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Article

Water Footprint Assessment of Seawater Desalination: A Methodological Approach and Case Studies

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Abstract: Growing concerns have been raised regarding the potential environmental impacts of seawater desalination. While numerous studies have employed Life Cycle Assessment (LCA) to evaluate impacts, these efforts have primarily focused on quantifying the carbon footprint (CF) of seawater desalination; a tailored methodology for water footprint (WF) quantification remains notably lacking. To address this gap, this study presents a comprehensive WF assessment method for desalination plants, including system boundary definition, process-level inventory analysis, a quantification framework, and results interpretation. The methodology was applied to case studies on Reverse Osmosis (RO) plant. From a water scarcity footprint (WSF) perspective, the RO seawater desalination system consumes only 0.0366 m^3 of freshwater to produce 1 m^3 of product water. Furthermore, two methods assessed the water degradation footprint (WDF). The critical dilution volume method yielded a WDF of 0 m^3 H_2O eq for both plants, indicating no dilution was needed. Using the equivalency factor method, the acidification and eutrophication footprints for RO brine were 2.74×10^{-3} kg SO_2 eq/m^3 and 9.81×10^{-4} kg PO_4^{3-} eq/m^3, respectively. Additionally, analysis showed that electricity and steam consumption are the main drivers of water scarcity in RO and MED plants, respectively, with slight seasonal variations. This study resolves long-standing ambiguities in WDF accounting by clarifying the distinct application conditions of the critical dilution volume method and the equivalency factor method, while providing transparent, step-by-step calculation procedures often omitted in previous studies; practically, it offers industrial managers and environmental regulators a robust tool to precisely identify water-intensive hotspots.

Keywords: seawater desalination; water footprint; water scarcity footprint; water degradation footprint

1. Introduction

The accessibility of freshwater resources is a challenge in modern society because of increasingly serious water pollution caused by human activities. It is estimated that nearly half of the world's population experiences severe water scarcity for at least one month per year, and one-quarter faces extremely high levels of water stress [1–6]. As a vital approach to ensuring freshwater sustainability and safety, seawater desalination is playing a crucial role in satisfying the water demands of households, agriculture, and industries, particularly in arid, drought-prone, and coastal regions [7–13]. According to the report published by the International Desalination and Reuse Association (IDRA) [14], by the end of 2024, approximately 22,000 seawater desalination plants were operational globally. These plants had a cumulative contracted capacity of around 115 million m^3 per day, which was sufficient to provide for more than 300 million people. Current desalination technologies are divided into two categories: membrane separation technologies (such as RO, nanofiltration (NF), electrodialysis (ED), and forward osmosis (FO)) and thermal distillation technologies (such as MED and multi-stage flash (MSF)) [14–20]. Although these desalination technologies are relatively mature methods which positively contribute to relieving the water shortage, they will inevitably bring environmental impact issues [21–27]. Therefore, it is necessary to conduct environmental impact assessments on seawater desalination plants to quantify their relevant impacts.

One of the methods to evaluate how desalination processes change or impact environmental parameters is the LCA. It is a tool to quantify environmental aspects and potential impacts through the life cycle from raw material to disposal of the product system [28–30]. Over the past two decades, the application of LCA in the seawater desalination field has garnered substantial attention. Related reviews have summarized the use of LCA to evaluate desalination systems. Lee et al. [31] reviewed the desalination LCA cases published from 2000 to 2018. The results showed that most of these articles conduct evaluations focusing on the CF and energy consumption. Aziz et al. [32] reviewed 62 previous studies on LCA of desalination technology from 2004–2019, which emphasized the application of LCA to desalination by means of research objective, scope of study, life stages, and impact assessment. In addition, this review indicated that LCA research focused on RO technology for seawater desalination has surpassed other desalination technologies. Regardless of the desalination technology employed, its associated impacts can be primarily categorized into two aspects: CF assessment and WF assessment. In recent studies, the majority of research has focused on CF or green-house gas (GHG) emissions as a key environmental impact factor. Wang et al. [33] conducted a comprehensive review of 34 studies (2004–2023), through which they analyzed CF data from 211 desalination scenarios. The results indicate the CF advantage of RO and low-carbon heat-driven MSF, MED, and membrane distillation (MD). They further emphasize that operational energy consumption plays the dominant role in the desalination CF, specifically regarding its amount, form, and source. Liu et al. [34] used hybrid input-output life-cycle analysis (IO-LCA) to compare the lifecycle GHG emissions of wind-driven and traditional thermal seawater desalination systems for low-carbon assessment. Cabrera et al. [35] proposed an approach enabling low-carbon operation of large-scale desalination on arid islands with weak electrical grids, and the results indicate a potential reduction of 77.4% of the CF. Pandit et al. [36] observed that hybridizing the desalination system results in substantially lower carbon emissions than conventional methods. Specifically, the PVT-integrated FO and MED achieve a specific energy consumption (SEC) of 1.8 kWh/m^3–4 kWh/m^3, reducing carbon emissions by 40–60% and outperforming traditional RO and MSF/MED. Leon et al. [37] investigated the ecological footprint of seawater reverse osmosis (SWRO) desalination plants in the Canary Islands (Spain) and reported a carbon footprint of 0.2 t CO_2 per person per year. Kabiri et al. [38] compared various energy recovery devices (ERDs), including turbochargers, Pelton wheels, Francis turbines, and pressure exchangers, applied in a SWRO unit. Their economic analysis revealed that the pressure exchanger led to the lowest cost savings among the devices considered.

