Screening Life Cycle Environmental Impacts and Assessing Economic Performance of Floating Wetlands for Marine Water Pollution Control

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Abstract: The growing environmental awareness of society, the advancement of nature-based solutions (NbSs), and the need for reliable and cost-effective solutions create a favorable environment of opportunities for floating wetlands as alternative solutions for marine water pollution control. The aim of this work was to screen, through OpenLCA, the environmental impacts of floating wetlands for marine water pollution control at various life cycle stages of the system, and assess its economic performance and contribution to the welfare of society. The stage of raw materials production and acquisition was found to be responsible for the main environmental impacts of the floating wetlands, especially on global warming potential, whereas the main impact of the operational stage was related to the eutrophication potential due to N and P residuals in the effluent. The economic performance indicators of economic net present value (ENPV), economic rate of return (ERR), and benefits/costs ratio (B/C ratio) indicate, although marginally, that floating wetlands may constitute a viable investment with potential positive socioeconomic impacts. However, there are still several scientific challenges and technical issues to be considered for the operational application of such systems at full-scale in marine environments.

Keywords: floating wetlands; marine pollution; life cycle assessment; environmental impacts; economic performance indicators; nature-based solutions; sustainability

1. Introduction

Marine life is heavily threatened by pollution and habitat degradation, mostly due to human activities, such as ship traffic, fisheries, and coastal zone activities (agricultural, industrial, and urban), whose impacts are exacerbated by climate change. Thus, the treatment of marine water for pollution control and contaminants removal, especially petroleum hydrocarbons, is an urgent task in many countries and especially around the Mediterranean, which is more polluted by oil than any other sea in the world [1]. Conventional treatment plants and techniques offer feasible methods for treating saline wastewater. However, these methods are ineffective in controlling diffuse pollution, and most of them use sophisticated equipment, which require large economic investments, and consume vast amounts of energy [2]. These drawbacks urge the scientific community to search for creative, cost effective, and environmentally sound ways to control marine water pollution [3].

In the last decades, there has been a growing interest about constructed wetlands as nature-based solutions (NbSs) of pollution control. Indeed, constructed wetlands are inspired and supported by nature, they are cost-effective, and simultaneously provide
environmental, social, and economic benefits while helping build resilience. Floating wetlands (FWs) are a promising and cost-effective eco-engineering tool to restore waterbodies [4]. They are gaining acceptance as a type of constructed wetland that can be applied within an existing natural or artificial water body, and withstand water level fluctuations and adverse conditions. FWs consist of emergent vegetation established upon a buoyant substrate that supports plant growth, and floats on surface waters. The upper parts of the vegetation grow and remain primarily above the water level, and the roots extend down in the water column, developing an extensive beneath water-level root system [5]. The main components responsible for pollutant removal efficiency in FWs are macrophytes and the biofilm attached to the roots, and the matrix of the floating substrate [4,6]. As physical and biochemical processes take place, the system functions as a natural filter for water quality improvement. Compared to conventional types of constructed wetlands, the biological processes are more effective in FWs due to the free suspension of roots in the water column—this establishes direct contact between contaminants and the root-associated microbial community [7].

FWs have successfully been applied for pollution control mostly in rivers, lakes, and ponds with fluctuating water levels, and are a suitable green solution for urban water management [8]. Currently, they are being used to treat hypertrophication, sewage and domestic wastewater, combined sewer overflow, contaminated groundwater, stormwater, acid mine drainage, poultry effluent, piggery effluent, boron-enriched water, polluted river water, and nutrients-enriched water [7]. However, the FWs technology in coastal and marine environments has been mainly used for erosion control, food web support, aesthetic improvement, and sediment trapping, but solid scientific documentation of these applications is limited [5,9]. Especially concerning the treatment of hydrocarbons and other marine contaminants with floating wetlands, a gap in scientific research and knowledge has been identified, and even the treatment of these contaminants with conventional constructed wetlands has been sparse [5,9,10], but with increasing research interest in recent years [11].

The growing environmental awareness of society, the advancement of NbSs, and the need for reliable and cost-effective solutions create a favorable environment of opportunities for the FWs, which appear as an attractive ecotechnology able to address marine environmental problems in a sustainable way. In this perspective, there is a need not only to document the treatment efficiency of FWs in marine environments, but also to evaluate the environmental and economic performance of these systems, and optimize their design and construction in order to capitalize their potential for operational application, and address the needs of society. Life cycle assessment (LCA) is an internationally standardized tool for assessing the environmental impact of products, processes, and activities over their entire life cycle by interpreting potential impact, and evaluating by categories [12–14]. Until present, LCA has been applied to varying degrees in water treatment technologies, including constructed wetlands. However, in the case of the marine oriented systems of floating wetlands, there is a significant gap of information. The aim of this work is the initial screening, through OpenLCA, of the environmental impacts of a floating wetlands system for marine water pollution control at various life cycle stages, and the assessment of the system’s economic performance as a contribution to the welfare of society.

