Alternative seagrass wrack management practices in the circular bioeconomy framework: a life cycle assessment approach

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Abstract

Despite providing important ecological functions, seagrass accumulation causes environmental and economic issues, including eutrophication and tourism reduction. Nowadays, seagrass wrack is commonly removed from the beaches and landfilled. In this study, alternative management strategies for seagrass valorisation, including anaerobic digestion (AD), composting, ecological restoration, and landfill, were considered using a life cycle assessment (LCA) perspective. The aim was to provide a robust evaluation method for public and private companies, allowing them to reduce the environmental impacts and fulfil the circular bioeconomy principle. An economic assessment was conducted to give further insights regarding the applicability of the proposed solutions, considering both direct and indirect impacts with a life cycle costing (LCC) approach. A selected beach located in the Northeast Mediterrean Sea was considered as a relevant case-study. The results highlighted a decreasing seagrass production in the analysed area (mean values of 267 Mg/km shoreline). A significant amount of inerts (up to 62.4% of the total mass) were present in the collected seagrass wrack. The environmental impacts of the seagrass management scenarios were evaluated with the method ReCiPe 2016 H, using both midpoint and endpoint levels. LCA results showed that the impacts of the alternative management strategies were in all cases significantly lower than the current landfill strategy, -70% considering the categories of human health, ecosystems and resources, and -95% considering global warming potential category. The LCC analysis proved that composting was the best alternative (NPV>1.27 M€), due to lower operating costs and higher fertilizer value. Tourists' reduction rates higher than 0.20% negatively impacted the economic balance for the municipality, so giving proper information to the stakeholders appears fundamental when adopting restoration solutions. The obtained results can help beach management companies and public administrations to select the best operational strategies to reduce the environmental and economic impact of seagrass collection and treatment.

1. Introduction

Seagrass meadows are important habitats that provide significant ecosystem services to our planet, including carbon storage, support of world fisheries production and prevention of beach erosion (Unsworth et al., 2019). When marine seagrasses lose their old leaves, part of seagrass production is exported to the adjacent beaches, where seagrass wrack accumulates (Cucco et al., 2020). The effects of seagrass wrack lying onshore in rotting piles are multiple and include the hindering of tourism in the affected areas (Corraini et al., 2018), as well as the contribution to greenhouse gas (GHG) emissions generation (Liu et al., 2019). On the other hand, seagrass wrack is an important component of coastal environments bringing ecological benefits (Vacchi et al., 2017) such as providing food and habitat to sandy beach fauna (Ince et al., 2007), protecting coastal dunes and supplying nutrients for vegetation (Del Vecchio et al., 2017).

In many administrative districts with touristic interest, it is very common to remove the material from the beach, although this practice is often associated with the use of heavy machinery (Simeone et al., 2013), a significant regression of the shoreline and a strong impact on the beach morphology (Boudouresque et al., 2016). Beach replenishment with traditional coastal engineering operations is the most common solution to compensate for beach regression. These techniques are expensive, provide unstable and temporary solutions, disrupt natural sediment transport, and can have negative impacts on proximal seagrass meadows (James et al., 2019) by slowing down their recovery and impacting sediment features (González-Correa et al., 2008). As observed by (Leiva-Dueñas et al., 2021) highly anthropized coasts can have a negative influence on long-term seagrass production with ecological impacts onshore and offshore. Hence, it is important to find appropriate and sustainable seagrass management strategies which can ensure a balance between the ecological benefits and the ecosystem services provided by the biomass left onshore, on one hand, and the social and economic interests behind its removal, on the other hand.

For sustainable management, a paradigm shift should be made, considering seagrass wrack no longer as a waste, but as a resource. Seagrass leaves have been used as a resource throughout history (Emadodin et al., 2020), but the interest in harvesting beach-cast seagrass wrack to produce biogas and fertilizers has been growing only in recent years due to technological and social advancements. Recently, the material has been studied to unveil its potential for biogas production in anaerobic digestion (AD) process (Balata and Tola, 2018), showing a good methane potential and a low concentration of heavy metals in the digestate (Misson et al., 2020), which can therefore be reused in agriculture. As an alternative to AD, seagrass wrack can be used as raw material for composting in combination with other organic materials (Cocozza et al., 2011). As an example, seagrass-based compost showed good performances in green-house tomato and lettuce cultivations (Grassi et al., 2015), and as a replacement for mineral fertilizers and peat in potted basil production (Mininni et al., 2015; Parente et al., 2014). Moreover, biochar production from seagrass wrack has also proved to be a promising and climate-friendly alternative material use (Macreadie et al., 2017). Notably, the use of marine-based organic fertilizers could reduce the use of chemical fertilizers and their associated environmental impacts (Emadodin et al., 2020).

Life cycle assessment (LCA) is a widely known and standardised methodology used to investigate the impact of a product or technology over its lifetime (Muralikrishna and Manickam, 2017). Due to its wide applicability in different technological fields, LCA has become a core topic in the environmental management field to support the choice between different technological options according to their environmental impacts (Lewandowska et al., 2013). By covering and modelling all the activities related to a product or a function, LCA aims to avoid problem shifting (Simonen, 2014). In the field of organic waste valorisation, several studies presented applications of the LCA methodology to alternative management systems such as AD or composting in order to compare the environmental performances of these valorisation scenarios to landfilling (Bernstad Saraiva Schott et al., 2016; Evangelisti et al., 2014; Slorach et al., 2019). In terms of substrates to be treated, most of the papers were focused on food waste (Lamnatou et al., 2019; Yu et al., 2020), garden waste (Lee et al., 2020) or organic residues of industrial processes (Batuecas et al., 2019; González et al., 2020; Timonen et al., 2019). To the best of our knowledge no papers were published applying LCA to alternative management systems alternative management systems for seagrasswrack.

While environmentally focused LCAs are crucial to determine the environmental impacts of processes, they do not address the economic aspects which can strongly influence decision-making (Estevan, 2018). Considering this framework, the application of LCA and economic analysis as two complementary assessment methods can be a robust tool to properly evaluate between several alternative investment options. The economic aspects can be assessed using a life cycle costing (LCC) methodology, which evaluates the life cycle costs of a product, process, or system. In the LCC methodology, two types of costs apply: internal costs, which a directly involved stakeholder pays, and external costs, or ‘externalities’, which represent the impacts indirectly ‘paid’ by the environment or society (Rebitzer and Hunkeler, 2003). Externalities may be considered when performing a LCC assessment “provided their monetary value can be determined and verified” (Estevan, 2018). For situations involving ecosystems, where the economic aspects are a main driver behind decision making, it is important to consider the inherent value of the services that the ecosystem provides. The valuation of the environment has been attempted for decades and is ethically controversial (Beder, 1997). Nonetheless, such valuations have merit in stressing the importance of ecosystem services and highlighting the risks associated with the degradation of those ecosystems (Picone et al., 2017).

In this study, three alternative seagrass wrack management scenarios to current landfill conferral were considered for the coastline of Grado Municipality (Italy), evaluating their different impacts from an environmental and economic perspective using a life-cycle approach. The scenarios included the following management strategies: AD, composting, ecological restoration. To the best of our knowledge, this is the first study which assesses in a thorough manner the environmental and economic impacts linked to seagrass management options. It can be a useful tool for beach managing companies and public authorities stimulating a paradigm shift from waste to resource, following circular economy and sustainability principles. The obtained results can be exported to different locations throughout the world, considering that seagrass wrack represents an issue to be tackled in many touristic areas, and can be integrated with other technical scenarios. Energy and resource recovery from seagrass can help in fully exploiting its natural and technical features, leading to an overall environmental improvement and to ecosystem restoration, together with generation of useful energy vectors or recovered material for agricultural purposes.

2. Materials and methods

Section 2.1 describes the case study, including seagrass wrack characteristics, sampling procedure and composition analysis. Section 2.2 illustrates the investigated scenarios for seagrass management, while Section 2.3 is focused on the adopted LCA methodology. Finally, Section 2.4 describes the implementation of the LCC analysis.

2.1 Seagrass characterization

2.1.1 Area under study

The investigated area was the coastline of Grado Municipality, overlooking the Adriatic Sea (Friuli Venezia Giulia, Northeast of Italy). A beach length of about 1.6 km was considered in the analysis. The seagrass meadows in the area were composed of different patches of *Cymodocea nodosa* (51.2%), *Zostera marina* (28.6%) and *Nanozostera noltii* (20.2%) (Misson et al., 2020). The beach managing company provided data on the average amount of produced seagrass wrack in the period 2004-2020 (Fig. S1), together with operating data related to seagrass collection and disposal.