When compared to extensive studies on CF quantification, research on measuring how water is used and evaluating its environmental impacts in seawater desalination is relatively rare. One widely used method for this life-cycle water use assessment is the "water footprint" approach created by Hoekstra and colleagues [39]. Conceptually, it's similar to the "CF" used in LCA. But there's a key difference: while CF is part of LCA, the WF concept developed independently from LCA methodology and has prioritized volumetric water use quantification [40]. Moreover, its practical applications still lack scientific methods and unified assessment standards, particularly in desalination fields where boundaries and methods for WF quantification are absent. A survey of water footprint-related articles published in the past decade reveals that WF assessment has been primarily applied to agricultural water resources and environmental evaluation, regional water resources and urbanization development, industrial water use and cleaner production management, as well as social consumption and transboundary water resource allocation [41–50]. However, there is a scarcity of literature focusing specifically on the water footprint of seawater desalination systems. Therefore, how to accurately analyze and quantify WFs, and how to develop applicable quantification methods suitable for assessing seawater desalination engineering that cover water quantity, quality, environment, and resources, remains a critical challenge requiring urgent attention.

To address the lack of standardized methodologies, this study provides a feasible WF assessment methodology in seawater desalination and quantifies the effects of WF covering the complete process chain, including seawater intake, pre-treatment, desalination, post-treatment, and brine discharge. Although recent studies have begun to apply the water footprint concept to desalination systems [51–55], these existing efforts exhibit critical limitations that hinder their direct applicability and methodological rigor. First, although Zhang et al. [52] and Ding et al. [51] provided baseline WF values for RO and MED plants, their studies lacked a transparent unit-process breakdown distinguishing the contributions of intake, pretreatment, desalination, and brine discharge. Second, Meneses et al. [53], Zhou et al. [54] and Muñoz et al. [55] focused on environmental impacts within the

LCA framework but failed to provide distinct and operational quantification models for the two core dimensions of WSF and WDF, more critically, overlooked the fundamental conceptual confusion between the critical dilution volume method and the equivalency factor method when calculating WDF. To rigorously address these gaps, the novelty of this study is threefold: (1) Methodological clarification—for the first time in the desalination context, we systematically distinguish the two WDF calculation approaches and explicitly specify their respective application conditions based on discharge compliance status, thereby resolving a long-standing ambiguity in implementing ISO 14046; (2) Computational transparency—unlike prior studies that merely report final aggregated results, we provide a complete, step-by-step inventory calculation procedure, including the derivation of WF characterization factors specific to membrane modules, chemicals, and energy carriers that are often missing in commercial databases; and (3) Process-level attribution—the developed framework enables precise attribution of water scarcity impacts to specific unit operations, offering actionable insights for targeted process optimization rather than generic conclusions. Additionally, this study offers a computational demonstration via a case study, thereby proposing an operable technical pathway for practical engineering applications.

This methodology defines system boundaries through full-process analysis, incorporating direct and indirect water consumption from energy use, material inputs, chemical dosing, and brine discharge impacts, etc. Quantification models were developed for representative desalination systems, with case studies demonstrating their application to RO and MED technologies. These practical examples not only provide methodological guidance but also verify the framework's capability to address critical gaps in WF accounting for desalination systems. At the same time, the case results show the difference between WSF and WDF in different technologies. This study aims to establish a scientific and effective methodology for assessing the WF of seawater desalination, enabling objective assessment of its environmental impacts, and supporting sustainable development of the desalination industry while promoting ecological conservation.

2. ISO Standards

Internationally, there exist two principal methodologies for WF evaluation: the Water Footprint Assessment (WFA) established by the Water Footprint Network (WFN) [56,57], and the ISO 14046 [58] WF standard published by the International Organization for Standardization (ISO) in 2014. These two frameworks demonstrate distinct methodological orientations and assessment objectives. The WFN approach emphasizes quantifying freshwater appropriation within specific temporal and spatial boundaries, employing a tripartite classification system comprising blue water (surface and groundwater consumption), green water (rainwater utilization), and grey water (pollution assimilation volume) to evaluate the magnitude of freshwater impacts exerted by organizations or production systems. In contrast, ISO 14046 [59] adopts a LCA perspective to systematically examine water-related environmental impacts throughout a product's value chain, from raw material extraction to end-of-life disposal. Based on the full-process characteristics of the seawater desalination process, the methodology introduced in this paper is developed with reference to the relevant standards of the ISO. Therefore, this section elaborates on ISO standards relevant to LCA and WF quantification, which serve as foundations for establishing the WF assessment approach of seawater desalination in this paper.

