter, sand, stones and agrochemicals such as fungicide, herbicides, and insecticides contaminate the plastic waste), and the presence of additives (e.g., UV protectants, thermal resistance additives, pro-oxidants). The two latter factors, in particular, have lead to high cost for proper collection, recycling, reusing, or disposal process. High disposal costs have even lead to illegal practices, such as on-site burning, on-site dumping, or plowing into field. These practices release harmful substances and air pollutants, contaminate the soil with consequent quality degradation, and degrade the whole agricultural Eco-system (Scarascia-Mugnozza et al., 2011). These practices are therefore forbidden according to the Landfill Directive (99/31/EC), the Incineration Directive (2000/76/EC), and the Revised Waste Framework Directive (2008/98/EC).

Levitan & Barros (2003), who described the same issues related to the waste management of plastic for the State of New York, illustrated recycling options for agricultural plastic stream towards a systematic analysis of kind of plastics, solutions, and markets for the recycled outputs. The main four kind of plastic indicated by the authors in agriculture are low density polyethylene (greenhouse films, nursery films, mulch films, bunker silo covers, bale wraps, and silage bags), high density polyethylene (pesticide containers and nursery containers), polystyrene (nursery trays and flats), and polypropylene (nursery pots, row covers, and woven tarps and sacks).

In Italy, the main problem for recycling plastic is related to the presence of stones and soil, which can damage the blades for cutting sacks within mechanical treatment in municipal facilities, and the presence of sand, which can cause erosion of the same blades (Briassoulis et al., 2012). In order to remove these contaminants before the recycling processes, preliminarily practices are required from industries, with relative high costs for treatment. To tackle this problem in Canada, Muise et al. (2016) explored scenarios that promote cooperative actions of farmers involved in preprocessing of post-consumer plastics.

For the reasons presented, plastic containers can be only used once in horticulture. Therefore, many alternatives, which can be introduced in the horticultural supply chain, have been proposed. The use of alternative substrates made of by-products and the use of bio-based containers can improve the *resource efficiency* towards a *cascade utilization* of resources in agriculture. These two concepts are introduced in the following section.

2.1.4 Cascade utilization and resource efficiency

The concept of *cascade utilization* (or cascade use or cascading) was introduced to reduce the consumption of resources by reusing and incrementing their value, as long as possible. The first formulation of cascade utilization was presented by Sirkin & Houten (1994). Recently, Olsson (2017) argued that resource quality plays an important role to boost cascade utilization. In Figure 2.7, a typical cascade utilization is presented with different paths: the final use of a resource (use n) can be converted to other chains (secondary cascade chain) or to a precedent use of the same resource (primary cascade chain). In this way the resource's intrinsic value can be preserved or, even better, elevated (Sirkin & Houten, 1994).

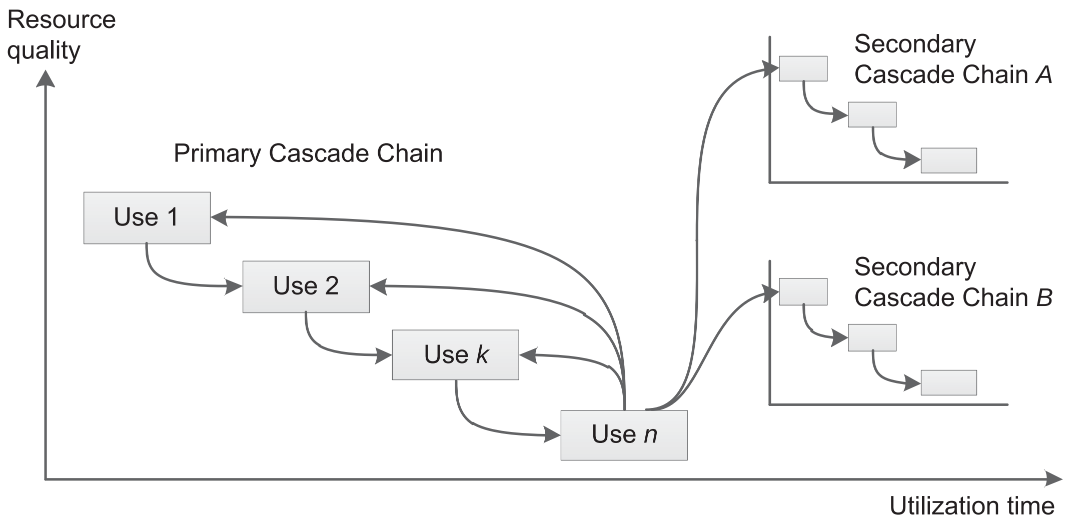

Figure 2.7: Cascade utilization and possible paths, i.e., cascade chain A and cascade chain B, for reusing resources (adapted from Sirkin & Houten, 1994, p.233)

The cascade utilization concept can be linked to the guiding theme "Reduce-Reuse-Recycle" used by the European directives that established waste recovery as the first choice in the waste management hierarchy (EC, 1999; EU Council, 1991, 1994). Renewable materials, which represent an alternative for resource scarcity and their exploitation, can increase the *resource efficiency* of a supply chain—a *supply chain* is defined as "system of organizations, people, activities, information, and resources involved in moving a product or service from supplier to customer; supply chain activities involve the transformation of natural resources, raw materials, and components into a finished product that is delivered to the end customer" (EU, 2017).

By-products discarded after the first use (for example of wood from the construction or furniture industry) can be reused firstly by chemical use and later on for energetic purposes, which should be the last option, the cascading concept suggests. Höglmeier et al. (2015), for example, presented paths of wood's cascade utilization from secondary uses of wood products into waste wood recovery and other residues, as well as Hesse (2015), who developed a model for exploring cascade utilization effect of wood with scenarios of different intensity.

In that context, the question arises whether by-products from agriculture could be reused in horticulture as peat substitute and thus contribute to resource efficiency. For the investigation of the consequences of cascade utilization of compost, the affected parts of the supply chain need to be analyzed. The phases which correspond to the processing and selling of a product, from the research & development (R&D) phase to the market, can be performed within the same company or externalized. In general, the preliminarily process, for the plastic industry for instance, is performed in laboratory. The whole process till the disposal phase of the supply chain can be depicted with Figure 2.8. Charpentier (2002) indicated in this figure the steps for the production phase with length and time scales for the chemical industrial research, but the same idea can be extended to any other processing activity. From the molecule scale (small scale, laboratory) to the enterprise one (large scale, corporate), Charpentier (2002) highlighted the steps in R&D process and reported a vision whose application is challenging. Experiments and trials—operations and actions to conduct experiments in laboratories—are indeed often necessary to fundamental research in chemistry or physics (where reactions are investigated), but an overall vision of further real-world applications might be missing. And, in general, a complete integration between disciplines towards a common aim is challenge task.

This thesis follows this approach: firstly, experimental trials to detect GHG emissions have been conducted on a small portion of compost and peat, secondly, these emissions are used inside an environmental analysis of such materials (which elaborates the environmental implications of their use), and thirdly, the outcomes of this investigation have been upscaled to a potted plant, in order to give recommendations to plant nurseries.

2.2 Alternatives to conventional materials

To reduce the environmental impacts of peat and plastic use in agriculture, alternative materials made of renewable resources or by-products have been used in horticulture. Many agro-waste substrates have been used as substitutes for peat, while biodegradable pots that can be directly buried into the soil with the plant has been developed to replace conventional plastic containers.

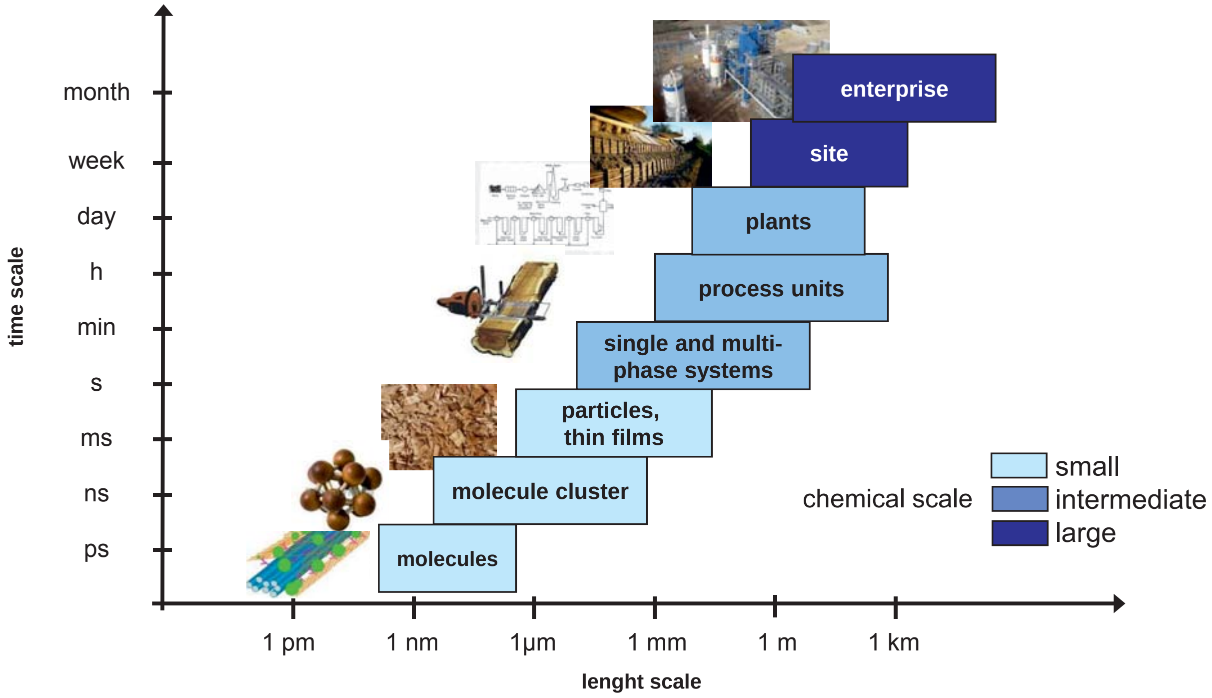

Figure 2.8: Upscaled vision of reactions in the wood industry (adapted from CHARPENTIER 2002, who reported the previous graph by GROSSMANN & WESTERBERG 2000).

However, less bio-based materials are used in agro-industry because of their higher costs of extraction/processing and technical aspects that are related to case-specific conditions (DEWULF ET AL., 2016). In some cases, indeed, alternative materials cannot guarantee the same standard agronomic quality of a plant, and selling price should be lower. On the other hand, bio-based materials can attract the attention of customers due to their environmental friendliness, which might become driving force for the green economy (ELLISON ET AL., 2015). Better environmental profiles may improve company's image in the market (RIGGI ET AL., 2011), let a company achieve product environmental certificates (RUGANI ET AL., 2013), and increase the contribution margin whether consumer willingness to buy environmental friendly products is assessed (VAN DAM ET AL., 2004).

In this section, alternative resources are introduced. These are made from biomass and can directly displace fossil fuel-derived products; therefore, both cascade utilization and resource efficiency's improvement can be performed.

2.2.1 Compost as peat substitute

Many materials have been studied and experimented as substitutes for peat. The quantity of alternative growth media used in EU is shown in Figure 2.9—the figure can be still considered representative of the current situation (see LAZZERINI ET AL., 2016), although the situation reported is dated 2008. Example for peat's substitution substrates are coconut fiber (NOGUERA ET AL., 2000), bark and wood substrates (BOHNE, 2001), rice hulls (EVANS & GACHUKIA, 2004), animal manures (INBAR ET AL., 1993), and compost (RAVIV, 2005).

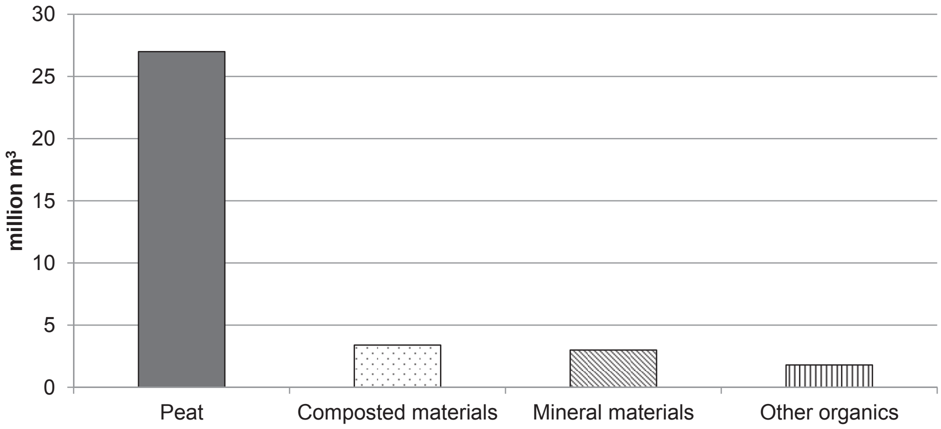

Figure 2.9: Types of growth media used in Europe (by volume) (adapted from ALTMANN, 2008)

Compost is the result of thermophilic and mesophilic activity of microorganisms which naturally decompose the organic matter, that is contained in the initial product (e.g., by-product of another value chain) (HAUG, 1993). A *by-product* is a residual material, inevitably produced at the end of an industrial supply chain. The transformation of carbon within composting is described as (STENTIFORD & DE BERTOLDI, 2010):

$$[C] + O_2 + \text{microbial activity} = \text{new biomass} + CO_2 + H_2O + \text{heat} \tag{2.1}$$

A material is indicated as *compostable* when biodegradation occurs with a rate comparable to known compostable materials without diminishing the value/utility of the outcome product resulting from the composting process (RIGGI ET AL., 2011). RAVIV (2014) described composting as suitable option to reduce the environmental impacts of the initial material and one of the most valuable options to reach a proper maturity of the organic matter for agricultural utilization.

The composting process requires, however, proper conditions in terms of pH, temperature, moisture, oxygenation, and nutrients, in order to permit the efficient degradation of the organic matter (ALBURQUERQUE ET AL., 2004). Although in horticulture settings peat could be best replaced by compost, beyond a certain substitution rate the agronomic quality of the plant becomes unwelcome (PAPAFOTIOU ET AL., 2005). Compost and peat are indeed different in terms of bulk density, salinity, phytotoxycity, pH, and maturity. Thus, plant nurseries still use peat as main component of the growth substrate to produce commercial plants (as seen in Section 2.1.2). Investigation is indeed required to evaluate varying levels of quality of plants cultivated in compost versus conventional growth medium, like peat. Many research projects have been conducted in Europe regarding horticultural use of compost obtained from different matrices, such as agro-industrial waste (Garcia-Gomez, 2002), sewage sludge (Perez-Murcia et al., 2006), municipal solid wastes (Herrera et al., 2008; Ingelmo et al., 1998), distillery wastes (Bustamante et al., 2008), pruning wastes (Benito et al., 2005), or olive-mill waste (Altieri et al., 2010).

In this thesis, compost obtained from olive-mill waste is investigated and compared with peat, because it has appropriate characteristics as growth medium (Arvanitoyannis & Kassaveti, 2007). However, olive-mill waste compost utilization in horticulture settings is still at R&D phase. Olive-mill waste compost is obtained from the industrial production of olive oil for the food industry. It is estimated that 3.05 millions tons of olive oil per year are produced worldwide, and Europe has 76% of share (year 2014)[1]. The manufacturing process yields olive oil (18-28%) and residuals (olive-mill waste), which are composed by aqueous liquor (40-50%, i.e., vegetation water) and semi-solid waste (30-35%) (Morillo et al., 2009). The aqueous liquor (*olive-mill wastewater*) comes from the vegetation water and soft tissues of the olive fruits, while the semi-solid waste is composed of a mixture of olive husk (*pomace*, made of pulp and peel) and olive pits. It is estimated, that 10-12 million m^3 of olive-mill wastewater are annually produced by Mediterranean countries (Niaounakis & Halvadakis, 2006). Figure 2.10 shows the quantity of olive-mill waste generated from olive oil processing by Spain, Italy, Greece, Tunisia, and Portugal.

Quantity and type of olive-mill wastes generated depend on the type of oil extraction system. Three extraction systems are used in Europe (see Figure 2.11): traditional press-cake system, three-phase system, and modern two-phase system. The *three-phase system* has been introduced to improve the extraction yield of the traditional ones, and it generates three outputs (as the traditional system): pure olive oil, olive-mill wastewater, and pomace. The *two-phase system* was introduced to reduce input water and energy consumption during the process, and it generates pure olive oil and a “two-phase olive-mill wastewater”, which includes the pomace and the aqueous liquor.

Remediation processes have been proposed to treat olive-mill wastes (Niaounakis & Halvadakis, 2006; McNamara et al., 2008): thermal processes, physico-chemical treatments, extraction of valuable compounds, agronomic applications (e.g., land spreading), animal-feeding methods, and biological treatments (e.g., composting). Potential outputs of these processes can be used in agriculture (compost, microbial biomass, and edible fungi), in energy sector (biogas,

[1] This is the latest database update on crops processed worldwide, provided by Food and Agriculture Organization (FAO) of the United Nations, Statistics Division, available at `http://faostat.fao.org`. Data retrieved in February 2018.

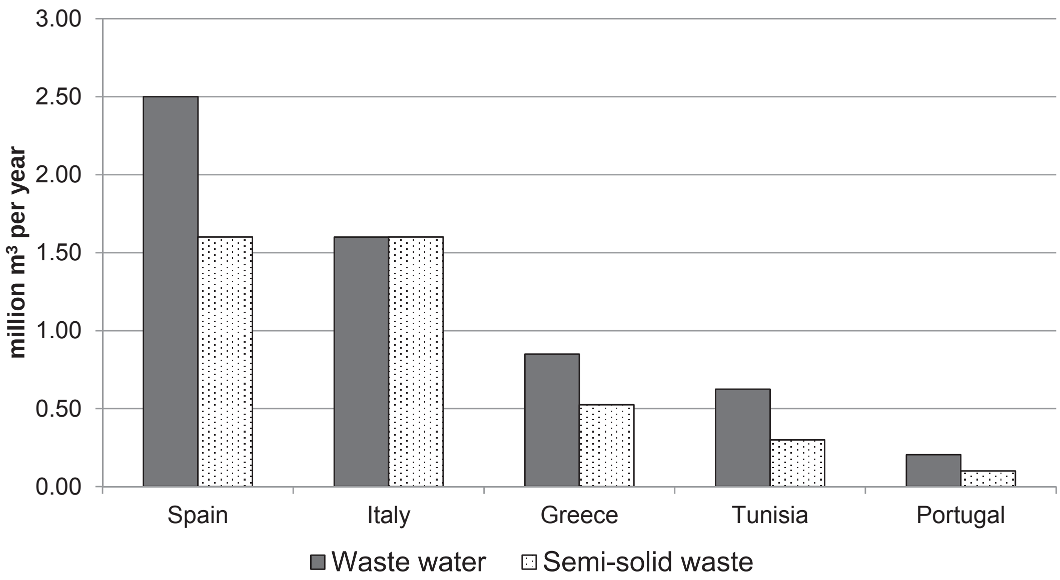

Figure 2.10: Quantity of olive-mill wastewater and olive-mill semi-solid waste generated from olive oil processing (adapted from Niaounakis & Halvadakis, 2006, p.11)

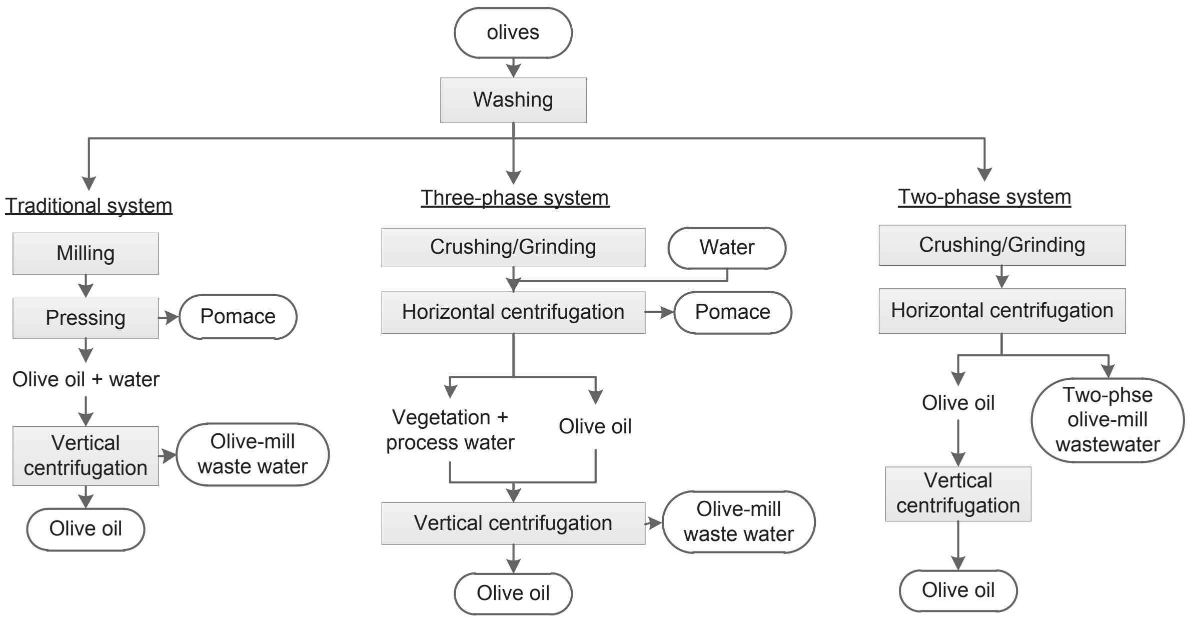

Figure 2.11: Industrial-scale olive oil extraction systems: traditional press-cake, three-phase decanter process, and two-phase centrifugation process (adapted from Morillo et al., 2009)

biofuel), and in industry (bio-polymers and enzymes), as shown in Figure 2.12. Among these processes, compost obtained from olive-mill waste has been investigated for decades (Miller & Jones, 1995; Chilosi et al., 2017).

Composting is a suitable low-cost option to treat, in particular, pomace (from traditional and three-phase systems) or two-phase olive-mill wastewater (from two-phase system), in order to produce a substrate with high fertilizing power, due to the presence of large amounts of proteins, polysaccharides, mineral salts, humic acids (Arvanitoyannis & Kassaveti, 2007). However, the aqueous liquor (in the two-phase olive-mill wastewater) contains phytotoxic and biotoxic substances, which can inhibit microbial growth, germination, and vegetative growth of plants (Linares et al., 2003). Moreover, since olive-mill wastewater is made by long chain fatty acids and phenolic compounds, it is resistant to degradation, and consequently, it can reach the water bed under the soil and pollute it (Morillo et al., 2009). To equilibrate the nutrient balance of olive-mill waste (specially nitrogen content, for agronomic purposes), to increase the porosity of the growth substrate (for agronomic purposes), and to avoid reaching of lethal temperature (for the composting process itself), bulking agents should be added up (Miner et al., 2001). Since uncontrolled disposal of olive-mill wastewater into soil or water has become an urgent environmental issue (Sierra et al., 2001), regulations (Italian Law n. 574, 1996) limit in Italy the spreading on land as disposal method.

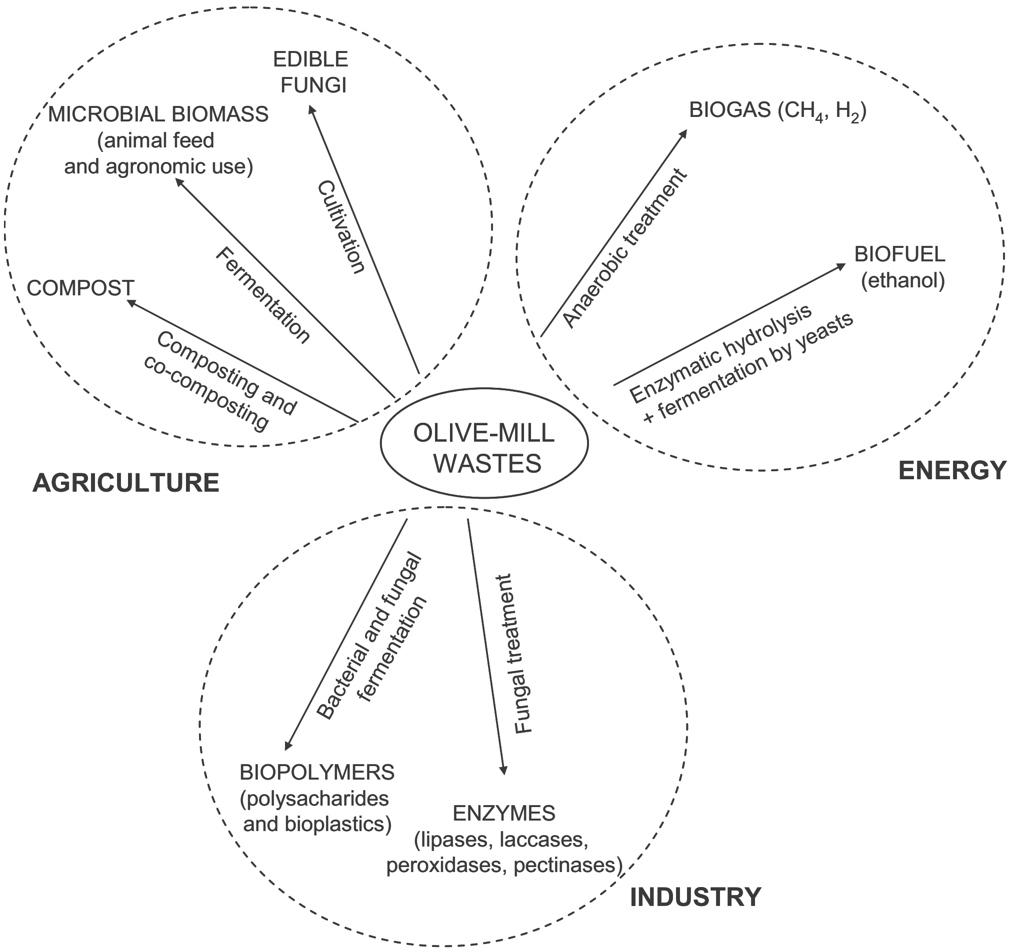

Figure 2.12: Remediation processes of olive-mill waste and potential outputs (Morillo et al., 2009)

In general, composting is a suitable method to be performed in small and medium size olive-mill facilities, e.g., where the annual olive oil production is less than 1000 t (Alfano et al., 2008). Different methods have been developed to compost olive-mill waste, e.g., open-windrow composting (Hellmann et al., 1997), dynamic turned pile (Ranalli et al., 2001), inside bio-reactor (Principi et al., 2003), static-pile composting with forced aeration and controlled temperature (Finstein et al., 1983), static-pile composting with natural passive aeration (Altieri et al., 2011). In this thesis, olive-mill waste compost is prepared following a simplified, low-cost procedure of static-pile composting with natural aeration. This procedure, described in Section 3.4, is further compared towards open-windrow composting.

2.2.2 Bio-based plastics used in horticulture

A first generation of bio-based, biodegradable plastics was introduced in the 1980s (Huang et al., 1990; Vroman & Tighzert, 2009). A bio-based material is produced by the extraction of polymers from organic resources (e.g., dedicated crops, agricultural residues, seaweeds), by chemical synthesis of polymers from monomers derived from organic resources, or by microbial production of polymers or monomers (Mohanty et al., 2002; Bastioli, 1998). Bio-based plastics (or bio-plastics, or bio-based polymers) are defined as “biodegradable plastics whose components are derived entirely or almost entirely from renewable raw materials” (Stevens, 2002, p.104). Bio-based polymers can be processed and blended with other compounds to improve their properties, such as strength, melt viscosity, sensitivity to water, biodegradability, compostability, thermal and tensile properties (see, e.g., Seggiani et al., 2018). Additionally, bio-based materials reduce disposal costs and save production and use of fossil-based resources (Azzone et al., 2014, p.153).

The behavior of biodegradation of bio-based plastics—into composting conditions or in the soil—is investigated since decades. A biodegradable material is able to decompose into CO_2, CH_4, water inorganic compounds, and biomass in which the main mechanism is the enzymatic action of microorganisms, that can be measured by standard tests (Mohanty et al., 2002). Two phases are identified: disintegration (with significant deterioration of mechanical structure) and mineralization (the process by which polymers breakdown into other compounds by physical and microbial agents).

The life cycle of bio-based, biodegradable plastic is shown in Figure 2.13. From the extraction and processing phases, similar to any other resource, the ideal bio-based material can be composted—in other words decomposed by microbiota into CO_2, H_2O, and eventually CH_4—and the biodegradation can occur. The resulting biomass can be re-used as agricultural feedstock to close the life cycle, and the decomposition rate cannot be *a priori* predicted; therefore, methods to assess this parameter have been developed (see Section 3.3). It can be noted that the life cycle can only "virtually" be closed because the degradation depends on some parameters, such as water content, temperature, organic content, carbon-to-nitrogen ratio (C/N ratio), aeration rate, composting mass (Haug, 1993).

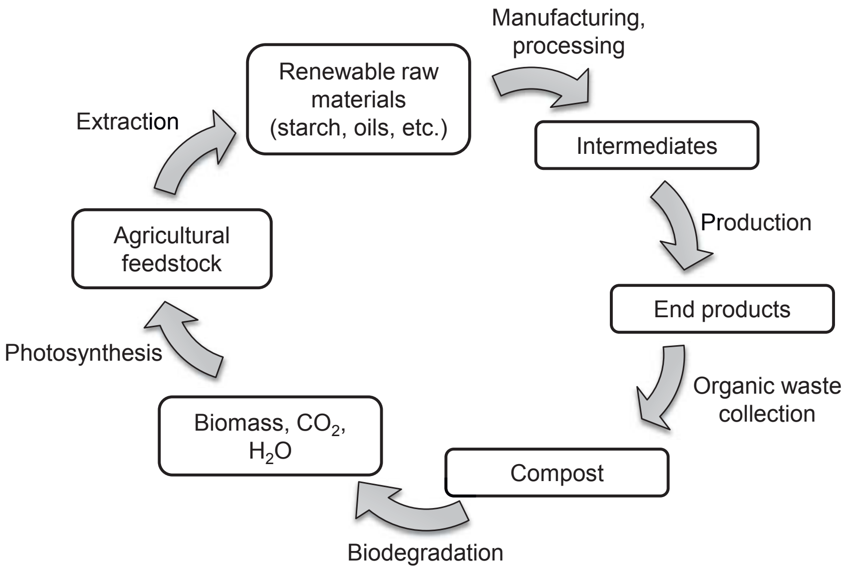

Figure 2.13: Life cycle of biodegradable materials (adapted from Wojtowicz, 2009, p.72)

Many commercial bio-based alternatives have been introduced into the market as petroleum-based plastic substitutes (Platt, 2006, chapter 4). Table 2.1 reports a selection of trade names, suppliers, composition, and application of different commercial bio-based polymers available in the market (year 2018).

Table 2.2 shows a classification of current and emerging plastics differentiated by the capability to be biodegradable (fully biodegradable or non-biodegradable products) and by the basis of composition (fully based on fossil fuels, partially bio-based, or fully bio-based).

In general, containers that are at the same time fully bio-based and fully biodegradable can be used in horticulture. They present positive and negative aspects. On the one hand, these products can be preferred over the petroleum-based ones because they are able to:

- promote growth of plants (Koeser, 2013);
- partially or totally reduce plastic waste once laid underground (mineralization of organic compounds by microorganisms start, see Unmar & Mohee, 2008);
- and provoke less disturbance of roots during transplanting (the "transplant shock" is the consequence of removing plants from their pot, once their growth exceeds the measure of the container, see van de Wetering & Athalage, 2010).

Table 2.1: A selection of biodegradable, commercial polymers and their applications

Product	**Society (Country)**	**Main component**	**Applications**
BAK 1095®	Bayer (Germany)	Polyester amide	Disposable items, flower containers
Ecovio®	BASF (Germany)	Co-polyester	waste bags, or agricultural films
Curlex®	American Excelsior Company (USA)	Aspen fibers	Erosion control blankets
Mater-Bi®	Novamont (Italy)	Starch and polyester	Collection bags for green waste, agricultural films, disposable items
Biomax®	DuPont (USA)	Polylactic acid (PLA)	3D printing filament, food service-ware, packaging
Ingeo®	Natureworks (USA)	PLA	Packaging, 3D printing filament, food service-ware, health and personal care
Expansorb®	PCAS (France)	PLA	Pharmaceutics

On the other hand, these products have properties (e.g., biodegradability rate) that might affect the following agronomic parameters (Nambuthiri et al., 2015):

- plant growth and quality (aesthetic parameter),
- water use (which modifies the irrigation potential of a plant),
- temperature of the substrate,
- container nutrient source,
- pot integrity and appearance,
- biodegradability post-transplanting (which is related to the organic matter content, rooting pattern, weather, cultural practices, and carbon:nitrogen ration),
- and pot's lifespan.

Moreover, overall impacts of prototyped bio-based containers are mostly unknown (Koeser et al., 2014), they are currently not yet introduced at industrial scale (because still at R&D phase, see, e.g., Riggi et al., 2011), and they cannot be used due to their instability against changes in external factors, such as temperature (Platt, 2006, p.94).