2.1.2 Sampling procedure and composition analysis

The sampling campaign lasted from winter 2015 to summer 2018. The data collection activities were diversified between summer and winter periods due to the different beach management practices throughout the seasons. During the summer period, 5 samples of seagrass wrack were collected once per week for each one of the 7 selected shore sectors. The sampling activities were carried out at dawn before cleaning operations. Instead, during the winter season, the material deposited by the sea was kept on-site to protect the beach. In these months, the sampling was performed along fixed transects perpendicular to the shoreline once every two weeks. Transects' position was determined according to the abundance of material deposited by the sea over the months.

The collected material was transported to the laboratory without delay and carefully washed, avoiding dispersing the sand present in the samples. Subsequently, the samples were visually sub-divided into the following categories: sand, wood, seagrasses, algae, other (e.g. shells, plastic, stones). Seagrasses were isolated from the rest of the material and taxonomically identified. The samples of seagrass green litter were cleaned by epiphytes and washed with deionized water. All the wrack categories were weighted, dried at 30°C until constant humidity was reached and weighted again to determine the relative moisture content. The detailed physicochemical characterization of the seagrass material was reported in (Misson et al., 2020) and (Misson et al., 2021). The total seagrass wrack mass (expressed as t/y) was finally split into different classes (Tab. S1) to draw up the mass balance. The mean composition of the collected seagrass wrack throughout the year was: 49.7% sand, 35.7% seagrass, 7.6% algae, 5.9% wood, 1.7% other.

2.2. Description of seagrass management scenarios

According to the Italian legislation, seagrass residues, like other beached materials, are classified as urban waste and can be discarded as such if necessary. The seagrass managing route for the selected beach nowadays includes material landfilling in Slovenia, as this material is still considered as waste. The distance between the beach and the selected landfill was 199 km; in addition, no biogas collection and energy recovery were applied in the landfill site. Landfill represents the current management scenario (S0_L) for seagrass wrack.

Alternatively to this management option, three alternative scenarios for seagrass valorisation were defined: (1) anaerobic digestion (S1_AD); (2) composting (S2_C); (3) ecological restoration (S3_ER).

2.2.1 Seagrass collection and transportation

Seagrass collection was a necessary phase for all the considered management scenarios and thus represented the first treatment step. The annual amount of collected seagrass wrack on the beach was 1,985 t in the years 2019-2020. Mean seagrass collection over a longer period (2004-2020) was calculated as 3,297 t/y (Fig. S1).

The material was collected every day (3 hours per day in the early morning) for about 6 months per year (from April to October), to keep the beach clean for tourists. The material was transported with truck, and each truck was supposed to be filled with the seagrass wrack and directly sent to the processing plant. Seagrass collection in the winter was forbidden, as it helped to preserve the beach (Misson et al., 2020). Beach cleaning was carried out using heavy mechanical equipment run by non-renewable energy sources (diesel fuel).

The total number of truckloads throughout the collecting season was calculated based on interviews with the cleaning company: considering that seagrass consistently accumulated in the winter period, an average of 15 trucks were filled during the first beach cleaning operation (typically executed in April). Subsequently, 1 full truck of material was collected every day until the end of the season (October). A total yearly number of 195 trips were thus considered for the following analysis and calculations. Given the extremely low density of the collected seagrass (Oldham et al., 2014), a compaction factor of 4, coherent with the commonly used mechanical equipment installed on waste collecting trucks, was supposed to allow increasing material’s density before treatment.

2.2.2 Scenario 1: Anaerobic digestion (S1_AD)

The material was assumed to be transported to an existing municipal wastewater treatment plant (WWTP) located nearby (distance of 22 km). Co-digestion with sewage sludge in the existing anaerobic digester was considered as the main operating scenario. Seagrass wrack treatment included the following steps: washing, grinding, heating, AD. The washing phase was aimed at eliminating sand and other inert material; a removal efficiency of 97% was considered, consistent with the authors' experience in the field. The wastewater (including the inert material) produced by the washing process was sent to the WWTP water line, while the washed seagrass material was grinded. Primary sedimentation in the selected WWTP was supposed to remove sand and inerts from the produced wastewater.

The electricity needed for grinding was calculated considering commonly available commercial devices (Monster Industrial, 2016). 320 days of operations throughout the year and 4 operating hours per day were assumed for the shredder.

Regarding the digester, the thermal energy request was calculated considering 2 distinct contributions: the energy needed for heating the incoming material to the desired mesophilic temperature (35 °C), and the energy requested to balance the losses throughout reactor walls. These two terms were calculated as proposed by (Cottes et al., 2020), considering the real geometric dimensions of the reactor, while the specific heat coefficients for seagrass and sludge were taken respectively from (Caldana, 2012) and (Cottes et al., 2020). The mean environmental temperature was obtained from Regional databases (Osmer FVG, 2020). Biogas yield from seagrass was calculated considering the results of laboratory scale trials (Misson et al., 2020), which showed that the salinity included in seagrass did not substantially alter the kinetics of anaerobic archaea. A 55% mean percentage of CH_4 in the produced biogas was considered in accordance to other studies (Brudecki et al., 2015; Lafratta et al., 2020). The AD biogas was used in a combined heat and power (CHP) system to produce the thermal and electric energy needed for the AD treatment. Only the surplus electricity was exported to the national electricity grid, while heat was only used for the AD reactor.

The digestate production was estimated from the amount of input material, considering 20% reduction in the overall mass, coherently with other studies (Pognani et al., 2012) and WWTP managers' experience. Digestate mechanical dewatering was supposed to be accomplished by using a centrifuge, in accordance with common WWTP operations. A final content of 25% total solids (TS) in the dewatered sludge was considered; the electricity consumption was calculated considering the available commercial devices (Hubertech, 2021). Nutrients (i.e., N and P) distribution between the solid and liquid phases was estimated as done by (Barampouti et al., 2020).

Digestate composition, especially nutrient content, was obtained from (Angelidaki et al., 2017), considering that seagrass digestion showed not to significantly alter heavy metal concentration in the digestate, when compared to sewage sludge alone (Misson et al., 2020).

2.2.3 Scenario 2: Composting (S2_C)

In the second scenario, the seagrass wrack was supposed to be transported to an existing composting plant, treating the organic fraction of municipal solid waste (OFMSW) (distance of 35 km). Co-composting with OFMSW was assumed to be the main operating strategy for the waste management company, considering that seagrass proved to be potentially usable as soil fertilizer in agriculture (Grassi et al., 2015), also in co-composting processes (Provenzano et al., 2015). Seagrass washing and grinding phases were kept the same as described for the AD process in S1_AD (Section 2.2.2). Composting in piles was selected as the main stabilization technique, due to the reduced electricity request in comparison with closed reactor processes (Diaz et al., 2007).

Compost production was estimated from the amount of input material considering a yield of about 20% from the input seagrass (Oldfield et al., 2016). Nutrient amounts in the seagrass compost were obtained from a previous study (Grassi et al., 2015). As for the digestate, it was assumed in addition that seagrass composting did not significantly alter heavy metal concentration in the compost, when compared to OFMSW alone.

2.2.4 Scenario 3: Ecological restoration (S3_ER)

In the third scenario, it was hypothesized to collect only part of the produced seagrass wrack (50%) throughout the season, to restore the original beach habitat, and beach replenishment activities were halved compared to the other scenarios. The residual part of the material was assumed to be treated through AD. In this scenario, the GHG emissions from material's degradation were calculated using the experimental assays conducted on the beach. The mean carbon fraction emitted to the atmosphere was 33% of the initial C content reported by (Misson et al., 2020). CO_2 generation was subsequently calculated considering a ratio of 12% between the produced CO_2 and the measured C loss, as reported by (Liu et al., 2019). Temperature showed to be the main factor affecting the degradation process. GHG emissions were estimated also considering CH_4 contribution: the total CH_4 emission from the analysed seagrass was obtained from the composition analysis (Section 2.1.2) and from the emission data summarized in (Misson et al., 2021).

2.3. Life Cycle Assessment

The LCA study was carried out in compliance with ISO standard (International Organization for Standardization (ISO), 2006a, 2006b) requirements. The following subsections describe the goal and scope definition, the inventory and the impact assessment.

2.3.1 Goal and scope definition

The main goal of the LCA analysis was to quantify the environmental benefits obtainable through adopting alternative seagrass management strategies to the current scenario (landfilling) as well as to compare the environmental performances of the three alternative scenarios (AD, composting and ecological restoration) in order to identify the most environmentally sound management system. To provide a better comparison between the treatment systems, the functional unit (FU) of the study was defined as the treatment of 1 t of collected material on the beach, having the composition reported in Table S1.