LCA is an ISO-standardized method that has been commonly used to evaluate the environmental performance of desalination. According to ISO 14040 [60] and ISO 14044 [61], LCA consists of four phases: goal & scope, life cycle inventory (LCI), life cycle impact assessment (LCIA), and interpretation. As defined in ISO 14040 [60], assessing the environmental impacts of products with LCA is an approach of "cradle-to-grave", and this concept has been applied in desalination. The environmental impact of seawater desalination is not only limited to the process of producing fresh water, but also involves the consumption of natural resources and the emission of pollutants through indirect demands, such as infrastructure construction, energy production, chemical production, membrane and equipment manufacturing, and waste management, etc. Following the impact assessment, "interpretation" with checking methods in ISO 14044 [61] could guide decision-makers to better understand uncertainties and assumptions. As the international standard for WF assessment, ISO 14046 [58] establishes LCA-based principles, requirements, and guidelines for quantifying and reporting WF of products, processes, and organizations. By providing a unified methodological framework, it enables producers, consumers, and policymakers to identify water-intensive activities across value chains, thereby promoting sustainable resource management. The standard's core objectives are to: (1) offer quantitative analysis instruction for WF evaluation, (2) standardize impact communication, (3) support environmentally informed decision-making through comparable WF disclosures, and (4) help organizations to identify and manage environmental impacts related to water usage, improve water-use efficiency, reduce unnecessary water consumption, and advance sustainable development. The main contents include: (1) clearly defining WF as water-related emissions generated throughout the entire life cycle of a product or service; (2) providing methodologies such as LCA, water balance analysis, and WF quantification; (3) suggesting that reports include details such as types of WF, volume, and spatial distribution; (4) encouraging organizations to incorporate environmental impact assessments (e.g., water scarcity, pollution) into their evaluations. The ISO WF methodology, grounded in the principles of LCA, comprises four principal phases. These phases include: (1) goal and scope definition, establishing the analytical boundaries and functional units; (2) WF inventory analysis, quantifying water inputs, outputs, and quality changes across the product system; (3) WF impact assessment, evaluating hydrological consequences through spatially explicit indicators; and (4) interpretation, synthesizing results to inform water stewardship strategies.

In addition, ISO/TR 14073 [62] outlines how to select appropriate WF assessment types (e.g., water consumption footprint, water quality footprint, WSF) based on factors such as assessment goals, system boundaries, water resource types, and hydrological changes. Specifically, this technical report provides multiple case studies covering diverse sectors and scenarios, including power plants, rice cultivation, and municipal water management, etc. Each case study details the assessment's goal, scope, inventory analysis, impact assessment, and interpretation of the results. By demonstrating how to quantify and report WFs across varying contexts for products, processes, and organizations, the report aims to foster broader adoption and interpretation of ISO 14046 [59].

3. Model Construction

3.1. Goal and Scope Definition

The life cycle comprises three stages: (1) construction, (2) operation and maintenance, and (3) disposal. According to ISO 14046 [59], a WF assessment can focus on specific life cycle stages based on its purpose, the availability of representative data for those stages, and the significance of their impact on the results. Four types of system boundaries have been considered in previous LCA studies: cradle-to-cradle, cradle-to-gate, gate-to-gate, and cradle-to-grave. Cradle-to-grave assessment covers the construction, operation and maintenance, and disposal stages of the desalination system's entire life cycle. Key activities within these stages include seawater extraction and processing, treatment, plant infrastructure, transportation, plant operation, distribution and use, dismantling, and final waste disposal. Cradle-to-cradle covers only the raw material extraction process. Cradle-to-gate includes the process from raw material extraction to the plant operation phase. Gate-to-gate focuses specifically on the plant operation phase itself. Numerous case studies on the environmental impact assessment of desalination have consistently shown that the operation and maintenance (O&M) phase is the dominant contributor across all desalination technologies and assumptions [32–36], whereas the environmental impacts of construction and disposal are significantly smaller than those of O&M [16,37,38,51,52]. This finding is further supported by a systematic review conducted by Kyungsun et al. [31], who examined 38 studies comprising 295 LCA scenarios for desalination systems published between 2000 and 2018. In their analysis of life cycle phases, Kyungsun et al. concluded that O&M is the predominant contributor regardless of the technology employed (i.e., MSF, MED, or RO) and other underlying assumptions, while construction and disposal contribute relatively minor impacts.