Although research on biodegradable plastic for many applications has grown, the use of bio-based containers is still at infancy at industrial scale in agriculture. According to a review on bio-based horticultural containers (Riggi et al., 2011), many patents have been applied for the

Table 2.2: Classification of plastics and sources (Riggi et al., 2011)

	Fully based on fossil fuels	**Partially bio-based**	**Fully bio-based**
Fully biodegradable	- Polybutyrate succinate (PBS)	- Starch blends (with biodegradable fossil fuel-based copolymers)	- Cellulose acetate
	- Polybutyrate succinate-co-adipate (PBSA)	- PLA blends (with biodegradable fossil fuel-based copolymers)	- Polyhydroxyalkanoate (PHA)
	- Polybutyrate succinate-coadipate-terephthalate (PBSAT)		- PLA
	- Polybutyrate succinate-co-lactate (PBSL)		- PLA-PHA blends
	- Polycaprolactone (PCL)		- Starch acetate
	- Polytetramethylene adipateterephthalate (PTMAT)		- Starch blends (with bio-based and biodegradable copolymers)
			- Thermoplastic starch (TPS)
Non-biodegradable	- Acrylonitrile butadiene styrene (ABS)	- Starch blends (with polyolefins)	- Bio-based polyethylene (PE)
	- Epoxy resin	- Polyamide (PA) PA610	- PA11
	- PA6,66 (nylon)	- Polytrimethylene terephthalate (PTT) from bio-based 1,3-PDO	- Bio-based polybutylene (PB)
	- Polybutyrate terephthalate (PBT)	- PBT from bio-based succinic acid	
	- PE	- Polyethylene terephthlate (PET) from bio-based ethylene	
	- PET	- Polyethylene-co-isosorbite terephthlate (PEIT) from sorbitol and bioethylene	
	- Polypropylene (PP)	- Polyvinyl chloride (PVC) from bio-ethylene	
	- Polyurethane (PUR)	- PUR from bio-based polyols	
	- PVC	- Epoxy resin from bio-based glycerol	

production of bio-pots but scarce information is available on commercial utilization. Schettini et al. (2013) developed, for example, biodegradable pots made of recycled wastes of agri-food industries, claiming a complete degradation within two weeks and an enhancement of the plant growth and roots development. van de Wetering & Athalage (2010) proposed a patent for a biodegradable planter container made from rice straw and coconut fiber, whose blend is claimed to disintegrate into the soil in two months. Muriuki et al. (2014) investigated biodegradable pots made from cellulose papers and banana sheaths, which are part of the Ellepot System produced by Ellegard A/S (Denmark), in order to test growth rate of plants under different conditions. Commercial products can be found in non-professional segment. These products are made of wood fibers, rice hulls, starch, coir, peat, grasses, vegetable oils, or cow manure. A selection of products is reported in Table 2.3.

Table 2.3: A selection of commercial bio-pots (Santagata et al., 2017)

Company	**Country**	**Web-site**
William Sinclair Horticulture Ltd.	England	`http://www.william-sinclair.co.uk`
Enviroarc	Australia	`http://www.enviroarc.net`
Fertil SA	France	`http://www.fertilpot.com`
CowPots	Connecticut, USA	`http://www.cowpots.com`
Ecoforms	California, USA	`http://ecoforms.com`
Jiffy Products International AS	Norway	`http://www.jiffygroup.com`

In this thesis, a container made of PLA is taken into account, due to its possibility to be fully biodegradable in soil, and to its promising technical characteristics, which can be suitable for potting plants. PLA is considered the main biodegradable and compostable bio-polymer used in agriculture (mainly film mulching) and available on the market (Guerrini et al., 2017). PLA is a fully biological, biodegradable, aliphatic polyester made of poly(L-lactide) and poly(D-lactide) (Garlotta, 2001). The production of PLA is divided in two phases (Razza & Degli Innocenti, 2012): enzymatic or chemical hydrolysis of various carbohydrate-based (starches) feedstock (e.g., corn, sugar cane, beets, tapioca, millet), cellulose, and lignocellulose materials; anaerobic fermentation.

PLA has some positive characteristics: it is derived from renewable resources (Ghofar et al., 2005), it is claimed that it can be composted in municipal biological treatment facilities, or decomposed in soil, or in water (Rudnik & Briassoulis, 2011), it can be manufactured on existing plastic-processing equipment, it does not produce toxic fumes if incinerated (Lamb, 2008). However, negative characteristics are: it may depend on large fields of crops (due to the field used to grow plants producing starches), it can contaminate recycling processes when mixed with other plastics if the product is not correctly sorted, it should be blended with fillers or protein polymers to reduce its costs. The rate of biodegradation of PLA depends on shape, size, proportions of the two enantiomers, and temperature (Gross & Kalra, 2002).

2.3 Summary

In summary, in this chapter the sustainability concept and possible substitutes in horticulture for peat and plastic containers have been presented.

Firstly, environmental impacts related to the horticulture supply-chain have been illustrated, as well as specific studies related to plant nurseries in Pistoia, focus of this thesis. The horticulture sector is thus described, with its particularities. Specifically, peatland and climate protection have been presented with the description of the European context in the substrate industry, as well as GHG emissions of peat's extraction, processing, and transport to the market. Here, it has been highlighted that exploitation of the peatlands can be reduced. In the same way, utilization of plastics in agriculture and their waste management have been presented. The improvement of the current situation in horticulture has been addressed by introducing the aspects related to the resource efficiency of resources and their cascade utilization, in order to minimize the use and the environmental impacts of a horticultural supply-chain.

Secondly, alternatives to conventional materials have been presented. Compost as peat substitute has been described with its peculiar characteristics, its different kind of composition, and different methods to obtain it. Biodegradable, bio-based plastic materials have been then introduced, with insights on bio-pots novel patents and research reported in the scientific literature.

In the next chapter, a method to assess the environmental emissions of the life cycle of peat, compost, plastic pots, and bio-based pots will be presented and used. Environmental technical parameters of the decision problem will be therefore yielded. The following aspects will be discussed: definition of LCA method (with its steps and discussion of the scientific literature referred to bio-based materials), experimental detection of GHG emissions for bio-based materials, results of the environmental analysis for the four case-specific technical solutions of the decision problem of this thesis.

3 Environmental aspects

In this chapter, case-specific technical solutions are introduced (Section 3.1), theoretical background of an environmental analysis is given (Section 3.2), theoretical foundations for an experimental analysis are described (Section 3.3), the outcomes of such environmental analysis are then reported for the case-specific technical solutions of the decision problem (Section 3.4 and Section 3.5), and finally, the environmental impacts of a potted plant, output of the horticultural supply chain of this thesis, are given (Section 3.6). These environmental aspects are necessary to describe the environmental objective of the decision problem of this thesis to be minimized[1].

3.1 Case-specific technical solutions

For ease of understanding, two different growth media and two kind of plastic containers are chosen as case-specific technical solutions of the decision problem of this thesis (Table 3.1). A generic peat sold from an Italian company is taken into account, since it is one of the most common in central Italy. Olive-mill waste compost obtained with natural aeration is considered as substitute, because it has promising agronomic properties to grow ornamental plants, and much research has been conducted at the R&D stage. A generic plastic container made of PP is considered, since it is one of the most used in horticulture, while a bio-based container made of PLA and produced from a commercial plastic is chosen as alternative, since it has suitable characteristics for cultivating plants, and much research has been conducted on PLA use in agriculture, which can be transferred to horticulture.

On the one hand, peat and petroleum-based plastic containers are considered as *conventional* technical solutions in this thesis, because they are established in the market for decades and well-known by practitioners. On the other hand, olive-mill waste compost and bio-based containers are considered *innovative* technical solutions, such as materials that still need experimentation on the field and further studies to be successfully introduced in the horticultural supply chain.

[1]The content of this chapter is partially included in Castellani et al. (2015), Castellani et al. (2016).

Table 3.1: Case-specific technical solutions

Type of alternative	Case-specific alternative
Peat	Commercial Baltic sphagnum peat (Vigorplant, Italy)
Compost	Experimental olive-mill waste compost from static-pile procedure (according to Altieri et al., 2011)
Plastic pot	Commercial plastic pot made of polypropylene (PP) (generic, manufactured in Italy)
Bio-pot	Bio-pot mainly based on PLA (Natureworks®, USA)

3.2 Life Cycle Assessment

General background on LCA is presented in this section, in order to define relevant basic aspects required to present the analysis of the case-specific technical solutions.

The environmental burdens can be assessed with specific methods, and among them, LCA is one of the most used to evaluate, develop, and improve products and systems (Finnveden, 1999; Guinée et al., 2002). LCA is defined as the "compilation and evaluation of the inputs, outputs, and potential environmental impacts of a product system throughout its life cycle" (ISO 14040, 2006; ISO 14044, 2006). The International Organization for Standardization (ISO), formed a Technical Committee (TC) 207 in 1993 to standardization the environmental management tools and systems of the ISO 14000 series. During the last meeting of TC 207 held on 15–16th August 2016, the Committee voted "to not revise or amend either ISO 14040 or ISO 14044"—the current normative is therefore dated 2006 (see AIHA, 2016).

LCA is a tool to assess (and track) environmental impacts of value chains and to identify paths for improvement (Hellweg & Milà i Canals, 2014). LCA analyzes the impacts from the extraction phase of the raw components to the disposal phase (end-of-life) and it can guide decisions between alternatives. LCA is performed in four steps, as indicated in Figure 3.1. The first step is characterized by the *definition of goal and scope* of the study. Here, the reasons for carrying out the study, the functional unit adopted, the product system, the system boundaries, the allocation procedures (allocating data to unit processes), and the methodology of impact assessment are clarified. The second step is the *Life Cycle Inventory* (LCI). Here, detailed descriptions of each unit process are presented, and flow diagrams are requested to obtain a clear presentation of the system. Data sources, collection techniques, and calculations are presented with input and output of the life cycle. Assumptions for missing data are justified, and allocation of flows are considered. The third step is the *Life Cycle Impact Assessment* (LCIA), where a product system is analyzed from an environmental perspective using selected impact categories

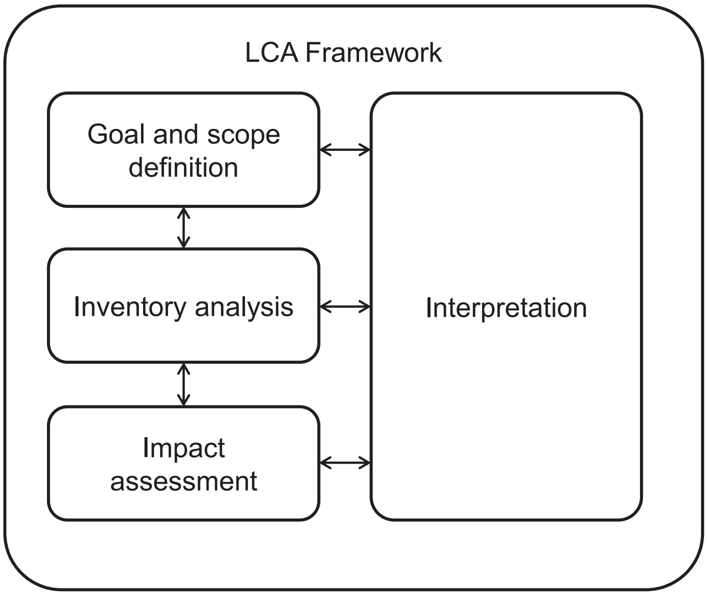

Figure 3.1: LCA steps (ISO 14044, 2006)

and category indicators linked to the LCI results. For each impact category, such as global warming potential (GWP), a category indicator is selected and environmental impacts are calculated. Results are then presented in profiles associated with the inputs/outputs of the product system. Contribution analysis is performed to evaluate the effect of different components of the product. The fourth step is the *Interpretation*, which provides results, limitations, and recommendations based on the previous LCA steps. Here, significant issues of the analysis emerge, sensitivity analysis is performed (determining uncertainties of data which can affect the reliability of final results and conclusions), consistency check is conducted (whenever assumptions, methods, and data collected are consistent with the goal and scope of the study), and conclusions are drawn. An assessment of opportunities and needs to reduce environmental impacts are also discussed.

3.2.1 Goal and scope definition

Within the first step of an LCA, the aim of the study should be declared (e.g., R&D, green marketing), as well as the functional unit of the system, which is the "measure of the function of the studied system and it provides a reference to which the inputs and outputs can be related. This enables comparison of two essential different systems" (ISO 14044, 2006).

For the analysis of environmental impacts of the technical solutions, the boundaries of the system should be defined. In general, studies have been conducted to present a holistic approach of LCA. Ekvall et al. (2007), for instance, identified possible improvements on LCA method by considering the following aspects: economic analysis (e.g., cost-benefit analysis, Life Cycle

Costing, or technology assessment), dynamic linear and non-linear modeling, site-dependent modeling of environmental impacts, and effects on background system. Of course, whenever more aspects are introduced, complexity (which involves more data), and therefore costs for the analysis might increase. In particular for bio-based materials, there are emissions that can be included inside the boundaries of the system. When focusing on waste, this inclusion can be performed with an extension of the system boundaries. Finnveden (1999) suggested that upstream (start of life) and downstream (end-of-life) phases within the boundary system "may have to be changed if one of the systems to be compared produce more or less waste than others". Blengini (2008) explains why the boundaries for the waste management of plastic bags, and avoided fertilizers and steel can be included inside the analysis, for instance, of composting municipal facilities. These aspects can be relevant whereas the outcomes of a LCA would be highly influenced by boundaries, especially for end-of-life stage. In this thesis, system boundaries have been expanded for the olive-mill waste compost, which is a by-product of another supply chain, the olive oil ones.

3.2.2 Inventory analysis

Within the second step, input data collection and selection of environmental indicators and methods are essential steps. Collecting data is an iterative process, which can be conducted step by step by defining a flowchart of qualitative and quantitative description of unit processes that determine the life cycle—fluxes of resources/materials and energy are here indicated. Inflows, expressed in terms of energy requirement (MJ) and primary resources (kg), and outflows, expressed as substances emitted into atmosphere, water, and soil, are in this phase reported.

Different sources can be used for collecting input data: primary data, i.e., site-specific direct measurements, and secondary data, i.e., indirect data retrieved through interviews and databases—for a review of LCA issues that include potential unresolved problems related to many aspects, such as database selection, see Reap et al. (2008). An example of database is Ecoinvent (2013), largely used in Europe. Other relevant databases are the European Life Cycle Database[2] and Probas, included in openLCA project[3]. The Ecoinvent database includes processes which represent in most cases the representation in operation, and "in few cases, the average of technologies offered on market, the best available technique (BAT), or even near future BAT" (Frischknecht et al., 2005)—there are few studies on the combination of BAT and LCA; an example is described by Geldermann et al. (1999a).

[2]Website: `http://lca.jrc.ec.europa.eu/`.
[3]Website: `http://www.openlca.org`.

Inputs and outputs are listed for each product and compiled in this phase. Different sources of data, such as primary (field measurements) and secondary (literature, databases, and Internet web-resources) are used for the calculations.

In this thesis, sub-phases of the life cycle are used to aggregate inputs and outputs: in Figure 3.2 four sub-phases are identified (upstream phase, processing/use phase, transport phase, and downstream phase), as well as inputs (upper part: raw materials, fuel, electricity) and outputs (lower part: emissions into the atmosphere, residuals, fugitive emissions, biogenic emissions, end-of-life emissions). This particular flow chart is used in this thesis to describe the life cycle of peat and compost, and to present the outcomes grouped by sub-phases, for ease of understanding.

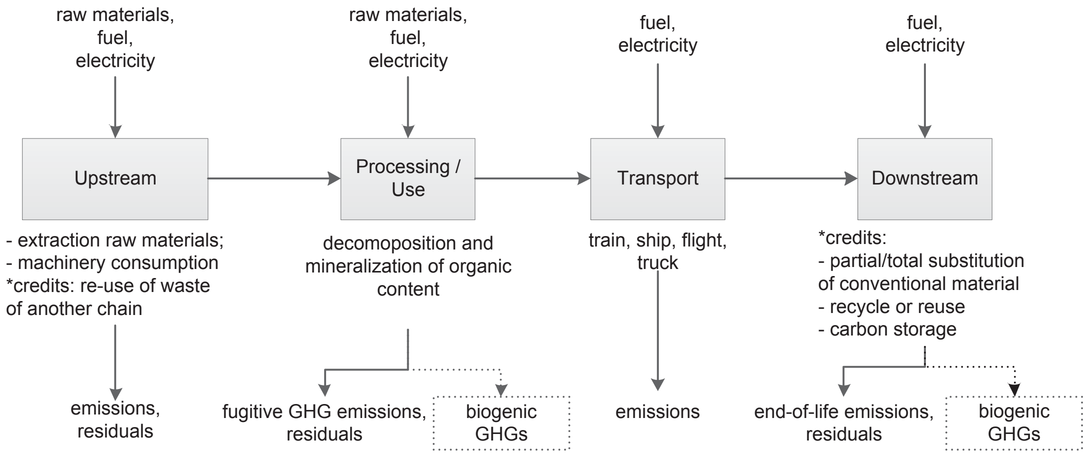

Figure 3.2: Example of life cycle sub-phases.
Notes: Specific outputs for peat and compost are: fugitive GHG emissions (processing phase), biogenic GHGs (downstream stage). Dotted boxes indicate possible inclusion of emissions

The *upstream* phase includes electricity use (in kWh), fuel consumption (in MJ), and direct emissions (GHG emissions) generated when extracting input raw materials to be processed. On the one hand, boundaries of the system can be extended to include end-products of another chain, which became input for the investigated supply chain. For instance, when studying olive-mill waste used as input for a recycling process, pre-processing should be included within the LCA. On the other hand, environmental credits of avoided emissions when reusing or recycling by-products can be subtracted from the system. For instance, when investigating the life cycle of a recycled post-consumer pot, the inputs for the avoided manufacturing of virgin material for a non-recycled pot (e.g., avoided fuel consumption and GHG emissions) is accounted as credit.

The *processing/use* phase includes decision-relevant processes for each product:

- mechanical operations to manufacture/process (electricity use of machinery and fuel consumption, e.g., to pump the wastewater);

- maintenance operations (necessary to preserve the product until the transport phase or the downstream phase; when calculating impacts of compost, for instance, operations to preserve the efficiency of a composting facility can be included, e.g., control, repair, substitution of out-of-order components);
- fugitive emissions (due to the degradation of organic content of peat and compost);
- disposal of residuals discarded from the manufacturing (linked to the kind of end-of-life, i.e., disposal in landfill, re-processing, or recycling).

Fugitive emissions of the GHGs CO_2, CH_4, and N_2O can be accounted here, as follows. CO_2 is emitted into the atmosphere when organic matter oxidizes in two forms: biogenic or fossil. Biogenic CO_2 emissions are related to the natural oxidation of organic matter within the carbon cycle, and they can be accounted and reported separately to ensure transparency—see e.g., PAWELZIK ET AL. (2013); BLENGINI (2008) and see specific subsection on biogenic emissions on p.36. Fossil CO_2 emissions, in contrast, are linked to carbon loss within the global cycle, therefore included in LCA (e.g., for the carbon loss of peatlands, see Section 2.1.2). Following the approach by BOLDRIN ET AL. (2009), fossil CO_2 emissions are estimated in this thesis from the degradation of carbon (C) content of the material ("loss of carbon"). Therefore, the C content before the degradation (C_{input}, in kg, via elemental analysis of the product) and fractional C loss after degradation ($C_{loss\%}$, in %, via elemental analysis or literature data) are calculated as follows:

$$C_{release} = C_{input} \cdot C_{loss,\%} \tag{3.1}$$

By using Equation 3.1, the following equation can be used for the release of CO_2 (in kg):

$$CO_{2,\mathrm{release}} = C_{\mathrm{release}} \cdot CO_{2\mathrm{emitted},\%} \cdot \frac{44}{12} \tag{3.2}$$

where $CO_{2\mathrm{emitted},\%}$ is the percentage of C lost as fossil CO_2; 44 is the molar weight of CO_2 and 12 is the atomic weight of C.

CH_4 is emitted with the production and transport of coal, natural gas, and oil, with agricultural practices, and with the decay of organic matter of waste in landfill. By using Equation 3.1, CH_4 emission (in kg) can be expressed as:

$$CH_{4,\mathrm{release}} = C_{\mathrm{release}} \cdot CH_{4\mathrm{emitted},\%} \cdot \frac{16}{12} \tag{3.3}$$

where $CH_{4\mathrm{emitted},\%}$ is the percentage of C lost as CH_4; 16 is the molar weight of CH_4 and 12 is the atomic weight of C.

N_2O is emitted with agro-industrial activities and with the combustion of fossil fuels or solid waste. The loss of N ($N_{loss,\%}$, in %) can be expressed from the input N (N_{input}, in kg) with:

$$N_{release} = N_{input} \cdot N_{loss,\%} \tag{3.4}$$

and by using Equation 3.4, N_2O emission (in kg) can be estimated as:

$$N_2O_{,\mathrm{release}} = N_{\mathrm{release}} \cdot N_2O_{\mathrm{emitted},\%} \cdot \frac{44}{14} \tag{3.5}$$

where $N_2O_{\mathrm{emitted},\%}$ is the percentage of N lost as N_2O; 44 is the molar weight of N_2O and 14 is the atomic weight of N. To sum up, the fugitive emissions of fossil CO_2, CH_4, and N_2O can be also directly measured with experimental analysis, described in Section 3.3.

The *transport* phase includes shipment of the product via truck, train, flight, or ship. Distances of transport can be assumed or determined via interviews or direct communications with the parties involved in the life cycle of the product. This calculation is made by considering the capacity of vehicle (t, weight in tons) multiplied by distance (km).

The *downstream* phase includes disposal of the product and its degradation until the end of the time horizon. The definition of time horizon deserves further explanation, as follows. In LCA, 'time horizon' is considered the time frame of the life cycle of a product, which is defined with its functional unit. For example, for the determination of impacts of a composting plant, the *functional unit* can be "1 kg of final compost, which is considered biologically mature and stable for commercial use in horticulture". All processing/manufacturing operations which lead to such final product should be therefore taken into account. Here, time horizon is defined by the time necessary to perform such processing operations. In this thesis, environmental emissions caused by the decomposition of peat and compost after utilization are included. This decomposition, which is linked to the organic content of peat, compost, or bio-pot, can be estimated as *decay rate* of its organic matter over 100 years. This estimation can be done by simulating decay rate of the organic matter. By using Equations 3.2, 3.3, and 3.5, and expressing the GHGs emissions with a generic term (GHG_{release}, in kg), the downstream emissions are as follows (in kg):

$$GHG_{\mathrm{downstream}} = d \cdot GHG_{\mathrm{release}} \tag{3.6}$$

where d (factor expressed in % and calculated as subsection on p.43) is the degradation rate estimated via simulation, within the time horizon.

3.2.3 Impact assessment

In the third step, the environmental effects associated with the substances indicated in the previous step are determined. Emissions and resources calculated in the previous section are therefore grouped here according to their impact units. It is important to underline that from the quantitative determination of emission of substances (*objective* analysis, as seen in the previous step), the environmental effects are here evaluated by choosing between impact indicators and methods of calculation (*subjective* analysis). The latter involves therefore a "judgment" of the potential damage to the environment (soil, water, air).

In general, most LCAs cover the following impact indicators: GWP, stratospheric ozone depletion (ODP), acidification potential (AP), eutrophication potential (EP), photochemical smog, terrestrial toxicity, aquatic toxicity, human health, resource depletion, land use, and water use. These indicators convert life cycle inventory data into equivalent emissions expressed, for instance, with carbon dioxide (for GWP), hydrogen (H+) ion (for AP), phosphate (PO_4) (for EP).

It is however important to note that the inclusion or not of one or more specific impact indicators (or the use of a method) is arbitrary and depend on the performer of a LCA. Avoiding redundancy of impact indicators has been proposed in the "streamlined LCA" (Hochschorner & Finnveden, 2003). With this regard, Steinmann et al. (2016) stated that GWP is strongly correlated to all other indicators, and they suggested that decision-making performed with too much indicators simultaneously can be, in many cases, impracticable. The authors used Principal Component Analysis (Yoder, 1936) and an optimization algorithm to find the optimal set of indicators between 135 impact indicators, calculated for 976 products reported in the Ecoinvent database. Klöpffer (1997) suggested that Cumulative Energy Demand (CED), the most applied life cycle based energy indicator, is a "sum parameter which implicitly indicates the environmental interventions due to the energy consumption connected with the system analyzed" and that "the accuracy of the well-established calculation of CED is much greater than that of most (emission-based) characterization procedures".

Following these considerations, a LCA can be reduced to few impact indicators. The analysis performed in such way can still show an overall picture of environmental burdens of a product/system, pursuing a business administration perspective. In this thesis, decision-relevant impact indicators, chosen for their relevance to agriculture (Torrellas et al., 2012), have been selected: GWP, CED, EP, and AP.

According to IPCC (2007), GWP is the radiative forcing caused by the release of a unit of mass of a given GHG integrated over a prescribed time period (usually 20, 100 or 500 years), relative to that of a unit mass of CO_2, and measured in kilograms of carbon dioxide equivalent per kilogram of GHG (kg CO_2-eq kg^{-1}). CO_2-eq is "the concentration of carbon dioxide that would cause the same amount of radiative forcing as a given mixture of carbon dioxide and other GHGs" (Johnston, 1999). Although this is the most widespread definition, it is not the only one (Shine, 2009). Fisher et al. (1990) defined the GWP as the ratio of the surface temperature change due to a sustained emission of a gas, relative to the temperature change due to a sustained emission of CFC-11. Although this definition is equivalent to the IPCC's definition where an infinite time horizon is adopted, according to Shine et al. (2005), this equivalence is quite close whether a time horizon of 100 years is considered. Rotmans & Den Elzen (1992) defined the GWP as a time-integrated version of the pulse global temperature change potential. In LCA, GWP is calculated over a specific time frame, which is typically 100 years (GWP_{100}). Therefore, the consequences of a specific action taken in a specific time are calculated by delaying emissions of GHGs in Impact Assessment by 100 years. This approach is consistent with many studies, e.g., Clift & Brandão (2008), Brandão & Levasseur (2011), Brandão et al. (2013). Specifically, CO_2 and CH_4 are expressed as CO_2-eq by using their GWP—see the characterization factors given in Table 3.2, used in this thesis.

Table 3.2: GWP characterization factors (100 years)

	Characterization factor [kg CO_2-eq]	**Reference**
Carbon dioxide, fossil (fossil-CO_2)	1	Solomon et al. (2007)
Methane (CH_4)	25	Solomon et al. (2007)
Nitrous oxide (N_2O)	298	Solomon et al. (2007)
Carbon dioxide, biogenic (biogenic-CO_2)	0	Christensen et al. (2009)
Avoided carbon dioxide, fossil	-1	Christensen et al. (2009)

CED is an indicator of indirect and direct energy use that includes fossil, non-renewable resources, and renewable resources, and highlights energetic consumption of bio-based materials; EP is an indicator of surface water eutrophication; AP is an indicator of the phenomenon of acid rain. It should be noticed, that the calculation of impact potentials "largely removes spatial and temporal considerations, resulting in analytical and interpretative limitations" (Geldermann et al., 2003). Therefore, these considerations should be included when compiling LCA inputs and expressing them with their impact potentials. Moreover, in order to make comparisons between the outputs of a LCA, the same method, environmental indicators, functional unit, boundaries of the system, assumptions, and life cycle stages should be taken into account.

These impact indicators can be used in a *contribution analysis* (for further description of this analysis, see Section 3.3), which enriches the study by identifying and comparing strengths and weaknesses (in terms of environmental impacts), associated with a specific component of the potted plant (e.g., a component is the growth substrate, the plant, or the pot) or to a stage of the life cycle (e.g., for the compost, a stage is the processing of input agro-waste, the composting process, or the transport). By doing so, components of the life cycle that relevantly contribute to the total environmental emissions can be further discussed for improvement in the supply chain, and, eventually, substituted with other components with lower impacts.

In the impact assessment, calculating impacts for bio-based materials which contain organic matter is not an easy task, since the scientific community does not have found a common path for accounting—databases (e.g., Ecoinvent) do not yet include this accounting. Therefore, this topic deserves further explanation, as described in the following subsection.

Biogenic emissions and end-of-life options of agro-industrial wastes

The carbon cycle (C-cycle) is crucial to understand the source of emissions. Modification, degradation, and transformation of C can yield CO_2 and CH_4 emissions, in particular for agro-industrial wastes, and specifically for compost. The most comprehensive work on composting emissions in Europe was presented by Smith et al. (2001). The authors defined approaches and methodologies to guide the accounting. In Figure 3.3, C-cycles for landfill and composting options are shown with relative fugitive emissions.

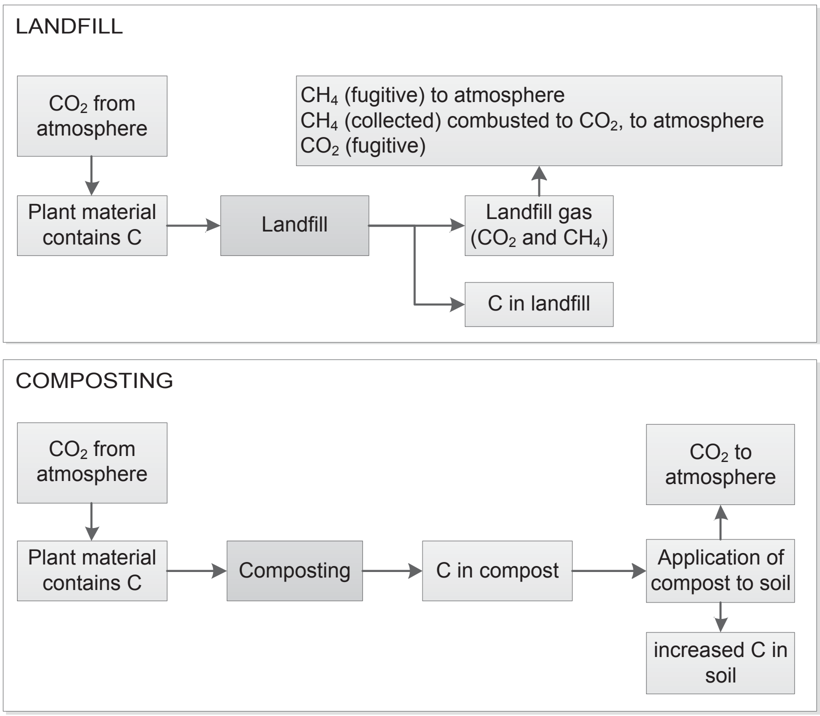

Figure 3.3: Carbon flows of biodegradable waste (adapted from Smith et al., 2001)

The emissions, in general, can be of fossil or biogenic origin. Fossil emissions are related to the increment of GHGs in the atmosphere and are in general accounted in LCA (Guinée et al., 2001). Conversely, biogenic emissions require further consideration, and there is no consensus among the LCA scientific community (Brandão & Levasseur, 2011). Many methods and approaches have been developed to handle biogenic emissions for bio-based products. Extensive reviews are reported in Levasseur et al. (2012), Brandão et al. (2013), and Pawelzik et al. (2013)—see also the review for temporary carbon storage made by Weiss et al. (2012), where three methods of accounting are reported: ignoring the biogenic contribution (Guinée et al., 2009), allocating the contribution, and crediting (Brandão & Levasseur, 2011).

Biogenic carbon dioxide emissions are denoted as CO_2 emissions linked to natural carbon cycle and other sources, such as combustion, harvest, fermentation, and natural decomposition of bio-based feedstock. A 'natural carbon cycle' is a flow of carbon in various forms, for instance as CO_2, through the atmosphere, terrestrial biosphere, ocean, and lithosphere (IPCC, 2007). And 'bio-based feedstock', as defined by U.S. EPA (2014), are biodegradable, organic materials not from fossil origin, which derive from plants, animals, and microorganisms, and include residues (by-products) from agriculture, horticulture, forestry, and the municipal waste organic fraction. C-fluxes can originate from different sources, as shown in Figure 3.4.