All the life cycle phases in each management scenario were considered in the system boundaries of the study, including material collection and transportation, system construction and operation, and product valorisation.

Figure 1 shows a schematization of the system boundaries for Scenario 1 (S1_AD) and Scenario 2 (S2_C). For S3_ER (ecological restoration scenario), the same system boundaries and the same assumptions as in the AD scenario were considered for the fraction of collected material, while only GHG emissions were analysed for the residual part of the material left on the beach.

Regarding the current management scenario (landfill), material collection and transportation, and successive landfilling (without production of valuable by-products) were the only considered phases. No material pre-treatment was necessary, since the seagrass wrack is directly landfilled as it is.

FIGURE 1

For the three alternative scenarios (S1_AD, S2_C, S3_ER), the production and use of all the resources required by all the phases of seagrass treatment process were included in the analysis, as well as the final disposal of the produced waste and wastewater.

Regarding the energy requirement for AD and ecological restoration scenarios, biogas combustion in the CHP system was considered. However, the electricity production of the CHP system was shown to be higher than the treatment process needs, and therefore the amount of electricity surplus, normally withdrawn from the electricity grid, was considered as an avoided product.

For the composting scenario (S1_C), the production of the electricity needed for the treatment process was considered, assuming that the composting plant obtained the electricity from the national grid. Seagrass composting and AD allowed to obtain valuable final products, respectively compost and solid digestate that, used as fertilizer, could allow to reduce mineral fertilizer production. In this study, the substitution of inorganic fertilizers (as avoided product) was considered although the application of these organic fertilizers into the soil was not taken into account.

The environmental emissions due to the composting and AD processes were considered as well, as the emissions produced by the seagrass wrack left on the beach (only for S3_ER).

Furthermore, for all the alternative scenarios, the environmental emissions due to energy production and usage were also included in the analysis.

2.3.2 Inventory

Life cycle inventory is the second phase of the LCA methodology, where all the data are collected and expressed in terms of the selected FU. The scenario modelling was performed using SimaPro 8 (Pre Consultans, Amersfoort, The Netherlands) software.

Primary inventory data about seagrass wrack characteristics, material collection and transportation, main treatment process parameters, were provided by laboratory analysis, interviews with experts and analytical calculations. Ecoinvent database v.3 was the main source of the background data used for the modelling of plant infrastructures and equipment, but also for the production of raw materials, fuels and vehicles used for material collection and transportation. Scientific literature, instead, was the main data source regarding the environmental emissions originating from the treatment processes and biogas combustion as well as to assess the amount of mineral fertilizers substituted by the solid digestate (S1_AD) and the compost (S2_C). GHG emissions due to the seagrass wrack left on the beach (only for S3_ER) were estimated from the values reported by (Misson et al., 2021).

Table 1 reports the primary data about seagrass wrack characteristics in terms of composition and density (before and after material compaction on collecting vehicles) and seagrass beach collection operations.

TABLE 1

Tables S2-S4 report the main inventory data about inputs and outputs of the S1_AD, S2_C and S3_ER management scenarios. The modelling of vehicles for material transportation to the treatment plants was performed adopting Ecoinvent database processes, considering freight lorries of 7.5 – 16 metric tons. The vehicle's size was selected by considering the daily material amount to be transported (about 10.3 t/d). The vehicle’s emissions were estimated calculating the average values of Euro 4, Euro 5 and Euro 6 lorries.
The modelling of the infrastructure and the equipment of the treatment plants was performed adopting the data of Ecoinvent database processes and considering plants with a treatment capacity of 10,000 t/y and a lifetime of 25 years (for the infrastructures) and 10 years (for the equipment).
No primary data was available for the modelling of seagrass landfill scenario (S0_L); therefore, landfill conferral was modelled using data and information from Ecoinvent database v.3 processes about the sanitary landfill of organic waste. The Slovenian energy mix was adopted regarding the energy requirement; furthermore, regarding the amount of leachate produced and treated, only the organic fraction of material was considered, thus excluding the inert fraction (as sand) to avoid an overestimation.

2.3.3 Impact assessment

The estimation of the potential environmental impacts of the diverse seagrass management scenarios was carried out adopting ReCiPe 2016 evaluation method with hierarchist perspective (H) using both endpoint and midpoint approaches. This is the most up-to-date and widely used method by LCA practitioners (Slorach et al., 2019); the H perspective is based on the most common policy principles concerning the time frame and other relevant issues (Huijbregts et al., 2017).

The midpoint level (problem-oriented) contains 18 impact categories, while the endpoint level (damage oriented) considers 17 categories grouped into three macro-categories: damage to human health, damage to ecosystems and resources consumption (De Feo and

Ferrara, 2017). The analysis at midpoint level was mainly focused on the most relevant midpoint categories, selected adopting the same procedure reported by (Ferrara and De Feo, 2020), i.e. those that provided the greatest contribution (at endpoint level) to the three macro-categories of the method. The selected midpoint categories were global warming, fine particulate matter formation, human carcinogenic toxicity, terrestrial acidification, land use and fossil resource scarcity.

2.4 Life Cycle Costing

The LCC analysis was performed in two parts: in the first step, only direct costs and revenues associated with the different management options were considered. The same processes were considered as done in the LCA. A combination of primary data, secondary data and calculations using the LCI inputs and outputs were used to calculate the costs and revenues. In the second part of the analysis, indirect costs were evaluated as well and integrated into the analysis. The investigated indirect costs included both internal costs (with a direct monetary value), such as beach replenishment operations, and external costs (without a direct monetary value), such as those associated with environmental impacts.

To assess the indirect external costs, two methods were utilized: i) environmental costs were evaluated through weighting of the most relevant LCA impact categories for the case; ii) environmental benefits (ecosystem services) were estimated through a natural capital assessment. The considered direct and indirect costs/added value for each of the investigated scenarios (S1_AD, S2_C, S3_ER, S0_L) were summarized in Table 2.

TABLE 2

Seagrass collection and transport to the landfill was provided as primary data by the beach managing company. The provided transport cost was inclusive of the landfill dumping fee;

thus, to calculate the transportation cost for the other scenarios, the following relation was used, where the costs are evaluated in € and the distance is expressed in km:

$$\text{cost}_{\text{transport to AD/compost site}} = (\text{cost}_{\text{transport to landfill}} - \text{cost}_{\text{landfill fee}}) * (\frac{\text{distance}_{\text{beach to AD/compost site}}}{\text{distance}_{\text{beach to landfill}}}) \quad (1)$$

The landfill fee was assumed to be 11 €/ton, consistent with what was reported by (Aleksic, 2013). The calculated transport cost to the WWTP/composting facility was subtracted from the municipality's original cost for transporting to the landfill, and the difference was then considered as a revenue for the WWTP/composting facility. This assumption accomplished two issues: from the municipality perspective, the financial aspect was the same for each scenario; thus, for the direct costs, the economic assessment could be simplified to focus on the remaining stakeholders (i.e., WWTP and composting facility). Furthermore, by considering a high willingness-to-pay for the municipality, an increased profitability for the processing companies was obtained. A sensitivity analysis was later conducted on the transport costs to the AD and composting sites, considering a lower payment fee by the municipality for access to the biological treatment. A breakeven point for both the AD and composting scenarios was found for the lowest payment they could accept in order to make profit.

As transportation to an existing facility in both the AD and composting scenarios was considered, it was assumed that the existing equipment was used and only the additional operating costs would apply. Operating costs would include electricity, water and labour costs needed in the grinding and washing phases, and the augmented operating costs for both the WWTP and composting facility (due to seagrass wrack inclusion). The water tariff considered was 1,41 €/m^3 (Irisacqua, 2019), while for electricity the average price paid by the WWTP (0,15 €/kWh) was used and the same cost was assumed, in addition, for the composting plant (Cottes et al., 2020). As the grinding phase was supposed to be mechanical, the additional labour cost was associated only with the washing process. An assumption of 4 working hours/day was made, with a worker wage of 7 €/h (Carlini et al., 2017). The revenues from both scenarios were based on the increase in production (either of compost or biogas) due to seagrass wrack treatment. As (Saveyn and Eder, 2013) report a high variation of compost prices (1-60 €/ton), an arbitrary selling price of 25 €/ton was assumed for calculating the revenues from S2_C.

The indirect internal costs associated with seagrass management were: i) beach replenishment, in the case of seagrass collection and ii) tourism reduction, due to ecological restoration (S3_ER). Beach replenishment costs were received as primary data from the Municipality. As for tourism reduction, an arbitrary value of -1% of tourists to the region was assumed, using the data from (ISTAT, 2018). To account for the uncertainty, a sensitivity analysis was later performed, using a wide tourism reduction (0-5%) range.