2. Materials and Methods

The screening of environmental impacts at various life cycle stages of a floating wetlands system for marine water pollution control was based on the life cycle assessment approach, which consists of the following steps: system description; setting the goal and scope of LCA; identification of the system boundaries; inventory of inflows and outflows; assessment of impacts; and sensitivity analysis of results.

The assessment of the economic performance of the system was based on the calculation of three economic performance indicators, which are the economic net present value (ENPV), economic rate of return (ERR), and benefits/costs ratio (B/C ratio).
2.1. Floating Wetlands Set-Up and Description

The development and operational use of FW for marine water pollution control in the Mediterranean may have a considerable environmental and socioeconomic impact on the area. In this perspective, the research project ATLANTIS was launched, aiming at the operational development and evaluation of floating wetlands, adapted to Mediterranean ecoclimatic conditions, as an environmental-friendly and eco-innovative solution for marine pollution control and petroleum hydrocarbons treatment [3]. The two stage research activities included (Figure 1): (a) the establishment of a lab-scale experimental system consisting of 18 floating wetlands for assessing the mechanisms and the ability of floating wetlands to reduce the concentration of petroleum hydrocarbons in a marine environment; and (b) the establishment of a pilot-scale FWs system in Ierissos Port for assessing the FW system in an operational environment.

![Figure 1. Floating wetlands for marine water pollution control within the framework of the ATLANTIS project. (a) Lab-scale units of floating wetlands, (b) Pilot-scale floating wetland established at Ierissos Port in Northern Greece.](image)

The lab-scale units cover an area of 50 m$^2$ with a total treatment capacity of 1.74 m$^3$ day$^{-1}$ and hydraulic residence time of 21 days. The floating wetlands were established in 18 containers of 300 L each. For the construction of the wetlands, in each container, a 0.5 × 0.5 m BIOHAVEN floating matrix was used, on which fine gravel and organic potting soil was placed as substrate for the establishment of wetland vegetation. Seedlings of salt tolerant species of Phragmites australis and Suaeda maritima were planted on the floating islands. The open area of each island was covered with burlap textile to protect the matrix from UV sunlight and prevent the soil leaching in the initial stages of plant establishment and development. Natural sea water enriched with petroleum hydrocarbons (diesel) was applied in the systems, and the observed reduction of total petroleum hydrocarbons (TPH) reached an average value of 73.58%, and TOC was reduced by 56.21%.

2.2. Goal and Scope of LCA

The targeted product in the present study is a pilot-scale floating wetland system established for sea water pollution control. The initial function of the floating wetland was to treat seawater contaminated by petroleum hydrocarbons and the production of purified effluent. Considering the actual situation of the eco-technology, as well as previous studies [12,15], 1 m$^3$ of water volume purification was used as the functional unit (FU).

The aim of the LCA is to explore the types of environmental impacts caused by inputs/outputs during the various life cycle stages of the targeted product and, thus, to support decisions for reducing impacts by design, comparing products to choose the least harmful, and providing options to prioritize environmental or product solutions. Although other tools have been designed for sustainability assessment (environmental impact as-
2.3. System Boundaries

System boundaries are delineated into three phases which include raw material production/acquisition, construction, and operation (Figure 2). The floating wetlands eco-technology lies in the removal and degradation of pollutants by physical chemical and biological processes with minimum external inputs. The study does not consider the release of pollutants in water due to plant decay, the decomposition and desorption of N and the reabsorption of micro-organisms in the aged and fallen sludge, or the absorption and release of N and P occurring in the water by the floating bed block. It is worth noting that these factors have little influence on the input and output of the entire LCA system.

![Figure 2. System boundaries.](image)

2.4. Life Cycle Inventory

The life cycle inventory (LCI) includes the main inflows and outflows in the raw materials, construction, and operation phase [6]. Primary data for the LCI were collected from field research, practitioner interviews, literature studies, and chemical analysis. Background data (i.e., data of materials, transportation) were obtained from the ELCD database. Based on the selection of the functional unit, the life cycle inventory was calculated per 1 m³ water body.

2.4.1. Acquisition and Construction Phases

The acquisition phase refers to the use of raw materials in terms of extraction, processing, and acquisition for the needs of the system. Given the limitations of the database, the burlap textile, as well as the twine, were calculated as biomass.