Given that seawater desalination systems exhibit long life cycles, obtaining reliable data on the production, installation, and decommissioning phases is often difficult in practice. Consequently, a "gate-to-gate" boundary covering all stages of the seawater desalination production process is defined as the system boundary for quantifying the WF of a desalination plant. This approach is consistent with common practice: among the 18 case studies reported in ISO/TR 14073 [62], 11 adopted the gate-to-gate system boundary. Therefore, for both membrane and thermal desalination processes, the system boundary for WF quantification includes the following stages: seawater intake, pre-treatment, desalination, post-treatment of desalinated water, and concentrated brine discharge. The schematic diagram of the membrane system boundary for quantifying the WF of seawater desalination is presented in Figure 1.

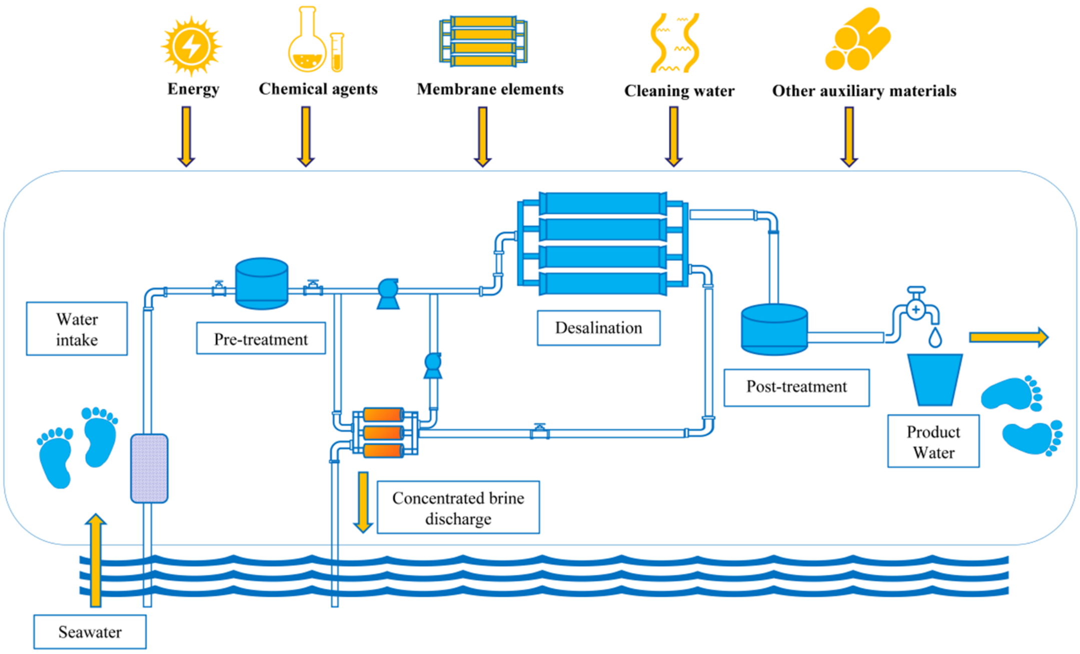

Figure 1. Schematic diagram of the membrane desalination system boundary for WF quantification in seawater desalination.

3.2. Inventory Analysis

Within the defined system boundary, a comprehensive inventory analysis of water-related items encompassing the system's input and output streams must be conducted. For the input side, it mainly includes continuous energy consumption, chemical agents and periodically replaced membranes, etc. Equipment such as high-pressure pumps and water tanks is categorized as one-time inputs. Compared to other continuous inputs, its contribution to the total WF is minimal, and it is excluded from the WF boundary. Therefore, the relevant data must be investigated and collected, including raw seawater, energy sources, chemical dosing, membrane elements replacements, cleaning water and other auxiliary materials.

The output side primarily consists of product water and concentrated brine. For product water, this analysis focuses on the production volume of product water. For concentrated brine, its water quality parameters and discharge method are critical. For the concentrated brine directly discharged into water bodies, the WDF is calculated using the critical dilution volume method first. When the concentrated brine is discharged only after meeting the required standards, the equivalency factor method is applied. The specific calculation formulas for these WDF methods will be introduced in the subsection WDF Calculation Method.

In summary, the LCI stage in this study strictly follows the requirements of ISO standards [59,60]. The LCI covers the gate-to-gate operational phase of seawater desalination, including seawater intake, pre-treatment, desalination (RO/MED), post-treatment, and brine discharge. Main input data for LCI: (1) Water input: raw seawater intake volume, product water used for chemical preparation and membrane cleaning. (2) Energy input: electricity (for pumps, membranes, and auxiliary systems) and steam (for MED thermal process). (3) Material input: coagulants, disinfectants, antiscalants, reducing agents, cleaning chemicals, and periodic replacement of membrane elements (ultrafiltration (UF), RO) and filter cartridges. Main output data for LCI: (1) Product output: volume and quality of desalinated fresh water. (2) Waste output: concentrated brine (volume, salinity, and pollutant concentrations), solid waste from pre-treatment, and process wastewater.