The indirect external costs and revenues were calculated from the environmental impacts of all scenarios and the environmental benefits of the ecological restoration scenario. To assess the environmental costs, the Ecovalue 2012 valuation set was used (Finnveden et al., 2013). Although several environmental impact categories can be quantified and monetized through this method, in this study the methodology was applied by weighing only the GWP impact category. Thus, the environmental costs were evaluated in terms of GHG emissions for each scenario.

To analyse the environmental benefits, the natural capital concept was used. A natural capital assessment quantifies the ecological benefits through emergy accounting, followed by a translation to monetary value through the use of an appropriate emergy-money ratio (EMR) (Picone et al., 2017). However, performing an emergy accounting analysis was outside of the scope of this work. Instead, secondary data were taken from (ten Brink et al., 2015) to determine the natural capital value of the seagrass wrack.

The LCC was performed with consideration to the different stakeholders, i.e., the municipality, the WWTP, the composting plant and the environment. The final economic comparison was done by aggregating the costs for each of the different life cycle steps per stakeholder, and then by quantifying the annual costs through the net present value (NPV) method (Žižlavský, 2014). The NPV (€/y) index was calculated through the following equation:

$$\mathrm{NPV} = \sum_{t=0}^{n} \frac{\mathrm{NCF_t}}{(1+r)^t} \quad (2)$$

Where NCF (€/y) is the annual cash flow, n is the number of years of operation, t is the year and r is the discount rate. A discount rate of 6% was used (Carlini et al., 2017).

3. Results and discussion

Subsection 3.1 reports and discusses the outcomes related to seagrass production and characterization, while subsection 3.2 deals with the LCA results. Subsection 3.3 reports the outcomes of the LCC analysis.

3.1 Seagrass production and characterization

A significant variability in the produced seagrass amounts was observed (Fig. S1) in the analysed coastline due to non-controllable environmental factors such as temperature, storm frequency, seagrass natural cycles (Misson et al., 2020). Seagrasses have been recognized to provide several ecosystem services, such as: i) stabilization of shoreline resilience; ii) increased biodiversity habitat (particularly regarding recolonized fishery nurseries); iii) carbon sequestration, that helps fighting climate change; iv) nutrient cycling; v) improvement in water clarity (Thorhaug et al., 2020). However, legislators and resource managers still do not fully appreciate seagrass importance, and a significant degradation of seagrass meadows is being observed worldwide, particularly in Southeast Asia (Thorhaug et al., 2020). A globally decreasing trend was observed also in the analysed location, particularly when considering long-term data (years 2004-2020, Fig. S1). However, the reported seagrass generation appears to be higher than other literature values: (Orr et al., 2005) claimed a summer wrack deposition of about 140 Mg dry mass/km shoreline in Northwest of Canada, while the actual specific value for the investigated shoreline was estimated as 267 Mg dry mass/km shoreline. It was recognized that the produced seagrass amount strongly depended on beach type, hydrodynamics, and wrack buoyancy characteristics (Orr et al., 2005).

Due to the non-specific equipment used for seagrass collection, consistent amounts of sand and other unwanted materials (plastics, inerts) were gathered with seagrass (Table S1). In addition, by analysing the seasonal variation in the merceological composition of seagrass wrack, it could be seen that seagrasses and wood fractions were more abundant in the winter season, while sand and algae content increased in the summer. The development of efficient and cost-effective technologies for seagrass collection appears mandatory in the near future, with the aim of reducing the inorganic content of the collected material (sand but also plastic) (Chubarenko et al., 2021), and consequently transportation costs. Recent literature studies in the topic highlighted that litter significantly accumulates in Portuguese sandy beaches, urging for proper measures to reduce seagrass contamination, as it threatens seagrass' role in ecosystems (Guerrero-Meseguer et al., 2020). It should be considered, anyway, that nowadays from a legal point of view seagrass immediately becomes waste as soon as it is collected, irrespectively of its composition (Chubarenko et al., 2021).

3.2 Life Cycle Assessment

Table 3 reports the environmental performances of the alternative seagrass management systems (AD, S1_AD, composting, S2_C, ecological restoration + AD, S3_ER) evaluated with the three endpoint categories of the ReCiPe 2016 method.

The results showed that the three alternative scenarios had significantly better environmental performances when compared to the current landfill management (S0_L), both for the AD-based scenarios (S1_AD; S3_ER, -90%) and the composting-based scenario (S2_C, -70%). S3_ER turned out to be the most environmentally sound seagrass management alternative, because only 50% of the material was collected, transported, and treated, while the remaining fraction (left on the beach) generated negligible impacts.

TABLE 3

Comparable results were obtained also adopting the midpoint level of the impacts evaluation method (Figure 2); seagrass landfilling (S0_L) was confirmed to be the worst environmental alternative for 16 out of the 18 analysed impact categories, while the AD scenarios (S1_AD; S3_ER) always showed the best environmental performances. These results confirm (Slorach et al., 2019) and (Lee et al., 2020), who showed that AD is the most sustainable solution for organic waste treatment, while landfilling should always be avoided even when considering landfill gas recovery.

FIGURE 2

In order to compare, in more detail, the three alternative scenarios from an environmental point of view, Figure 3 shows the contribution of the main hotspots to the total impacts of each scenario, calculated with the most relevant midpoint impact categories (see Subsection 2.3.3). In terms of global warming and fossil resource scarcity, S1_AD and S3_ER demonstrated lower impacts than the composting scenario (S2_C), mainly because of the biogas production used as a renewable energy source (Lin et al., 2018; Takata et al., 2013). This aspect significantly affected the results, also considering that the Italian energy mix is currently based on the use of fossil fuels for more than 50% (International Energy Agency (IEA), 2020). According to (Oldfield et al., 2016), the environmental performances of AD and composting scenarios were similar for the treatment of wasted food and food residue, although AD was preferred over composting for most categories, consistent with the actual reported outcomes (Fig. 2-3).

FIGURE 3

Looking more specifically at the composting scenario, remarkable literature results showed that the hotspots that mostly contributed to the total impacts in terms of global warming and terrestrial acidification were the process air emissions (as produced impact) and the fertilizers production (as avoided impact) (Blengini, 2008). Accordingly, as shown in Figure 3, the avoided impacts due to mineral fertilizers production were more consistent for the composting scenario due to the greater amount of produced compost compared to the solid digestate, and also to the higher nutrient content (see Tables S2-S3).

Another hotspot that substantially affected the environmental performances of both S1_AD and S2_C scenarios was the wastewater treatment phase (consistent with (Takata et al., 2013); wastewater was produced mainly during the seagrass washing phase and was subsequently treated in a dedicated process). Finally, also the diesel consumption for machines operation provided a significant contribution to the total impacts of seagrass management systems; this was due to both the material collection phase on the beach, and its handling in the treatment plants.

The robustness of LCA application to waste management has been recently claimed by (Christensen et al., 2020), as it is able to provide a consistent, comprehensive and transparent overview of the flows, quantifying the environmental profile of the analysed systems. No other tools can provide similar technical information in such a detailed way.

3.3 Life Cycle Costing

The results of the LCC analysis related to the annual cost are displayed in Tables S5 and S6 included in the supplementary material. Table 4 shows the final NPV calculations per each stakeholder by considering a time period of 10 years and a discount rate of 6%.

TABLE 4

The results indicate that the S2_C was associated with the highest revenues when compared to the other scenarios due to the lower relative operating costs for composting as indicated by the secondary data (Pergola et al., 2018). Composting in piles, in fact, is commonly characterized by minimum energy and labour requirements, particularly when compared to more sophisticated systems such as composting in closed vessels (Diaz et al., 2007).

The largest costs were incurred by the municipality in the S3_ER scenario, due to the assumption of tourism reduction related to leaving part of the seagrass material on the beach. Hence, the results show that by limiting seagrass wrack removal to 50%, the economic losses due to tourism reduction were much more significant than the costs reduction, even considering the assumed wrack natural capital.

The municipality paid about 95 €/ton for seagrass wrack transport and disposal to the Slovenian landfill. In the overall LCC, the municipality was considered to pay the same amount to transport the seagrass wrack in all scenarios, providing a source of revenue both to the WWTP and the composting plant. Charging a fee for managing and treating waste is a common practice which can be referred to as access to “biological treatment” (Aleksic, 2013).

The sensitivity analysis conducted in the successive step of the work (Figure 4) showed that the WWTP and the composting plant could still make profit if the willingness-to-pay by the municipality was reduced. For S1_AD, the cost paid by the municipality could be reduced to 24 €/ton for the WWTP to reach the breakeven point, while in S2_C the municipality could even sell the wrack to the composting plant for a breakeven price of 6 €/ton. The possibility for the municipality to sell the wrack as opposed to having to pay for its disposal would causes a significant shift in the perception of this material, from waste to resource.