In the construction phase, all the transportation was based on a lorry with a capacity of 7.5 t total weight and 3.3 t max payload. Given the nature of the system, no power tools were applied for the construction, and the energy consumption was negligible, thus, it was not included as input. The life cycle inventory data during the raw materials acquisition stage and construction stage are presented in Table 1.

2.4.2. Operational Phase

The sea water under treatment by the floating wetlands (FW) contained organic compounds, which are transformed during their degradation into CO₂, CH₄, and N₂O. In water treatment units, these conversions are accelerated, and they are considered as a GHG...
emission source [17]. The inputs and outputs of the FW operational phase are presented in Table 2.

Table 1. Life cycle inventory data during raw materials acquisition stage and construction stage.

| Raw Materials Acquisition Stage | Composition      | Material                                      | kg/m³ |
|---------------------------------|------------------|-----------------------------------------------|-------|
| Burlap textile                  | Jute or Cotton   | 0.828                                         |       |
| Twine                           | Jute natural fiber | 0.033                                        |       |
| Floating matrix                 | Polyethylene     | 2.91                                          |       |
| Water treatment containers      | Polyethylene     | 26.64                                         |       |
| Rope                            | Polyethylene     | 0.20                                          |       |
| Pipes and elbows                | Polyvinylchloride | 2.36                                          |       |

| Construction Stage              | Composition      | Transport Distance (km) | kg/m³ |
|---------------------------------|------------------|-------------------------|-------|
| Plants                          |                  | 40                      | 13.32 |
| Planting medium                 |                  | 5                       | 9.99  |
| Fine Gravel (4–8 mm)            |                  | 5                       | 6.66  |
| Fertilizer                      |                  | 5                       | 0.083 |

Table 2. Inputs and outputs of the operational phase.

| Parameter          | Concentration (mg/L) |
|--------------------|-----------------------|
| TOCin              | 58.5                  |
| TNin               | 4.3                   |

| Parameter          | Emissions (mg m⁻² h⁻¹) |
|--------------------|------------------------|
| CO₂-C              | 146.9783               |
| CH₄-C              | 5.5599                 |
| N₂O-N              | 0.0140                 |

The carbon and nitrogen emissions from FWs in marine environments are not sufficiently studied. On the other hand, there are several studies regarding the effect of salinity on carbon and nitrogen emissions from coastal wetlands, as well as from wetland sediments treated with high salinity water, in research efforts to predict the impact of climate change and seawater rise. The results of these studies, however, are very contradictory, as was also noted by Ardon et al. [18], who provided a summary of studies on the effects of salinity on greenhouse gas emissions from wetland soils. In this perspective, and to avoid a biased assessment based on a specific study, the assessment of GHGs emissions within the framework of this study was based on the work of Mander et al. [19], as they provided an approximation for the assessment of emissions that is based on the analysis of a broad data set from wetland systems that are used for water quality improvement. More specifically, the carbon and nitrogen emissions assessment was based on the proposed models provided by Mander et al. [19] for free water surface (FWS) wetland systems, while taking into account the research outcomes of Vinasco [17] for similar systems. Mander et al. [19], in their study for greenhouse gas emissions in constructed wetlands, defined FWS as shallow and low flow velocity wetlands which have areas of open water and floating, submerged, and/or emergent plants. Based on the above, the proposed equations for the calculation of emissions are used under the assumption that floating wetland systems fall into the definition of FWS, and the models provided by the authors could be applied for such systems. Thus, the emission of carbon dioxide, methane, and nitrous oxide were calculated using the following Formulas (1)–(3):

\[
\text{CO}_2\text{-C emission (mg m}^{-2}\text{ h}^{-1}) = 5.4 + (5.869/\text{TOCin}) \tag{1}
\]

\[
\text{CH}_4\text{-C emission (mg m}^{-2}\text{ h}^{-1}) = 0.11 \text{TOCin} + 1 \tag{2}
\]

\[
\text{N}_2\text{O-N emission (mg m}^{-2}\text{ h}^{-1}) = 0.001 \text{TNin} + 0.011 \tag{3}
\]
where TOCin and TNin values in g m\(^{-2}\) h\(^{-1}\) correspond to the inflow total organic carbon and total nitrogen, and were calculated based on the area, the hydraulic load, the residence time, and the inflow TOC and TN concentrations using the experimental data.