Data collection should prioritize obtaining precise original data. However, practical investigations reveal that only average consumption rates of various materials and energy over specific periods (e.g., monthly, quarterly, or annual averages) are typically obtainable, and the appropriate averaging period should be determined based on the specific operation status of the desalination project. The finalized WF inventory for the seawater desalination system should encompass, but is not limited to, the contents detailed in Table 1.

Table 1. WF inventory of seawater desalination.

Data Type	Items	Data Inventory
Input	Seawater	Water intake volume, water quality, water intake method
	Energy	Types and quantities of energy; Pressure, temperature, and quantity of steam
	Membrane elements	Quantities, area, and replacement cycle of ultrafiltration/microfiltration and RO membrane elements
	Chemical agents	Types, quantities, and water consumption of bactericides, flocculants, antiscalants, cleaning agents, etc.
	Cleaning water	Cleaning water volume and cleaning cycle for membrane elements and heat exchange tubes, etc.
	Other auxiliary materials	Quantities and replacement cycle of materials such as gratings, security filter cartridges, activated carbon, etc.
Output	Product water	Water volume, water quality
	Concentrated brine	Discharge volume, water quality, discharge method; other discharged water: quantity and quality of system flushing and cleaning discharge

3.3. Impact Assessment

3.3.1. Selection of Impact Types

When assessing the potential environmental impacts associated with water capacity variations, it is essential to evaluate the WSF of seawater desalination systems. The assessment framework should encompass the following contents. (1) The WSF of seawater desalination systems can be divided into direct WSF and indirect WSF according to whether water consumption occurs in the production process. Direct WSF comes from operational water inputs such as membrane cleaning water, chemical agents dosing water, and product water output; while indirect WSF includes indirect water use throughout the entire supply chain, including chemical production, energy generation (steam/electricity), and water consumption during membrane module manufacturing. (2) Quantify the proportions of cleaning water, chemical agents dosing water, and other components in direct WSF, and analyse their contribution to WSF based on relative proportions. (3) Quantify the proportions of water usage for energy generation, chemical agents production, membrane element manufacturing, and other auxiliary materials in indirect WSF, and analyse their contribution to WSF based on relative proportions. (4) Based on the quantification results and contribution analysis, assess the impacts of each process stage (including water intake, pre-treatment, desalination, and post-treatment) on WSF. (5) In addition, the effects of factors such as climate, geographical location, and accounting time span on WSF can be assessed based on the quantification results and contribution analysis.

When assessing the potential environmental impacts associated with water quality variations, a systematic assessment of the WDF in desalination systems should be implemented. The assessment framework should encompass the following contents. (1) The WDF of seawater desalination systems can be categorized into direct WDF and indirect WDF according to whether the negative water quality changes are directly caused by the output water. Direct WDF primarily comes from concentrated brine discharge, i.e., the result of seawater concentration. Indirect WDF includes chemical agents dosing (e.g., antiscalants, bactericides) and their cascading ecological impacts. However, residual agents and by-products generated during the chemical agents dosing process generally become incorporated into the concentrated brine. Therefore, for WDF quantification purposes, this concentrated brine is defined as the aggregate of process-generated brine, chemical cleaning wastewater, and flushing wastewater. (2) Based on the accounting results and contribution analysis, the impacts of each adverse element on the water degradation footprint are evaluated. (3) In addition, the effects of influencing factors such as climate, geographical location, and accounting time span on the WDF could be assessed according to the quantifying results and contribution analysis. The detail environmental impacts of seawater desalination WF are shown in Figure 2.

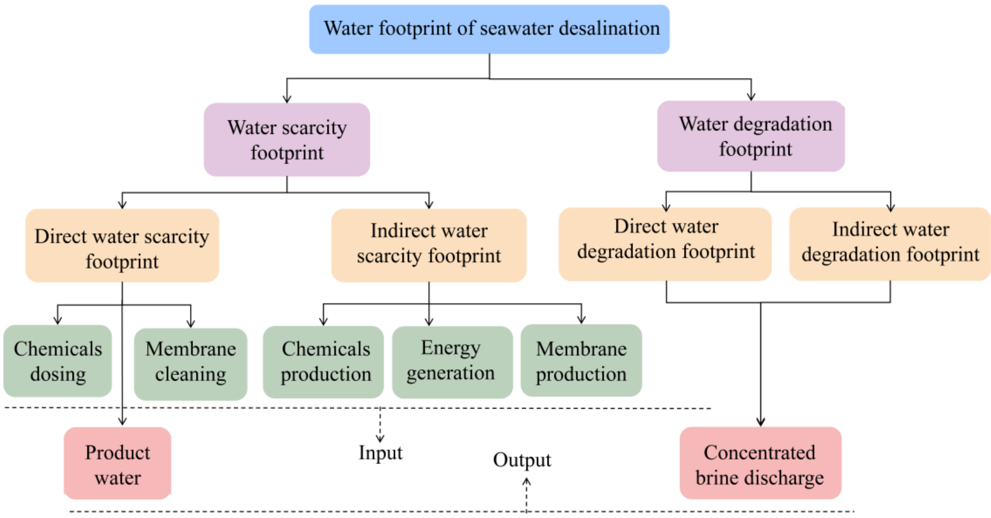

Figure 2. Schematic diagram of the types of environmental impacts for seawater desalination WF.