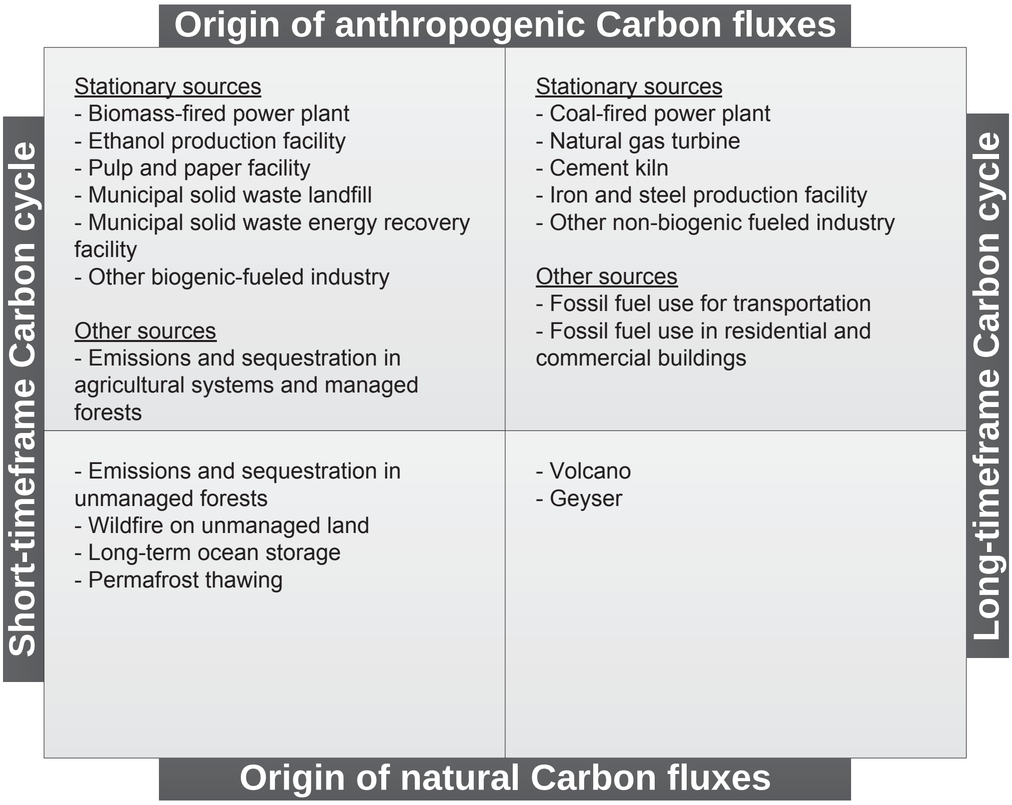

Figure 3.4: Carbon fluxes: types and time frames (adapted from U.S. EPA, 2014, p.8)

The bio-based feedstock category does not contain coal, petroleum, natural gas, and other materials that are considered "non-renewable during time-horizon relevant for policymaking" (U.S. EPA, 2014). Materials from fossil origin could remain isolated from the biogeochemical C-cycle without human intervention. It should be noticed, that the bio-based feedstock category do not include peat.

The effect of excluding biological CO_2 from processes can be studied to avoid distortion of the outcomes (see, e.g., the investigation on agro-wastes by Blengini, 2008). Christensen et al. (2009) suggested to measure biogenic emissions for processing via composting, due to its potential in substituting fertilizers and sequestrating C into the soil, when used as growth substrate. This concept is equivalent to the carbon sink effect presented for peatlands. An interesting work for estimating carbon sequestration potential was published by Smith et al. (1997). The authors investigated different scenarios with long-term experiments, by taking into account a 20-years old database, where data retrieved from experiments and research programs have been collected. Generalizing, when long-term experiments for case-specific materials are missing, simplifications are introduced. And imprecisions that are consequently introduced in the model should be therefore included in LCA (for further discussion on this topic, see Section 3.3). Other important works have been done by Guest et al. (2012) and Cherubini et al. (2012). Bovea & Powell (2006) estimated biogenic CO_2 emissions (250 m^3 per ton of biodegradable material emitted from landfill and 100 m^3 per ton of material from composting process) from the putrescible fraction of municipal solid waste disposed in landfill. The same concept was summarized by Weiss et al. (2012), who included in his investigation the potential reduction of agro-industrial residues when used for manufacturing other materials or for energy—the production of second generation biofuels (e.g., bioethanol) and biochemicals by biorefineries is an example of cascade utilization of agro-waste, i.e., lignocellulosic crop (Cherubini & Jungmeier, 2010).

As for agricultural residuals, also for biodegradable plastic materials the accounting of biogenic CO_2 emissions is still controversial. Weiss et al. (2012) reported a case study described in Hermann et al. (2011), where the analysis was focused on several biodegradable polymers, in order to define carbon credits and differences between end-of-life options, such as industrial composting, home composting, municipal incineration, incineration for energy production, and anaerobic digestion. As the author suggested, when compost is used to replace petroleum-based materials, its beneficial effect in terms of CO_2-credits can surpass the benefits of final incineration. In general, nevertheless, the cascade utilization of bio-based materials within the supply chain with final incineration for energy recovery maximize the avoided environmental emissions (see Bringezu et al., 2009).

3.2.4 Interpretation

The interpretation phase is used to indicate, suggest, point out, or summarize paths for reducing environmental impacts, and guide towards the choice of alternatives. By doing so, the objectives of the study can be answered. In this phase, problems related to the accounting and limitations on the choice of the environmental indicators should be pointed out. Furthermore, an evaluation through sensitivity analysis, as well as significant direction for the improvement of the emissions.

Specifically, the interpretation phase defines the following aspects: *normalization*, *grouping*, and *weighting*. Normalization is used to compare the outcomes with a particular geographical area that is considered relevant for the aims of the study. Grouping is used to rank the environmental impact categories and group them into categories. Weighting is used to restrict the result on a single environmental score that can encompass all the other environmental impact indicators. These actions are, however, affected by subjectivity and may lead decision-makers to misunderstand the outcomes.

In this thesis, the outcomes are compared with literature data, and parameters which may have positive effect on the emissions related to a potted plant are discussed—this approach follows other studies on ecological assessment, e.g., the work on kerosene burning and its improvement done by Geldermann et al. (1999b). The choice of a specific commercial type of peat transported from northern European Countries, for instance, can be compared to peat produced in Canada and transported to a plant nursery in Europe. The transportation phase, in this case, relevantly contributes to the total environmental emissions. This comparison can help to understand the influence of a specific product to the overall environmental impacts of a potted plant. Finally, recommendations can be suggested. Results of a study should be further checked and reviewed by experts, as soon as more data and information can be retrieved.

Summarizing, in this section LCA has been described with its four steps to define basic aspects and prepare the field for an application of the method with materials with organic content (e.g., peat and compost used in horticulture, focus of this thesis). However, listing inputs and outputs for these materials is challenging, because their emissions are case-specific, no literature data is available to assess the environmental burdens, and no appropriate databases of emissions can be used. For these reasons, experimental detection of case-specific emissions could be of help in this particular case. The following section describes therefore the method pursued in this thesis for environmental accounting of these emissions.

3.3 Experimental detection of GHGs

To further analyze case-specific environmental emissions of materials with organic content, experimental analysis is presented in this section. As seen in Section 3.2.2, experimental analysis can be necessary to determine fugitive emissions and evaluate the downstream phase of case-specific products to provide data for the inventory analysis in LCA. This is required in particular for innovative materials that are at R&D stage. After conducting experimental trials, LCA can use their outcomes and quantify the environmental consequences.

Experimental analysis is able to determine GHGs emitted by organic materials in the use phase and predict the end-of-life phase. The 'use phase' is related to the transformation of the C content of a bio-based material into other compounds, and it is associated with GHG fugitive emissions. In this thesis, emissions of the use phase for a potted plant are, while emissions of the 'end-of-life phase' are related to the decomposition of C content until a prefixed lifespan.

Three kinds of experimental analysis can be conducted:

- **In situ analysis** is aimed at directly evaluating GHG emissions of resources with equipment and chemical analytical methods (e.g., infrared analysis, gas-chromatography)—*equipment* indicates an experimental apparatus developed for research purposes, and its development is referred to hardware implementation and software programming. For GHG accounting and extending the results to the chosen time horizon, long-term experimentations (years) have been conducted (see, e.g., the experimenal trials reported by Smith et al., 1997);
- **Simulation analysis** is related to mathematical modeling of emissions and parametrization of outcomes with empirical equations (see, e.g., the work on mathematical simulation for the composting process done by Petric & Selimbašić, 2008);
- **Laboratory analysis** is related to micro- or meso-scale determination of GHG emissions through equipment, in many cases developed for the aim (for the connection between experimental trials in laboratory and carbon storage deteremination see, e.g., the work by Barlaz, 1998).

It should be noticed, that a discussion on GHG accounting via experiments is still ongoing, and much research have been conducted in many disciplines. Regarding the determination of GHG emissions, comprehensive studies have been done for compost (Boldrin et al., 2009), for peat (Cleary, 2004), and general frameworks have been proposed (EPA, 2016). Many methods have been developed and adopted to assess GHGs and derive databases of emissions for case-specific products. Moreover, boundary conditions of an experimental trial could influence the outcomes of a study, which are therefore considered case-specific. In general, studies have been conducted with commercial apparatus (see the review by BARRENA GÓMEZ ET AL., 2006), in accordance with international directives (e.g., ISO 14855-1:2012), evaluating the respiration of organic, bio-based materials with the detection of oxygen (e.g., GARCIA-OCHOA & GOMEZ, 2009) or carbon dioxide (e.g., ADANI ET AL., 2006), at different scales (e.g., YAMADA & KAWASE, 2006), with different matrices/substrates (WONG ET AL., 2011), etc. Other studies have been conducted to directly determine emissions and use them as basis for LCA. One of the most relevant work about the experimental analysis for the determination of emissions of peat and compost was done by BOLDRIN ET AL. (2009), with a subsequent LCA by BOLDRIN ET AL. (2010). Since these works are related to the Danish context and specific compost made of green waste, and since LCA for bio-based materials is site-specific, their outcomes cannot be generalized or directly transferred to Italy.

In this thesis, experimental trials in situ and at laboratory scale have been conducted to:

- estimate fugitive GHG emissions (accounted in the processing phase),
- predict the end-of-life (accounted in the downstream phase),
- determine the carbon storage effect (accounted as avoided emissions in the downstream phase).

These aspects are further described in the following three subsections, followed by an additional subsection about uncertainty and sensitivity analysis. Sensitivity analysis has been used in this thesis for evaluating fluctuation of inputs and outputs in LCA, and it deserves further explanation before investigating the case-study.

Estimation of fugitive GHG emissions

In this thesis, fugitive GHG emissions of compost as main component of the growth substrate have been measured, and two laboratory scale prototypes have been developed to simulate composting conditions and detect GHG emissions[4].

One of the aims for prototyping these prototypes is the determination of fossil CO_2 emissions under controlled conditions. It should be noticed that, although experimental detection could be

[4]In this thesis, a *white box* is implemented for conducting the trials at laboratory scale, which is contrast with a black-box. In the discipline of computing, defined as "the systematic study of algorithmic processes that describe and transform information: their theory, analysis, design, efficiency, implementation, and application" (DENNING ET AL., 1989), a *black-box* is intended as device (or program) whose inputs and outputs are known, but its internal structure could be unknown due to its confidential nature.

useful to upscale laboratory results and simulate (via prediction) end-of-life of products, more investigation is required. Here, an introduction of the topic is given.

The fugitive emissions of biogenic CO_2, fossil CO_2, CH_4, and N_2O are inevitably emitted by bio-based products and can be detected with sensors in situ or at laboratory scale. In this thesis, other GHGs are excluded due to technical issues related to sensors' limit of detection, e.g., in the case of fluorinated gases, namely hydrofluorocarbons, perfluorocarbons, sulfur hexafluoride, and nitrogen trifluoride, which are synthetic, powerful GHGs emitted by several industrial processes (EPA, 2011).

When dealing with peat, *in situ analysis* can be performed by detecting, for instance, CH_4 emissions due to the degradation of the organic matter during stock-piling after excavating from peatlands and before transporting (see Section 2.1). When dealing with compost, in situ analysis can be necessary to detect fugitive emissions during the composting process itself—it should be noted that the composting system adopted influences the results. Figure 3.5 shows schematically a composting pile, which is analyzed with probes collocated inside the mass at different positions. In this thesis, following this approach, GHGs detection has been performed for olive-mill waste during static-pile composting.

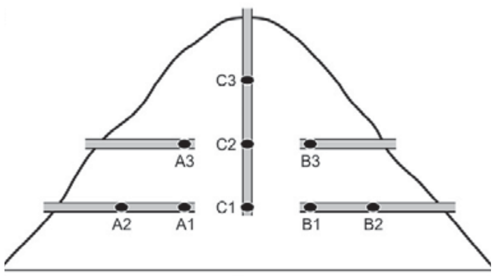

Figure 3.5: *In situ* analysis of composting pile (Andersen et al., 2010b).
Notes: black dots represent sampling locations of gas and gray lines represent gas probes in a cross-section of the composting pile

In general, this approach can be performed for any bio-based material, which, during the processing/use phase (see Figure 3.2), may emit fugitive GHGs into the atmosphere—the composting phase is indicated as the most important phase where GHGs can be emitted (van Haaren et al., 2010).

Laboratory analysis is performed, on the other hand, by simulating processing activities with forced aeration of oxygen, to stimulate the degradation of organic matter (*mineralization*). The simulation is performed as follows. GHGs which are emitted from this degradation activity are measured by controlled decomposition inside reactors in presence of oxygen (aerobic degradation). To simulate composting conditions at laboratory scale, forced aeration at the same conditions of municipal composting facilities is performed, and respiration of bacteria that naturally occur is investigated by detecting GHGs. In particular, the detection of CO_2 concentration of the exhausted air coming from the reactors can be performed via infra-red analysis, while the detection of CH_4 and N_2O via gas chromatography—it should be noted that N_2O detection is hardly performed at laboratory scale due to its volatility and low emission rate from the mass, and, consequently, studies neglected this problem (see, e.g., Sommer & Dahl, 1999). The following equation has been used in this thesis to determine CO_2 emissions [g] of the exhausted air escaping from a reactor:

$$\mathrm{CO_{2,sample}} = \frac{\mathrm{Q} \cdot \mathrm{CO_2} \cdot \Delta \mathrm{t} \cdot \mathrm{M}\,(\mathrm{CO_2})}{\mathrm{V}_m} \tag{3.7}$$

where Q is the flow rate measured with a gas mass meter (of the inlet air to the reactor) [m^3/min], CO_2 is the concentration of carbon dioxide measured with infra-red sensor in the exhausted air [ppm], Δt is the period of the measurement cycle [min], $M\,(CO_2)$ is the molar mass of CO_2 [g/mol], and V_m is the volume occupied by one mole of CO_2 at the exhaust-air temperature, as determined by the sensor [m^3/mol]. This general equation can be used to determine discrete emissions of CO_2 over time. The cumulated emissions during the trial's time simulate CO_2 emissions in real-scale. When dealing with compost, trial's time can be various months, until a steady-state of emissions is reached (further details are reported in the next chapter). Following the approach of Boldrin et al. (2009), in this thesis the experimental detection of fossil CO_2, CH_4, and N_2O emitted from compost and peat are expressed as CO_2-eq emissions with the characterization factors of Table 3.2.

End-of-life prediction

The prediction of the end-of-life of peat and compost can be performed by measuring the decay rate of organic matter (parameter d, introduced in the previous section). GHGs detected at the laboratory scale, which approximate the degradation of organic matter in situ, are influenced by temperature, moisture, particle size, and chemicals (Zeman et al., 2002). These parameters should be taken into account when performing trials. Furthermore, since the duration of trial (in general, some months) cannot coincide with the selected time horizon of the analysis (time horizon for the GWP_{100} in LCA is indeed 100 years), GHG emissions measured at the laboratory scale must be extended with a decay function. This approach presented for peat decomposition by Cleary (2004) has been used for this thesis to determine the degradation of peat. A single exponential decay equation can be used for the determination, as shown in Figure 3.6.

At the end of the selected time horizon (100 years), almost the total amount (around 99%) of the initial carbon content of peat is emitted following an exponential decay rate, as shown in

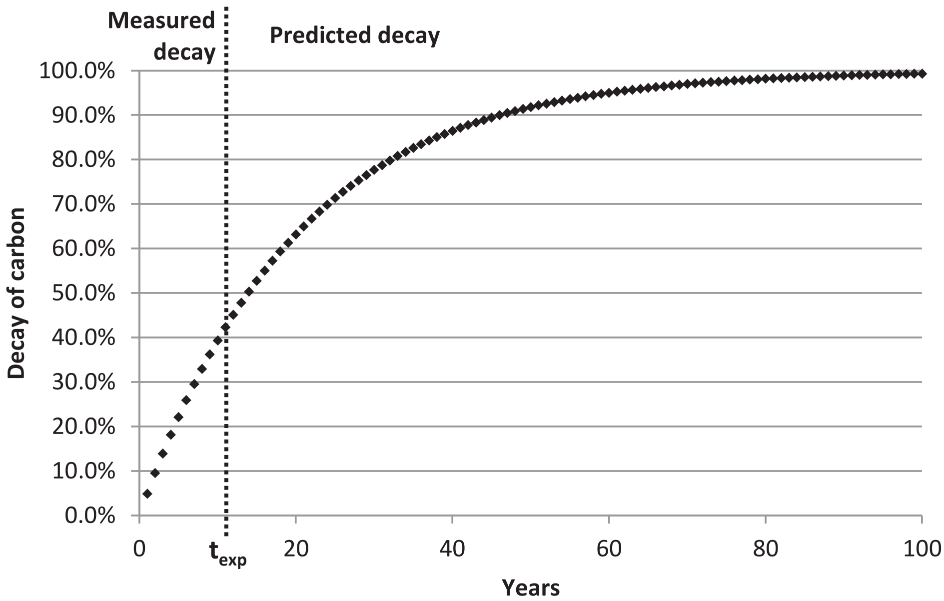

Figure 3.6: Example of single decay function for peat (5% decay per year).
Notes: Each dot indicates the cumulated decay every year; t_{exp} is the time of the experimental trial. The decay of carbon (C) is measured until t_{exp}, and after the decay is predicted with the same decay rate per year.

the figure. This decay rate (parameter d, as in Section 3.2.2, in this case equal to 5% per year) can be experimentally determined via regression analysis of discrete CO_2 emissions measured experimentally (see Barlaz, 1998). The decay rate is indeed measured until the time of the trial (t_{trial}, in Figure 3.6). From this time, the rate of decay can be assumed constant ("predicted decay", in Figure 3.6), or reduced by a corrective factor that simulates in situ conditions. C-emissions at the end of the selected time horizon are then transformed with Equation 3.6 to fossil-CO_2 emissions (if the product has fossil origin, e.g., peat or plastic pot), and to biogenic–CO_2 emissions (if the emissions of the product are included in the natural carbon cycle, e.g., compost or biological content of bio-pot).

Determination of carbon storage effect

Following the same procedure, the carbon storage effect can be in theory determined. In this case, conversely, it should be addressed the percentage of C (and therefore the avoided emissions as fossil-CO_2 from petroleum-based materials) that is bound into the soil at the end of the selected time horizon. This is particularly evident with compost from agro-waste, whose use as growth substrate contribute to the carbon content of the soil (Favoino & Hogg, 2008).

However, prediction and approximation of carbon storage into the soil is particularly critical, because many factors influence it. For practical reasons and for ease of analysis, in this thesis, carbon storage effect of compost and peat are taken into account from the scientific literature (Cleary, 2004; Smith et al., 2001). Long-term studies (20-40 years) have been used as main reference for the calculations (e.g., Murayama et al., 1990).

Sensitivity analysis

In general, since variation of environmental inputs can produce different outcomes, the heterogeneity of resources investigated is reflected on different emissions. In LCA and when determining GHG emissions via experimental analysis with bio-composite materials, uncertainties are introduced in the modeling. It is widely acknowledged, that there are many sources of uncertainty in LCA, such as data inaccuracy and gaps, unrepresentative data, model uncertainty, uncertainty due to choices, spatial and temporal variability, variability between sources and objects, epistemological uncertainty, measurement errors, and estimation of uncertainty (Björklund, 2002). Therefore, a sensitivity analysis helps to evaluate the effects of these variations and present transparent outcomes.

Fluctuations introduced into the results by the cumulative effects of input uncertainty (and data variability) can be estimated, in particular, by following the general procedure suggested by Björklund (2002): (1) scoping the uncertainty analysis, (2) selecting a method for modeling the uncertainties, (3) assessing the uncertainties in input data, (4) propagating the uncertainties though models, (5) reporting the uncertainty of output data. Following this general principle, LCA of agro-waste can handle uncertainties related to (Clavreul et al., 2012): biological treatment (e.g, composting), use of waste on the soil (e.g., spreading agro-waste), and recycling (e.g., substituting petroleum-based resources). Specifically, biological treatment is affected by the degradation rate of organic matter (contained in the bio-based material) and by variation of CH_4 emissions, use of waste on the soil is affected by the substitution rate of compost/fertilizers and carbon storage effect, and recycling is affected by the substitution rate of raw resources (e.g., peat).

Sensitivity analysis, which evaluates the effects on the results due to the modification of one single input at once, is performed in this thesis (and reported in the interpretation LCA step) by means of (Clavreul et al., 2012):

- *contribution analysis*. This method (presented in Section 3.2.3) concerns the "decomposition" of LCA outputs into single processes, in order to clearly present the contribution of a single element of the analysis. The LCA of a construction building, for instance, can include a contribution analysis that is referred to impacts of heating system versus others elements, such as pavement, shell, etc. (see, e.g., PROIETTI ET AL., 2013);

- *scenario analysis*. This method (presented in the example in Section 3.4.4) evaluates general effects on the outputs due to a modification of a set of conditions (a *scenario* is defined as the sum of particular input parameters that simulate a real-world situation or prediction, e.g., the change of heating system for a building, the change of LCA method, or the change of substitution rate of a bio-based material).

Moreover, outcomes of LCA can be only understood when the results have been displayed. Therefore, scenario analysis is used to illustrate variation of LCA outcomes (see, e.g., Figure 3.15). An example is the work done by GELDERMANN & RENTZ (2004a), who presented a LCA linked to cost analysis with different scenarios for the vehicle refinishing sector, in order to quantify changes between several spray processes and stages compared to the initial situation. A further example is about the high variability of emissions related to growth substrate: GHG emissions of peatlands have indeed been investigated and they still present high uncertainty (see Section 2.1.2), as well as fugitive GHG emissions during composting; their accounting in LCA introduce therefore uncertainties.

Summarizing, the theoretical background given so far is used as a basis for the LCA presented in the next two sections, where environmental impacts of the four case-specific technical solutions are calculated and reported.

3.4 LCA of peat and compost

In this section, the environmental technical parameters for the alternatives indicated in Table 3.1 (on page 28) are determined. The peat chosen is a commercial product sold in large bales having properties suitable for growing plants. The experimental compost is produced following the static-pile composting procedure by ALTIERI ET AL. (2011).

3.4.1 Goal and scope definition

In the first step, goal and scope of the study is defined. In this thesis, the scope of the study is to find decision-relevant environmental emissions of the case-specific alternatives in the horticultural supply chain.

The functional study, which is the basis comparison in LCA, is “a potted plant of a specific species cultivated inside a plant nursery for one year”. This functional unit has been chosen because it is representative of the decision problem. The system boundaries of the LCA of each product are defined, and non-relevant processes are excluded and justified on the basis of: previous studies (e.g., from literature), neutrality to LCA, or because not decisive for the study or outside its scope.

This LCA has been performed for the following reason. Many LCA studies have highlighted the environmental impacts of compost produced from different sources (Zeman et al., 2002) and different technologies (van Haaren et al., 2010). Each of the many commercial and experimental composts available requires specific data, however, to assess its environmental impacts. This is a challenge in LCA, where a lack of primary data can lead to inaccurate results and debatable conclusions. Although passively-aerated static-pile composting has been under study for decades (see, e.g, Leton & Stentiford, 1990; Fernandes & Sartaj, 1997; Veeken et al., 2002), an LCA of olive-mill waste is still missing in the literature (Bong et al., 2016). Therefore, in this thesis a "cradle-to-grave" LCA is performed for such experimental compost per ISO 14040 (2006) and ISO 14044 (2006) using Umberto NTX 7.1 software[5] for modeling purposes. References (secondary data) are databases (i.e., Ecoinvent, 2013) and case-specific literature. The functional unit is 1 t of ready-to-use material (peat or compost). The impacts are therefore compared here on a 1:1 weight basis.

Static-pile composting has been compared on the basis of experiments (primary data) conducted during a research project[6], using olive-mill by-products derived from a three-phase olive oil extraction system. To predict the nutritional and suppressive potential (against diseases) of olive-mill waste compost, its physical-chemical and biological characteristics were investigated, along with its performance as a peat substitute in horticulture.

To present the outcomes of the LCA, the composition of the initial mixture is given in Table 3.3 and the main chemical characteristics of peat and the initial mixture of compost are shown in Table 3.4.

Here, total carbon and total nitrogen are determined via a carbon, hydrogen, nitrogen, sulfur analyzer (CHNS) (Macro Cube, Elementar, Germany); moisture content is determined by drying the samples at 105°C for 24 h; and ash content was determined as loss on ignition using a muffle furnace at 650°C for 24 h.

The system boundaries of the life cycles of peat and compost—with relative inputs, outputs, avoided impacts, and processes–are shown in Figure 3.7a and Figure 3.7b.

[5] Website of Umberto software: `https://www.ifu.com/en/umberto`.

[6] The research project has been funded by Agricultural Agency, Tuscany Region, Italy, Framework "Multi-call for integrated supply chain projects (in Italian: *Progetti Integrati di Filiera*, PIF), Ordinance n. 604/2011. Reg. CE 1698/05-PSR 2007/2013", Project: "Making new amendments from olive mill waste as peat substitute for potted plant production in nurseries", Acronym: SANS-OIL.

Table 3.3: Composition of initial mixture, before composting in static-pile

	Composition [%, volume]
Olive pomace	35.2
Olive-mill wastewater	40.9
Leaves	2.9
Straw	10.5
Waste wool	10.5

Table 3.4: Composition of initial mixture, before composting in static-pile

	Moisture content [%]	**Ash content** [%, dry matter]	**Total carbon** [%, dry matter]	**Total nitrogen** [%, dry matter]
Peat	59.0	1.9	47.7	1.1
Compost	65.0	5.7	47.2	3.5

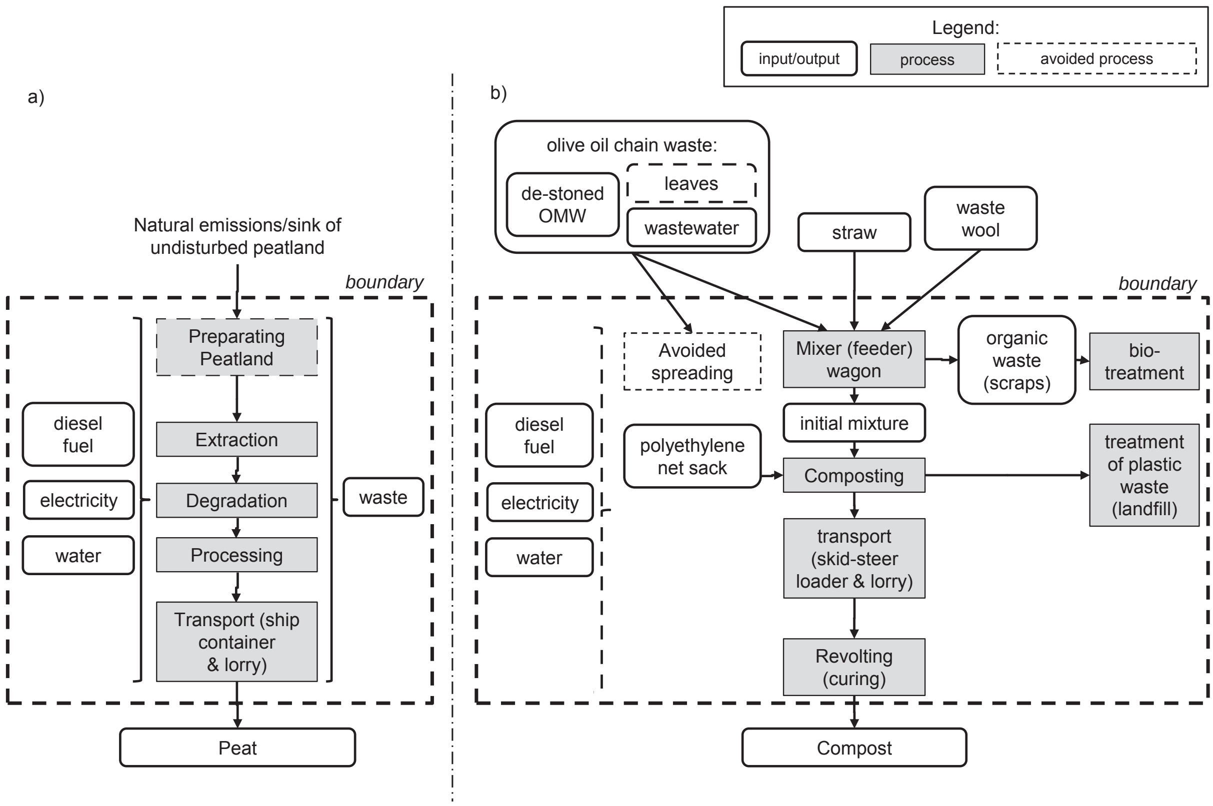

Figure 3.7: Peat and compost system boundaries

As suggested in Figure 3.2, the following sub-phases of the life cycle have been included (see Table 3.5): upstream, processing, transport, and downstream.

Table 3.5: Life Cycle sub-phases of peat and compost

Sub-phase	**Peat**	**Compost**
Upstream	Preparing peatland	Manufacturing of gas-permeable bags; *Avoided emissions: spreading on land, reuse of olive pits*
Processing	Stock-pile, processing (screening and bagging)	Skid-steer loader, mixer wagon, fugitive GHGs, forklift
Transport	Truck and train	Truck
Downstream	Decomposition in situ	Organic waste to mechanical biological treatment, bags to landfill; *Avoided emissions: carbon storage, 3-times bags reuse, reduced fertilizers*

The upstream phase includes peatland preparation before excavating and potential avoided GHG emissions before processing the composts. The avoided emissions are related to the olive-mill wastes that would otherwise be spread on land. The processing phase includes the operations needed to obtain the product (fuel and electricity consumptions as inputs) and fugitive GHG emissions from decomposing organic matter (see EPA, 2011). Fugitive emissions relevant for the LCA are the released GHGs CH_4, N_2O, and NH_3. The transport phase includes the shipment of products from the extraction/production site to the plant nursery, where they are used to prepare the growth media for plant cultivation. The downstream phase includes the emissions avoided by substituting peat with compost ('carbon sequestration potential'). These assumptions are drawn from literature that highlights CO_2-savings for compost.

Other potential reductions in GHG emissions as further improvements (avoided emissions) in the life cycle of compost are discussed. These include the hypothetical re-use of olive pits for energy production in the upstream phase and the reduction of fertilizer use due to the substitution for peat in the downstream phase.

Excluded from the LCA are the following:

- natural peatland emissions, due to the extreme variability in the results of LCAs that include such emissions (Couwenberg, 2011);

- fugitive emissions of biogenic CO_2 from compost, due to its carbon neutrality in the LCA (Brandão & Levasseur, 2011);
- construction of processing facilities and the effect of decreased soil erosion (outside the scope of this study);
- fugitive GHGs other than CH_4, fossil-CO_2, and N_2O (not decisive for LCAs studies on compost, see Favoino & Hogg, 2008);
- the preparation phase of the growth media inside the plant nursery, since all products (peat or compost) use the same operations.

This approach follows procedures undertaken in similar studies on nurseries (see, e.g., Koeser et al., 2014).

The life cycle of peat (Figure 3.7a) is investigated by taking into account the following stages: peatland preparation, extraction, degradation in situ during storage in stock-pile, processing, and transportation to the plant nursery. The life cycle of static-pile composting is shown in Figure 3.7b. Again, some assumptions are necessary: The starting mixture is prepared with a feeder-mixer wagon (Unifeed, Seko, Italy). It is immediately packaged in gas-permeable plastic bags (composting units) and stored outdoors, protected from rain, for about three months. Following the naturally occurring composting phase in bags, there is a maturation phase (called "curing" phase). Photos that illustrate the procedure for composting olive-mill waste are shown in Figure 3.8.

3.4.2 Life cycle inventory for GHG emissions

GHG emissions are here reported for peat and olive-mill waste compost. Calculations required for the determination of impacts are reported, as well as assumptions.

GHG emissions caused by peat

GHG emissions of peat are caused by harvesting, processing, decomposition, and transportation. Harvesting emissions for peat are accounted as average values between the impacts calculated by Cleary et al. (2005) and Boldrin et al. (2010). The first study estimated peat emissions on 124 km^2 of peatland. Assuming an annual peat yield of 10.4 t per km^2 of peatland with a vacuum-harvesting technique (Waddington et al., 2009), and taking 1152 t CO_2 per year per km^2 for peatland exploitation and 295 t CO_2 per year per km^2 for peat extraction and processing, Cleary et al. (2005) calculated a value of 144.7 kg CO_2 per ton of peat. Boldrin et al.

Figure 3.8: Processing of olive-mill waste compost

(2010), on the other hand, referred to the impacts of this phase as 105.4 kg CO_2 per ton of peat. Since an estimation of peat emissions—in particular related to the harvesting phase—is difficult to give due to the variability of many parameters (Courchesne et al., 2010; Höglund & Martinsson, 2013), an average value of 110.31 kg CO_2 per ton of peat is taken into account in this thesis.

Cleary et al. (2005) subdivided the emissions by phase: 52.1% for peatland preparation, which includes heavy machinery operations, such as clearing vegetation and draining bogs; 13.3% for extraction and processing (28.00 t CO_2 per ton of peat is taken for processing, retrieved from Cleary et al., 2005), mainly due to diesel fuel consumption; and 34.6% for transportation—assuming an average distance of 250 km by truck and 250 km by ship. In this thesis, peat used in Italy for horticulture is assumed to be extracted in Sweden, packed in 700–1000 kg bales, shipped by truck for 50 km, by train for about 2200 km, and then by truck again for 20 km. This input data is used to model the impacts of peat for the transport sub-phase.