FIGURE 4

Table S7 shows the sensitivity analysis conducted varying the percentage of tourism reduction in S3_ER and comparing the total annual costs for the municipality to those of S0_L. The analysis revealed that even a 0.20% tourism reduction to the region would result in costs for ecological restoration which would already exceed the cost of seagrass landfilling. Thus, providing proper information to tourists (and more in general to all relevant stakeholders) appears crucial to obtain an overall positive economic outcome, beside analysing environmental aspects (that were previously shown to be significantly improved).

The effect of seagrass wrack on tourism is highly contested in literature; though wrack is often removed under the assumption that its presence will negatively impact tourism (Corraini et al., 2018), some authors appear more sceptical (Boudouresque et al., 2016). Nowhere in literature has a quantifiable value been given to beach wrack's aesthetic impact on tourism. Furthermore, it is uncertain whether or not removing wrack from one beach in a municipality will reduce the overall seasonal tourism, as tourists could simply move to another wrack-free beach in the region. Nonetheless, it is clear in literature that economic drivers such as tourism often influence the management of an ecological resource (Falco et al., 2008; Ruiz-Frau et al., 2019).

For example, in the case of seagrass meadows in the Mauritian bay, the environmental value of the meadows was shown to be overshadowed by the touristic value of a vegetation-free beach and the wrack was therefore removed (Daby, 2003). However, the effects of ecosystem dynamics can often be missed, and thus these economic drivers can be somehow misleading. (Daby, 2003) showed that by removing the seagrass meadows, the underwater sediment was no longer anchored, and the beach waters became murky, subsequently reducing in an indirect way beach touristic value. While the present study did not consider the regional seagrass meadows, seagrass meadows are threatened and decreasing globally (Unsworth et al., 2019), and the size and health of seagrass meadows directly correlates with the amount of produced wrack (Cucco et al., 2020). As previously discussed, seagrass wrack quantities collected annually from the Grado region have also been trending downwards (Misson et al., 2020). On the other hand, as the natural habitats are threatened by economic activities and extreme climatological events, their intrinsic value is expected to augment in the near future (Tyllianakis et al., 2019). Thus, the long-term sustainability of seagrass wrack as a resource influences both the environmental and economic evaluations, and could be explored in future works.

The significant improvement of energy and resource recovery from seagrass was proved with this study, giving a useful insight for both beach managing companies and other relevant stakeholders in the field. Future research should investigate in a detailed way tourists' perception of ecological restoration scenario, to better underline the real feasibility of applying such natural habitat conservation strategies even in touristic areas such as the investigated one.

4. Conclusions

In this work, different seagrass management practices were investigated from an environmental point of view by means of a life cycle assessment (LCA), complemented by an economic analysis through life cycle costing (LCC). The Northeast Mediterranean basin was selected as a meaningful case study, and a beach length of about 1.6 km was considered in the analysis. Three different solutions, alternative to the current landfill disposal, were investigated, including anaerobic digestion (AD), composting and ecological restoration. A significant long-term variation in seagrass production was highlighted, due to natural plant cycles and environmental conditions. The seagrass material was characterized by a consistent inert presence (>50%), due to the unspecific mechanical equipment used for wrack collection. The LCA model showed that ecological restoration (50%)+ AD (50%) was the scenario with the lowest environmental impacts, while both AD (-90%) and composting (-70%) proved to be significantly less impacting on the environment when compared to landfill. The LCC analysis showed that the ecological restoration (50%) + AD (50%) decreased the direct costs, but an overall negative outcome was obtained due to potential tourism reduction. Even a reduction of 0.20% in seasonal tourism would negatively impact the economic balance. Composting yielded the highest positive economic income (NPV>1.27 M€), considering the low operating costs and the high value of compost as fertilizer. Municipality willingness-to-pay could be reduced to 24 €/ton (AD) and 6 €/ton (composting) to reach the breakeven point. The proposed method can be applied to other relevant locations throughout the world, considering that seagrass wrack is an issue for a huge number of touristic areas. Proper attention should be given to the stakeholders when adopting natural-based solutions, such as ecological restoration. Further studies are required to properly assess the impact of ecological restoration on touristic fluxes and tourists' willingness to accept natural seagrass management solutions.

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Tables

Table 1. Primary inventory data about collection of seagrass wrack and its characteristics. All the data are referred to the functional unit.

Item	Unit	Value
Seagrass[a]	t	0.68
Inert	t	0.32
Material initial density	t/m^3	0.06
Volume of collected material	m^3	18.18
Diesel consumption	L	1.93
Material density after compaction	t/m^3	0.22

[a] Moisture amount was allocated to the seagrass content

Table 2: Direct and indirect costs associated with each seagrass management scenario.

Scenario	Direct costs	Source of data direct costs	Indirect costs	Source of data indirect costs
S1_AD (Anaerobic digestion)	Seagrass collection	Primary data	Beach replenishment	Primary data
	Transportation to WWTP	Estimated from primary data	Emissions from collection, transport to WWTP, WWTP operations, CHP plant operations and digestate management	Estimated from LCA output; (Finnveden et al., 2013)
	WWTP plant operations	Estimated from (Dave et al., 2013) and primary data		
	Revenue from produced electricity (added value)	Estimated from primary data and (Carlini et al., 2017)		

S2_C (Composting)	Seagrass collection	Primary data	Beach replenishment	Primary data
	Transportation to composting facility	Estimated from primary data	Emissions from collection, transport to composting facility, composting operations, and compost management	Estimated from LCA output; (Finnveden et al., 2013)
	Composting operations	Estimated from (Pergola et al., 2018) and primary data		
	Revenue from compost (added value)	Estimated from (Saveyn and Eder, 2013) and primary data		
S3_ER (Ecological restoration 50%+ AD 50%)	50% seagrass collection	Estimated from primary data	Seagrass wrack ecosystem services (added value)	Estimated from (ten Brink et al., 2015) and primary data
	50% transportation to WWTP	Estimated from primary data		
	50% WWTP plant operations	Estimated from (Dave et al.,	Tourism reduction	Estimated from (ISTAT, 2018)

		2013) and primary data		
	50% revenue from produced electricity (added value)	Estimated from primary data and (Carlini et al., 2017)	CO_2 and CH_4 emissions from seagrass wrack decomposition (on beach)	Estimated from LCA output; (Finnveden et al., 2013)
S0_L (Landfill)	Seagrass collection	Primary data	Beach replenishment	Primary data
	Transportation to landfill	Primary data	Emissions from collection, transport to landfill, and landfill decomposition	Estimated from LCA output; (Finnveden et al., 2013)

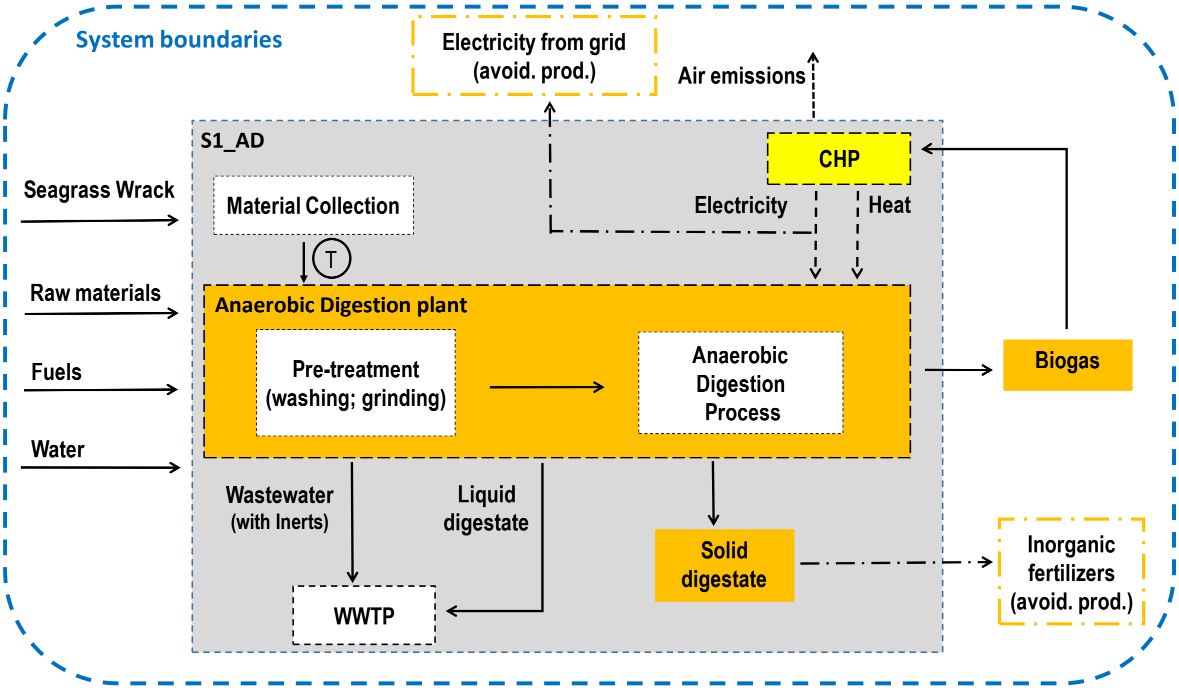
Table 3. Life cycle impacts of the considered seagrass management scenarios, estimated with the three macro-categories at endpoint level of ReCiPe 2016 (H) evaluation method.