3.3.2. The Calculation Methods of WF

(1) WSF Calculation Method

The daily total WSF of seawater desalination is calculated by summing the daily freshwater consumption of each unit process, which can be categorized into direct water scarcity and indirect water scarcity components.

The WSF of seawater desalination is calculated according to Equation (1):

$$WF_{sc}=\sum_{i=1}^{n} V_i+\sum_{j=1}^{n} a_j \times M_j \tag{1}$$

In the formula: $WFsc$ represents daily total WSF (m^3), i represents unit process within the system boundary contributing to direct WSF, V is freshwater consumption volume for i (m^3), j represents energy or material input within the system boundary contributing to indirect WSF, a is the WSF characterization factor of energy/material j (m^3/mg), and M represents daily consumption quantity of j (mg).

(2) WDF Calculation Method

Two primary methodologies existed can be used to quantify the WDF of seawater desalination: the critical dilution volume method and the equivalency factor method [63,64]. If there are locally established discharge standards for desalination concentrated brine, the critical dilution volume method should be employed first when calculating the water degradation footprint. This involves computing the theoretical volume of water required to dilute the pollutant to the concentration specified by relevant emission standards [63–67]. For instance, when quantifying the WF of seawater desalination projects in China, the relevant parameters stipulated in the published Chinese standard "Requirements for Discharge of Seawater Desalination Brine (HY/T 0289-2020)" can be referenced and utilized. For concentrated brine that has already met the discharge standards, the equivalent factor method can continue to be used for calculation [68–70]. Calculated using the equivalency factor method, the WDF primarily comprises the eutrophication footprint, the acidification footprint, and the freshwater ecotoxicity footprint [56,71–73]. Therefore, the selection of the appropriate WDF accounting method depends on the specific pollutant discharge pathway, and WDF analysis can be conducted using one or both of these methods. The critical dilution volume method shall be calculated using Equation (2):

$$WF_{deg}=\sum_{j=1}^{m}\left(\sum_{i=1}^{n} a_{deg,\,i} \times M_{deg.i}\right) \tag{2}$$

In the Formula (2): WF_{deg} represents the seawater desalination WDF value, and the unit is cubic meters of equivalent water (m^3 H_2O eq); i indicates the pollutant type; j is the unit process within the system boundary contributing to WDF; $a_{deg,\,i}$ is the WDF characterization factor for pollutant "i", defined as the reciprocal of the concentration limit for target pollutant "i" in the applicable desalination brine discharge standard, with units of cubic meters of water equivalent per kilogram (m^3 eq H_2O/kg); $M_{deg,\,i}$ is the emission amount of pollutant i (kg).

The equivalency factor method shall be calculated using Equation (3):

$$WF_{deg,\,k}=\sum_{j=1}^{m}(\sum_{i=1}^{n}a_{deg,\,ik}\times M_{deg.ik}) \tag{3}$$

In the Formula (3): k is the impact category of the WDF (e.g., water eutrophication footprint, water acidification footprint, or water ecotoxicity footprint). Since different categories represent fundamentally different environmental impact mechanisms, their quantified results cannot be directly summed. $WF_{deg,\,k}$ represents the seawater desalination WDF value for category k, with units specific to each impact category (e.g., kg PO_4^{3-}-eq for eutrophication); i indicates the pollutant type; j is the unit process within the system boundary contributing to WDF; $a_{deg,\,ik}$ is the WDF characterization factor for pollutant i in category k; $M_{deg,\,ik}$ is the emission amount of pollutant i (kg).

It should be noted that the above two WDF calculation methods assume that the concentrated brine is discharged directly into the receiving water body without further utilization or treatment. However, if the brine is partially or fully utilized as a source for mineral recovery (e.g., extraction of magnesium, lithium, or calcium) or as a medium for carbon sequestration (e.g., carbon dioxide sequestration), via mineralization with CO_2, the conventional WDF framework may overestimate the actual water degradation impact. In such cases, the WDF should be recalculated based on the net pollutant load entering the environment after utilization. Specifically, for mineral recovery, pollutants removed from the brine (e.g., heavy metals or scale-forming ions) should be excluded from $M_{deg,\,i}$ in Equations (2) and (3). For carbon sequestration applications, if the process alters brine pH or reduces its ecotoxicity, the characterization factors $a_{deg,\,i}$ or $a_{deg,\,ik}$ may need to be adjusted according to the post-treatment discharge standards. Furthermore, if the brine is entirely consumed in a utilization process (e.g., zero-liquid discharge systems coupled with mineral recovery), the WDF could be considered negligible for the discharged residual stream. Therefore, future LCA studies on desalination should explicitly report whether brine utilization is applied and, if so, provide a modified WDF based on the ‘utilization-adjusted emission load.