2.5. Tools and Impact Assessment Method

The LCA model was set up using the OpenLCA software v1.10.3, with life cycle impact (LCI) data sourced from the European reference life cycle database (ELCD) of the Joint Research Center Version 3.2. OpenLCA is an open-source tool used to collect, analyze, and monitor the sustainability performance data of the product. The ELCD is a LCI database containing over 300 unique datasets covering a wide array of products, services, and processes from energy carriers and technologies to transport services, and from materials production to end-of-life treatment. The life cycle impact assessment (LCIA) was carried out with the CML-IA baseline method, which was used to evaluate the following impact categories: abiotic depletion potential (ADP); acidification potential (AP); eutrophication potential (EP); global warming potential (GWP); photochemical oxidation potential (POCP); ozone layer depletion potential (ODP); human toxicity potential (HTP); freshwater aquatic eco-toxicity potential (FAETP); and terrestrial eco-toxicity potential (TETP). All impact categories were analyzed including climate change as the main target, since the results are easier to communicate due to the current political interest in this field [20].

2.6. Sensitivity of LCA Results

Sensitivity analysis highlights the most important model parameters that determine whether data quality needs to be improved, and, thus, enhances the interpretation of results. In this perspective, and following the approach of similar research studies [8], a sensitivity analysis was conducted regarding the key variables of the system which represent the main assumptions of the study. According to Garfi et al. [21], a variation of 10% was considered for the parameters, and the sensitivity coefficient was calculated using Equation (4).

\[
\text{Sensitivity coefficient} = \frac{(\text{Output}_{\text{high}} - \text{Output}_{\text{low}})/\text{Output}_{\text{default}}}{(\text{Input}_{\text{high}} - \text{Input}_{\text{low}})/\text{Input}_{\text{default}}}.
\]  

(4)

where Input is the value of the input variable (i.e., emissions of CO\(_2\), CH\(_4\), N\(_2\)O, the TOC load), whereas Output is the value of the environmental indicator (ADP, AP etc.).

2.7. Economic Assessment

The main purpose of the analysis was to assess the project economic efficiency and contribution to social welfare through the calculation of three economic performance indicators: the economic net present value (ENPV); economic rate of return (ERR); and benefits/costs ratio (B/C ratio). Following the EU guidelines for cost-benefit analysis of Investment Projects 2014–2020 [22], the analysis was based on the following assumptions: (a) all amounts stated in constant EUR; (b) real discount rate of 5% in economic analysis as the social discount rate (SDR), which reflects the social view on how future benefits and costs should be valued against present ones; and (c) a reference period of 30 years. Based on relevant experience, it was assumed in the analysis that the construction phase will last 1 year, and after this period, the system will be fully operational. Investment data were gathered from the detailed engineering design, and prices were provided by local companies.

The value of environmental benefits as projections of revenues was calculated as the sum of: (a) the ecosystem and aesthetic value of the system, 22.483 $/ha/year and 76.03 $/year, respectively, as referred by Steer et al. [23] in their life-cycle economic model of small treatment wetlands, after taking into account an average inflation rate of 2.23% per year between 2002 and 2021; and (b) the avoided costs, which represent an estimation of the economic value of the environmental benefits obtained from the cleaning process. Based on the methodology of Hernández-Sancho et al. [24], the avoided costs were calculated
using the yearly volume of pollutant removal in the treatment process (1.06 kg of N, 0.9 kg of P, 22.35 kg of BOD$_5$, and 109.23 kg of COD), and the shadow prices for the pollutants (€/kg), considering that the final receiver of the treated water is the Mediterranean Sea.

As a final step, a sensitivity analysis of the results was performed to identify the ‘critical’ variables of the economic performance. The analysis was carried out by varying one variable at a time and determining the effect of that change on the net present value. ‘Critical’ variables have been considered those variables for which a variation of ±1% of the value adopted in the base case gives rise to a variation of more than 1% in the net present value.

3. Results and Discussion

3.1. Environmental Impact Characterization

LCA can be used to identify which life stage carries the most significant environmental impacts for particular designs or classes of designs [25]. The characterization results of LCA enable the identification of the most positive contributing stages to the environmental degradation within each impact category, whereas the normalization results present the predominant damage category within each stage [26]. In the FWs system under study, the characterization results (Table 3) indicate that ADP, AP, GWP, ODP, and HTP are mainly affected by the stage of raw materials production and acquisition. The highest values have been detected in the global warming potential (GWP) and the human toxicity potential (HTP)—a similar effect of the raw materials stage on these two parameters was also reported by Yao et al. for combined ecological floating beds [27]. In the categories of FAETP, TETP, and POCP, the construction stage was the most positive contributor, whereas in the case of EP, there was an almost equal contribution of the raw materials stage and the operational stage of the system. The raw material stage was responsible for the main environmental impacts of the FWs. The former is in line with the findings of Dixon et al. [28] in their LCA impact assessment of a reedbed system; however, in the case of FWs under study, the construction stage had a lower environmental impact due to the lack of excavations, and the use of local materials which minimize the transportation emissions.