In this thesis, it is estimated that decomposition of peat extracted from undisturbed peatland release fossil CO_2 emissions. The maximum theoretical quantity of fossil CO_2 (in kg) that can be emitted by peat was calculated by considering its total carbon content, a time horizon of 100 years, and a prefixed decay rate of the organic matter. The decomposition of peat was therefore estimated with a first-order exponential function, following general assumptions for the decay rate of organic matter according to Cleary (2004), with a 5% annual decay rate. This average value is in accordance with other studies. Murayama et al. (1990), for instance, estimated the decomposition rate of peat between 3.7% and 11% with long-term experimental trial (around 1 year). The theoretical quantity of CO_2 was calculated adopting the Equation 3.6 for peat:

$$CO_2 = M \cdot C \cdot \frac{44}{12} \cdot d \tag{3.8}$$

where M is the total dry solids, C is the total organic carbon fraction (dry basis), 44 is the molecular weight of CO_2, 12 is the atomic weight of carbon, and d (%) is the decay over 100 years. The CO_2 from peat is calculated using its carbon content (47.73%, Table 3.4) and moisture (30%, assumed), which yields 525.03 kg CO_2 emitted per ton of material.

Olive-mill waste compost

The GHG emissions of compost are calculated following the experimental results on the preparation of olive-mill waste compost, as follows. Olive mill pomace and bulking agents are transported 10 km to the mill facility, where they are collected and transported via skid-steer loader, and then blended in a mixing feeder wagon commonly used for agricultural operations to obtain a homogeneous mixture. Olive mill wastewater is pumped into the wagon with a 200 W centrifugal pump for about 30 minutes per batch. The initial fresh mixture was composed of 1160 dm^3 of olive-mill wastewater, 1000 kg of olive pomace, 80 kg of olive leaves and twigs, 300 kg of straw, and 300 kg of waste wool, thus forming a 2.84 t batch with a volume of 4.5 m^3 and 60–65% moisture content. The mixing capacity of the feeder wagon was 4 t of material per hour. It is assumed, that the mixing wagon electricity consumptions to be equivalent to those of a hydraulic digger (5–10% deviation is reported from real fuel consumption data). All the processes are complied and reported in Table 3.6.

The mixture unloaded from the wagon was placed via skid-steer loader into 1 m^3 gas permeable bags, filling them to 550–600 kg weight, which yields a bulk density of 0.5–0.6 t/m^3. The production and extrusion of gas permeable bags made of PP is included in this thesis. The desired static composting process was assured by adequate aerobic conditions even in the heart of the bags, as reported in previous trials (Altieri et al., 2011). Each batch resulted in about 5 filled bags (composting unit). It is assumed 0.5% wastage after the mixing procedure and another

Table 3.6: Life Cycle sub-phases of compost

Sub-phase process	Activity description	Database entry	Unit	Amount
1. Mixing wagon	Loading the mixer wagon (1 worker)	Excavation, skid-steer loader	m^3	1.62
	Pumping olive-mill wastewater (200 W pump for 30 min)	Electricity, low voltage, production IT, at grid	kWh	0.05
	Operating mixer wagon (Vol = 20 m^3)	Excavation, hydraulic digger	m^3	1.62
2. Composting	Unloading the mixer wagon and filling large bags (2 workers)	Excavation, skid-steer loader	m^3	1.62
	Manufacturing of bag (PP raffia)	PP, granulate, at plant and Extrusion, plastic film	kg	1.80
3. Transport	Transporting (truck, 20km)	Transport, Truck 16–32, Euro 3	t·km	30.46
	Forklift for loading large bags into the truck	Excavation, skid-steer loader	m^3	2.54
4. Curing	Emptying large bags from the truck; turning the mixture over 3 times	Excavation, skid-steer loader	m^3	4.5
	Irrigating compost in curing phase	Tap water, at user	dm^3	58.0
5. End-of-Life	Disposal of bags (PP)	Treatment of PP, sanitary landfill	kg	1.80
	Treatment of residual waste from mixing and unloading	Treatment of residual waste, composting	kg	29.97
Avoided emissions (credits)	Wastewater loading and spreading	Solid manure loading, spreading by hydraulic machine	kg	395.64
	Wastewater transport with tractor	Transport, tractor and trailer, agricultural	t·km	15.83
	De-stoned olive-mill waste spreading (Tractor 80 kW, manure spreader)	Solid manure loading, spreading by hydraulic machine	kg	998.70
	De-stoned olive-mill waste transport with tractor	Transport, tractor and trailer, agricultural	t·km	39.95
	Reuse of bag made of PP raffia (3 times)	Polypropylene, granulate, at plant and Extrusion, plastic film	kg	20.00

0.5% after filling the bags. The wastage residues were assumed to be fed to the local mechanical-biological treatment facility. Figure 3.9 depicts a typical static-pile composting mass balance for one ton initial mixture.

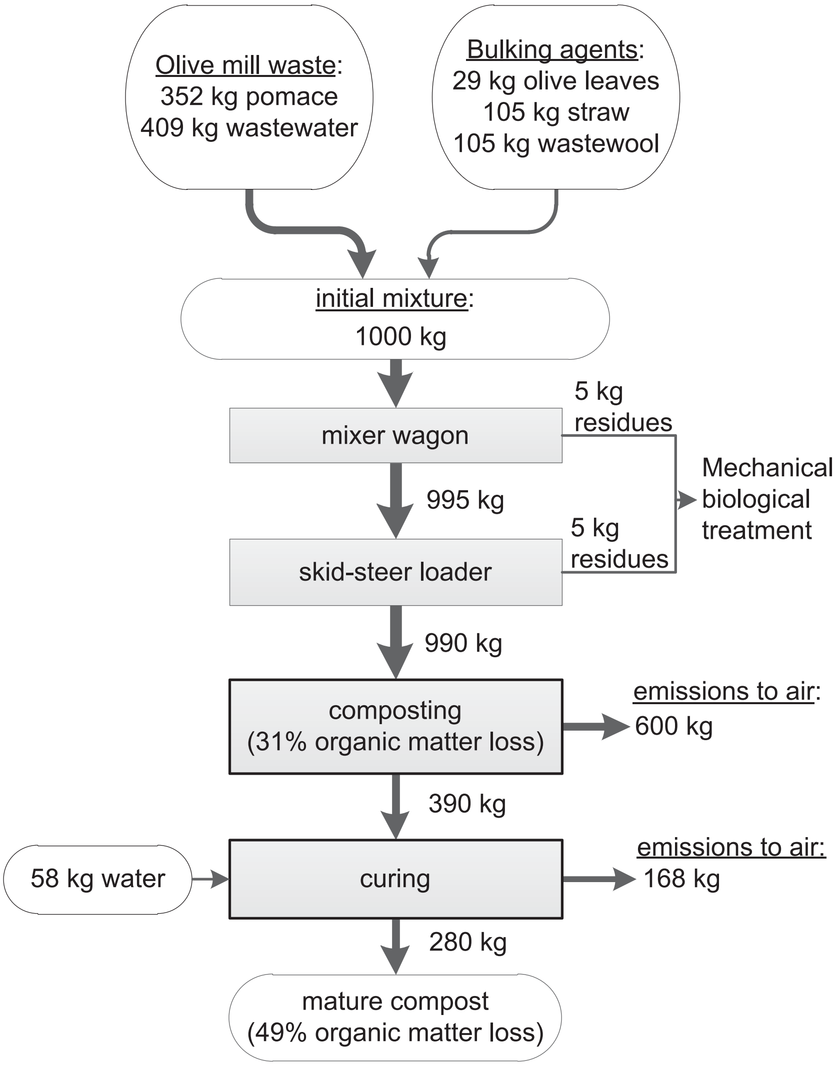

Figure 3.9: Mass balance of olive-mill composting in static-pile

During static storage, each bag started a typical "thermal" composting phase as soon as it was filled, without needing forced aeration and/or mechanical turning—therefore, no fuel or electricity was consumed. During composting, the temperature of the mass was measured to monitor the static-pile method, to assess the rate of degradation of organic matter, and to determine where an inhibition of microbial activity might occur (Liang et al., 2003; Miyatake & Iwabuchi, 2006). The latter is a risk when the temperature in the composting reaction exceeds 80°C, as reported in Leton & Stentiford (1990). Monitoring the temperature was therefore necessary to determine whether the size of the composting unit was suitable for the static-pile method. To detect the temperature of the mass, and following the approach described on Section 3.3 (i.e., subsection on page 41 about the detection in situ of fugitive emissions), probes (platinum resistance thermometer, Pt100) have been placed at the centers of the bags at about 40 cm from the ground—assuming that this sampling point was not influenced by external air. Another probe (Pt100) was placed outdoors to measure the ambient temperature. A multi-gas detector (MX6 iBrid, Industrial Scientific) was used to detect CO_2 (infrared sensor), volatile organic compounds (VOC) including methane (10.6 eV photoionization), and NH_3 (electrochemical sensor). Gas was sampled by means of plastic tubes inserted into the center of the mass. The limits of detection for CO_2, VOC, and NH_3 were 0.01% vol., 0.1 ppmv, and 1 ppmv, respectively. The O_2 concentration inside the mass was measured with an electrochemical sensor (range 0–30% vol.). These gas parameters were measured once per day during the three-months composting period; temperatures were collected every 15 minutes via data logger. (The composting period was chosen until a steady-state of degradation—e.g., inner temperature of the mass is equal to the room temperature—was reached.)

All non-biogenic GHG emissions investigated were below the detection limits; similar results were found on home static composting by Colón et al. (2010). Here, the lack of GHG emissions seems related to the fully aerobic conditions established by the small composting units used in their study (0.4 m^3), which are similar in size to the ones used in this thesis (1 m^3). Since no VOCs were detected and oxygen was continuously detected at over 12% vol., even during the thermal phase, it is assumed that the entire composting process was carried out under appropriately aerobic conditions. This indirectly confirms the observations of other authors, who report GHGs emissions only when anaerobic processes are activated (Haug, 1993). (However, these results look quite different from those reported by He (2000)—who worked on the organic fraction of municipal waste in a lab-scale composting facility (0.016 m^3)—and from those reported by Nasini et al. (2016)—who examined olive-mill waste under static-composting composting with a larger composting unit (26 m^3).)

Given the uncertainty of the results reported in the literature, and given that the analysis of this thesis reported GHG emissions below the detection limits, it is assumed half the values of those limits for CH_4 and NH_3—as a precautionary measure—to evaluate an average scenario. Levels of N_2O fugitive emissions from static-pile compost were adopted from the work of Colón et al. (2010) on home composting, where a similar composting volume, temperature profile (maximum temperature detected 65°C, linked to heat loss), assumptions on the detection limits, and absence of leachate prevailed.

After static-pile composting, a forklift loaded the bags into a 16-ton truck (Euro 3), which then drove to the plant nursery. One truck could transport 22 bags and 20 km was the distance to the final destination. The bags were unloaded, and the compost, being still immature, was stacked to allow the curing phase to proceed. This was carried out until the appropriate compost maturity was reached and the mass temperature matched that of the environment. The initial mixture of one ton (65.0% moisture, wet basis) yielded about 280 kg of mature compost (33.0% moisture, wet basis, see Figure 3.9).

Avoiding the use of peat resulted in CO_2-savings, calculated as follows. In the upstream phase, the starting processes were modeled to account for Italian Law n. 574 (1996), which limits the annual direct spreading of raw olive-mill waste on the soil to 50 m^3 per hectare, in order to avoid pollution of the groundwater. This limitation has been reported also in the Italian Legislative Decree n. 152 (1999), which is a transposition of the European Directives 91/271/CEE and 91/676/CEE. The calculation was done for wastewater and pomace separately, since consumption due to the operations are assumed different. For the spreading of 20 m^3 of wastewater, it is assumed 1.53 kg diesel consumption per hectare with a 65 kW tractor (typical for the Tuscany Region of Italy) used in combination with a centrifugal spreader (assumed from Recchia et al., 2011, p.63). Therefore, 0.076 kg diesel consumption is estimated per m^3 of wastewater. To spread 395.64 kg wastewater (see Table 3.6), whose average density is assumed 1.1 t/m^3 (Bouknana et al., 2014), 0.027 kg diesel consumption is compiled. For the spreading of 30 m^3 of de-stoned olive-mill waste, it is assumed 6.55 kg diesel consumption per hectare with a 80 kW tractor and manure spreader (Recchia et al., 2011, p.63). Thus, 0.218 kg diesel consumption is estimated per m^3 of de-stoned olive-mill waste. To spread 998.7 kg de-stoned olive-mill waste (see Table 3.6), whose average density is assumed 1.45 t/m^3 (Rizzi et al., 2017), 0.150 kg diesel consumption is compiled. The transportation distance of both wastes between the olive-mill facility and the field is assumed 20 km. The input data for the calculations are given in Table 3.7.

In the downstream phase, the re-use of plastic bags for three seasons and the avoided impacts due to the carbon storage effect are assumed. The PP bags were assumed to be re-used and then disposed in a sanitary landfill. In modeling the carbon storage effect, the potential carbon that was bound for the time horizon of 100 years, including a first-order exponential decay function, has been taken into account. In the many long-term field experiments that have been conducted, values between 2% (minimal content, e.g., for highly putrescible feedstock) and 10% (maximal content, e.g., for feedstock with high lignin content) have been estimated as the percentage of carbon bound relative to the initial compost carbon content (Smith et al., 2001). Consequently, it is assumed an average of 6% carbon bound. This assumption was based on literature analysis and was used because emissions of compost present high variability, and it is very difficult to estimate the carbon storage effect via experimental analysis.

3.4.3 Life cycle impact assessment

In this LCIA, the environmental impact indicators GWP, CED, AP, and EP are considered for peat and compost (see Section 3.2.3).

Table 3.7: Avoided spreading on land of wastewater and de-stoned olive-mill waste

	Unit	**Wastewater**	**De-stoned olive-mill waste**	**Reference**
Diesel consumption for spreading	[kg/ha]	1.53	6.55	Recchia et al. (2011)
Tractor power	[kW]	65	80	Recchia et al. (2011)
Limit per legislation	[m^3/ha/year]	20	30	Italian Law n. 574 (1996)
Diesel consumption	[kg/m^3]	0.076	0.218	
Input material	[t]	395.64	998.70	Table 3.6
Density	[t/m^3]	1.1	1.54	Bouknana et al. (2014); Rizzi et al. (2017)
Total diesel consumption	[kg/input]	0.027	1.50	

By using the characterization factors reported in Table 3.2 (on page 35) and the GHG emissions caused by peat (calculated in the previous section, on page 50), the impacts of one ton of *peat* in terms of GWP_{100} are as follows: 110.3 kg CO_2-eq for upstream, 28.0 kg CO_2-eq for processing, 525.0 kg CO_2-eq for decomposition in situ (downstream), and 154.8 kg CO_2-eq for transportation. The latter is modeled with Umberto NTX software by using the inputs reported for peat transport (see page 50) with freight lorry (> 32 ton, Euro 5), train, and truck (16–32 ton, Euro 3), and Ecoinvent (2013) is used as reference. This data (i.e., upstream, processing, transport, and downstream impacts) sum up to 818.2 kg CO_2-eq per ton of peat.

The impacts of *compost* are shown through Sankey diagram—which illustrates the quantity of flows (kg CO_2-eq) as proportional to the width of arrows. It describes the fluxes of the model (Figure 3.10). The contribution to the GWP_{100} of the upstream phase of compost was calculated as follows. Since 1 liter of diesel emits 3.908 kg CO_2-eq (Maciel et al., 2015) and by using its density (density of diesel assumed 0.832 kg/liter), the spreading on land of wastewater and de-stoned olive-mill waste via tractor is calculated as 0.129 kg CO_2-eq and 0.705 kg CO_2-eq, respectively. By adding up these emissions with the transport emissions between the olive-mill facility and the land, a sum -12.484 kg CO_2-eq/t is yielded as environmental credits for the upstream phase. The processing phase, which does not take into account biogenic CO_2

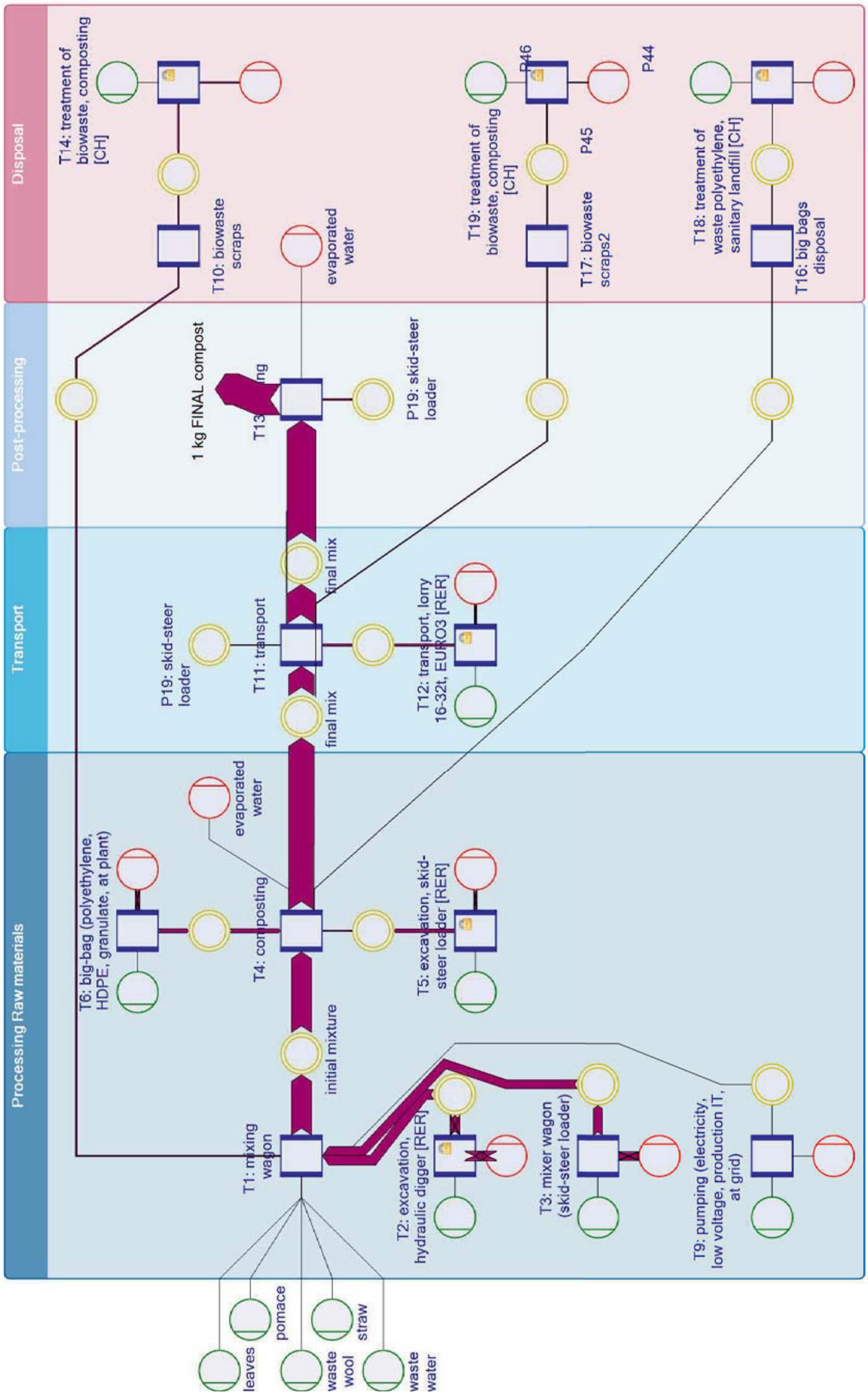

Figure 3.10: Sankey diagram of olive-mill waste compost (LCA elaborated with Umberto NTX 7.1 software; RER = activities located in Europe)

contribution to the GWP_{100}, contributes to the GWP_{100} 20.7 kg CO_2-eq/t, while the fugitive emissions contribute 66.0 kg CO_2-eq/t to the GWP_{100}.

Comparing the emissions of peat and compost, Figure 3.11 shows the results: peat contributes 818.2 kg CO_2-eq/t to the GWP_{100}, while static-pile composting accounts for 44.7 kg CO_2-eq/t. The upstream impacts of peat are due to fuel and energy consumption, with a contribution of 110.3 kg CO_2-eq/t; CO_2 savings due to the avoided spreading of raw olive-mill olive waste account for 12.5 kg CO_2-eq/t. Fuel and electricity consumption have a minor impact on the processing phase, where it contributes only 2-3% of the total emissions. During the transport phase, however, this factor ranges between 19% for the peat—due to the long transport distance—and 4% for the compost, due to their shorter delivery paths.

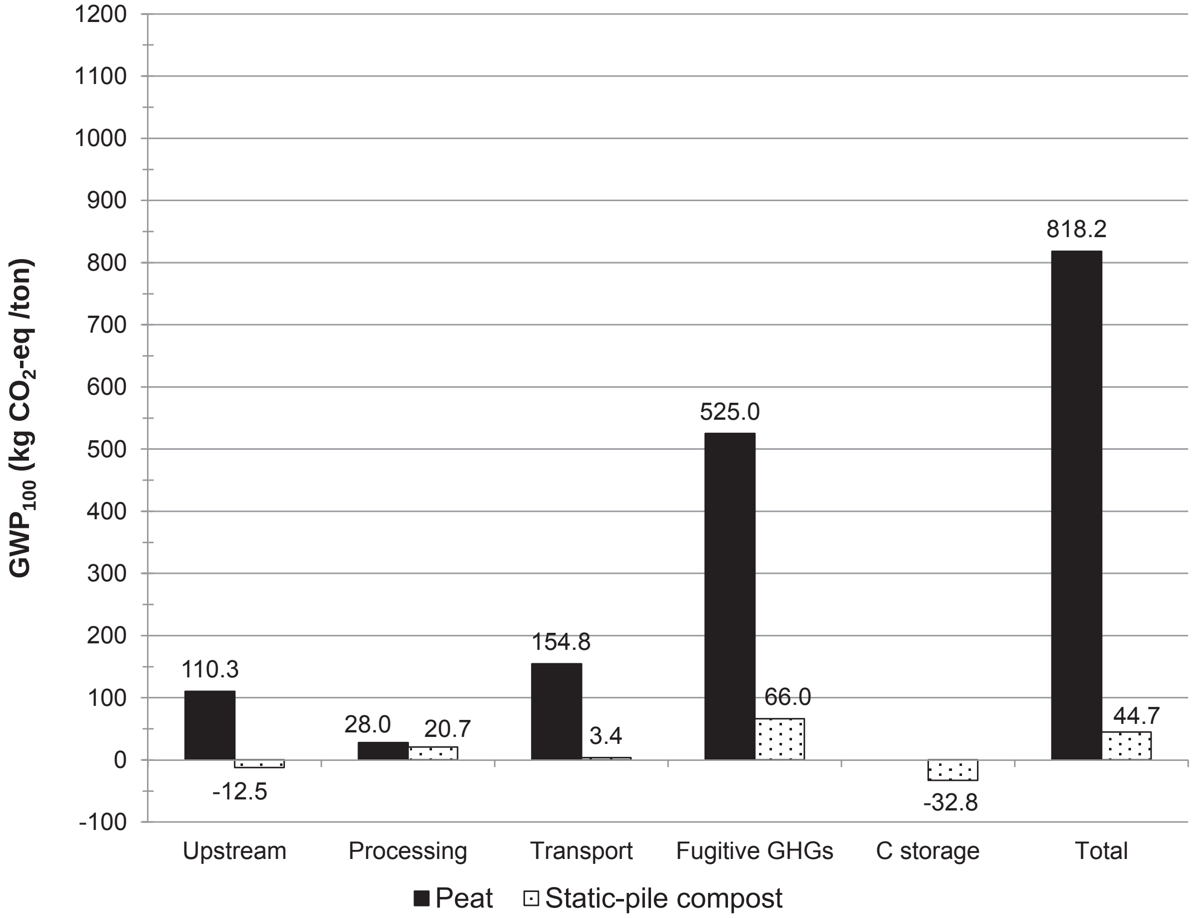

Figure 3.11: Global warming potential of peat and olive-mill waste compost

The factor most relevant to the overall impacts is the fugitive GHG emission: here, the peat value is 525.0 kg CO_2-eq/t and the compost value is 66.0 kg CO_2-eq/t (13% of the peat value). This is due to the assumptions made about the fugitive emissions of the excavated peat (fossil origin) and the composting (biogenic). Peat harvesting, indeed, releases CH_4 accumulated within the peatland, increases the natural decomposition rate of organic matter, and promotes fossil CO_2

emissions into the atmosphere. On the other hand, emissions due to compost decomposition were considered neutral to the LCA, while other GHGs cannot be neglected and were therefore included in this thesis. By looking at the total GHG emissions, the difference between peat and composting is accrued due to the CO_2-savings from carbon storage for composts: the theoretical emissions accounted for 32.8 kg CO_2-eq per ton of mature compost.

These results indicate that the upstream and the C-storage contribute significantly to the impacts of compost, whereas the processing phase exhibits negligible differences (28 kg CO_2-eq for the peat versus 20.7 kg CO_2-eq for the compost). By excluding the CO_2-savings, the fugitive emissions of peat and compost contribute 64% and 73% to the GWP_{100}, respectively. This result is confirmed by Saer et al. (2013), a study based on a literature review of compost emissions, where average fugitive emissions were calculated (74.0% contribution to the GWP).

To evaluate the effects of different impact indicators on composting in gas-permeable bags, a contribution analysis—expressed in percentage and adopted from the method proposed by Clavreul et al. (2012)— is carried out on the four most relevant impact indicators: CED, GWP_{100}, AP, and EP. The results are shown in Figure 3.12. Here, the processing phase contributes between 34% and 93% of the overall impacts, where the maximum represents the CED indicator. Transport emissions contribute 3% to 6%, and disposal operations, 1% to 8%, considering bag disposal and the biological treatment of organic residues.

The transport phase contributes only slightly to the overall impacts, whereas C-storage appreciably decreases the GWP_{100} (30%). Avoiding the spreading of wastewater also significantly reduces the overall impacts: between 13% (GWP_{100}) and 35% (AP). Including multiple indicators in the contribution analysis results in a more accurate view of the problem, especially for organic based materials.

As seen in Figure 3.11, composting (44.7 kg CO_2-eq) has much lower environmental impacts than peat (818.2 kg CO_2-eq). This is due to low fugitive GHGs, avoided emissions for carbon storage, and the avoidance of wastewater spreading. Comparable results on home composting (46.3 kg CO_2-eq/t) were reported by Colón et al. (2010) for raw fruits and vegetables. The assumptions made in this thesis, however, are specific to olive-mill waste and cannot be applied generally. Other studies have reported higher values of GWP for windrow composting of garden waste (about 130 kg CO_2-eq per ton of compost) or green waste (about 210 kg CO_2-eq per ton of compost), for example, as reported by Boldrin et al. (2010), who took into account the leachate effect and offered no CO_2 credits for avoided wastewater spreading.

The fugitive emissions considered in this thesis are CH_4 (GWP_{100}), N_2O (GWP_{100}), and NH_3 (AP, EP). In this thesis, CH_4 and NH_3 emissions are measured during the static-pile composting phase with an accuracy of 10 ppmv and 1 ppmv, respectively. Some studies have reported,

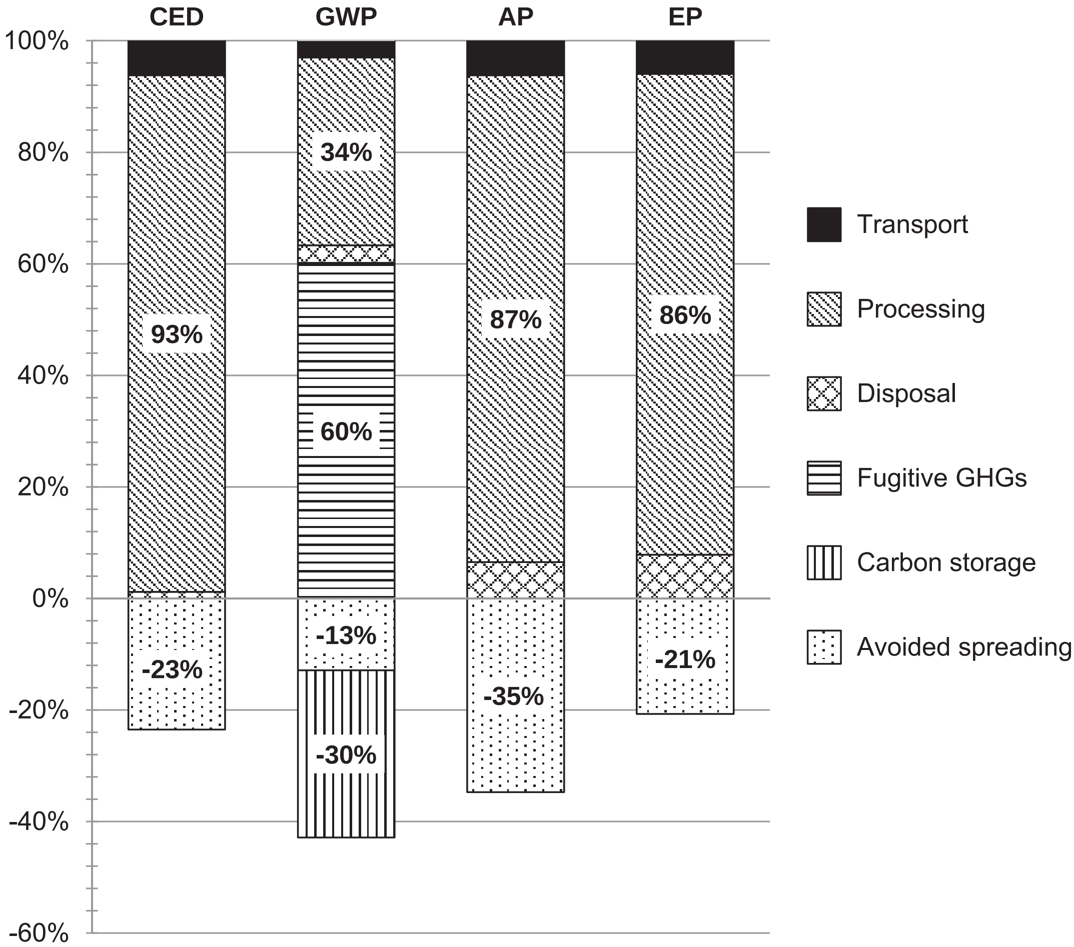

Figure 3.12: Contribution analysis for static-pile composting of olive-mill waste. Environmental indicators: cumulative energy demand (CED), global warming potential (GWP), acidification potential (AP), and eutrophication potential (EP)

however, that these accuracy levels may not be sufficient for small composting units (Amlinger et al., 2008) and that N_2O emissions may have a great impact on the overall results (Andersen et al., 2010a). In particular, Andersen et al. (2010a) evaluated CH_4 and N_2O for the home composting of organic household waste with a high accuracy photo acoustic gas monitor, using a composting unit of $0.3\,m^3$ and a static flux chamber to detect such emissions. Their analysis revealed that the GWP of home composting was indeed close to that of industrial composting. An example of static-pile composting of olive-mill waste is also reported in Nasini et al. (2016), where VOC and NOx were detected. However, these results are specific to particular composting conditions and materials, making comparisons difficult. Moreover, accurate NH_3 measurements can also contribute to the overall impacts, as reported for example by Cadena et al. (2009). They adopted a novel NH_3 detection method that uses a matrix of sampling points and applied it to an mechanical-biological treatment facility that composts the organic fraction of municipal solid waste. The authors detected 0.35 kg SO_2-eq per ton of compost produced (1 kg NH_3 corresponds to 1.88 kg SO_2-eq according to HAUSCHILD & WENZEL 1998), twice as much as estimated in this thesis for static-composting (0.17 kg SO_2-eq per ton of final compost).

Although fugitive GHGs may have been underestimated (as other researchers have also done), mitigation impacts included in this thesis are also due to the use of bulking agents and the fact that the static-piles were not turned mechanically. These two aspects may promote more homogeneous (i.e., passive) ventilation of the mass and a consequent reduction in CH_4 emissions (BONG ET AL., 2016).

3.4.4 Interpretation

Further CO_2-savings might be achieved by considering: the reuse of olive pits for energetic purposes, the reduction of fertilizer needs (due to the substitution of peat), the reduction of watering and pesticides.

When *reusing the olive pits* after oil extraction (for energy production), a credit of 1.327 kg CO_2-eq per kg of olive oil produced in Tuscany can be compiled (according to RECCHIA ET AL., 2011, p.127). This value was calculated by the authors by introducing in the value chain a pit stone separator, by considering a mill working capacity of 200–500 kg of olives per hour. Assuming 15% extraction efficiency of olive oil from the olives processed inside an olive-mill facility (typical for central regions in Italy), 0.09 kg CO_2-eq per kg of olives processed is calculated. By considering the mass balance for static-pile composting indicated in Figure 3.9, from an initial quantity of compost processed (1 t) a reduced quantity of final compost is yielded (280 kg), therefore 0.09 kg CO_2-eq per kg of olives processed corresponds to about 19.20 kg CO_2-eq per ton of final compost. Although this estimate excludes the impacts of field operation and transport, it does include the impacts of extracting the pit from the olive pomace.