3.3.3. Functional Unit

Due to the varying production capacities of different seawater desalination plants, a functional unit for seawater desalination must be defined to ensure comparability. The functional unit is established as the unit product water of seawater desalination, i.e., 1 m^3 of product water. Therefore, the WSF or WDF of seawater desalination is expressed as the footprint generated during the production of a unit volume of product water, as shown in Equation (4):

$$WF_{up}=\frac{WF_d}{V_d} \tag{4}$$

where WF_{up} represents the WSF or WDF per unit product; WF_d represents the daily total amount of either WF_{sc} or WF_{deg}, and V_d is the daily water production volume of the desalination plant (m^3).

3.4. Interpretation of WF Assessment Results

The interpretation of seawater desalination WF assessment results should include: (1) analyzing the calculation results of seawater desalination WF; (2) identifying the key impact factors of the seawater desalination WF; (3) discussing the limitations and uncertainties of the seawater desalination WF assessment.

3.5. Data Acquisition for Seawater Desalination WF

According to the seawater desalination WF calculation methodology, the required data primarily include: (1) actual operational data from seawater desalination plants, and (2) the characterization factors for WDF and WSF. Operational data can be collected through methods such as on-site investigations and literature reviews. A standardized questionnaire survey administered to plant operators/manufacturers is highly recommended, as this approach facilitates the direct and accurate acquisition of key process-specific parameters (e.g., energy consumption, chemical usage, water recovery rate). The WDF and WSF characterization factors can be acquired via the Efootprint platform by accessing authoritative LCI databases, such as the Chinese Reference Life Cycle Database (CLCD), the European Reference Life Cycle Database (ELCD), and Switzerland’s Ecoinvent commercial database. When specific characterization factors are unavailable in existing databases, they may be approximated by using factors from other materials exhibiting comparable chemical structures and/or production methods.

4. Case Studies

4.1. Quantifying WFs of Seawater RO Desalination Project

The WF of an RO desalination plant with a water production capacity of 100,000 m^3/d was quantified using this method. The operation data of an RO seawater desalination plant is shown in Table 2. The UF membrane modules adopted in this desalination system have a membrane area of 50 m^2 with a replacement cycle of 60 months. The RO membrane elements have a membrane area of 37 m^2 and a replacement cycle of 48 months. The cartridge filter has a replacement cycle of 3 months. The detailed quantification process and results are as follows. Based on the boundary-setting principles outlined in Sections 3.1 and 3.2, and by combining them with the plant's specific process characteristics, the WF system boundary encompasses the following stages: water intake, pre-treatment, UF treatment, RO treatment, and concentrated brine discharge.

Inputs include: seawater intake, electrical energy, chemical agents, and membrane elements requiring replacement (including security filter cartridges, UF membrane elements, and RO membrane elements). Since cleaning water is sourced from UF-treated desalinated water, and chemical preparation water is sourced from RO product water, these internal water flows (cleaning water and chemical preparation water) are excluded from the system inputs. Outputs comprise product water and concentrated brine.

The WF coefficients of the materials involved were primarily sourced from the Ecoinvent, CLCD-China-ECER, and ELCD databases. In these databases, due to the lack of WF data on UF membrane modules, RO membrane elements, and safety filter cartridges, relevant coefficients were obtained through manufacturer investigations. Based on the collected production data and process modeling, the WF coefficients for these membrane components were calculated as follows: 175.5 kg H_2O/m^2 for UF membrane elements, 241.7 kg H_2O/m^2 for RO membrane elements, and 0.67 kg H_2O per cartridge for safety filter cartridges (specification: 30 m^3/h). The WF inventory analysis results for the RO seawater desalination plant are presented in Table 3. Since the WSF calculation requires annual average inputs, the inventory data in the table are annual averages. Using Equation (1), the WSF was calculated as 3.66×10^{-2} m^3 per m^3 of product water.

As introduced above, to regulate this environmentally sensitive effluent, China published the marine industry standard "Requirements for Discharge of Seawater Desalination Brine (HY/T 0289-2020)" in 2020. This standard specifies the maximum permissible concentrations of pollutants in desalination brine.

For calculating the WDF of brine released through direct discharge, the critical dilution volume method is first applied. The method requires comparing the brine's pollutant concentrations against HY/T 0289-2020 thresholds. These regulatory limits are compared with monitored water quality data from the RO desalination brine in Table 4. Results indicate full compliance with all parameters with HY/T 0289-2020. According to the critical dilution volume method, the theoretical dilution volume needed to reduce pollutant concentrations to regulatory standards is zero, resulting in a WDF of 0 m^3 H_2O eq. It is important to recognize that the WDF quantification methods described above, rely on compliance with regulatory concentration limits as the sole criterion for determining water quality degradation. This approach, while standardized and operational, inherently simplifies the complexity of environmental impacts. A calculated WDF of 0 m^3 H_2O eq, which occurs when all pollutants in the discharged brine are below the specified standard thresholds, does not signify the absence of environmental consequences. Rather, it indicates that, from the perspective of this specific regulatory compliance-based metric, no additional dilution volume is theoretically required to meet the discharge permit.