| Impact Categories | Unit       | Raw Materials Production/Aquisition | Construction | Operation |
|-------------------|------------|-------------------------------------|--------------|-----------|
| ADP               | kg Sb eq   | $7.51956 \times 10^{-7}$            | $2.18155 \times 10^{-13}$ | -         |
| AP                | kg SO$_2$ eq | 0.01766                             | 0.00017      | -         |
| EP                | kg PO$_4$ eq | 0.00219                             | $3.77543 \times 10^{-5}$ | 0.00181   |
| GWP               | kg CO$_2$ eq | 20.62733                            | 0.02008      | 0.00095   |
| ODP               | kg CFC-11 eq | $1.24847 \times 10^{-7}$            | $2.40473 \times 10^{-10}$ | -         |
| HTP               | kg 1.4-DB eq | 11.52605                            | 0.00074      | -         |
| FAETP             | kg 1.4-DB eq | $-0.01288$                          | $1.30037 \times 10^{-5}$ | -         |
| TETP              | kg 1.4-DB eq | $-0.00177$                          | $2.09400 \times 10^{-6}$ | -         |
| POCP              | kg C$_2$H$_4$ | 0.00860                             | $9.28044 \times 10^{-6}$ | $1.31710 \times 10^{-7}$ |

ADP: abiotic depletion potential, AP: acidification potential, EP: eutrophication potential, GWP: global warming potential, POCP: photochemical oxidation potential, ODP: ozone layer depletion potential, HTP: human toxicity potential, FAETP: freshwater aquatic ecotoxicity potential, TETP: terrestrial eco-toxicity potential.

The normalized values (Table 4) are used to determine the most significant environmental impact of each stage. In this perspective, the primary environmental impact of the raw materials stage is on global warming potential (GWP), and the secondary, on human toxicity potential (HTP) and photochemical oxidation potential (POCP). The construction stage affects mainly the acidification potential (AP), and the secondary, the global warming potential (GWP). As far as the operational stage concerns, the analysis revealed that the main impact of the system is related to the eutrophication potential (EP), which was also
detected as a potential impact of tidal flow constructed wetlands by Wang et al. [29], since N and P residuals in the effluent are the main culprits of EP.

Table 4. Normalized values for floating wetland impact during raw material acquisition, construction, and operational stages.

| Impact Categories | Unit | Raw Materials Production/Acquisition | Construction | Operation |
|-------------------|------|--------------------------------------|--------------|-----------|
| ADP               | year | $8.87308 \times 10^{-15}$           | $2.57422 \times 10^{-21}$ | -         |
| AP                | year | $6.26881 \times 10^{-13}$           | $6.20551 \times 10^{-15}$ | -         |
| EP                | year | $1.66344 \times 10^{-13}$           | $2.86177 \times 10^{-15}$ | $1.36896 \times 10^{-13}$ |
| GWP               | year | $4.10484 \times 10^{-12}$           | $3.99583 \times 10^{-15}$ | $1.88941 \times 10^{-16}$ |
| ODP               | year | $1.39828 \times 10^{-15}$           | $2.69330 \times 10^{-18}$ | -         |
| HTP               | year | $1.48686 \times 10^{-12}$           | $9.58838 \times 10^{-17}$ | -         |
| FAETP             | year | $-2.48529 \times 10^{-14}$          | $2.50971 \times 10^{-17}$ | -         |
| TETP              | year | $-3.65239 \times 10^{-14}$          | $4.31364 \times 10^{-17}$ | -         |
| POCP              | year | $1.01447 \times 10^{-12}$           | $1.09509 \times 10^{-15}$ | $1.55418 \times 10^{-17}$ |

ADP: abiotic depletion potential, AP: acidification potential, EP: eutrophication potential, GWP: global warming potential, POCP: photochemical oxidation potential, ODP: ozone layer depletion potential, HTP: human toxicity potential, FAETP: freshwater aquatic ecotoxicity potential, TETP: terrestrial eco-toxicity potential.

The relative contributions of the life cycle stages per category of impact are illustrated in Figure 3. For most impact categories, the raw materials stage was found to have a predominant effect. The eutrophication potential (EP) is affected by all three stages of the life cycle, but mainly by the raw materials stage and the operational stage. The construction stage was also found to have a notable effect in the acidification potential (AP). In addition to the above, the analysis gave negative values in freshwater aquatic eco-toxicity potential (FAETP) and terrestrial eco-toxicity potential (TETP). Negative effects on ecotoxicity have been reported by Yao et al. [26] for the operational stage of floating beds in inland waters.