Endpoint category	Scenario			
	S1_AD	S2_C	S3_ER	S0_L
Human health (mDALY)	0.013	0.055	0.007	2.116
Ecosystems (species.1E+06yrs)	0.006	0.140	0.004	1.968
Resources (USD2013)	0.807	2.327	0.403	8.379

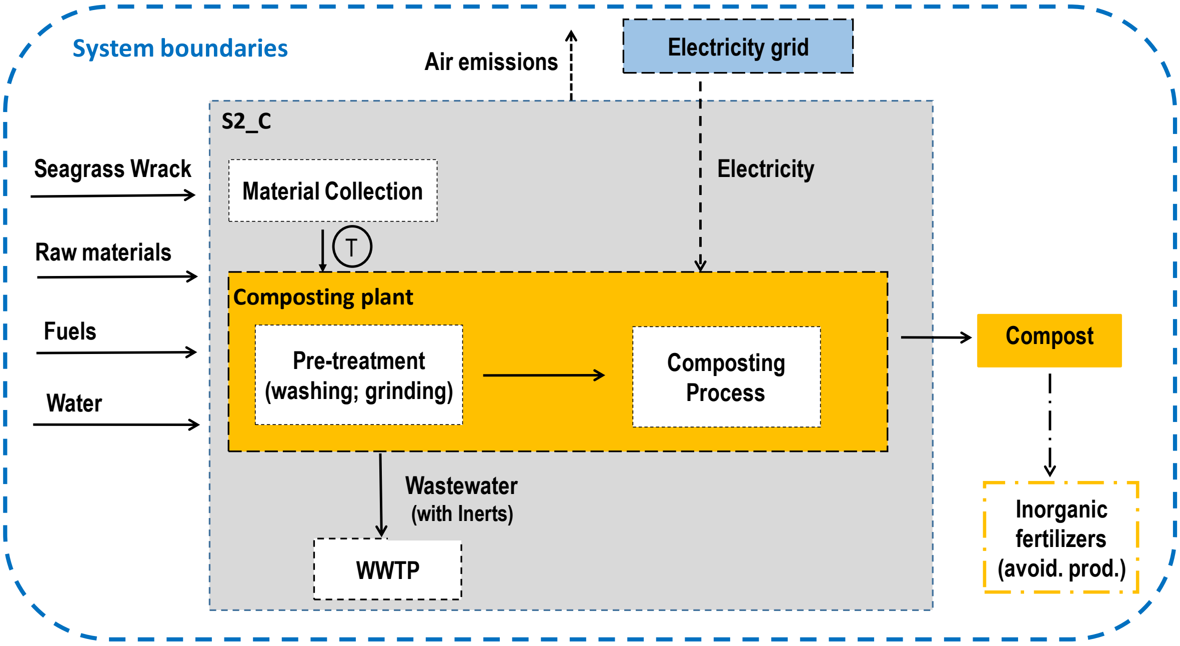
Table 4: Life Cycle Costing results for each stakeholder and scenario.

Total for municipality: S0_L, S1_AD, S2_C	-€ 2.944.034,82
Total for WWTP: S1_AD	€ 910.268,94
Total for composting plant: S2_C	€ 1.274.688,45
Total for municipality: S3_ER	-€ 23.911.082,81
Total for WWTP: S3_ER	€ 455.134,47

935 **Figures**



(a)



(b)

936 Figure 1. System boundaries of the study for scenarios S1_AD (Anaerobic digestion
937 scenario) (a) and S2_C (Composting scenario) (b).

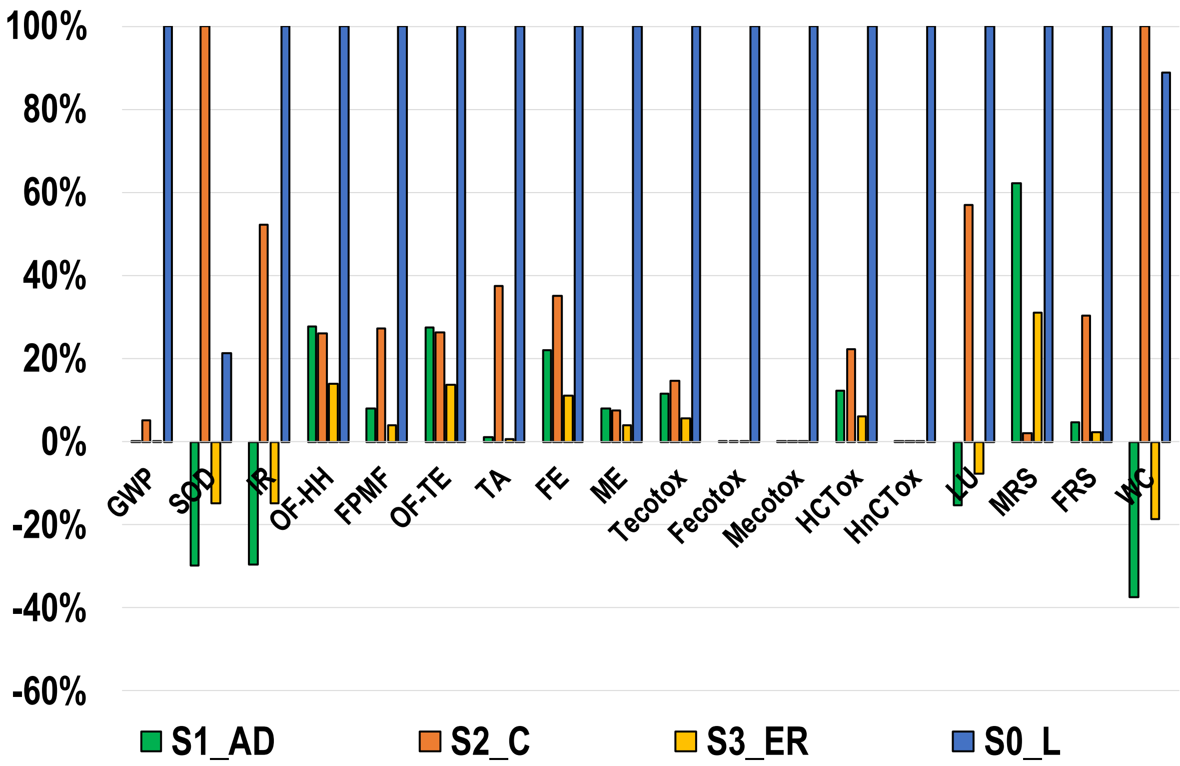


Figure 2. Environmental comparison of the alternative seagrass management strategies considered in the analysis, for the midpoint impact categories of ReCiPe 2016. Impact categories acronyms: GW = Global Warming Potential; SOD = Stratospheric Ozone Depletion; IR = Ionizing Radiation; OF-HH = Ozone Formation, Human Health; PMF = Fine Particulate Matter Formation; OF-TE = Ozone Formation, Terrestrial Ecosystems; TA = Terrestrial Acidification; FE = Freshwater Eutrophication; ME = Marine Eutrophication; TEcotox = Terrestrial Ecotoxicity; FEcotox = Freshwater Ecotoxicity; MEcotox = Marine Ecotoxicity; HCTox = Human Carcinogenic Toxicity; HnCTox = Human non-Carcinogenic Toxicity; LU = Land Use; MRS = Mineral Resource Scarcity; FRS = Fossil Resource Scarcity; WC = Water Consumption.