The CO_2-savings that accrue from *reducing fertilizer use* by replacing peat with compost in plant nurseries have been calculated previously (see, e.g., HANSEN ET AL., 2006). The results of these calculations with emission and substitution factors for fertilizers are given in Table 3.8, where: the compost nutrient contents are determined from experiments (on wet waste), the emission factors are retrieved as average values from ECOINVENT (2013), and the substitution factors are adopted from HANSEN ET AL. (2006) and estimated during the experiments performed for this thesis.

The effects of these improvements are shown in Figure 3.13, which depicts three scenarios: the basis calculation presented in the previous sections, the reuse of olive pits, and the simultaneous reuse of pits and reduction in fertilizers. The second scenario results in a net

Table 3.8: Fertilizer substitution with olive-mill waste compost (wet waste) in horticulture

	Compost nutrient content [kg/ton wet waste]	**Emission factor** [kg CO_2-eq/kg nutrient]	**Substitution factor** [fraction of applied nutrient]	**Avoided impacts** [kg CO_2-eq/ton wet waste]
Nitrogen (N) total	24.16	8.65	0.2	41.80
Phosphorus (P)	0.21	2.00	0.9	0.37
Potassium (K)	13.96	1.03	1	14.38

plus of 25.5 kg CO_2-eq per ton of final compost, while the third scenario yields a net minus of 32.7 kg CO_2-eq/t. This latter result is due to the avoided emissions associated with fertilizer reduction, which is equal to 56.5 kg CO_2-eq per ton of final compost (sum of the values indicated in the third column of Table 3.8). These results should be viewed with caution, however, due to the uncertainties related to the effective reduction in fertilization.

Another aspect of substituting compost for peat in plant nurseries is the possible CO_2-savings from a reduction in *watering*. This could be related to the compost's good micro-porosity (which may be even higher than that of peat), since this is strictly linked to water retention. However, more investigation with case-specific data is needed to evaluate these potentially avoided emissions. And there is another interesting aspect that was not investigated in this thesis. When olive-mill waste compost is used as peat substitute, it may help suppress plant pathogens—mainly due to the presence of *antagonistic activity* of microorganisms (fungi and bacteria) that hinder diffusion of plant diseases (in phytopathology, antagonistic activity is referred to the action of microorganisms that suppress the activity of a plant pathogen, see Chilosi et al., 2017). Undertaking this analysis would be useful to understand CO_2-savings due to a reduction in *pesticides* and to quantify the correlation (Hansen et al., 2006).

Comparison with open-windrow composting

In order to understand the effective CO_2 reduction potential by substituting peat with compost from static-pile procedure, the LCA is compared with another procedure for composting. Here, a comparison between compost produced in static-pile and open windrows is thus reported, since their evaluation might be used for optimization purposes.

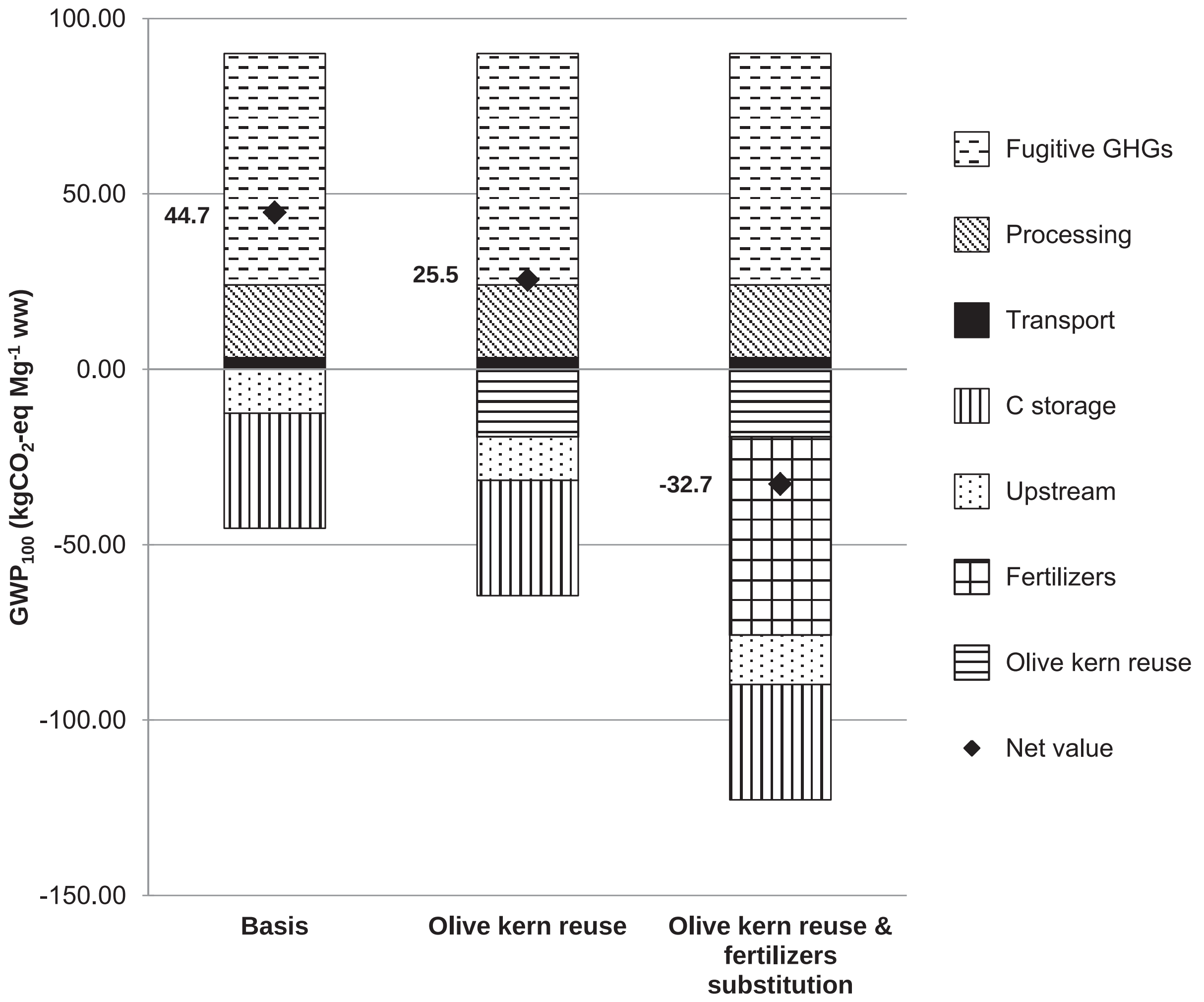

Figure 3.13: Improvements from the basis scenario to the olive pit reuse scenario and the combined reduction of N-P-K fertilizers scenario.
Notes: Ww = wet waste

Open windrow composting is a widely-used procedure undertaken outdoors, in piles up to 3 meters high, several meters wide, and hundreds of meters long. In general, different quantities of GHGs are released with these two procedures into the atmosphere, due to different levels of fuel and electricity consumption and different potentials for emitting CH_4 from anaerobic pockets within the mass (Smith et al., 2001).

The life cycle of open-windrow composting is shown in Figure 3.14. Here, the following assumptions are made: The transportation of bulking agents to the mill facilities is accounted for. The initial mixture is processed with agricultural shredders, loaded into the open-windrow facility, and composted. During the composting phase, the mass is periodically turned mechanically and irrigated, until a suitable maturation is reached. No bio-filters are installed in the open-windrow facility to treat fugitive GHGs. After the thermal composting phase, the mass is matured during the curing phase, which requires further periodic mechanical turning and irrigation. When mature, the compost is transported to the plant nursery to prepare the growth media.

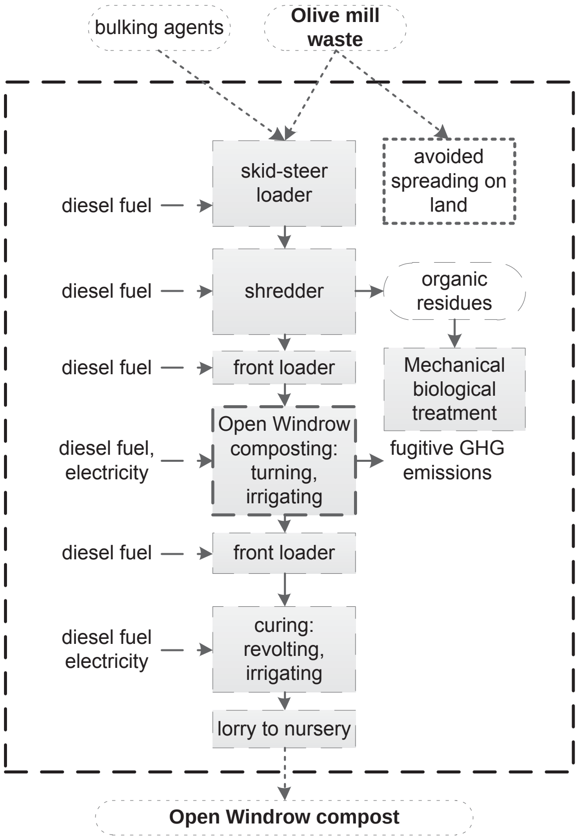

Figure 3.14: Open-windrow composting system boundaries

Open-windrow composting compost emissions are generated from diesel fuel and electricity consumption for machinery operations, i.e., shredders, skid loaders, and turning machinery. When one ton of wet waste mixture is treated in an open-windrow composting facility, consumption of 26.03 kWh_{el} and 1.06 kg diesel fuel can be assumed (average values in Italy, according to a review by Buratti et al., 2015). Since 1 kWh_{el} and 1 kg of diesel fuel emit 0.56 kg CO_2 (Blonk et al., 2010) and 3.17 kg CO_2 (IPCC, 2007), respectively, the total equals 17.94 kg CO_2. This value is close to the 19.28 kg CO_2 average reported in Boldrin et al. (2009). Fugitive emissions in open-windrow composting facilities have been considered in many studies, but there is no consensus on how to estimate them. It is assumed that CH_4 and N_2O emissions are generated during open-windrow composting (Saer et al., 2013), due to undesired, sporadic anaerobic digestion in small pockets of the mass. This occurs when the mass is not properly turned and aerated and the oxygen level drops below 5% (vol./vol.) (Leton & Stentiford, 1990). Here, the average values for CH_4 (3.415 kg CH_4 per ton of wet waste, reported by Boldrin et al. 2009, a literature review conducted for residues and garden waste treated in open-windrow composting facilities) and for N_2O (0.1297 kg N_2O per ton of wet waste, reported in Boldrin et al.

2010) emissions are adopted, whose sum equals 124 kg CO_2-eq by using their characterization factors (see Table 3.2).

The results show that open-windrow composting (101.4 kg CO_2-eq/t) have much lower environmental impacts than peat (818.1 kg CO_2-eq/t) and higher impacts than static-pile composting (44.7 kg CO_2-eq/t). Table 3.9 shows these differences of impacts for peat, open-windrow composting, and static-pile composting.

Table 3.9: Different impacts of peat, open-windrow composting, and static-pile composting

	Unit	**Peat**	**Open-windrow compost**	**Static-pile compost**
Upstream	[kg CO_2-eq]	110.3	-12.5	-12.5
Processing	[kg CO_2-eq]	28.0	17.9	20.7
Transport	[kg CO_2-eq]	154.8	4.7	3.4
Fugitive GHGs	[kg CO_2-eq]	525.0	124.0	66.0
C-storage	[kg CO_2-eq]	0	-32.8	-32.8
Sum	[kg CO_2-eq]	818.2	101.4	44.7

Scenario analysis

Scenario analysis helps to explore the uncertainties related to composting organic by products (see Section 3.3 on page 45). Two scenarios with more pessimistic data than the base case are investigated (scenario S1 and scenario S3), as well as one scenario with maximum CO_2-savings due to carbon storage (scenario S2). Scenario S1 represents the worst case for fugitive emissions during the composting phase. For open-windrow composting, it considers the maximum GHG emissions found in a literature review of 15 studies made by Saer et al. (2013), who listed the CH_4 emissions (0.021–7.0 kg CH_4) and N_2O emissions (0.0003–0.252 kg N_2O) for one ton of mixtures similar to olive-mill waste. For static-pile composting, this worst case scenario takes into account the maximum detection limit of sensors, adopting the approach presented in Colón et al. (2010) for home composting. Scenario S2 explores the maximum C-storage by assuming 10% of carbon bound (Smith et al., 2001) for 100 years. Therefore, by considering the carbon content of the final compost (44.7%) and its moisture (33.4%), it can be calculated the theoretical emissions of CO_2 of the compost (547.4 kg CO_2-eq/t) and its 10% carbon bound into the soil (54.7 kg CO_2-eq/t). Scenario S3 investigates the worst case for transport contribution to the GWP by increasing the average transport distance of the bulking agents (10 km, in basis scenario) to represent lack in Tuscany of waste wool (500 km) or straw (20 km).

The results (Figure 3.15) indicate that fugitive GHGs contribute greatly to the LCA of organic waste (scenario S1). The GWP can vary from 44.7 kg CO_2-eq (basis) to 68.8 kg CO_2-eq (+54%)

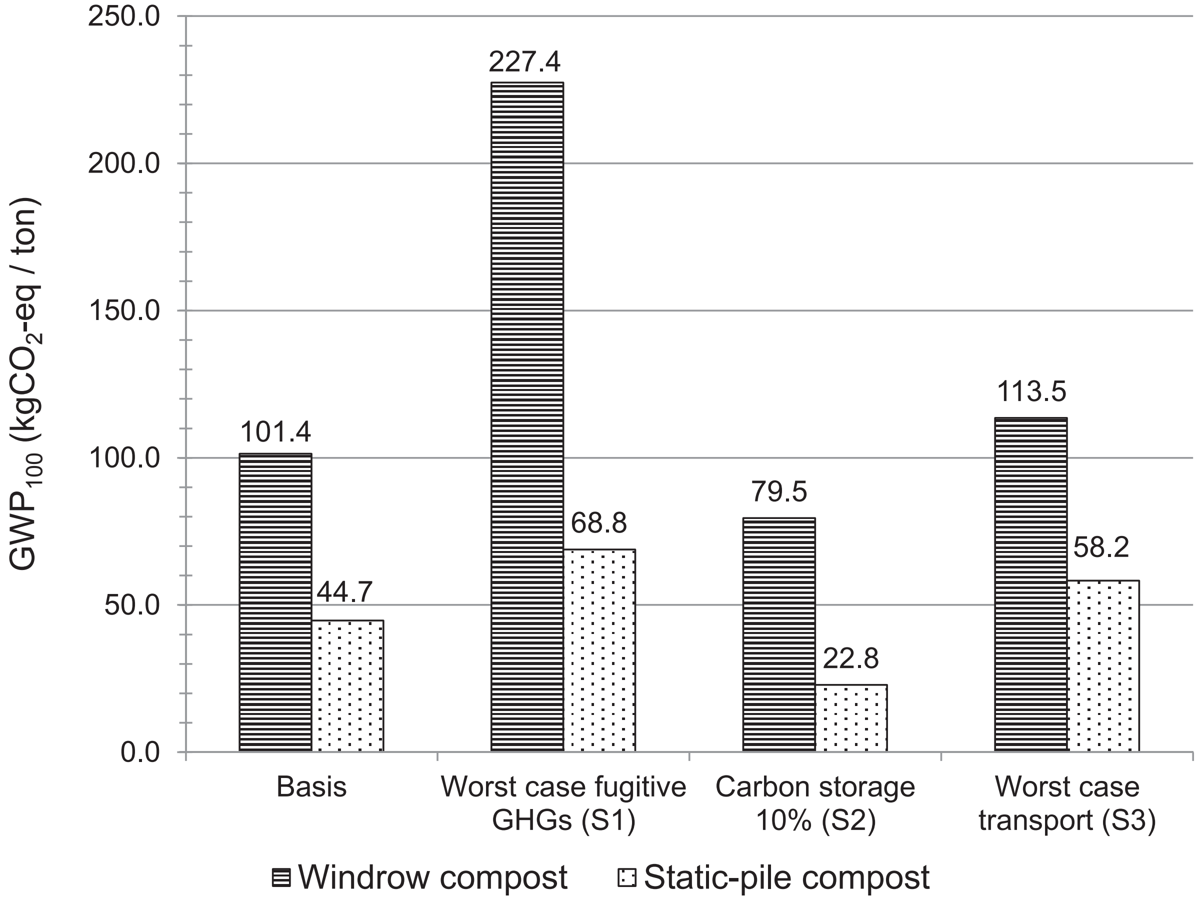

Figure 3.15: Scenario analysis for open-windrow (left bar) and static-pile (right bar) composting, in terms of GWP_{100}, per ton of final compost. S1 explores maximum fugitive emissions (from literature for windrow compost and from maximum detection limit of sensor for static-pile compost); S2 explores maximum carbon storage (from literature analysis); S3 explores maximum transportation distance for bulking agents.

for static-pile compost and from 101.4 kg CO_2-eq to 227.4 kg CO_2-eq (+124%) for open-windrow compost (scenario S1). Moreover, when carbon storage reaches the maximum value found in the literature (scenario S2), the GWP_{100} is reduced from 101.4 kg CO_2-eq to 79.5 kg CO_2-eq (−22%) for open-windrow compost and from 44.7 kg CO_2-eq to 22.8 kg CO_2-eq (−49%) for static-pile compost. Carbon storage should also be considered very uncertain and variable, and appropriate identification of secondary data may increase the accuracy of the outputs. Finally, when input materials must be transported for long distances (scenario S3), the GWP increases from 101.4 kg CO_2-eq to 113.5 kg CO_2-eq (+12%) for open-windrow compost and from 44.7 kg CO_2-eq to 58.2 kg CO_2-eq (+30%) for the static-pile compost. These three alternative scenarios illustrate the degree of potential fluctuation in the LCA results for olive-mill waste compost. However, even the worst of these scenarios (scenario S1 for open-windrow compost) still leads to an 70% reduction in GWP compared to peat.

Conclusions on the LCA of peat and compost

The results of this LCA illustrate that olive-mill waste compost show that a significant reduction of emissions in the horticultural sector can be achieved under favorable circumstances. It should be noticed, however, that the results presented previously are highly sensitive to a variety of factors connected to the waste management of organic materials. In particular, assumptions about the C-storage factor and the accounting of fugitive GHG emissions strongly influence the results, as illustrated in the scenario analysis. On the positive side, the contribution analysis suggests that using olive pits as fuel and reductions in fertilizer use for the cultivation of plants may result in a drastic decrease in the environmental impacts (delta $GWP_{100} = 32.7\,kg\,CO_2$-eq per ton of static-pile compost, see Figure 3.13).

Although this study did include the detection of fugitive GHGs and an analysis of the C-storage due to the substitution of peat as growth media for horticulture, more research is required to understand the influence of these factors on the overall results. Although many long-term trials have indeed been conducted to evaluate the effective C-storage, further work is required to accurately detect the emissions over the time horizon of 100 years, since sizable fluctuations in organic material emissions may influence outcomes. In situ studies would also improve the estimation accuracy of all GHG fugitive emissions in the static-pile composting of olive-mill waste.

Although the LCA proposed in this thesis shows that peat substitutes significantly reduce CO_2 emissions, the actual consequences of using olive-mill waste compost when cultivating plants must also be investigated. More importantly, plant growth performance (e.g., aerial part length, number and length of lateral roots) can be affected by a complete substitution of peat with compost. From a horticulturalist's perspective, the quality of the plants' growing conditions must always have priority. Naturally, the economic consequences and changes in buying and selling prices of the plants must also be taken into account in making substrate decisions. Only when these two fundamental factors are incorporated into the analysis can sound recommendations be offered for the practical usage of compost in horticulture. For these reasons, Chapter 4 describes the economic estimation of costs for the substitution of peat with compost and takes into account agronomic quality of a potted plant.

3.5 LCA of plastic pot and bio-pots

In this thesis, plastic pots made of PP and bio-pots made of PLA are taken into account for a horticultural supply chain (see Table 3.1). Environmental technical parameters of the decision problem of these products are here reported.

LCAs of several plastics have been proposed, but the analysis of their agricultural use is still limited. Generalizing, inputs and outputs of bio-based plastics, compiled in the inventory analysis, present indeed some issues (Niaounakis, 2013, p.278):

- the compilation of input and output data in LCA is time-consuming, and therefore economically prohibitive for many companies;
- a universally accepted method to calculate the impacts for bio-based materials is missing;
- case-specific outcomes of specific bio-plastics cannot be generalized;
- the environmental indicator "land use change" is excluded from studies, because it is difficult to generalize;
- current availability, consistency, and stability of raw materials are not taken into account;
- a single, specific product is under investigation; therefore, the overall impacts of systems are not taken into account;
- LCA performed "today" might be not valid in the future, because of preliminary R&D data, which might be better in future after process optimization (this aspect might discourage performing LCAs towards decision-making).

The most relevant LCAs on plastics used in agriculture were reviewed by Razza & Cerutti (2017) and here reported. Weiss et al. (2012) elaborated a meta-analysis to survey 44 LCAs on bio-based materials, by concluding that a cubic meter of bio-based material can save 3 ± 1 t CO_2-eq emissions. The author suggested that drawing general conclusions is almost impossible due to the variability of many parameters, and that uncertainties are generally high on life cycle inventory data compilation (which are affected by 40% error) and on allocation procedures (up to 90% error). Such uncertainties are related to a different way of compiling inputs, outputs, and environmental impacts when treating agricultural residues, farming practices, temporary carbon storage, and end-of-life waste. (The authors suggest, for instance, that the use of PLA can decrease 0.1–3.8 t CO_2-eq the environmental emissions of one ton of petroleum based equivalent material.)

Hottle et al. (2013) reviewed LCAs of bio-based polymers, and their results on PLA, polyhydroxyalkanoate (PHA), and thermoplastic starch (TPS) showed similar environmental impacts compared to petroleum-based plastic in terms of GWP. They stated that the environmental impacts of bio-based polymers remain unclear due to the large variety of products, uses, and disposal. The authors stressed the fact that a universal judgment is almost impossible, and that analysis is always case-specific. Although most bio-based materials are claimed as compostable, the authors stated that it is unclear how end consumers are going to dispose these products. Thus, a detailed analysis of end-of-life scenario is in most cases needed.

Yates & Barlow (2013) made a review of LCA of bio-based polymers and estimated that PP has GWP (per kg of polymer) equal to 2.0 kg CO_2-eq (PlasticEurope, 2008), while GWP of PLA ranges between 0.5 kg CO_2-eq (PLA pellets, see Groot & Borén, 2010) and −4.0 kg CO_2-eq (PLA thermoform boxes). Different end-of-life options can be taken into account in the analysis, and their choice can influence the results, the authors suggest. Again in this study, no general consideration can be drawn, since variability of such parameters highly influences the outcomes. The only general question that arises is whether these fluctuations make the environmental benefits of biodegradation questionable. And another important aspect is the missing experimental data of emissions of such degradation into different environments. The authors concluded that the outcomes of the survey report conflicting results.

Razza & Degli Innocenti (2012), pointed out the benefits of biodegradability for bio-based materials and underlined the importance of organic recycling (composting and anaerobic digestion) for different materials, such as mulch films, catering items, and biodegradable carrier bags.

Taking into account the works so far presented, the analysis of GWP of plastic pot and bio-pot is in this thesis performed via literature analysis. The system boundaries of PP pot is shown in Figure 3.16a and of PLA pot is shown in Figure 3.16b. Functional unit is 1 kg of material ready to use for agriculture.

For the GWP of plastic pots made of PP, an average value from European studies on LCA of plastics (PlasticEurope, 2016) is adopted (1.6 kg CO_2-eq per kg). The life cycle of PP takes into account the phases: heavy crude oil separation with distillation process, naphtha fraction processing via polymerization in reactor and polycondensation, thermoplastic polymer PP production, extrusion and injection molding, and transportation.

Reviews on environmental emissions of PLA production (Yates & Barlow, 2013; Hottle et al., 2013) indicated that many LCAs have been performed by using data retrieved from Vink et al. (2007). The latter work (and its updated version, see Vink & Davies, 2015) is the most comprehensive study performed with direct inputs data. Thus, the GWP of PLA

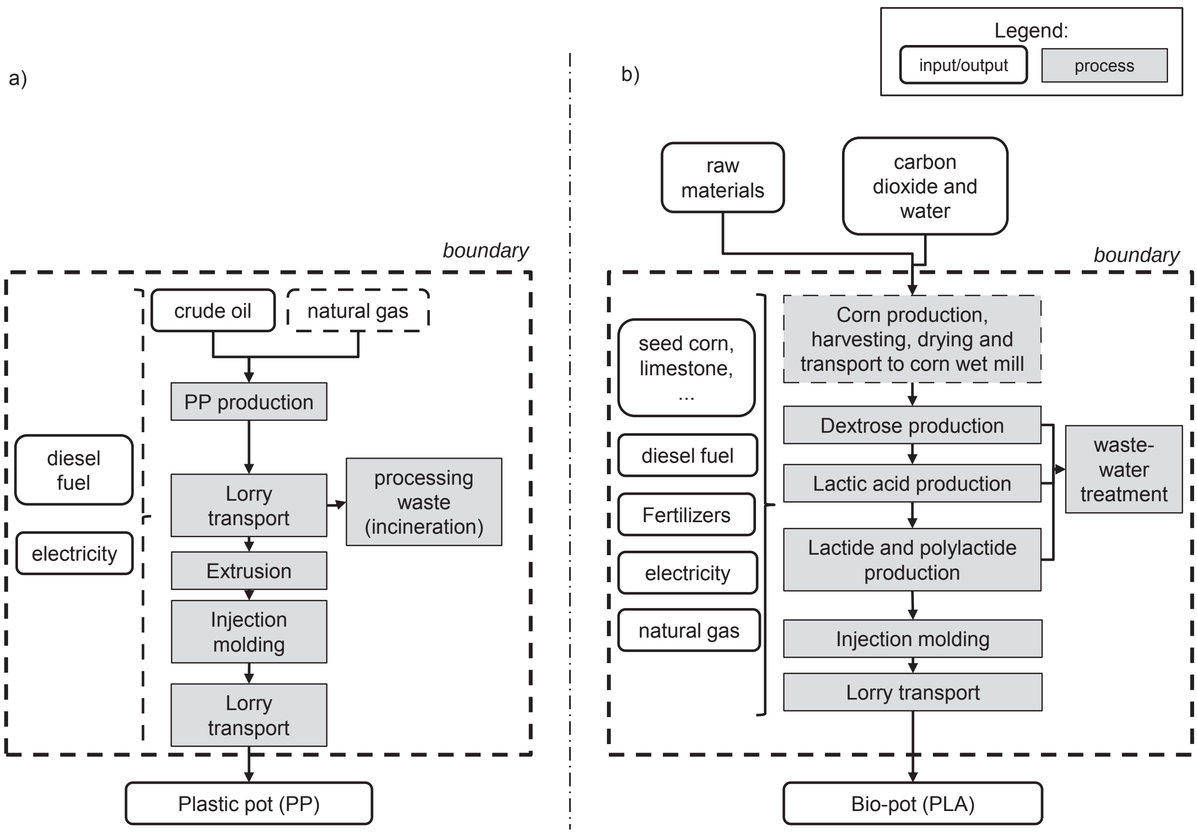

Figure 3.16: System boundaries of PP and PLA plastic polymers.
Note: PLA's system boundaries are adapted from the NatureWorks® PLA production system (adapted from Vink & Davies, 2015)

production system (0.62 kg CO_2-eq per kg) reported in Vink & Davies (2015) is used in this thesis. Life cycle stages of manufacturing of PLA are included: corn production, transport to corn wet mill facility, dextrose and lactic acid production, lactide production and polylactide production, injection molding, and transportation.

3.6 Environmental impacts of a potted plant

The results of the LCAs presented in the previous two sections are here referred to a potted plant cultivated in 5 dm^3 container potted plant.

For the growth substrate, peat contributes 818.2 kg CO_2-eq/t to the GWP (see Figure 3.11). The impacts of compost are calculated as difference between the overall compost contribution to the GWP (44.7 kg CO_2-eq/t, see Figure 3.11) and its carbon storage effect (-32.8 kg CO_2-eq/t), since the latter has been considered controversial in many LCAs, and it is therefore excluded from the optimization model in this thesis. Thus, compost contributes 77.6 kg CO_2-eq/t to the GWP. To calculate the contribution to the GWP of the growth substrate of a potted plant, the following quantities are taken: average compost density for the growth substrate (330 kg/m^3, adopted from Altieri & Esposito 2008), average peat density (181 kg/m^3, standard commercial value), and average filling factor (0.8) of the maximum capacity of the a 5 dm^3 pot. It is assumed the substitution of peat with compost on 1:1 volume basis, in accordance with the literature (see e.g., Boldrin et al., 2010). This means that to substitute a certain volume of peat, the same volume occupied by compost is taken into account (thus, the share of compost directly depends on the densities fo the growth media). Therefore, for a potted plant, a growth substrate composed of 100% peat or by 100% olive mill-waste compost contributes 0.59 kg CO_2-eq or 0.10 kg CO_2-eq to the GWP, respectively.

Table 3.10: Environmental technical parameters of environmental objective function

	Global Warming Potential (GWP) [kg CO_2-eq] 1 kg	5 dm^3 container
Peat	0.818	0.59
Compost	0.077	0.10
Plastic pot	1.60	0.48
Bio-pot	0.62	0.31

For the container, 30% increment of weight is assumed (based on experimental observations) for the substitution of PP pot with PLA bio-pot, according to average values for plastics (PlasticEurope, 2016). Assuming 0.3 kg as plastic pot's weight (with 20 cm diameter), GWPs are yielded: 0.48 kg CO_2-eq for the PP pot and 0.31 kg CO_2-eq for the PLA bio-pot.

The environmental impacts (given in Table 3.10) will be used in the optimization model reported on Chapter 5 to describe the environmental objective.

3.7 Summary

To sum up, in this chapter the case-specific technical solutions of the decision problem of this thesis have been presented, as well as theoretical aspects related to an environmental analysis of emissions and outcomes of the LCAs.

Firstly, LCA has been introduced, with specific aspects related to the analysis of bio-based materials. In particular, the four steps of a LCA have been described. In the *goal and scope* step, it has been highlighted that an expansion of the boundaries of the system is required for by-products of agro-industrial streams, as compost from olive-mill waste. In the *inventory* step, sub-phases for a typical life cycle of bio-based materials have been illustrated with a flow chart. Focus has been given to the use phase and downstream phase of a LCA. Here, accounting of GHG emissions has been described by means of equations. In the *impact assessment* step, critical aspects which are related to the evaluation of environmental effects of the emissions of peat, compost, and bio-pots have been introduced and briefly discussed. Characterization factors used in this thesis have been given. Moreover, special focus has been given to the effects of biogenic emission accounting in LCA for bio-based materials, because it is still considered a crucial aspect on the current LCA debate. In the *interpretation* step, final aspects for recommendation to LCA practitioners have been suggested.

Secondly, an introduction to the theoretical background for the experimental detections of GHGs of bio-based materials has been given. In situ and laboratory experimental trials, which can be used when databases of emissions for case-specific materials are missing, have been described. Here, fugitive GHG emissions accounting, end-of-life prediction, and carbon storage effects have been introduced with technical aspects, equations, hypothesis, and suggestions for further research.

Finally, the outcomes of LCAs conduced on peat, compost, plastic pot, and bio-pot have been presented. For compost and peat, a study, which combines experimental analysis in situ and literature data, has been reported following the four steps of LCA. Assumptions, implications, contribution analysis, scenario analysis, and conclusions have been reported. For plastic pots and bio-pots, a literature analysis has been used to present their contributions to the GWP. And finally, the environmental impacts in terms of GWP of a potted plant, which are required for the environmental objective to be minimized, are given.

To complement the environmental assessment of potted plants, the next chapter will present economic aspects of the decision problem. The following aspects will be described and discussed: accounting with bio-based materials of decision-relevant costs, qualitative grade of potted plants, dependency of sales revenues on the quality grade, willingness to pay for green products, costs of a potted plant.

4 Economic aspects and quality assessment in horticulture

In this chapter, economic aspects of the supply chain of peat, compost, plastic pots and bio-based pots are described. Firstly, accounting of decision-relevant costs with insights on the combination between environmental and economic aspects are introduced (Section 4.1). Secondly, quality grades related to the different levels of potted plants raised inside innovative mixture of substrates are illustrated, as well as the dependency of sales revenues on quality levels (Section 4.2). Finally, additional costs which incur due to the substitution are determined for the technical solutions (Section 4.3).

4.1 Cost analysis

To assess economic parameters (in particular costs) within the supply chain, Life Cycle Costing (LCC) was developed by the USA Department of Defense (U.S. Department of Defense, 1970b,a). The aim was to help in investment decisions. The following stages of the supply chain were taken into account: R&D, processing, installation, operation, maintenance, and end-of-life. In Figure 4.1, the general framework of LCC is shown, from the general idea presented in Rebitzer & Hunkeler (2003). As explained by the authors, the differentiation between the costs are as follows. The internal costs are related to the stakeholders (e.g., supplier, manufacturer) who pay for the stages of the life cycle (e.g., production/extraction, processing, end-of-life). The external costs are related to the "monetized effects of environmental and social impacts not directly billed to the firm, consumer, or government" (Rebitzer & Seuring, 2003), while the externalities are outside the economic system boundaries but inside the natural and social system. The authors recommend to include only internal and internalized costs within the analysis, e.g., energy consumption costs, maintenance costs, selling costs, to avoid double counting of external factors (Korpi & Ala-Risku, 2008), which can introduce "additional uncertainties" into the system (see Geldermann, 2014, p.133ff.).