3.2. Sensitivity Analysis of LCA

Sensitivity analysis enables the identification of the ‘critical’ variables of the project, and provides a solid base for studying the robustness of results, and their sensitivity to uncertainty in LCA factors. The results of the sensitivity analysis are shown in Table 5. Results showed that ADP, AP, EP, ODP, HTP, FAETP, and TETP were not sensitive to...
any of the parameters considered (sensitivity coefficient < 0.1). On the contrary, POCP was found to be highly sensitive to CH\textsubscript{4}; however, it is underlined that the calculated POCP values are, by definition, not absolute values. As data, for e.g., the chemical and photochemical reactions are often not known in great detail, and their representation in the model will often be a compromise. Therefore, even for the same scenario, the POCP values can be calculated with higher precision when more accurate input data and more powerful computer tools are available [30]. Finally, GWP was found sensitive to both CO\textsubscript{2} and CH\textsubscript{4}, since a 10% increase in CO\textsubscript{2} would increase GWP by 3.1%, and a 10% increase in CH\textsubscript{4} would increase GWP by 7.3%.

Table 5. The sensitivity analysis of the considered variables in relation to the environmental impact categories.

|        | ADP | AP | EP | GWP     | ODP | HTP | FAETP | TETP | POCP |
|--------|-----|----|----|---------|-----|-----|-------|------|------|
| CO\textsubscript{2} | ±0.000 | ±0.000 | ±0.000 | ±0.315 | ±0.000 | ±0.000 | ±0.000 | ±0.000 | ±0.000 |
| CH\textsubscript{4} | ±0.000 | ±0.000 | ±0.000 | ±0.736 | ±0.000 | ±0.000 | ±0.000 | ±0.000 | ±1.001 |
| N\textsubscript{2}O | ±0.000 | ±0.000 | ±0.000 | ±0.000 | ±0.000 | ±0.000 | ±0.000 | ±0.000 | ±0.000 |
| TOC  | ±0.000 | ±0.000 | ±0.000 | ±0.000 | ±0.000 | ±0.000 | ±0.000 | ±0.000 | ±0.000 |

ADP: abiotic depletion potential, AP: acidification potential, EP: eutrophication potential, GWP: global warming potential, POCP: photochemical oxidation potential, ODP: ozone layer depletion potential, HTP: human toxicity potential, FAETP: freshwater aquatic ecotoxicity potential, TETP: terrestrial eco-toxicity potential.

3.3. Economic Evaluation

The detailed construction and operational cost of the FWs system is presented in Table 6. The investment costs of the construction phase refer to materials, equipment, transportation, and staff effort for the establishment of the FWs. The typical operating cost items, as in the cases of other water investments, include materials and maintenance costs, as well as technical and monitoring services. Though the main purpose of the constructed wetlands is to remove pollutants, good management practices strive to achieve multiple aims by providing habitats, and increase the recreational value with improved aesthetics [4]. In this perspective, and based on the environmental benefits of the FW system establishment and operation, the yearly system revenues have been calculated at 239.66 Euros/year (Table 7), from which 32% is due to water treatment benefits, 41% to ecosystem value, and 27% to aesthetic value of the floating wetland.

Table 6. Cost analysis of the FW system construction and operation.

| Cost Analysis | Investment Cost | Yearly Cost |
|---------------|-----------------|-------------|
| A. Construction |                 |             |
| Floating matrix | 750.00 €        |             |
| Gravel         | 15.00 €         |             |
| Potting soil substrate | 40.00 € |             |
| Vegetation     | 80.00 €         |             |
| Burlap textile | 5.00 €          |             |
| Anchoring wire | 30.00 €         |             |
| Twine          | 10.00 €         |             |
| Pipes and elbows | 20.00 €   |             |
| Staff effort-works | 300.00 € |             |
| Transportation | 80.00 €         |             |
| Solar position lights | 20.00 € |             |
| SUM (A)       | 1350.00 €       |             |
| B. Operation-Maintenance |     |             |
| Maintenance and monitoring services | 165.00 € |             |
| Consumables   | 20.00 €         |             |
| SUM (B)       | 185.00 €        |             |
Table 7. Evaluation of yearly system revenues in terms of environmental benefits.