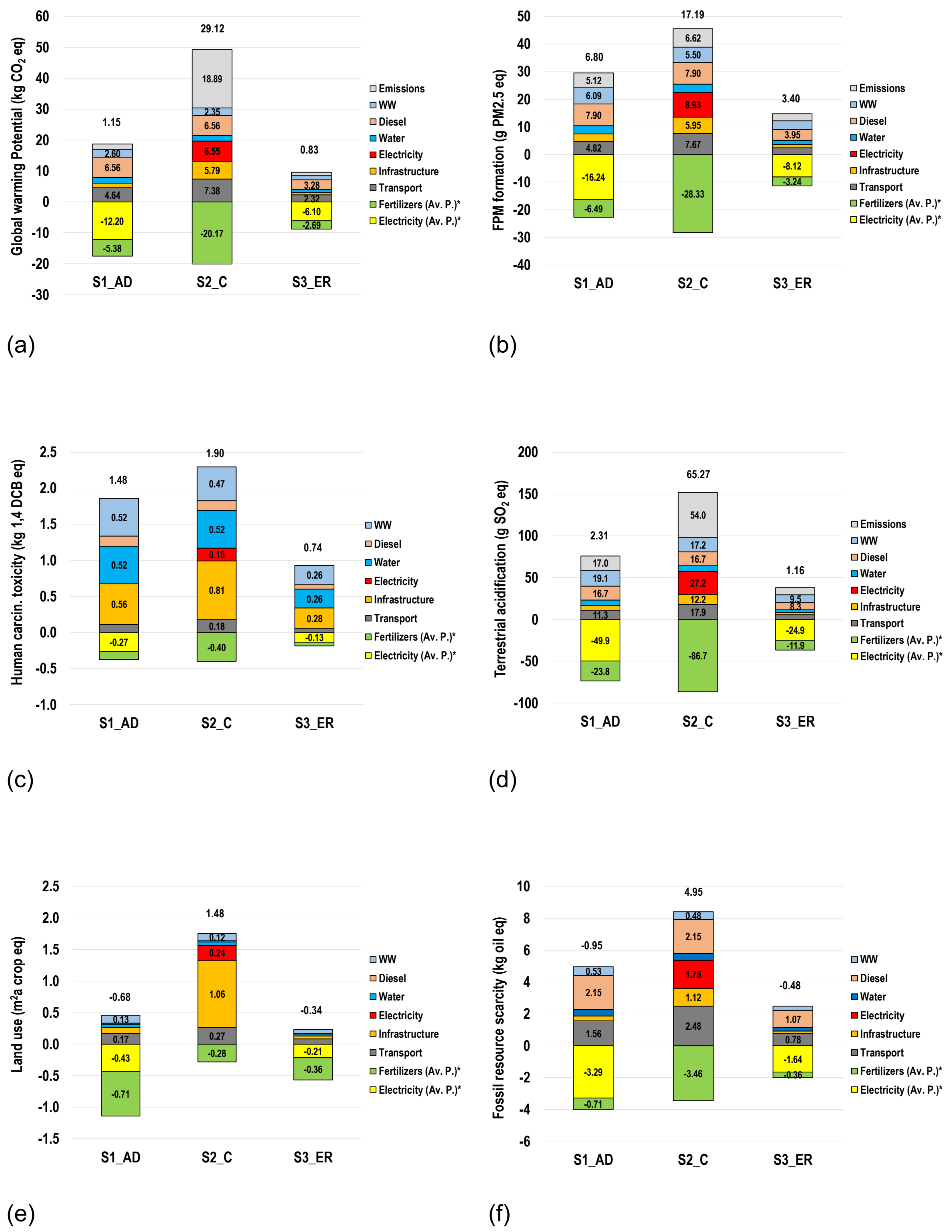


Figure 3. Contributions of each hotspot to the total life cycle impact of the three alternative management scenarios (AD, composting, ecological restoration + AD), evaluated with the following midpoint categories: global warming (a); fine particulate matter formation (b);

human carcinogenic toxicity (c); terrestrial acidification (d); land use (e) and fossil resource scarcity (f).

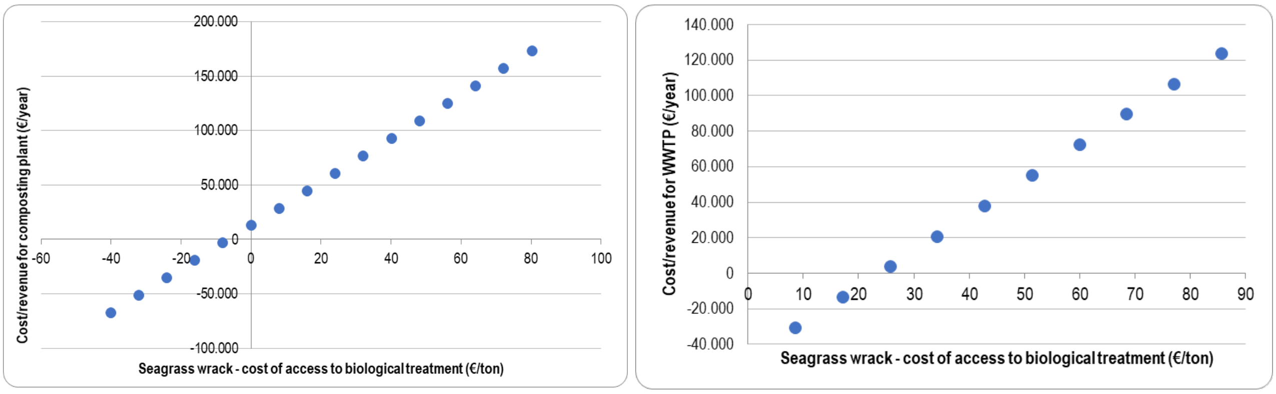


Figure 4: Breakeven point for WWTP and composting plant considering different costs of access to the biological treatment, expressed as €/ton.

Supplementary material

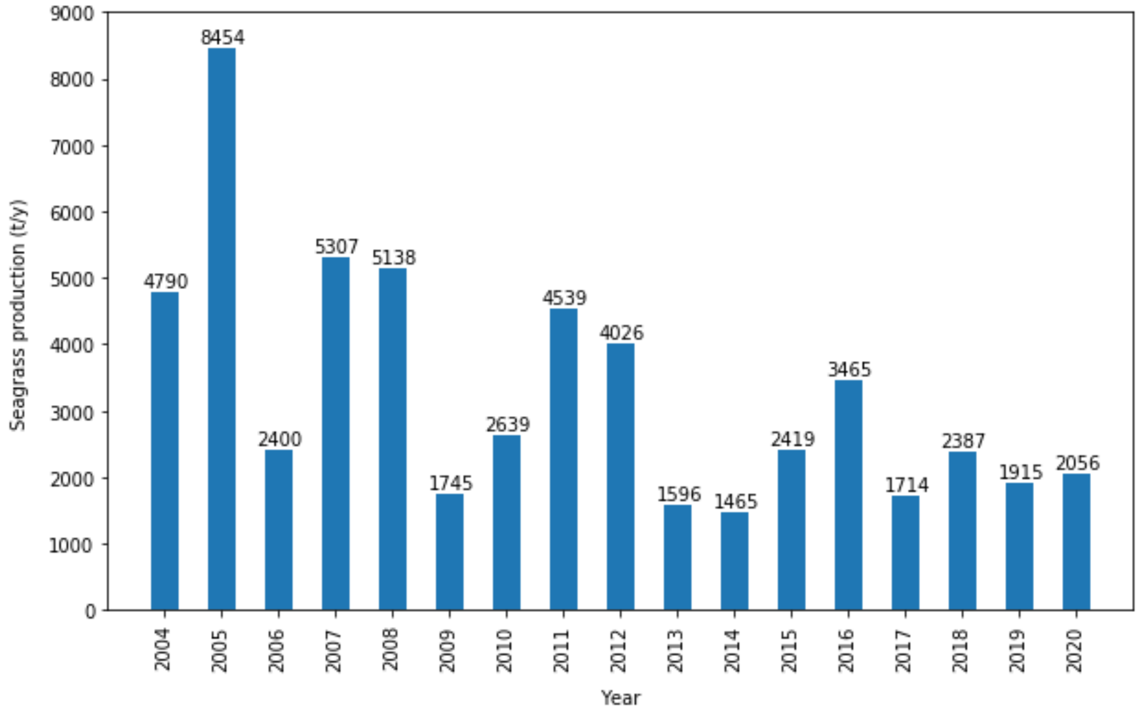


Figure S1: Seagrass wrack production in the analysed beach (years 2004-2020).

Table S1: Results of the merceological characterization of the collected material.

Merceological class	Spring	Summer	Autumn	Winter
Seagrass (% dry matter)	28.9±12.8	32.8±9.4	36.5±16.2	44.6±18.2
Sand (% dry matter)	43.6±14.2	62.4±18.6	53.1±15.2	39.6±13.6
Wood (% dry matter)	9.7±6.8	1.2±0.5	2.3±1.4	10.3±7.1
Algae (% dry matter)	6.4±5.9	9.2±3.1	8.6±4.3	6.1±4.3
Other (% dry matter)	1.6±1.4	0.8±0.6	1.3±1.0	2.9±1.4

Table S2. Inventory data for the anaerobic digestion scenario (S1_AD). All the data are referred to the functional unit.