Studies have been conducted to compare and combine economic and environmental aspects via LCC and LCA, but their "harmonization" is a challenge. More in general, when trade-off between

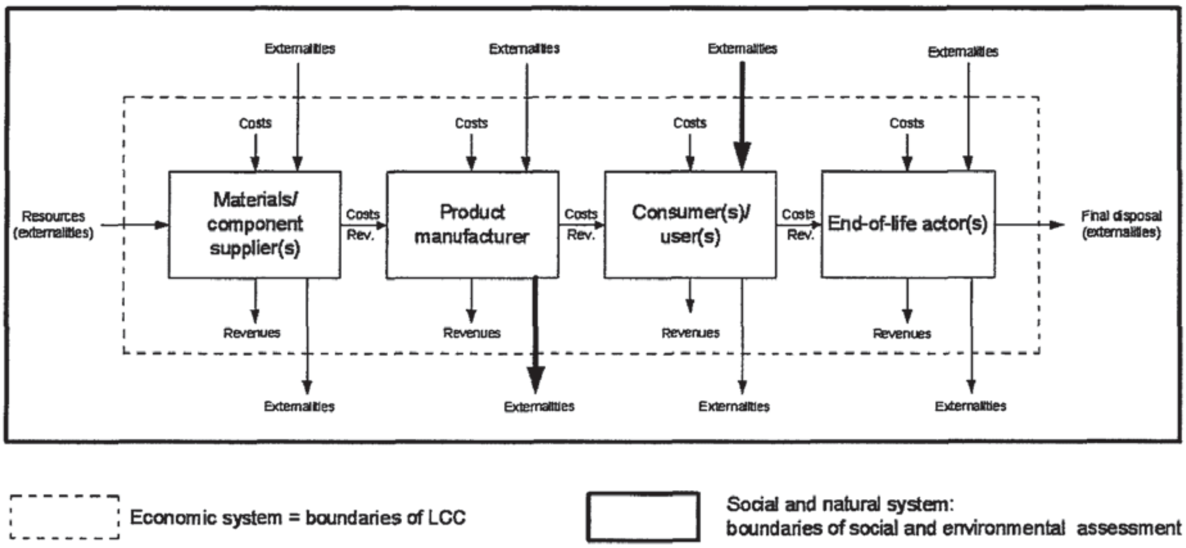

Figure 4.1: LCC general framework (Rebitzer & Hunkeler, 2003)

environmental and economic aspects should be determined, a novel topic of research has been emerged over the past years, the so called *life cycle thinking*, which has been discussed within the *circular economy* concept (see Bocken et al., 2016).

Both aspects of sustainability are indeed critical to define at the same time (Heijungs et al., 2013). The first formulation for coupling both methods was presented by Ciroth et al. (2008) and formalized by Swarr et al. (2011). With these two publications, the research group of the Society of Environmental Toxicology and Chemistry (SETAC) has defined the first steps to prepare the field of a systematic application of the method. A general view of potential bridge between LCA and LCC, with a review on the state-of-art, is presented by Hoogmartens et al. (2014), who concluded that most of the studies conducted have included "stand-alone" methodologies due to difficulty in implementing combined analysis of economic and environmental aspects. The method provided by SETAC remains barely applied in literature, although "LCA and LCC, when carried out in an integrated manner and from a system perspective [...], have a high potential for moving industrial practice towards sustainable development" (Rebitzer & Seuring, 2003). For the combination of LCC and LCA, same aspects, e.g., reference unit and system boundaries, should be considered. Moreover, many ways of presenting the results can be performed, e.g., integrated, separated, or parallel, graphical ways. Hunkeler et al. (2008) illustrated the idea of portfolio presentation, as well as Traverso et al. (2012) and Swarr et al. (2011), in order to present comparative studies and steer decision making with both aspects, monetary and environmental. Another approach was developed by Settanni (2008) and elaborated by Heijungs et al. (2013) with a "computational structure" to overcome inconsistencies between the methods.

Many case studies on the combination of economic and environmental aspects of a supply chain have been published. Ferreira et al. (2014), for instance, quantified the environmental impacts of the recycling system in Portugal converting the results of the LCA impact categories into monetary values with three different methods, in order to improve consistency of the results. The outcomes of the study present the trade-off between benefits and costs. The comparison was based on a deterministic approach by using same functional unit, same boundaries, and including avoided emissions. Nevertheless, the analysis still remains subjective, and the variability (or uncertainty) of input parameters, which produce sensible variation into the final outputs, solicits further investigation, the authors suggested. Another interesting case-study by Barrios et al. (2008) studies the drinking water production in the Netherlands by using cumulative values of environmental and financial impacts over two processes to find possible paths of improvement. In waste management, Accorsi et al. (2014) evaluated the reuse of plastic containers at end-of-life by developing a framework that includes both environmental emissions and economic returns, towards the definition of potential strategies with a multi-scenario analysis.

The following subsection introduces the accounting of costs that incur when substituting conventional materials with innovative ones.

4.1.1 Accounting of decision-relevant costs

The papers presented in the previous section account different types of costs, which are estimated in order to make decisions. Costs related to the choice of a product can be estimated by identifying, in particular, decision-relevant costs when substituting. When the decision is between peat and compost material, or between plastic container or bio-based ones, only those costs which are relevant for the decision should be considered. These costs are considered as *additional costs* when performing a substitution (see Geldermann et al., 2000). Following the approach for the techno-economic assessment of emission abatement suggested by Geldermann & Rentz (2004b), a limitation of analysis to decision-relevant costs (and inputs) can be performed by:

- cutting off identical system parts between the alternatives (e.g., the same costs related to the procedure of cultivating when using plastic containers or bio-based containers);
- cutting off inputs and outputs, which are identical for the alternatives (e.g., costs related to the transportation distances of different containers, when they are manufactured from the same supplier);
- cutting off inputs and outputs, which do not have enough relevance for the outputs (e.g., electricity use to irrigate plants may be slightly different when growing plants into plastic containers or bio-based ones, but it can be cut off because irrelevant).

Consequently, irrelevant costs that do not influence the decision are not taken into account in this thesis. The cutting off approach is in accordance with other studies in LCA (see WEIDEMA, 2000), in environmental incremental analysis (see AZZONE & NOCI, 1996), in LCC (see ASIEDU & GU, 1998), in OR problems (see SHIH & FREY, 1995; MÉNDEZ ET AL., 2006; KUMRAL, 2003).

Additional costs due to the substitution (C) can be identified as sum of annualized investment costs, varying when making a decision, and annual operating costs. The following equation can be used to calculate costs in a plant nursery (adapted from GELDERMANN, 2006):

$$C = \sum_{j \in J} \alpha_j (I + \Delta I_j) + \sum_{i \in I} \dot{m}_i \cdot u_{c,i} + p \cdot u_p + c_o + c_{od} - r_s \tag{4.1}$$

where:

C = annual costs [€/year]

I = investment expenses [€]

α = percentage used to determine costs $j \in J$ [1/year]

ΔI_j = tax additions and/or reductions of the investment I for the costs $j \in J$ [€]

$\dot{m}_i$ = utilities consumption of the material/energy $i \in I$ [q.ty/year]

$u_{c,i}$ = costs for the utilities consumption of i [€/q.ty]

p = personnel requirement [employees]

u_p = personell costs [€/year and employees]

c_o = other costs [€/year]

c_{od} = cost items depending on the decision [€/year]

r_s = sales revenues of products minus selling expenses [€/year]

Investment costs (first sum) occur when the use of novel substrate includes additional licenses and constructions for preparing it. Utilities consumption costs (second sum) occur for instance for the fuel consumption required to mix the substrate, for the electricity use to irrigate plants, and for the mechanical operations to place potted plants. In this thesis, investment costs, utilities consumption costs, and personnel costs are assumed not decision relevant and therefore excluded. Conversely, cost items (c_{od}) which depend on the decision of substitution, like higher costs of input raw materials (e.g., bio-based containers) and different sales revenues (r_s) that can incur when substituting, are taken into account.

Another way of describing the problem is by following the approach in OR presented by SAEDT ET AL. (1991) for potted plant cultivation. Adapting it for the decision problem of this thesis, contribution margin—the marginal profit per unit sold—when substituting materials in a plant nursery can be expressed as difference between selling price and decision-relevant costs of input raw materials, which both vary depending on the decision.

Since the costs for manufacturing bio-pots is generally higher than their conventional alternatives, their selling price is consequently higher. For this reason, consumers prefer not to buy them. However, some studies investigated willingness to pay a premium price for bio-based, innovative products in the USA. An introduction of this topic and results about these studies is given in the following subsection.

Willingness to pay for "green" products

The costs related to the manufacturing of bio-based materials are currently higher than petroleum-based ones. Despite that, consumers may perceive the value of such materials. Some studies have investigated the consumers' willingness to pay a premium price for bio-based pots. In the study of Yue et al. (2010b), the participants of the interview stated that they were willing to pay \$0.23–\$0.29 more for pots made from straw-wheat starch. The authors argued that a successful communication to end users of environmental benefits of biodegradable pots would increase revenue for a manufacturer firm by using premium price, even if total units sold decrease. Hall et al. (2012) confirmed these outcomes by investigating characteristics of biodegradable pots that consumers thought most desirable. With an internet survey with 535 observations in the USA, the authors concluded that, on average, rice hull or straw are preferred over plastic. Yue et al. (2010a) found that the so-called "green consumers" are acknowledged as being more likely to take into account environmental aspects when purchasing goods. The authors confirmed a willingness to pay a price premium for biodegradable containers made mainly of by-products between \$0.23 (wheat starch pots) and \$0.58 (rice hull pots). Ellison et al. (2015) studied six pots (four made of PLA and two made of PHA, see Table 2.2 for a classification) and indicated that consumers are willing to pay a premium of \$0.67–\$1.14 for a bio-plastic plant container, exhibiting also higher willingness to pay for containers that biodegrade quickly in the soil and have a fertilizer effect. The authors suggested that profit of firms can substantially increase, and that these bio-products can be easily incorporated into existing supply chains.

The studies presented so far might encourage firms to take sustainable strategies. Investing in "greening" their value chain could be indeed an option to improve their supply chain. However, the outcomes of such studies based on interviews can be debatable, because in many cases they might not match reality. There might be, indeed, discrepancy between something that is *thought* and something that is actually *done*. Moreover, these results are geographic-specific (referred to the USA) and their results and implications should be carefully understood, in order to be extended to Europe, due to the cultural difference of consumer's behavior.

In this thesis, for the sake of transparency, different scenarios are taken into account. A scenario includes consumer willingness to pay for green products and another one excludes this behavior. These scenarios are presented at the end of the next chapter.

4.2 Quality assessment in horticulture

The selling price of a potted plant depends on its agronomic quality, which is correlated to many parameters. Among them, growth of the plant is one of the most studied (Ferrante et al., 2015; Grigatti et al., 2007). When substituting compost for peat (see Garcia-Gomez, 2002) or when substituting bio-based pot for plastic pot (see Koeser, 2013), reduction of plant's quality can be observed on the final product. The quality can be estimated by experts (agronomists, horticulturists, or growers), who assign discrete grades (qualitative classes) to the plant at the end of the cultivation. This assessment is an index of visual quality, which is considered essential in horticulture (Niu & Rodriguez, 2006)—fruits or vegetables sold in the market are classified and labeled with classes, e.g., class I, class II.

Much research has been conducted on the food industry to evaluate qualitative grades of products (foods, vegetables) and optimize a production system. Plank (1948) proposed one of the first methods towards this end, Brosnan & Sun (2002) illustrated a non-destructive fully automatic procedure to grade agri-food products, and later on, Narendra & Hareesha (2010) reviewed methods in literature for the automated quality inspection via computer aided algorithms.

Regarding ornamental potted plants, focus of this thesis, the most comprehensive work on qualitative evaluation was done by Ferrante et al. (2015). With a literature review on ornamentals, the authors identified crucial parameters for qualitative grading: visual quality, plant longevity, quality index, leaf (number, drop, wilting, chlorosis), flower (number, diameter, healthy, wilted, drop, inflorescence longevity), bracts abscission, pest and disease. These aspects, all together, affect the grading of plants. The qualitative scale used by the authors classified also plants by general visual, aesthetic appearance: breeding, light, light quality, temperature, air humidity, growth retardants, plant nutrition, water deficit, CO_2 enrichment, ethylene inhibitors, supplemental irradiance, modified atmosphere packaging, etc. Another example of quality assessment is reported by Bañón et al. (2011) for ornamentals in South America. The authors described two agronomic species of potted plants that were investigated through different qualitative parameters (i.e., plant dry weight, shoot to root ratio, leaf area, visual quality index, leaching fraction and leachate electric conductivity). The plants were thus evaluated with a quality scale to identify classes. These results suggest that the visual quality depends also on the species investigated, and consequently, conclusions drawn for a specific specie cannot be easily generalized.

In literature, examples of automated grading are also reported. The work by Timmermans & Hulzebosch (1996) presented a sophisticated computerized system via color camera and insights on neural network techniques for grading potted plants (e.g., screening between flowered and cactus). Carvalho & Heuvelink (2015) provided a literature review of methods and presented a model for the prediction (and optimization) of the quality of cut flowers based on several parameters, i.e., stem morphology (length, diameter, strength), leaf morphology (number, size) and flowering aspects (number, size, position). Still, further research is requested, the authors suggested. Despite this selection of studies, optimization of horticultural production in OR by using grading system is barely covered in literature—a comprehensive review dated 1998 on optimization methods used in horticulture was done by Lentz (1998); however, grading system was not covered—due to the presence on numerous parameters, which are, in many cases, not easily predictable.

If the growth of ornamentals deviates from standard horticultural characteristics (required for the market), horticulturists can decide to associate each qualitative class with a selling price with a *quality function*. The lower the class, the lower is the selling price of the plant. This concept has been presented in the wine industry by Ferrer et al. (2008), who assessed the quality of grapes with four classes: best quality (*premium*), intermediate quality (*reserve* and *varietal*), and worst quality (*bulk*). By defining a quality function, the authors presented a graphical representation (Figure 4.2) of costs and deviation of harvesting date from an optimal harvest date, which leads to the premium quality (therefore, maximum contribution margin). Such functions are widely used for edible perishable foods or for horticultural plants devoted to the food industry, but they can be extended to ornamentals.

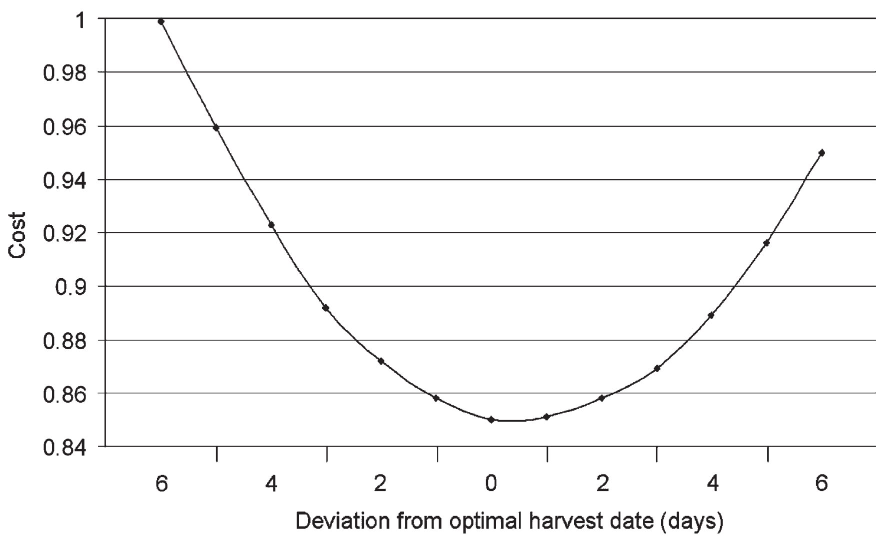

Figure 4.2: Example of quality function for wine industry (Ferrer et al., 2008)

This approach can also be seen as an extension of the general concept of 'quality loss function' introduced by Taguchi & Clausing (1990) in Business Administration. And it is also equivalent to the concept of 'penalty function' (Romero & Rehman, 1989, p.199) used for agribusiness problems: a penalty scale (e.g., between 0, no penalty, and infinite, maximum penalty) incurs due to the violation of specific requirements of a product. This approach seems to be more realistic, because it takes into account the *magnitude* of the fluctuation of a variable (these kind of problems was for the first time introduced by Kvanli, 1980).

In this thesis, studies which indicate and illustrate the dependency of growth of plants with different substrates have been considered, e.g., Mugnai et al. (2007); Bohne (2001); Perez-Murcia et al. (2006); Garcia-Gomez (2002); Guérin et al. (2001); Ostos et al. (2008); Aleandri et al. (2015); Mattei et al. (2017); Bustamante et al. (2008); Herrera et al. (2008); Zaller (2007), as well as relevant studies which focus on the influence of olive-mill waste substrate on the quality of plants, e.g., Garcia-Gomez (2002); Papafotiou et al. (2005, 2004); Bañón et al. (2011); Ouzounidou et al. (2010). The assessment of qualitative classes for potted plants is taken into account in this thesis with a common scale in horticulture: grades between 1 (non-commercial) to 5 (maximum quality, optimal growing, standard ones). A quality function is further used to depict the correlation between quality classes and selling price.

4.3 Economic analysis of peat, compost, plastic pot, and bio-pot

Decision-relevant costs of the technical solutions include the purchase of raw input materials (direct costs). These data was gathered during interviews to three plant nurseries in Pistoia, literature (Ghofar et al., 2005; Riggi et al., 2011) and web-pages of commercial products.

For the peat, sphagnum peat of three types (sold in big bales of 1–3 m^3 of volume) is chosen[1]. These products are prepared as different mixtures of white peat and black peat, whose minimum purchase cost (0.0322 €/dm^3) and maximum purchase cost (0.0431 €/dm^3) have been accounted. An average cost of 0.0377 €/dm^3 of peat is taken into account, which corresponds to 0.1885 € for a 5 dm^3 container.

The costs of the *compost* are mainly due to the energy consumption for mixing it and are based on one ton of material. The purchase price of mature compost for a plant nursery (10.52 €/t) is calculated by considering an olive-mill facility which decides to invest in purchasing a mixing wagon, in order to process by-products of its olive oil production, and selling the resultant compost, as follows (see Table 4.1).

[1] Commercial substrates for potted plant cultivation: "TS 1 fein Rez. 876", "TS 3 fein Rez. 416", "TS 4 med.Rez. 602" (Brill Substrate, Germany); "Tray 60/40 M Gramoflor-Containersubstrat R" (Gramoflor, Germany).

Table 4.1: Costs for composting 1000 t of olive-mill waste by static procedure and produce olive-mill waste compost

Annual fixed costs	[€]
Depreciation	2,000.00
Interests, maintenance, insurance	1,500.00
Storage costs (covering, moving)	15,600.00
Other costs	1,910.00
Annual variable costs	[€]
Straw	4,000.00
Waste wool	2,000.00
Big bags	6,668.00
Personnel	11,250.00
Energy requirement	1,912.50
Other unexpected costs	3747.24
Annual sales revenues	[€]
Selling of mature substrate	33,333.33
Selling of stones from olives	15,000.00
Avoided costs for spreading on land	12,750.00

The annual costs for an olive oil mill facility to produce a mature compost are: fixed costs (depreciation, interests, storage costs, other costs) and variable costs (straw, waste wool, big bags, personnel, energy requirement, other costs). It is assumed that 1,000 t of olives are milled per year, and the costs are based on this quantity then referred to a potted plant. In particular, annual fixed costs are assumed: to purchase a feeder-mixer wagon and a machine to separate olive pits (30,000 €), for the annual depreciation of the wagon (for 15 years), for interests, maintenance, and insurance (5% initial investment), for storage costs related to covering, placing, moving the gas-permeable bags (15.6 € are assumed per ton of material processed), other costs (10% fixed costs). Annual variable costs are assumed: to purchase straw (80 €/t, including transport to olive mill facility, calculated for 50 per ton of straw which is 5% of 1000 t initial mixture), to purchase waste wool (40 €/t, including transport to olive mill facility, calculated for 50 t of waste wool which is 5% of 1000 t initial mixture), to purchase gas-permeable bags (8 €per bag recycled twice, calculated for 1667 bags with density of 0,6 t/m^3, which are required to compost 1000 t of olive-mill waste), for personnel (15 €/h, considering 2 workers for 1.5 times the operating time of the mixing wagon, assumed as 31 days, 8 hours per day), to pump and process the initial mixture in the mixing wagon (0.17 €/kWh price for electricity use, considering: 31 days operating time of the mixing wagon, 8 hours per day, 45 kWh total consumption), for other unexpected costs (8% sum of annual fixed and variable costs). These costs sum up to 50,586 € total annual costs.

By further assuming as sales revenues: selling price for mature olive-mill waste compost (sold as peat substitute for 25 €/m^3, considering: 80% mature compost (in volume) yielded from 1,667 m^3 produced from initial 1000 t olives), selling price for olive pits from olives (150 €/t, considering: 10% of olive pits produced from initial 1000 t of olives), avoided costs for spreading on land olive-mill waste (15 €/t, based on 850 t olive-mill waste to process in the olive mill facility). These revenues sum up to 61,083.33 € total annual sales revenues. Therefore, 10,495.59 € are assumed as revenues for an olive mill facility when producing mature olive-mill waste compost. Finally, the purchase price for a plant nursery for a 5 dm^3 compost is 0.03 €.

The purchase cost of *plastic pot* made of PP has been calculated as average (1.07 €) of minimum (0.91 €) and maximum (1.16 €) purchase prices of four commercial products[2]. A 30% corrective factor is assumed for the purchase price of *bio-based pot* made of PLA (see Bastioli, 2001; Ghofar et al., 2005; Riggi et al., 2011). Concluding, the decision-relevant costs of growth substrate and container of a potted plant are given in Table 4.2.

Table 4.2: Economic technical parameters of economic objective function

	Costs [€] 5 dm^3 container
Peat	0.19
Compost	0.03
Plastic pot (5 liters, weight 360 g)	1.08
Bio-pot (5 liters, weight 600 g)	1.40

4.4 Summary

This chapter has introduced economic aspects of the value chain of growth substrate and containers. Firstly, the combination of LCA and LCC has been presented. Here, aspects currently debated have been introduced, such as definition of common system boundaries, differentiation between costs, and application of a combined method to real-world case studies. Furthermore, decision-relevant cost accounting for optimization problems has been described, with focus on

[2] Commercial containers for potted plant cultivation: "Online Vierkanttopf", PP, 5.7 dm^3, Germany (`http://www.gbk-shop.de`); "Viereckiger Topf HQ", PP, 5.8 dm^3, Germany (http://www.blumentoepfe.biz); "Stabiler Pflanzentopf", PP, 5 dm^3, Germany (`http://www.growland.net`); "Blumentopf", PP, 5 dm^3, Germany (`http://www.pflanzenbedarf.com`).

cutting off approach, additional costs accounting when substituting resources, contribution margin calculation, and willingness to pay for green products. Quality assessment used in horticulture to evaluate plant production has been introduced, with research papers that include statement of classes for vegetable, fruits, and ornamentals. Here, the concept of quality function has been described, with focus on horticulture.

Secondly, the results of an economic analysis of a potted plant have been presented, by describing decision-relevant costs of a plant nursery in Pistoia. In particular, since olive-mill waste compost is produced following a novel procedure (at R&D stage, following the results of a research project, described on page 47), its decision-relevant costs have been calculated for the case-study.

In the next chapter, existing approaches to optimize resources in agriculture will be described. OR methods will be introduced with focus on blending problems. Then, the decision problem of this thesis will be described by defining an environmental function and an economic function. Decision variables will be declared and optimal Pareto solutions will be illustrated for the case study of plant nursery in Pistoia. Finally, a scenario analysis will investigate the effects on the outcomes when varying input parameters.

5 Mixing problem

In this chapter, a modeling approach is developed and applied to a case study. Firstly, existing approaches for optimizing resources in horticulture are presented (Section 5.1) with a review of OR papers, as well as decision-making problems concerning the blending of input materials which meet certain requirements. Secondly, an optimization approach is illustrated (Section 5.2), where decision variables of the problem and objectives are described with mathematical functions. Here, environmental emissions and additional costs are taken into account, as well as quality levels of potted plants. Thirdly, a case study (Section 5.3) elucidates the decision-making problem by including technical parameters determined in the previous chapters. The description of these aspects of the problem are followed by a Pareto optimization and a final scenario analysis, which investigates the variation of input parameters into the results[1].

5.1 Existing approaches for optimizing resources

To help decision makers, many methods (and models) have been developed and used for several industrial applications, as well as in agriculture. According to Guinée et al. (2001), no model of 'full reality' exists. LCA, in particular, introduces 'crude simplifications' when it is used for decision-making, and for a 'broad applicability' the model should be as "simple and transparent as possible, without violating the fully reality too much".

Methods of OR, which is the discipline that applies mathematical techniques for decision-making, have been applied to the agriculture sector. Many reviews have been published about agri-food business problems related to perishable and non-perishable products (Ahumada & Villalobos, 2009), resource management (Hayashi, 2000), or crop planning, harvesting, and risk management (Lowe & Preckel, 2004). In general, focus of these agricultural decision problems are particular stages of the value chain (for ease of simplification) (Higgins et al., 2010).

The problem addressed in this thesis is a multi-objective ones. Multi-objective optimization, which is part of mathematical programming that provides solutions for improving real systems

[1]The content of this chapter is partially included in Chilosi et al. (2017), Krüger et al. (2018)

(Figueira et al., 2005), has attracted the attention of OR practitioners when conflicting goals are addressed. Many companies face indeed the dilemma to find the *trade-off* between environmental and economic objectives, which are often in contrast, especially in agriculture. Multi-objective problems, in contrast to single-objective ones, identify more than one objective. Typically, in multi-objective optimization no single global solution can be identified, but a set of optimal solutions, which is called *Pareto set*, when the solutions can be expressed with a function.

This first idea of multi-objective optimization goes back to Pareto (1906), who defined the so called "Pareto-front": there is no single global solution, on the contrary, there is a set of efficient solutions to a multi-objective problem (minimization or maximization). A graphical illustration of Pareto-front can be depicted and is immediately useful for decision makers. An example is reported in Figure 5.1, where a Pareto set of optimal points (B and C) satisfy at the same time two objectives ($U1$ and $U2$). In general, the *trade-off* between them can be identified.

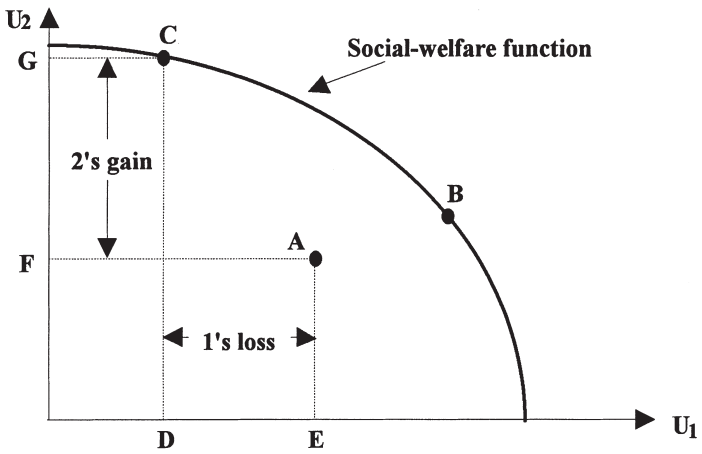

Figure 5.1: Pareto optimization

The example is reported for the case of "welfare" economics, where the questions arises whether resources should be allocated, in terms of production and consumption, to maximize the social welfare. By definition, a Pareto-set of efficient points are the ones where no individual welfare can be made better off without making at least one other individual worse of. Bi-objective problems, like the ones addressed in this thesis with environmental and economic targets, can be easily illustrated with a Pareto-front. The graphical illustration of a Pareto-front is used in this thesis as guide for decision-making in horticulture.

In multi-objective optimization, X is the feasible set (or the set of alternatives of the decision problem), x is a variable (or point), and $f = (f_1, ..., f_p)$ is vector of objective functions. In general, a minimization problem is described as:

$$\min \ (f_1(x), ..., f_p(x))$$
$$\text{subject to } x \in X$$

where p is the number of objective functions. Pareto optimality (or efficiency) is based on the fact that any x which is not efficient "cannot represent a most preferred alternative for a decision maker, because there exists at least one other feasible solution $x' \in X$ such that $f_k(x') \leq f_k(x)$ for all $k = 1, ..., p$" (Ehrgott, 2005). Therefore, x' should clearly be preferred to x.

The fundamental aim of multi-objective optimization is the determination of the Pareto set. Many methods in multi-objective optimization have been developed to find Pareto optimal solutions, e.g., weighting method (Zadeh, 1963), ε-constraint method (Haimes, 1973), normal-Boundary intersection (Das & Dennis, 1998), method of equality constraints (Lin, 1976). For an overview on methods of multi-objective optimization see Ehrgott (2005).

In this thesis, a decision-problem is addressed for the horticulture sector. In literature, the following aspects have been under investigation: planning of production in the wine industry (e.g., Szidarovszky & Szenteleki, 1987), configuration of farms (e.g., Tan & Çömden, 2012), crop optimization in the agri-food industry (e.g., Berbel, 1989), and planning of cultivations in plant nurseries (e.g., Hofstede, 1992). This thesis deals with the determination of a mixture of input materials for preparing growth substrate, and it includes decision-relevant aspects, such as environmental emissions, additional costs, and quality levels. These three aspects are investigated in literature, and OR has been used for solving decision-making problems. A review of the relevant literature will be presented in the following two subsections to lay the foundation for a development of the model and its solution approach. Firstly, relevant papers which use OR methods in the agriculture sector are reported, and secondly, papers which address the problem of choosing the optimal mix of input materials that meets certain requirements is presented.

OR models in horticulture

Papers related to the optimization of an *economic* parameter has been the majority since decades. One of the first investigations in horticulture was done by Szidarovszky & Szenteleki (1987), who proposed a model to find the optimal land area for growing grapes, such that net annual profit is maximized and manpower demand is minimized at the same time. Costs for annual cultivation and harvesting, as well as direct costs for the production of wine and income from selling wine, were taken into account. Subsequently, Berbel (1989) developed a model to define crop patterns with the following objectives: profit maximization, risk minimization, and seasonal labor minimization. SAEDT ET AL. (1991) optimized potted plant production in a plant nursery with a transition planning model, in order to find the optimal number of potted plants (and optimal spacing). The aim was to maximize the contribution margin, which incorporates decision-relevant costs. Another relevant example can be found in SISKOS ET AL. (1994), who modeled cropping patterns to find farm lands to be allocated to specific crops, such that gross margin profit is maximized, seasonal labor is minimized, tractor utilization is minimized, and forage production is maximize.

Taking a step forward from these four preliminary studies, other authors investigated decision-problems combining economic and *environmental* objectives. GROOT ET AL. (2012), for example, defined a model to find Pareto-optimal alternatives for farm configuration (e.g., areas of cultivated crops), by minimizing nitrogen soil losses (environmental objective), maximizing operating profit (economic objective), minimizing labor distance (social objective), and maximizing the organic matter balance of the soil structure. A range of variation in the optimization procedure is presented with minimum and maximum values for the optimal solutions by referring to a single crop. Subsequently, VARSEI & POLYAKOVSKIY (2016) elaborated the design of a supply chain network in the wine industry to determine the quantity of wine produced, such that: decision-relevant carbon dioxide equivalent emissions are minimized (environmental objective), supply chain costs of transportation are minimized (economic objective), and social impacts associated with the established bottling plants are minimized (social objective).

The inclusion of more objectives, in some cases, requires another aspect, such as the *quality* of the end-product. The inclusion of a qualitative objective requires, nevertheless, more sophisticated algorithms, depending on the aim of the optimization problem addressed. Two papers are reported here due to their innovative approaches, which have inspired the ones of this thesis. FERRER ET AL. (2008) introduced an approach to find optimal quantity of grapes to minimize costs, and at the same time maximize quality of the harvested grapes. The quality of wine was incorporated inside the model with a function that defines loss of revenue; therefore, wine was classified into quality groups (or categories/grades/levels, see Section 4.1). Another example is reported by TAN & ÇÖMDEN (2012), who proposed a model for the planning of annual fruit and vegetables to determine farm areas and seeding time, in order to maximize the total expected profit. Within the model, maturation time of fruits or vegetables were taken into account.

An overall vision of the aforementioned works is shown in Table 5.1. Although these works introduced the accounting of decision-relevant environmental emissions and decision-relevant costs (and only some of them include also product quality), none of them—and in general in literature—concern growth substrate formulation in horticulture, which is the focus of this thesis. To address this specific problem, OR papers have been published in other sectors, and they are

Table 5.1: Selected papers on optimization of resources in agriculture

Authors	**Topic**	**Environmental objective**	**Economic objective**	**Quality**
Groot et al. (2012)	farm planning (supply chain)	(✓)	✓	
Varsei & Polyakovskiy (2016)	wine industry (supply chain)	✓	✓	
Szidarovszky & Szenteleki (1987)	wine industry (area)		✓	
Berbel (1989)	agri-food industry (crop pattern)		✓	
Siskos et al. (1994)	farm planning (crop pattern)		✓	
Saedt et al. (1991)	plant nursery (spacing)		✓	
Ferrer et al. (2008)	wine industry (scheduling)		✓	✓
Tan & Çömden (2012)	agri-food industry (area and seeding time)		✓	(✓)

called blending models. In the following section, relevant studies that introduce important aspects to describe the decision-problem of this thesis are illustrated.