| Environmental Benefits      | Euros/Year |
|----------------------------|------------|
| Water treatment benefits    | 76.39      |
| Ecosystem value             | 97.44      |
| Aesthetic value             | 65.83      |
| Total Yearly Benefits       | 239.66     |

The summary cash flow analysis of the FWs in fixed base year prices, and the economic performance indicators of the system are presented in Table 8. The economic net present value (ENPV) represents the difference between the discounted total social benefits and costs, and is the most important and reliable indicator of social cost benefit analysis (CBA) as the main reference economic performance signal for project appraisal. Although the balance of costs and benefits in the early years of the FW project is negative, it eventually becomes positive after some years. The economic analysis of the system reveals that the FWs system is desirable from a socio-economic perspective, as is demonstrated by the positive economic net present value (ENPV). Based on the analysis, the economic rate of return (ERR) is 8.33%, whereas the benefits/costs ratio (B/C ratio) was calculated at 1.01. The B/C ratio exceeds the social discount rate of 5% and, thus, favors the project adoption. The B/C ratio is independent of the size of the investment, but it does not generate ambiguous cases, and for this reason, it can complement the net present value in ranking projects where budget constraints apply, and is used to assess a project’s efficiency.

Table 8. Economic performance analysis and indicators.

| Year          | Investment Costs | Operation Costs | Revenues | Net Cash Flow |
|---------------|------------------|-----------------|----------|---------------|
| 2021          | €1350.00         | €0.00           | €0.00    | -€1350.00     |
| 2022–2030     | €0.00            | €1665.00        | €2156.94 | €491.94       |
| 2030–2039     | €0.00            | €1850.00        | €2396.60 | €546.60       |
| 2040–2049     | €0.00            | €1850.00        | €2396.60 | €546.60       |
| SUM           | €1350.00         | €5365.00        | €6950.14 | €235.14       |

Discount Rate 5%
ENPV: 82.35 €
ERR: 8.33%
B/C Ratio: 1.01

The positive values of all economic indicators resulting from the analysis indicate that the implementation of such a project may increase social welfare, and, in this perspective, such projects are quite competitive in terms of EU financial support [22]. However, these results indicate only a marginal profitability of FWs.

The problem with current evidence for the cost-effectiveness of FWs is that in case of NbSs, appraisals underestimate the economic benefits of working with nature, especially over the long term. In this perspective, four major issues should be addressed [31]: (a) NbSs are multi-functional systems delivering a wide range of benefits. Yet, non-monetary benefits (e.g., carbon sequestration, education) are difficult to monetize, or there is high uncertainty about their non-market value [32,33]. (b) Appraisals rarely factor in trade-offs among different interventions and ecosystem services, or between stakeholder groups, which may experience the costs and benefits of NbSs differently. For example, the importance of marine FWs is different for the local fishermen, the cargo boat owners, the tourists-swimmers, etc., reflecting differences in the extent of dependency on natural resources [34].
Changes in the provision of ecosystem services over time, for example, under climate change and other stressors, are rarely considered, and there are major questions about how to balance future benefits with current costs [35]. Engineered solutions can usually be implemented with relative certainty in terms of type and timescale of benefits. On the other hand, NbSs generally offer more flexible long-term alternatives with benefits that might not be reaped when the costs are felt [31]. (d) An additional major challenge regarding the cost-effectiveness of NbSs relates to the variable levels of protection they offer, since the efficacy can vary with intensity and frequency of threats, the resilience of the ecosystem in which they are established, and the vulnerabilities of the local socioeconomic system. As a result, the response of ecosystems is much harder to predict and cost compared to conventional engineered solutions [36].

In relation to the above, the sensitivity analysis of the economic assessment variables (Table 9) for the FWs indicates the uncertainty behind the valuation of NbSs and, thus, the three components of environmental benefits are characterized as critical aspects of the economic performance.

Table 9. Sensitivity analysis of economic performance variables.

| Variation of the ENPV Due to a ±1% Variation | Critical Judgment |
|---------------------------------------------|-------------------|
| Floating matrix                             | 8.67% Critical    |
| Gravel                                      | 0.17% Not critical |
| Potting soil substrate                      | 0.46% Not critical |
| Vegetation                                  | 0.93% Not critical |
| Burlap textile                              | 0.06% Not critical |
| Anchoring wire                              | 0.35% Not critical |
| Twine                                       | 0.12% Not critical |
| Pipes and elbows                            | 0.23% Not critical |
| Staff effort-works                          | 3.47% Critical    |
| Transportation                              | 0.93% Not critical |
| Solar position lights                       | 0.23% Not critical |
| Maintenance & monitoring services           | 50.15% Critical   |
| Consumables                                 | 6.08% Critical    |
| Water treatment benefits                    | 23.22% Critical   |
| Ecosystem value                             | 29.61% Critical   |
| Aesthetic value                             | 20.01% Critical   |