Item	Unit	Value	Source of data
Transport to the plant	tkm	22	Primary data
Input			
Pre-treatment and AD process			
Water	m^3	5.00	Primary data
Diesel	L	2.50	(Oldfield et al., 2016)
Electricity (from CHP)	kWh	34.57	Estimated from literature and primary data*
Heat (from CHP)	kWh	21.42	Estimated from literature and primary data*
CHP Unit			
Biogas	m^3	34.62	(Misson et al., 2020)
Electricity prod.	kWh	62.17	(Cottes et al., 2020; Misson et al., 2020)
Heat prod.	kWh	71.05	(Cottes et al., 2020; Misson et al., 2020)
Output			
Digestate (solid fraction)	t	0.07	Estimated from literature and primary data*
Digestate (liquid fraction)	t	0.49	Estimated from literature and primary data*
Wastewater	m^3	5.49	Primary data
Emissions			
CH_4 biogenic	g	49.60	(Fei et al., 2021)
SO_2	g	5.3757	(Fei et al., 2021)
CO	kg	0.10	(Fei et al., 2021)
NOx	g	32.39	(Fei et al., 2021)
NMVOC	g	4.27	(Fei et al., 2021)
Avoided products			

Electricity (from grid)	kWh	27.60	(Cottes et al., 2020; Misson et al., 2020)
NP (Inorganic Fertilizers)			(Angelidaki et al., 2017; Barampouti et al., 2020)
N	kg	0.43	
P	g	6.45	

*. See Subsection 2.2.2

Table S3. Inventory data for the composting scenario (S2_C). All the data are referred to the functional unit.

Item	Unit	Value	Source of data
Transport to the plant	tkm	35	Primary data
Input			
Pre-treatment and composting process			
Water	m^3	5.00	Primary data
Diesel	L	2.50	(Oldfield et al., 2016)
Electricity (from grid)	kWh	15.13	Ecoinvent database v.3[a]; (Monster Industrial, 2016)[b]
Output			
Compost	t	0.14	(Oldfield et al., 2016)
Wastewater	m^3	5.00	Primary data
Emissions			
CO_2 biogenic	kg	151.6	(Oldfield et al., 2016)
NH_3	kg	0.028	(Oldfield et al., 2016)
H_2S	kg	0.014	(Oldfield et al., 2016)
N_2O	kg	0.063	(Oldfield et al., 2016)
Avoided products			
NP (Inorganic Fertilizers)			
N	kg	1.93	(Grassi et al., 2015)
P	kg	0.65	(Grassi et al., 2015)
K	kg	1.12	(Grassi et al., 2015)

[a]. For composting in pile process

[b]. For material pre-treatment

Table S4. Inventory data for the ecological restoration scenario (S3_ER). All the data are referred to the functional unit. Data sources were the same as reported in Table S3 (for S1_AD).

Item	Unit	CM[a] (50%)	MB[b] (50%)
Transport to the plant	tkm	11	-
Input			
Pre-treatment and AD process			
Water	m^3	2.50	-
Diesel	L	1.25	-
Electricity (from CHP)	kWh	17.29	-
Heat (from CHP)	kWh	10.71	-
CHP Unit			
Biogas	m^3	17.31	-
Electricity prod.	kWh	31.09	-
Heat prod.	kWh	35.53	-
Output			
Digestate (solid fraction)	t	0.03	-
Digestate (liquid fraction)	t	0.24	-
Wastewater	m^3	2.74	-
Emissions			
CH_4 biogenic [c]	g	24.80	7.52 c
SO_2	g	2.69	-
CO	kg	0.05	-
NOx	g	16.20	-
NMVOC	g	2.14	-
Avoided products			

Electricity (from grid)	kWh	13.80	-
NP (Inorganic Fertilizers)			
N	kg	0.21	-
P	g	3.23	-

[a]. Collected material (CM) on the beach (50% of the total amount)

[b]. Material left on the beach (MB) (50% of the total amount)

[c]. Value estimated as reported in Section 2.2.4

Table S5: Direct and indirect annual costs for different stakeholders considered in the LCC. M = municipality, W = WWTP, C = composting facility, E = environment.

Stakeholder	Life Cycle Costing	Specific cost (€/ton)	Annual cost (€/year)
S0_L			
Direct costs			
M	Seagrass collection	-45	-90,000
M	Transportation to landfill	-95	-190,000
Indirect costs			
M	Beach replenishment	N/A	-120,000
E	GHG emissions	-158.02	-316,042
Total for municipality		N/A	-400,000
Total for environment		N/A	-316,042
S1_AD			
Direct costs			
M	Seagrass collection	-45	-90,000
M	Transportation to WWTP	-95	-190,000
W	Revenue wrack treatment	85.71	171,427
W	Pre-treatment: grinding and	N/A	-34,182

	washing		
W	AD, biogas and digestate production	N/A	-46,586
W	Revenue electricity production	N/A	33,017
Indirect costs			
M	Beach replenishment	N/A	-120,000
E	GHG emissions	-0.322	-644
Total for municipality		N/A	-400,000
Total for WWTP		N/A	123,676.3821
Total for environment		N/A	-644
S2_C			
Direct costs			
M	Seagrass collection	-45	-90,000
M	Transportation to Ronchi dei Legionari plant	-95	-190,000
C	Revenue wrack treatment	80.23	160,452
C	Pre-treatment: grinding and washing	N/A	-34,182
C	Revenue compost production	23.46	46,919
Indirect costs			

M	Beach replenishment	N/A	-120,000
E	GHG emissions	-8.16	-16,324
Total for municipality		N/A	-400,000
Total for composting plant		N/A	173,189
Total for environment		N/A	-16,324
S3_ER			
Direct costs			
M	Seagrass collection	N/A	-45,000
M	Transportation to WWTP	N/A	-95,000
W	Revenue wrack treatment	N/A	85,714
W	Pretreatment: grinding and washing	N/A	-17,091
W	AD, biogas and digestate production	N/A	-23,293
W	Revenue electricity production	N/A	16,509
Indirect costs			
M	Beach replenishment	N/A	-60,000
M	Tourism reduction	N/A	-3,048,750
E	GHG emissions	-0.2324	-465
E	Natural value	1,055.172323	320,013

Total for municipality	N/A	-3,248,750
Total for WWTP	N/A	6,1838
Total for environment	N/A	319,548

Table S6: Net present value calculation of costs/revenue flows for each stakeholder involved in the different scenarios (discount rate: 6%, time period: 10 y).

Stakeholder	Cash flow (€/y)	NPV (€)
Municipality: S0_L, S1_AD, S2_C	-400,000	-2,944,035
Municipality: S3_ER	-3,248,750	-23,911,083
WWTP: S1_AD	123,676	910,269
WWTP: S3_ER	61,838	455,134
Composting plant: S2_C	173,189	1,274,688

Table S7: Comparison of total costs for municipality in S3_ER to S0_L with varying tourism reduction values.

% Tourism reduction	Economic losses (€/y)	Total costs for municipality: S3_ER (€/y)	Total costs for municipality: S0_L (€/y)
0,00%	0	- 399,548	-400,000
0,10%	- 304,875	- 94,673	-400,000
0,20%	- 609,750	- 210,202	-400,000
0,30%	- 914,625	- 515,077	-400,000
0,40%	- 1,219,500	- 819,952	-400,000
0,50%	- 1,524,375	- 1,124,827	-400,000
0,60%	- 1,829,250	- 1,429,702	-400,000
0,70%	- 2,134,125	- 1,734,577	-400,000
0,80%	- 2,439,000	- 2,039,452	-400,000
0,90%	- 2,743,875	- 2,344,327	-400,000
1,00%	- 3,048,750	- 2,649,202	-400,000
1,25%	- 3,810,937.5	- 3,411,390	-400,000
1,50%	- 4,573,125	- 4,173,577	-400,000
1,75%	- 5,335,312.5	- 4,935,765	-400,000
2,00%	- 6,097,500	- 5,697,952	-400,000
2,50%	- 7,621,875	- 7,222,327	-400,000

3,00%	- 9,146,250	- 8,746,702	-400,000
3,50%	- 10,670,625	- 10,271,077	-400,000
4,00%	- 12,195,000	- 11,795,452	-400,000
4,50%	- 13,719,375	- 13,319,827	-400,000
5,00%	- 15,243,750	- 14,844,202	-400,000

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