Blending models in OR

A blending (or mixing) problem concerns the blending or mixing of input materials, such that the obtained final product (output) meets certain characteristics or properties (Harvey, 1979, p.17). Common inputs of a blending problem can be chemicals, metal alloys, crude oils, or livestock feeds. Outputs are defined by quality and quantity of the resultant blend with particular restrictions. In general, an objective of a blending problem can be the minimization of processing costs, and common variables used in the model are the quantity of input materials used in the final blend. Blending problems can be mathematically formulated as linear problems. The conditions for a mathematical model to be linear are (Harvey, 1979, p.18): the decision variables are continuous (they can assume any value within a prefixed range), the objective function is linear, and the constraints are linear. Constraints are a set of functional equalities (or inequalities) that represent restrictions to the decision variables of the problem.

A generalized form for blending model constraints is the following (Rader, 2010, p.30). Suppose to combine input materials I in order to produce products J, and each product $j \in J$ is defined by combining inputs $i \in I$. Then, the quantity of product j is determined by adding the total quantity of the inputs. Therefore, by defining x_{ij} as quantity of inputs i in product j (for each

$i \in I$ and $j \in J$), by defining q_i as total quantity of each input i used, and by defining z_j as total quantity of each product j produced, the following equations are obtained:

$$q_i = \sum_{j \in J} x_{ij}, \;\; i \in I \tag{5.1}$$

$$z_j = \sum_{i \in I} x_{ij}, \;\; j \in J \tag{5.2}$$

Since there is a typical maximum quantity a for each input i available for use and a minimum required quantity r for each product j, the constraints are therefore:

$$q_i \leq a, \;\; i \in I \tag{5.3}$$

$$z_j \geq r, \;\; j \in J \tag{5.4}$$

Relevant papers concerning blending problems that are linked to the three aspects of the research question are illustrated here. Blending problems in OR have been mainly addressed in the food, petroleum, or chemical industry (Al-Shammari & Dawood, 1997). One of the first problems (Stigler, 1945) was a linear programming tool for the food industry to build a mix of several items with several nutritional requirements.

A blending problem related to the formulation of fertilizers in the chemical industry was investigated by Ashayeri et al. (1994). Objectives of the study are minimization of raw material costs and time of planning, such that ratio of nutrients, contained in the input materials, determine the nutritional content of the resultant blend—the nutritional content can be considered as *quality* of the final product. Many different blends which meet the requirements are possible, and therefore an optimization model is required. Decision-relevant costs for the purchase of raw materials were taken into account.

Optimal blending strategy for a paint and putties factory was reported by Al-Shammari & Dawood (1997), who optimized the quantity of raw materials used to manufacture 17 blends of paint from 29 input materials. Objectives of the study are cost minimization while meeting blending requirements, such as technical specifications that define the final *quality* of a blend (e.g., opacity level, capability of adhesion).

Kumral (2003) elaborated a model to determine optimal blending of iron ores from four different sources in the mineral industry sector. The aim is to find optimal combinations of different materials (e.g., Fe, SiO_2, and Al_2O_3), such that expected costs of the blending process are minimized and quality specifications are satisfied (e.g., minimum initial temperature before blending). Also in this kind of problems the choice of technical parameters influence the modeling, and those parameters are problem-specific. The determination of the blending ratio can be compared to the

Table 5.2: Selected works that applied blending models

Authors	Topic	Environmental objective	Economic objective	Quality
Ashayeri et al. (1994)	chemical fertilizers		✓	✓
Al-Shammari & Dawood (1997)	paint industry		✓	✓
Kumral (2003)	mineral industry		✓	✓
Méndez et al. (2006)	oil-refinery (scheduling)		✓	✓
Shih & Frey (1995)	power plants	✓	✓	(✓)

determination of input materials of the growth substrate. Technical parameters which influence the quality of a potted plant are, nevertheless, numerous, and deserve agronomic investigation.

A complex problem related to the oil-industry was presented by Méndez et al. (2006), who analyzed short-term blending and scheduling. The problem involves multiple production requirements while meeting "given product specification"—the latter is considered in the model as product *quality*; examples of specifications are: minimum octane number, maximum sulfur and aromatic content. By incorporating discrete and continuous variables in the model, the authors proposed a method for maximizing the net profit, which is defined as total product value minus decision-relevant costs.

The only study in literature—so far presented—which includes the three aspects of the research question is by Shih & Frey (1995), for coal blending. The aim is to reduce sulfur emissions in power plants by incorporating coal properties (e.g., sulfur content, ash content, heating value), such that decision-relevant costs (e.g., coal price, ash disposal costs, capital costs of plant modification, and incremental change in operating and maintenance) are minimized. Theses costs are *additional costs* that incur when coal is blended in addition to the costs of the base coal (initial situation, also called basis scenario/case). The same way of accounting can be performed when deciding between alternatives in horticulture, where an alteration of a basis case (the use of peat and petroleum-based containers) can determine additional costs of production.

The studies reported previously are summarized in Table 5.2. These papers introduce relevant aspects linked to the research question: formulation of a mixture (by using a continuous variable), meeting a certain product specification to achieve a specific final quality (which is determined via technical parameters), definition of decision-relevant costs (which vary between basis scenario and new ones). In this thesis, for the first time, these aspects are combined to develop a model that is used to answer to the research question and it is applied to a real-life case study. For this reason, the next section presents an approach which incorporates aspects of OR models applied in agriculture and blending models.

5.2 Optimization approach

In this section, an optimization approach to solve the decision problem of this thesis (introduced in Chapter 1) is presented. This approach is developed by including aspects that have been introduced in the previous section: optimizing a mixture, meeting quality requirements of the final product, accounting of decision-relevant environmental emissions and decision-relevant costs. These aspects are here specifically referred to a plant nursery. The model includes technical solutions, objective functions, constraints, and decision variables.

In general, objectives are decision maker's values, which can be expressed with a mathematical function $f(x)$ of the decision variables. Maximization or minimization of the functions (also called goal functions) represent an improvement of one of more of the values (Romero & Rehman, 2003)—it should be noticed that this kind of problems it is not new; since 1970, environmental issues have indeed determined many industrial strategies which included both economically and environmentally efficiency (French & Geldermann, 2005).

In this thesis, the decision problem is the suitable combination of the choice of pot and the share of compost. Therefore, two *decision variables* are considered: the kind of pot (β) and the share of compost (γ). The variable β represents a binary decision ($\beta \in \{0, 1\}$), and it differentiates if the pot is made of petroleum based plastic ($\beta = 0$) or if the pot is made of bio-composite material ($\beta = 1$). The linear variable γ is in the continuous solution space ($\gamma \in [0, 1]$), and vary from 0 (no substitution: substrate composed by 100% peat) to 1 (complete substitution: substrate composed by 100% compost). The growth substrate, as a mixture of compost and peat, is schematically depicted in Figure 5.2.

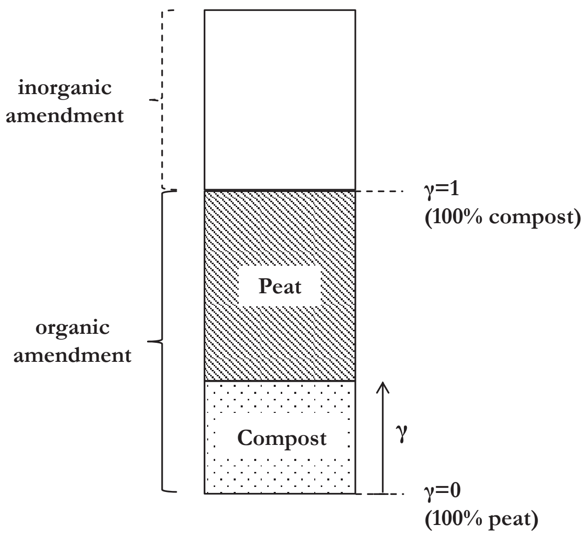

Figure 5.2: Share of compost as decision variable of the problem

Aim of the model is to find the optimal (β, γ), such that additional environmental emissions and costs due to the substitution are minimized. These two objectives are expressed as an environmental function (f_e, which includes the decision-relevant emissions, measured in CO_2-equivalents) and an economic function (f_c, which includes the decision-relevant costs, measured in €). The following general equations describe the optimization model:

$$\min f_e(\beta, \gamma) \tag{5.5}$$

$$\min f_c(\beta, \gamma) \tag{5.6}$$

$$\text{subject to}$$

$$\beta \in \{0, 1\}$$

$$\gamma \in [0, 1]$$

The *environmental function* to be minimized is described as:

$$f_e(\beta, \gamma) = \gamma(a_2 - a_1) + \beta(b_2 - b_1) + a_1 + b_1 \tag{5.7}$$

where a_1 is the GWP of peat, a_2 is the GWP of compost, b_1 is the GWP of petroleum-based plastic pot, and b_2 is the GWP of bio-pot. The first term of the sum identifies the mixture of the growth substrate as a linear function. If $\gamma = 0$, this term is equal to the emissions of peat, if $\gamma = 1$, this term is equal to the emissions of compost—this case identifies a complete substitution. All intermediate mixtures are possible. The second term of the sum identifies the choice of a plastic pot or a bio-pot.

The *economic function* (f_c) to be minimized denotes the additional costs that incur when substituting. It comprises the additional selling costs (f_{sc}, it is linked to the lower quality levels of plants raised inside compost, which can be sold at lower selling price) and the additional item costs (f_{ic}, due to the higher price of bio-pot when substituting). It is assumed that the additional selling costs function, f_{sc}, is a step function that correlates the selling price of a potted plant and the share of compost by means of a discrete number of constant pieces. The constant pieces can be determined by discretization of an experimental curve of qualitative points based on agronomic and aesthetic evaluation given by experts (see Section 4.2). A premium price (r_j) is assumed when bio-pots are used (when $\beta = 1$)—see Section 4.1.1. A general expression of f_{sc}, for each constant piece j, is:

$$f_{sc}(\beta, \gamma) = e_j(1 - \beta) + r_j e_j \beta \tag{5.8}$$

$$\text{subject to}$$

$$t_k \leq \gamma < t_z$$

$$j, t_k, t_z > 0$$

$$t_z > t_k$$

where e_j is the selling price associated with the step j, r_j is the premium (such as an extra price for the product quality associated with the step j), t_k and t_z are the minimum and maximum of the step j. The additional item costs function f_{ic} is a linear function that includes a mixture of substrate and pot, and it includes the additional costs of the four technical solutions when substituting.

Thus, the additional item costs function, f_{ic}, is described as:

$$f_{ic}(\beta, \gamma) = \gamma(c_2 - c_1) + \beta(d_2 - d_1). \tag{5.9}$$

The following additional costs function f_c is then obtained by combining Equation 5.8 and Equation 5.9:

$$f_c(\beta, \gamma) = f_{sc}(\beta, \gamma) + f_{ic}(\beta, \gamma). \tag{5.10}$$

5.3 Case study

The developed model is here applied to a case study in Pistoia, by using case-specific environmental and economic parameters.

5.3.1 Objective functions

The environmental function and the economic function are described by their case-specific technical parameters, previously obtained in Chapter 3 and Chapter 4. These parameters are shown in Table 5.3.

Table 5.3: Environmental and cost technical parameters of the model

Material	**GWP** [kg CO_2-eq]	**Costs** [€]	**for**
Peat	$a_1 = 0.59$	$c_1 = 0.19$	$\gamma = 0$
Compost	$a_2 = 0.10$	$c_2 = 0.03$	$\gamma = 1$
Plastic pot	$b_1 = 0.48$	$d_1 = 1.08$	$\beta = 0$
Bio-pot	$b_2 = 0.31$	$d_2 = 1.40$	$\beta = 1$

The additional item costs for the case study are displayed with the linear function f_{ic} in Figure 5.3, which is strictly decreasing. Here, the additional costs due the use of bio-pot are evident.

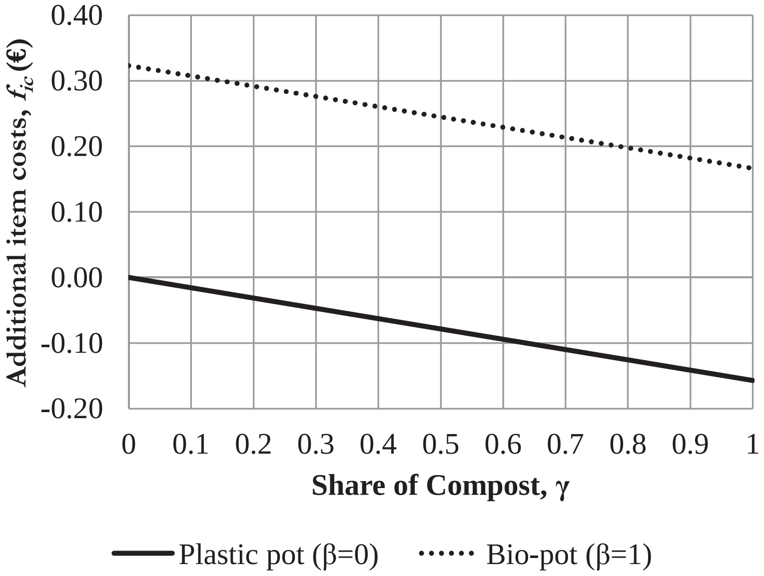

Figure 5.3: Additional item cost function f_{ic}

The additional selling costs are calculated with the correlation between the selling price of the potted plant, its quality levels, and the share of compost, as follows. The ornamental species selected is *Photinia fraseri* cv. "Red Robin", due to its commercial relevance in Italy (LANZI ET AL., 2009). Much research has been conducted on this species, and in particular on the agronomic quality of plants cultivated in different growth substrates (see, e.g., CHILOSI ET AL., 2017).

A quality function (f_q) is elaborated from the results of a study conducted on the propagation of 80 plants of *Photinia x fraseri* cultivated in $5\,\mathrm{dm}^3$ containers (CHILOSI ET AL., 2017). The authors performed agronomic trials that lasted four months to assess plants' growth on different substrates: peat not substituted (case base, $\gamma = 0$), partially substituted ($\gamma = 0.33$ and $\gamma = 0.66$), or completely substituted ($\gamma = 1$). At the end of the trial, three qualitative parameters have been measured, such as aerial part length (q_1), roots length (q_2), and number of lateral roots (q_3). One-way ANOVA, a standard statistical method (FISHER, 1973), was performed followed by Tukey's post-test (TUKEY, 1949) to identify pairs with significantly different means. To the three statistical-significant groups ("a", "ab", and "b", reported in CHILOSI ET AL., 2017), in this thesis a qualitative discrete rating is used: between 1 (dead plant) and 5 (maximum quality), but only three levels (3, 4, and 5) are taken into account to be suitable for the market. In particular, 5 is associated with the group "a", 4 to the group "ab", and 3 to the group "b". This data is reported in column q_1, q_2, and q_3 given in Table 5.4. A final aesthetic evaluation performed in this thesis is used (column q_4), and average values (column *Average q*) are calculated (see Table 5.4) for different shares of compost.

Table 5.4: Qualitative matrix score of peat substitution.
Notes: γ = share of compost, q_1 = aerial part length, q_2 = roots length, and q_3 = number of lateral roots (Chilosi et al., 2017); q_4 = aesthetic evaluation made by experts

γ	q_1	q_2	q_3	q_4	*Average q*
0	5	5	5	5	5.00
0.33	4	5	5	3	4.25
0.66	4	4	5	5	4.50
1.00	3	4	3	2	3.00

From the average values (column *Average q* on Table 5.4), q values are interpolated, and a quality function f_q is thus obtained (see Figure 5.4a). The quality function f_q is then discretized (Figure 5.4b) to simulate a common qualitative rating with discrete qualitative levels used for horticultural products in real-world situations. Thus, the resultant 7-steps quality function f_q is piecewise constant for each $\beta \in \{0, 1\}$. For the plastic pot ($\beta = 0$), f_q is described as:

$$f_q(\beta,\gamma) = \begin{cases} 5.0; & \text{for } \gamma = (0.00, 0.09) \\ 4.6; & \text{for } \gamma = (0.09, 0.24) \\ 4.2; & \text{for } \gamma = (0.24, 0.42) \\ 4.4; & \text{for } \gamma = (0.42, 0.57) \\ 4.5; & \text{for } \gamma = (0.57, 0.75) \\ 3.7; & \text{for } \gamma = (0.75, 0.91) \\ 3.0; & \text{for } \gamma = (0.91, 1.00) \end{cases} \tag{5.11}$$

Higher values of f_q are assumed (+1%) for the use of bio-pot ($\beta = 1$), since their experimental use has been seen as promising[2].

[2]These assumptions have been made following the results (elaborated for this thesis) of the research project funded by Agricultural Agency, Tuscany Region, Italy, Framework "Multi-call for integrated supply chain projects (in Italian: *Progetti Integrati di Filiera*, PIF), 2^{nd} step, Ordinance n. 161/2012. Reg. CE 1698/05-PSR 2007/2013", Project: "Promotion of a plant nursery net, based on the enrichment and expansion of the supply chain of plant nurseries (based on the ongoing PIF, called SAN-SOIL) and based on the enhancement of plant production through the MPS environmental certification (More Profitable Sustainability), the modernization of the plant nursery companies, and the investigation on innovative biodegradable pots (Eco-pot)", Acronym: IGAN (Italian Green Agri-Net) - ECO-POT.

The quality function f_q is then correlated to the selling price of the plants, as follows. A fixed price for the standard commercial functional unit is assumed (6.50 € if $\gamma = 0$, $\beta = 0$, based on personal communications with plant nurseries in Pistoia). A premium (r) is assumed for the selling price of bio-based materials (+10% of the selling price). A fixed price for the minimum quality ($q = 3$) is assumed as one third of the maximum price. The price function, f_p, is determined by linear interpolation, and the additional selling costs are therefore determined as difference between the selling price of each quality level and the selling price of the basis case ($\gamma = 0$ and $\beta = 0$), as shown in Figure 5.4c.

Summarizing, Table 5.5 elucidates selling prices (f_p) and additional selling costs (f_{sc}) for each range of share of compost (γ) linked to the 7-steps of the quality function (f_q) and for each choice of pot (β).

Table 5.5: Additional selling costs for the case study

	f_q		f_p		f_{sc}	
γ	$\beta = 0$	$\beta = 1$	$\beta = 0$	$\beta = 1$	$\beta = 0$	$\beta = 1$
0-0.19	5	5	6.50	7.15	0	-0.65
0.20-0.29	4.63	4.65	5.69	6.26	0.81	0.24
0.30-0.39	4.25	4.29	4.88	5.36	1.62	1.14
0.40-0.44	4.38	4.42	5.15	5.66	1.35	0.84
0.45-0.69	4.5	4.55	5.42	5.96	1.08	0.54
0.70-0.74	3.75	3.77	3.79	4.17	2.71	2.33
0.75-0.80	3	3.00	2.17	2.38	4.33	4.12

Finally, the additional costs function f_c is obtained by combining the additional selling costs function (f_{sc}) and the additional item costs function (f_{ic}), as indicated in Equation 5.10. The environmental function f_e and the economic function f_c are then depicted in Figure 5.5.

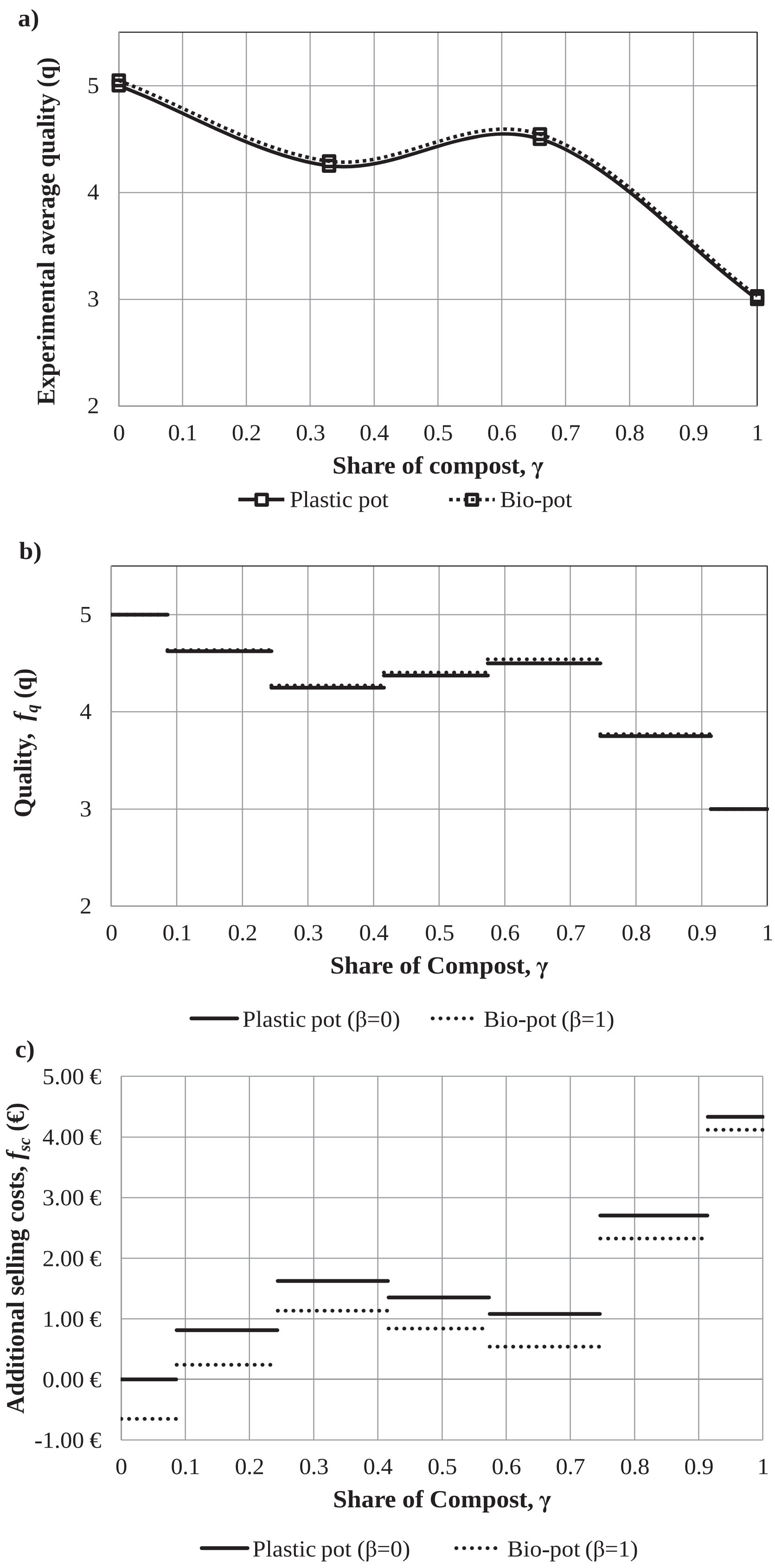

Figure 5.4: Correlation between share of compost γ and quality of a potted plant q: experimental points interpolated with linear function (a), discretization with piecewise constant function f_q (b), and additional selling costs function f_{sc} (c)

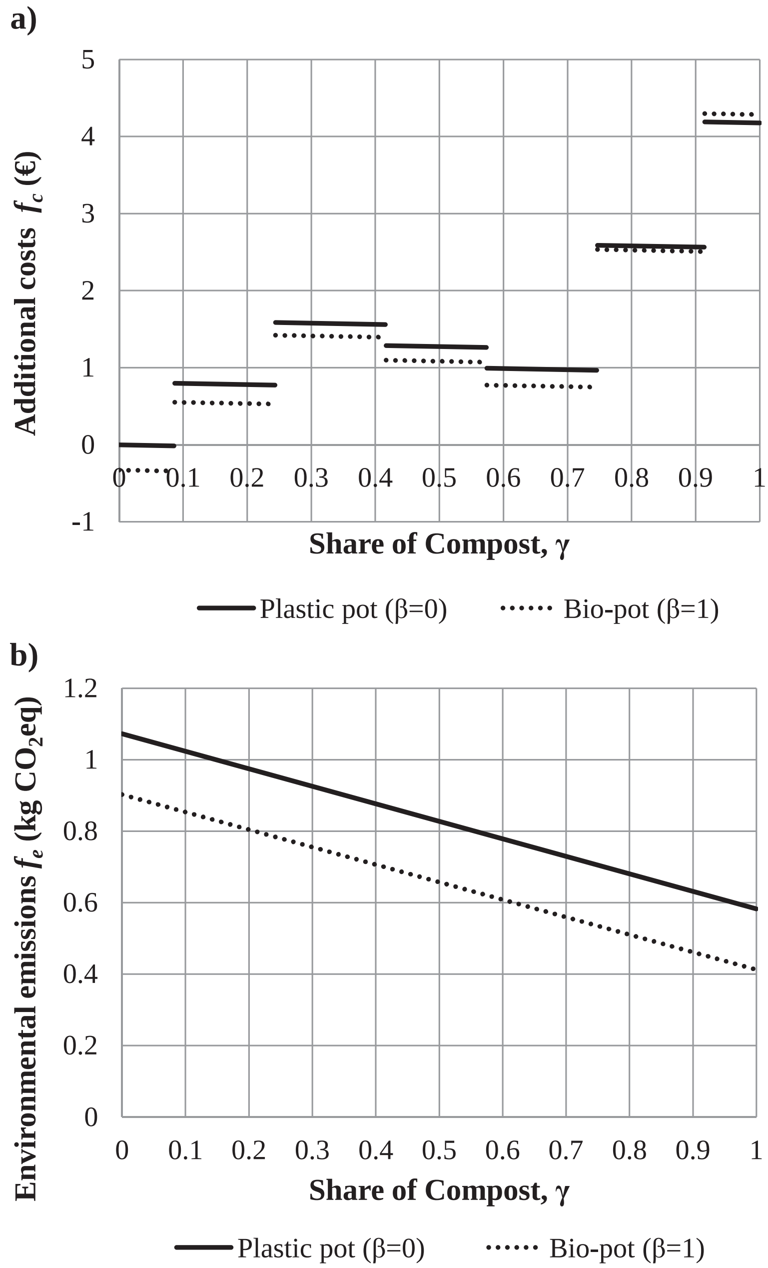

Figure 5.5: Additional costs f_c (a) and environmental emissions f_e (b) for the case study

5.3.2 Pareto optimization

In this thesis, to minimize both the environmental function (Equation 5.7) and the economic function (Equation 5.10), the optimization model is described as:

$$
\begin{aligned}
\max\ f_e(\beta,\gamma) \quad & = \gamma(a_2 - a_1) + \beta(b_2 - b_1) + a_1 + b_1 \\
\max\ f_c(\beta,\gamma) \quad & = f_{sc}(\beta,\gamma) + \gamma(c_2 - c_1) + \beta(d_2 - d_1) \\
& \text{subject to} \\
& f_q(\beta,\gamma) \geq 3 \\
& \beta \in \{0,1\} \\
& \gamma \in [0,1]
\end{aligned}
$$

where f_{sc} is described by Equation 5.8, and f_q is described by Equation 5.11.

A set of optimal solutions that at the same time minimize environmental emissions and additional costs can be yielded by performing a bi-objective model. The optimization of the objective functions f_e and f_c yields seven sets for the choice of the plastic pot ($\beta = 0$) and seven sets for the choice of bio-pot ($\beta = 1$), as shown in Figure 5.6.

The current situation (point ‘basis’, where $\beta = 0$ and $\gamma = 0$), located on the extreme left of the first qualitative level (if $\gamma < 0.09$ in this model) is located on the upper part of the graph. Five optima correspondent to the choice $\beta = 1$ are yielded. If $\gamma = 0.09$, the environmental impacts decrease (-20%), as well as the additional costs (-0.34 €), as indicated with point A (9% of the peat in the growth substrate is substituted by compost, and $\beta = 1$). The other non dominated solutions (points: B, C, D, E) provide a reduction in GHG emissions between 27% (point B) and 62% (point E), but they provide an increment of the additional costs between 0.53 € and 4.28 € (see Table 5.6 for an overview of efficient solutions).

Table 5.6: Pareto optima of the optimization model.
Notes: γ = share of compost, β = kind of pot, f_c = additional costs, f_e = environmental emissions, *basis* = current situation, before optimizing

	γ [0-1]	β (0,1)	f_c [€]	f_e [kg CO_2-eq]
basis	0	0	0	1.072
A	0.09	1	-0.34	0.860
B	0.24	1	0.53	0.783
C	0.75	1	0.75	0.537
D	0.91	1	2.51	0.455
E	1.00	1	4.28	0.412

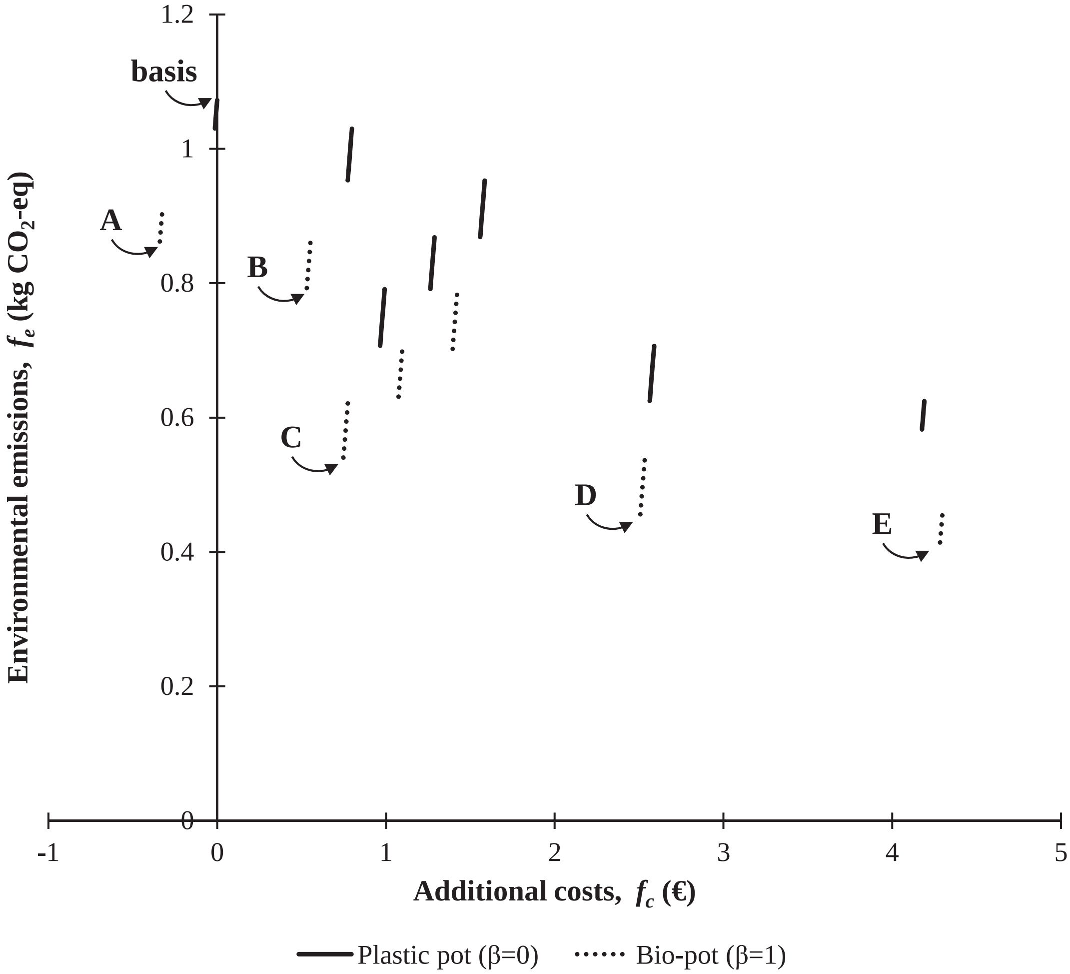

Figure 5.6: Pareto frontiers of the optimization model.
Notes: Efficient solutions are indicated with A, B, C, D, E. Reference: a potted plant in $5\,\mathrm{dm}^3$ container

These solutions represent efficient solutions to the decision problem. Nevertheless, these optima should be carefully evaluated by investigating different shapes of the quality function, different additional cost accounting, and different environmental emissions. In order to present a broader image of the outcomes, in the next section a sensitivity analysis that explores the variation of some input parameters is presented via scenarios. This analysis is helpful to investigate how the results can deviate from the non-dominated solutions already presented.

5.3.3 Scenario analysis

Input parameters used for modeling horticultural managerial problems are affected by fluctuation of quality, quantity, and changes in material/resource properties, but also by imprecisions of experimental data or measurements. These fluctuations can be called *parameter uncertainty*

(Lowe & Preckel, 2004; Geldermann et al., 2016b). For peat, for example, fluctuations of environmental emissions are related to the geographical definition of the boundaries, kind of experiments conducted to measure GHG emissions, assumptions, and methods for modeling. An investigation on fluctuation of these environmental parameters is required to present a scenario analysis. Therefore, minimum and maximum values of environmental emissions and additional costs that incur when substituting can be accounted and investigated.