The highest variation was detected in maintenance services, since this is affected by the environmental risks and potential extreme events in the coastal and marine environment, plus socioeconomic aspects related to the cost and availability of services in the long term. In addition to the above, the floating matrix was also identified as a critical factor in the sensitivity analysis of economic performance. This may be since the matrix plays a key role in the durability and life expectancy of the entire system. In this perspective, there is a need of evidenced-based analysis on the product’s durability operating in the real environment, since the interaction of several factors may affect its life expectancy. For example, animals roosting and foraging, as well as the penetration of roots through the matrix, may reduce its tensile strength, whereas wave action and constant movement may reduce the armoring. Thus, it is of primary importance to study the FWs system in simulated coastal conditions to assess its durability and lifespan. This research challenge was also identified in the bibliometric analysis of Colares et al. [37], and the research work of Karstens et al. [38] who indicate that most studies about FWs remain on the laboratory- or pilot-scale, and point out the need to assess the behavior of the floating carrier in the natural environment.

4. Conclusions and Future Recommendations

To achieve sustainable growth, it is necessary to invest in operations aimed at limiting emissions, and improving resource efficiency. The use of OpenLCA for the assessment
of impacts has limitations in the availability of databases and the lack of information about their contents; however, the analysis of data may provide valid results for the initial screening of environmental impacts in the system under study. Indeed, the data analysis revealed that the production and acquisition of raw materials is a key stage in the establishment of FWs, and responsible for the main environmental impacts of the system. In this perspective, further research is needed in terms of GHGs emissions from FWs operating in marine environments, and in terms of durability, materials, and acquisition processes that will allow the development of robust FWs with a lower environmental footprint.

Furthermore, it is underlined that this assessment was based on lab-scale experiments of FWs operating in a controlled environment. The introduction of real-scale FWs in port areas and coastal areas is by far a more complicated procedure, and results, as well as operating parameters, may vary depending on several factors. In terms of system durability for example, the Department of Environment and Science, Queensland estimates a 20–30 year lifespan for FWs [39], and the improvements in UV-resistance, tensile strength, and armor- ing allow the projection of 60-year design life for today’s bio-havens [40]. However, in case of extreme events, and depending on their frequency, potential issues with breakdown may occur, increasing the cost of maintenance, and decreasing the life expectancy of the system. In addition, the exposure of FWs in freezing and thawing cycles, depending on the climate zone of establishment, causes mechanical stress in the matrix, and poses the risk of macro and micro plastic pollution due to the degradation of the matrix material [41].

An additional consideration, in terms of real-scale application, has to do with the operational characteristics of FWs in terms of the hydraulic residence time that is needed for the effective treatment of pollutants in port areas. The water residence time in ports may vary significantly depending on the technical/design characteristics of the port and the environmental characteristics of the area. For example, in the case of Tarragona Harbor in Spain, the estimated innermost residence time (days) reaches 11.01 ± 23.42 days [42]. Currents and wind direction play a critical role in the transportation and accumulation of pollutants (including oil pollutants) in specific areas of a harbor with increased residence time, while also affecting the level of coastal impact. Hydrodynamic modelling in combination with pollutants diffusion studies in a port area allows the determination of which areas are suffering more from water quality degradation (e.g., those with almost stagnant waters, or subject to multiple pollution sources), and allow the planning of appropriate measures to address the problem [43]. This is considered as a necessary step before the establishment of a FW, since it will provide valuable data regarding the potential hot spots of pollution, as well as the direction and speed of water movement in different port areas. Using this information, the designers of the FWs will be able to select the best location for the establishment of FWs, and make decisions about the necessary size, design, and technical configuration of the system to effectively address the water quality issues, depending on the purpose of establishment (e.g., water treatment, pollution barrier). In this perspective, lab-scale experiments of FWs provide valuable information regarding the treatment efficiency in relation to the time and the size of the system, and, thus, may guide the technical design of the final FWs system.

Concluding, the overall economic performance of the FWs system under study indicates that such systems could have positive socioeconomic impacts, and contribute to public welfare. In this perspective, FWs have the potential to serve the environmental needs of public and private users (e.g., ports, marines, fish-farms, touristic facilities, and local authorities), while providing multiple benefits to local societies, and contributing towards the goals of the European Green Deal. However, for the establishment and operation of FWs at a full-scale level in coastal and marine environments, there are still several scientific questions to be answered, as well as technical challenges to be addressed.

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