The input parameters used in this section are chosen with some assumptions; literature and databases are used for the analysis to draw conclusions. These choices might steer the decision-making process, however, in a particular direction. To illustrate the effects on the results when changing such input parameters, a sensitivity analysis is here presented with different scenarios—this analysis results particularly interesting when facing with waste management, as described for LCA by Clavreul et al. (2012). The variations of input parameters are related to the environmental emissions and the additional costs, which yield Pareto sets of optimal solutions. The variation of input parameters for each scenario is described, as follows.

Five scenarios in this section are compared with the scenario with Pareto efficient solutions of the previous section (scenario S0, which is described by Table 5.6 and Figure 5.6). In particular, two scenarios explore modified technical parameters (from the ones indicated in Table 5.3) for the environmental function f_e: increased environmental emissions for the compost and bio-pots (scenario S1), and decreased environmental emissions for the peat and plastic pot (scenario S2). Two other scenarios explore modified technical parameters for the additional costs function f_c: no willingness to pay for bio-pots (scenario S3), and higher cost for compost or bio-pots (scenario S4). Moreover, a scenario (scenario S5) that investigates a different shape of the quality function f_q is presented to evaluate the influence of quality levels on the results.

Scenario S1 considers a different GWP of the compost (a_2) and a different GWP of the bio-pot (b_2). For the compost, maximum value of a_2 is accounted by considering maximum transport distance of raw materials and maximum fugitive GHG emissions. The transport distance was arbitrarily changed from 20 km (basis case) to 500 km, considered as maximum distance between the production site of the compost and the plant nursery. This calculation yields 16.9 kg CO_2-eq per ton of compost for the transportation. Fugitive GHG emissions are calculated considering maximum values of methane (15 kg CO_2-eq per ton of compost) and nitrous oxide (75 kg CO_2-eq per ton of compost) emitted during the composting phase. These emissions, upstream, and processing emissions of the basis, sum up to 114 kg CO_2-eq per ton of compost, which corresponds to a_2 as 0.150 kg CO_2-eq (+50% of a_2 indicated in Table 5.3). The GWP of the bio-pot is then arbitrarily increased (+30% of b_2 indicated in Table 5.3), therefore 0.403 kg CO_2-eq are considered here as b_2.

Scenario S2 takes into account different GWP for the peat (a_1) and different GWP for the plastic pot (b_1). 550 kg CO_2-eq emitted per ton of peat are considered as minimum value reported in the review by Boldrin et al. (2009), who cited the work of Brinkmann et al. (2004). Therefore, 0.398 kg CO_2-eq are accounted for a_1 (-33% of a_1 indicated in Table 5.3). The plastic pot emissions are arbitrarily decreased by 30%. From the value used in scenario S0 (0.48 kg CO_2-eq), here 0.336 kg CO_2-eq for b_2 are considered.

Scenario S3 evaluates no willingness to pay for bio-based pots, a real-world situation which affects the decision-making process whereas a higher price could influence purchasing behavior of customers. This scenario investigates a typical situation in business, since results of studies about willingness to pay for innovative, bio-based products can be debatable (see subsection in Section 4.1.1). This choice determines no difference between the additional selling costs f_{sc} of plastic pot and bio-pots, which are therefore identical in this scenario (i.e., f_p when $\beta = 1$ coincides with f_p when $\beta = 0$). This modification subsequently produces an increment of additional costs of bio-pots (in f_c).

In scenario S4, the costs of compost and bio-pots are arbitrarily incremented by 30%. Therefore, technical parameters c_2 (costs of the compost) and d_2 (costs of the bio-pot) are affected.

Scenario S5 explores a different quality function f_q, as shown in Figure 5.7. From scenario S0, this different f_q simulates specifically a decreased quality level (arbitrarily assigned), when 66% peat is substituted by compost. Subsequently, this scenario introduces a modified selling price function f_p and a different additional costs function f_c.

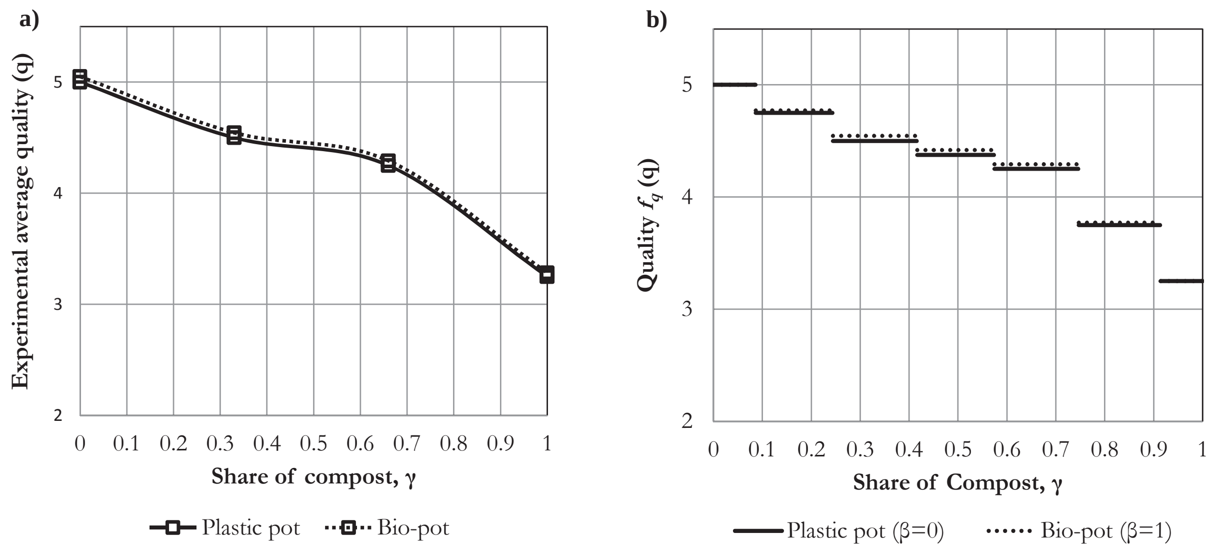

Figure 5.7: Modified experimental quality graph (a) and modified quality function f_q (b)

The modified technical parameters for the scenario analysis are listed in Table 5.7.

Table 5.7: Modified technical parameters for the scenario analysis.
Notes: a_1 = GWP of peat, a_2 = GWP of compost, b_1 = GWP of plastic pot, b_2 = GWP of bio-pot, c_1 = additional costs of peat, c_2 = additional costs of compost, d_1 = additional costs of plastic pot, d_2 = additional costs of bio-pot

Scenario	**GWP** [kg CO_2-eq]				**Costs** [€]			
	a_1	a_2	b_1	b_2	c_1	c_2	d_1	d_2
S0	0.59	0.10	0.48	0.31	0.19	0.03	1.08	1.40
S1	0.59	0.15	0.48	0.40	0.19	0.03	1.08	1.40
S2	0.40	0.10	0.34	0.31	0.19	0.03	1.08	1.40
S3 (different f_p)	0.59	0.10	0.48	0.31	0.19	0.03	1.08	1.40
S4	0.59	0.10	0.48	0.31	0.19	0.04	1.08	1.82
S5 (different f_q)	0.59	0.10	0.48	0.31	0.19	0.03	1.08	1.40

By using the input parameters presented so far (for scenario S1, S2, S3, S4, and S5), the sensitivity analysis yields the following *results* (see Figure 5.8 and Table 5.8), which are described as follows.

In scenario S1 (Figure 5.8a), the increment by 50% of a_2 (GWP of compost) and by 30% of b_2 (GWP of bio-pot) generate an increment of the environmental emissions for all Pareto optima of scenario S0. All optima shift here to higher environmental emissions with respect to scenario S0, i.e., increasing their f_e between 11% (point A) and 34% (point E). These results are directly influenced by the different technical parameters, but no particular conclusions can be drawn.

In scenario S2, by modifying a_1 and b_1 between 30% and 33%, all optima shift from scenario S0 to higher f_e, i.e., between 21% (point A) to 4% (point D), a part from point E which is not influenced by the initial choices (as expected, because a_1 and b_1 do not influence optima with $\gamma = 1$ and $\beta = 1$). And overall, these optima present lower values of GWP from the basis of scenario S0, between 36% (point A) and 62% (point E, as in scenario S0). This variation of the outcomes does not produce, nevertheless, high modification of the outcomes.

Following the considerations drawn for S1 and S2, Figure 5.8b shows scenarios where the technical parameters related to the additional costs function, f_c, are modified. Scenario S3 yields optimal points A, C, D, E located on the same extreme values of the relative qualitative levels—the point B in this scenario is not located on the Pareto frontier (detailed graphs for each scenario are reported in the Appendix). Here, the same GWPs of scenario S0 are yielded. Two other Pareto optima, indicated with point F ($\gamma = 0.75$) and point G ($\gamma = 0.09$), are related to the choice of plastic pot ($\beta = 0$). Point G, in particular, has f_e value as 1.03 kg CO_2-eq and the

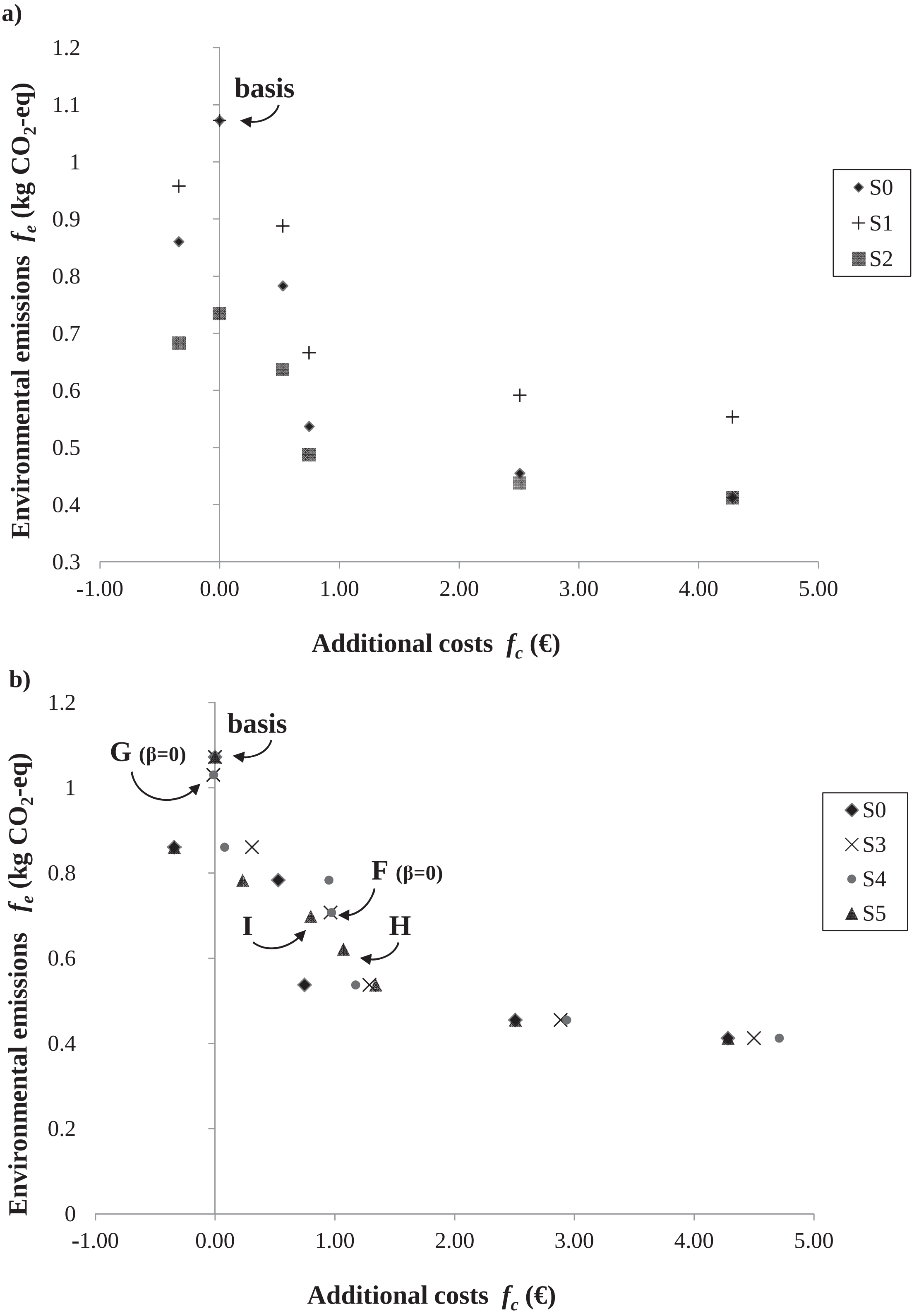

Figure 5.8: Pareto frontiers for the scenario analysis: scenario S1 (increased environmental emissions for compost and bio-pot) and scenario S2 (decreased environmental emissions for peat and plastic pot) (a); scenario S3 (non-willingness to pay), scenario S4 (increased costs for compost and bio-pot), and scenario S5 (modified quality function) (b)

Table 5.8: Pareto optima of the sensitivity analysis.
Notes γ = share of compost; β = kind of pot; f_c = additional costs [e]; f_e = environmental emissions, in GWP [kg CO_2-eq]
; *Basis* = current situation ($\beta = 0$, $\gamma = 0$)

	γ	β	f_e						f_c					
			S0	S1	S2	S3	S4	S5	S0	S1	S2	S3	S4	S5
Basis	0	0	1.07	1.07	0.73	1.07	1.07	1.07	0	0	0	0	0	0
A	0.09	1	0.86	0.96	0.68	0.86	0.86	0.86	-0.34	-0.34	-0.34	0.31	0.08	-0.34
B	0.24	1	0.78	0.89	0.64	-	0.78	0.78	0.53	0.53	0.53	-	0.95	0.23
C	0.75	1	0.54	0.67	0.49	0.54	0.54	0.54	0.75	0.75	0.75	1.29	1.17	1.34
D	0.91	1	0.45	0.59	0.44	0.45	0.45	0.45	2.51	2.51	2.51	2.89	2.94	2.51
E	1.00	1	0.41	0.55	0.41	0.41	0.41	0.41	4.28	4.28	4.28	4.50	4.71	4.28
F	0.75	0	-	-	-	0.71	0.71	-	-	-	-	0.97	0.97	-
G	0.09	0	-	-	-	1.03	1.03	-	-	-	-	-0.01	-0.01	-
H	0.57	1	-	-	-	-	-	0.62	-	-	-	-	-	1.07
I	0.42	1	-	-	-	-	-	0.70	-	-	-	-	-	0.80

same f_c value of the basis (current situation: $\beta = 0$ and $\gamma = 0$). In this scenario, it is interesting to notice that the additional costs due to the substitution range between 0.31 € (point *A*) to 4.50 € (point *E*). In particular, point *A* presents additional costs that were negative in scenario S0 (-0.34 €). This means that accounting willingness to pay for bio-based options may steer decision making and should be carefully taken into consideration. Moreover, two times of the additional costs in scenario S0 can be also accounted for point *B* (+1.10 € in scenario S3 versus -0.53 € in scenario S0).

Scenario S4 describes a situation where optima with share of compost less than 75% highly modified their values from scenario S0. In particular, point *A* is characterized by higher additional costs (from -0.34 € of scenario S0 to +0.08 €), point *B* increase its GWP (+80%), as well as point *C* (+57%). The other optima (*D*, *E*, and *F* that is located in the same position of scenario S3) do not present relevant alterations (between 10% and 17% from scenario S3). It can be noticed, that both scenarios S3 and S4 yield Pareto optima located closed to each others.

Scenario S5 shows a variation of additional costs from scenario S0 only for point *B* (-56%) and point *C* (+80%), and no changes can be observed for the other optima (*A*, *D*, *E*). Moreover, scenario S5 introduces Pareto optima with point *H* ($\gamma = 0.57$, $\beta = 1$) and *I* ($\gamma = 0.42$, $\beta = 1$)—see Table 5.8. In this scenario, the different quality function affects all the optima within the range of share of compost linked with the modified q values—and consequently modified f_p and f_c values.

Concluding, it is evident that accounting the variability of environmental emissions of peat and compost should be performed (see scenario S2 and S3), because their contribution to the GWPs can modify the outcomes. Moreover, willingness to pay for bio-pots (scenario S3) should be carefully considered, because its inclusion in the modeling could influence the results. An increment by 10% of the selling price of potted plants can indeed steer decision-making, and bio-alternatives can be preferable over the petroleum-based ones in some cases. Accounting of decision-relevant costs and their variation (scenario S4) is also important, and it can influence the results. Here, 30% variation produces a new efficient solution and can highly modify some of the optima (their f_c can be two times higher than the values reported in the initial scenario, S0). Finally, a variation of quality levels (scenario S5) can determine also a variation of results: a slightly different value of a particular quality level (-6% of the q correspondent to $\gamma = 0.66$) can determine a high variation of some optima (e.g., point C has 80% higher additional costs than the initial scenario, S0).

5.4 Summary

Summarizing, in this chapter a decision-making problem in horticulture has been presented, existing approaches in OR have been introduced, and a proposed original approach developed for this thesis has been illustrated.

Firstly, OR models in agriculture have been presented, which are linked to the research question: accounting of decision-relevant emissions in agriculture, determination of decision-relevant costs, and evaluation of quality levels within optimization models. A literature review of relevant works has presented these aspects. A research gap has been highlighted linked to this thesis.

Secondly, a proposal for an optimization approach is described. Decision variables for the share of compost and the choice of a plastic pot or a bio-pot have been defined, as well as two objective functions. The environmental function has been introduced with its dependency on the technical parameters, and the economic function has been described with its parts: the additional item costs (due to the substitution) and the additional selling costs. The latter, linked to the evaluation of the quality levels of a potted plant, has been defined with a step function. A quality function that includes additional selling costs and a function for the additional item costs due to the substitution are described in detail.

Thirdly, a case study has been presented with the technical parameters of the previous chapters, a case-specific quality function for the potted plant species *Photinia x fraseri* (which follows the investigations of a research project) has been defined, and correlations between the share of compost, the quality, and additional selling costs have been described with graphs. Pareto frontiers for the case study have been presented and a sensitivity analysis with five scenarios has elucidated the results with possible variations of the input parameters.

The next chapter will present general outcomes of the results of this thesis. Furthermore, it will introduce a discussion about the results, and it will suggest implications for the stakeholders of a horticultural supply chain where peat and plastic pots are substituted. Finally, further field of research is proposed.

6 Conclusions and outlook

In this chapter, the research question of this thesis is answered. Subsequently, the developed model that minimize environmental emissions and costs of a potted plant taking into account quality levels is discussed, and further investigation is proposed.

6.1 Conclusions

In this thesis, an optimization model for the minimization of environmental emissions and costs of potted plants is developed. The model yields Pareto efficient solutions that represent the optimal share of compost and kind of pot in a horticultural supply chain. The model is applied to a case study in Pistoia. By using the results of the previous chapters, the research question formulated in the introduction of this thesis can be here answered, as follows.

Can decision-makers of plant nurseries substitute innovative, bio-based materials for conventional ones and improve the environmental sustainability of potted plants and at the same time minimizing the additional costs?

Decision-making in horticulture is characterized by the presence of multiple choices of resources. Due to different kinds of production costs, uncertain labor supply, high international competition, limited access to global market, and scarce availability of land, water, and resources, decision-making in horticulture is a challenge (Zhang & Wilhelm, 2011). Bio-based materials can be petroleum-based substitutes if their properties match the requests of stakeholders (e.g., horticultural decision makers). In horticulture, there are many examples of resources used as peat substitutes (e.g., coconut fibers, by-products of the wood industry, see Section 2.2.1), and many examples of products used as petroleum-based pot substitutes (e.g., biodegradable and compostable plastics made of PLA, PHA, or TPS, see Section 2.2.2). However, these substitutes require experimentation on the field before being used. The quality of a potted plant could be indeed affected by the substitutes (see Section 4.2). Especially compost requires investigation due to the fluctuation of its properties (see Section 2.2.1).

Although reduction in GHG emissions can be achieved, substitutes for peat and plastic pots can increase additional costs in horticultural supply chains. Additional costs that incur when substituting, as seen in the model in Section 5.2, comprise additional item costs and lower selling prices. On the one hand, additional item costs incur if bio-pots are chosen, due their higher current price—low costs bio-pots with identical (or better) physical and chemical characteristics (e.g., tensile strength) than petroleum-based pots could change current horticulture management practices (see prospects for the future in Guerrini et al., 2017). On the other hand, lower selling prices incur because replacing peat with compost changes the agronomic quality of the plant, and the selling price of a potted plant depends on its quality—research might improve current horticulture settings, in order to produce compost obtained from by-products without making quality product worse of (see recent investigation by Mattei et al., 2017).

To answer to the research question, in this thesis the following steps are taken:

(a) A detailed analysis of peat and plastic pots and their substitutes has been conducted (Chapter 2).

(b) The environmental sustainability of conventional materials and their substitutes has been investigated via LCA (Chapter 3), followed by the analysis of their additional costs (Chapter 4).

(c) An OR-model has been then developed (Chapter 5), which has pointed out that the simultaneous minimization of decision-relevant GHG emissions and additional costs that incur when substituting can be achieved under particular circumstances. The results of the optimization model (see Figure 5.6) yield, only in one case, reduction in GHG emissions (−20%) and reduction of additional costs (−0.34 €): when substituting 9% peat with olive-mill waste compost and substituting plastic pot with bio-pot made of PLA. Other optimal non dominated solutions of the decision problem provide higher reduction in GHG emissions, but at the same time increment the additional costs. Additionally, scenario analysis presents a broader view of optimal solutions by varying input parameters used for modeling the decision-making problem.

(d) Furthermore, uncertainties are taken into account in the model by using a new concept of robust optimization, described in the following section.

By using the model developed in this thesis, for the first time growth substrate formulation in horticulture is optimized by minimizing decision-relevant environmental emissions, decision-relevant costs, and taking into account product quality. Modeling mixture optimization as a blending problem with by-products might improve horticultural planning of resources. Generally speaking, developing practical guidelines to improve an initial situation is not easy task in decision-making (Geldermann & Rentz, 2004b). By considering practitioners' perspective, indeed, many other factors like agricultural variables (e.g., weather influence on quality of plant) or biological variables (e.g., presence of pathogens on growth media) could significantly influence a decision.

6.2 Discussion and outlook

This thesis raises the question whether a **scale-up** of the results for a single potted plant might yield a drastic reduction in environmental impacts in the horticulture sector. What might happen if the whole Pistoia district, with its 5000 nurseries, would to modify their potting practices in favor of the more sustainable alternatives illustrated here? What would be the consequences on the carbon sink effect of the peatlands? Since the supply chain of peat is consolidated during the past decades, novel materials should undertake the approval of decision makers in horticulture. Therefore, a complete substitution cannot be foreseen. Although substituting for peat is an operational decision that can be immediately realized without building new nursery farm structures, if the whole horticultural sector in Pistoia abstains from using peat, it needs a sufficient quantity of peat substitute. The quantity of olive-mill waste compost, for instance, is related to the availability of sufficient by-products of the olive oil supply chain, which are yearly produced by olive-mill facilities during the harvesting period (two or three months per year). Moreover, although the method used for composting in this thesis (i.e., static-pile composting, see Altieri et al., 2011) is suitable to obtain mature and stable organic material, it requires on-site storage and proper logistics to distribute final compost to plant nurseries. The use of olive-mill waste compost for a larger horticultural supply chain needs, therefore, further investigation.

Another aspect correlated to the modeling presented in this thesis is the combination of **different disciplines**. This thesis accounts GHG emissions (useful to provide input data for the LCA), decision-relevant costs (which are nevertheless restricted to some commercial products and data retrieved from literature), and it takes into account agronomic quality of plants. These aspects are correlated to different disciplines, such as environmental engineering (e.g., experimental detection of GHG emissions), agronomy (e.g., evaluation of agronomic properties of growth substrate), and business administration (e.g., development of a blending model). Understanding which kind of approach has been used in this thesis is useful to determine future paths. Towards this end, definitions of different types of approach are reported, as follows (for a literature review, see Hadorn et al., 2008). According to Godemann (2006), *multidisciplinary* research "refers to a topic lying transversal to the disciplines. Different disciplines work on its different aspects with their respective methods", while *interdisciplinary* research "applies to a common problem that alludes to several disciplines and thus represent a disciplinary interface, [...] and new knowledge structures are established by the integration of different disciplinary perspectives, theories, and methods", and *trans-disciplinary* research refers to "problems outside the scientific world which may only be solved by scientists in cooperation with experts [practitioners in general] in possession of practical experience from outside the academic world". Nevertheless, 'true multi-disciplinary' can be achieved only bringing together experts from different disciplines (French & Geldermann, 2005). This dissertation addresses the problem of multidisciplinary research, since a combination of disciplinary particularities is pursued—in the sense of juxtaposition of disciplines, which remain separate, and "disciplinary elements retain their original identity, and the existing structure of knowledge is not questioned" (Klein, 2010). Taking a step forward from the multi-disciplinary approach of this thesis, further trans-disciplinary research might be used to bridge the gap between academia and industry.

This thesis evaluates the fluctuation of input parameters of the model via scenario analysis. This is a sensitivity analysis that is used to determine the influence of inputs into the results (see Section 3.3). The **uncertainties** that are introduced in the model are related to the nature of organic materials. In particular, carbon-based growth media decompose over time, thus contributing GHGs to the atmosphere. Modeling these contributions is complicated due to the varying of fractions of organic content and differences in the properties of peat and compost. Although oxidation rates could be obtained via laboratory analysis of the specific growth media components, they are often taken from the literature. Generally speaking, the quality of emissions modeling depends on the input data. Taking into account measurement errors, fluctuations in conditions for preparing growth substrate, and imprecise horticultural data by using value intervals are an option (Lowe & Preckel, 2004). In renewable resource processing, two types of uncertainty are typical: parameter uncertainty and decision uncertainty. *Parameter uncertainty* is introduced in the model by expressing costs and environmental emissions as ranges (or intervals), rather than as exact numbers (Geldermann et al., 2016b). *Decision uncertainty* occurs whenever solutions of a decision problem are not implemented exactly as calculated, but only within a range of the computed values. It is common in horticulture, that the targeted share of compost could not be implemented as planned, due to imprecisions in the mixing process. Growth substrate is indeed prepared for many potted plants at one time, and the mixing is done roughly in big heaps using skid-steer loaders—this is an example of decision uncertainty. In this situation, robust optimization provides a suitable framework for decision support, since it seeks solutions that are feasible for all scenarios and that perform best with respect to the worst case (Bertsimas & Sim, 2004). Robust optimization is especially suitable for problems in which the uncertainty set (i.e., the set of all scenarios that may occur) is known, and where knowledge about a probability distribution on the uncertainty set is not given (Bertsimas & Sim, 2004)—the latter is a typical situation in horticulture. Robust optimization has been applied to various problems in agriculture. For example, Bohle et al. (2010) applied it to the scheduling of wine grape harvesting in Brazil; Doole (2012) applied it to compare deterministic and robust approaches to evaluating perennial pasture species in Australia by using uncertain parameters within intervals; Munhoz & Morabito (2014) applied it to orange juice production; and Hombach et al. (2017) applied a robust and multi-objective optimization model with integrated risk attitude to a biodiesel supply chain in Germany. The decision problem of this thesis in presence of uncertainty is investigated by Krüger et al. (2018), following the approach by Krüger (2018). By investigating maximum hypothetical deviation for which the quality level of a potted plant does not deviate from the expected level, the authors highlight that including uncertainty in the problem leads to solutions that are robust against deviations in parameters and decision variables. Future research into horticulture with uncertainty could include combining different pot capacities, varying resources (both growth medium and pot), investigating different safety margins, and solving the problem for plant species while simultaneously taking into account that the supply of compost is ultimately limited. Including these parameters would help promote a strategic managerial perspective in horticulture, which is currently managed on the basis of tradition. This analysis implies that the used approach might be applicable to other fields having a broader class of uncertain optimization problems.

The model presented in this thesis for potted plant cultivation optimization could be applied to other case studies. The same optimization approach can be adopted, for instance, for any horticultural species defined by quality functions (see Section 5.3.1), for other field crops (e.g., wheat, cotton, rice, corn, or seeds), or for other specialty crops (e.g., fruits, vegetables, or grapes). Other aspects that could be further investigated are: broader evaluation of case-specific willingness to pay for bio-based resources, elaboration of quality grades for different species with several parameters, weighting the results, investigation on end-of-life scenarios with case-specific input data, use of case-specific geographical data, and consideration of several agronomic parameters that influence plant cultivation, such as fertilizing, irrigating, or using pesticides. Additionally, many other environmental impacts indicators could be taken into account (e.g., land use impact indicator). The research of this thesis could be further extended to areas whereas the demand for substitutes for petroleum based materials grows, in order to accelerate the transition towards sustainable, post fossil-carbon societies (Ingrao et al., 2016). In particular, case studies that include environmental and economic parameters could be modeled following the indications of this thesis. The following general questions might also arise: What would be the impact if the food industry (which can be affected by uncertainty in the exact recipe of ingredients) were to switch all products at the same time to more sustainable alternatives (e.g., biodegradable packaging) by modifying current supply chains? What role will bioplastics play in the future? To the last question, arose in 2002 by (Stevens, 2002, p.145), still no concrete answer can be given, in particular in agriculture. In the long term, increasing biomass production for producing agricultural bio-based plastics might still raise environmental concern. However, biotechnology research and by-product reuse could introduce novel environmental-friendly products with low additional costs into the market.

7 Summary

Fossil resources substitutes, such as bio-based materials, can be used in a horticultural supply chain. However, their use require the analysis of environmental and economic aspects, in order to make decisions. To define a support for decision-makers of plant nurseries, an OR-model has been developed in this thesis and applied to a case study. This dissertation can be resumed, as follows.

Chapter 1, which introduces the research question and background of the research, is followed by Chapter 2, which elucidates core concepts of this thesis and foundation for the development of the model. The chapter introduces the supply chains of peat, plastic pot, compost, and bio-pot. Here, paths for reducing GHG emissions and additional costs when processing are described. As a result, an overview of technical solutions of the decision problem is given.

Chapter 3 illustrates case-specific technical solutions of the decision problem and theoretical background and steps of LCA: (1) *goal and scope* step illustrates peculiar aspects linked to organic resources, (2) the *inventory* step defines sub-phases of the life cycle of resources, (3) the *impact assessment* evaluates the environmental effects of GHG emissions by using environmental impact indicators, and (4) the *interpretation* of LCA results is described. This background on LCA is enriched with the experimental detection of GHGs emitted into the atmosphere by carbon-based resources like peat, compost, and bio-pots. Three kind of analysis (i.e., in situ, simulation, laboratory) are described to define their field of application. Experimental analysis is used in this thesis to estimate fugitive GHG emissions, to predict the end-of-life, and to determine the carbon storage effect. Then, LCAs of Baltic sphagnum peat and olive-mill waste compost from static-pile procedure are analyzed, as well as LCAs of plastic pot made of PP and bio-pot made of PLA. Decision-relevant environmental impacts (in terms of GWP) of growth substrate (made by peat and/or compost) and plastic pot (made of PP or PLA) are finally compiled for a potted plant, reference of the optimization model.

Chapter 4 introduces cost accounting, LCC, and combination of environmental and economic aspects. Then, a general procedure that limits the analysis to decision-relevant costs is described, followed by an introduction on willingness to pay for environmental-friendly resources. Quality assessment in horticulture is then illustrated with a literature review, and a scheme for assessing qualitative classes for potted plants is given with a common scale. The chapter concludes with the determination of decision-relevant additional costs when substituting for peat and plastic pots.

In Chapter 5, a literature review on OR methods and blending models is reported, followed by the development of the decision-making model with its decision variables, constraints, and objective functions (i.e., environmental and economic). By using additional item costs (which incur when substituting PP pot with PLA pot) and lower selling prices (which incur when substituting peat with olive-mill compost), an economic objective function is defined. A quality function describes variation of the selling price of a potted plant when substituting for peat, by means of quality levels. Then, the model is applied to a case study in Pistoia, by using decision-relevant emissions and decision-relevant additional costs determined in Chapter 3 and Chapter 4 for a potted plant of the species *Photinia x fraseri*. Pareto optimal solutions are yielded, and a scenario analysis investigates the variation of the outcomes by using different input parameters, which are based on literature and case-specific fluctuation of input data. A broader image of optimal solutions is therefore given.

Chapter 6 discusses the results of the optimization model by answering to the research question formulated in Chapter 1. A discussion on further scale-up of the results, case-specific limitations of the model related to the use of olive-mill waste compost, multi-disciplinary research, and modeling with parameter and decision uncertainty in horticulture settings is reported. Further fields of research are then suggested to extended the model.

In this thesis, for the first time, growth substrate is optimized and kind of pot is chosen to reduce GHG emissions and additional costs of a potted plant, by taking into account its agronomic quality. Towards this end, in this thesis, agronomic properties are investigated, LCAs are performed, supply chains of technical solutions of the decision problem are examined, and a bi-objective model is developed. The decision problem is then addressed using Pareto optimization, and, in presence of uncertainty, by means of robust multiobjective optimization. The outcomes of this thesis can provide support for managerial decisions for a horticultural supply-chain.

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