Table 2. The operational data of RO and MED seawater desalination plants.

Seawater Desalination Plant	Data Type and Items		Summer	Spring/Autumn	Winter
RO seawater desalination plant	Electrical energy (kW·h/d)		323,000	333,900	371,400
	Disinfectant	NaClO (kg/d)	1300	810	470
	Coagulant	$FeCl_3$ (kg/d)	8080	7480	7510
	Antiscalant	$Na_4P_2O_7$ (kg/d)	502	427	464
	Reducing agent	$NaHSO_3$ (kg/d)	756	738	723
	Cleaning chemicals	NaClO (kg/d)	15	15	15
		H_2SO_4 (kg/d)	18.2	18.2	18.2
		NaOH (kg/d)	14.8	14.8	14.8
	Membrane elements	UF membranes	6400	6400	6400
		RO membranes	8190	8190	8190
		Safety filter cartridges	360	360	360
	Concentrated brine (m^3/d)		162,000	165,000	170,000
	Product water (m^3/d)		95,000	96,000	100,000

Table 3. WF inventory analysis results for RO and MED seawater desalination plants.

Seawater Desalination Plant	Data Type and Items			Summer	Spring/Autumn	Winter	Annual Average Data
RO seawater desalination plant	Input (m^3/m^3 product water)	Electrical energy		3.45×10^{-2}	3.53×10^{-2}	3.77×10^{-2}	3.56×10^{-2}
		Disinfectant	NaClO	5.92×10^{-5}	3.65×10^{-5}	2.10×10^{-5}	3.80×10^{-5}
		Coagulant	$FeCl_3$	3.59×10^{-7}	3.29×10^{-7}	3.17×10^{-7}	3.33×10^{-7}
		Antiscalant	$Na_4P_2O_7$	1.39×10^{-7}	1.17×10^{-7}	1.22×10^{-7}	1.24×10^{-7}
		Reducing agent	$NaHSO_3$	6.53×10^{-8}	6.31×10^{-8}	5.94×10^{-8}	6.27×10^{-8}
		Cleaning chemicals	NaClO	6.83×10^{-7}	6.76×10^{-7}	6.49×10^{-7}	6.71×10^{-7}
			H_2SO_4	1.53×10^{-8}	1.51×10^{-8}	1.45×10^{-8}	1.50×10^{-8}
			NaOH	1.71×10^{-7}	1.69×10^{-7}	1.63×10^{-7}	1.68×10^{-7}
		Membrane elements	UF membranes	3.24×10^{-4}	3.21×10^{-4}	3.08×10^{-4}	3.19×10^{-4}
			RO membranes	5.35×10^{-4}	5.29×10^{-4}	5.09×10^{-4}	5.26×10^{-4}
			Safety filter cartridges	2.82×10^{-5}	2.79×10^{-5}	2.68×10^{-5}	2.77×10^{-5}
	Output (m^3/d)	Concentrated brine		162,000	165,000	170,000	165,500
		Product water		95,000	96,000	100,000	96,750

Table 4. Concentrated brine discharge standard limits of HY/T 0289-2020 and water quality of RO and MED desalination plants effluent.

Item	Unit	Limit	Water Quality of RO Plant Concentrated Brine
pH	—	6.5~8.5	7.98
Fe	mg/L	≤0.3	<0.0045
Aluminum	mg/L	≤0.05	<0.04
Total Phosphorus	mg/L	≤0.5	0.098
Copper	mg/L	≤0.2	0.0027
Chromium	mg/L	≤0.05	0.00061
Nickel	mg/L	≤0.02	0.00163

When calculating the WDF using the equivalency factor approach, the WDF can be mainly categorized as follows: (1) the eutrophication footprint, which quantifies impacts from excessive nitrogen (N) and phosphorus (P) discharges, leading to algal blooms, oxygen depletion, and biodiversity loss, expressed in kg PO_4^{3-}-eq or kg N-eq; (2) the acidification footprint, measuring acidifying pollutants (e.g., SO_2, NO_x from industrial emissions) that reduce water pH, mobilize toxic metals, and impair aquatic ecosystems, with units of kg SO_2-eq; and (3) the freshwater ecotoxicity footprint, assessing chronic toxicity of synthetic chemicals (e.g., pesticides, heavy metals, pharmaceuticals) through bioaccumulation and trophic transfer. In this project, $FeCl_3$ was utilized as the coagulant during the pretreatment stage, with the objectives of removing suspended solids, colloids, reducing turbidity, and eliminating organic substances. However, residual coagulant has been transported off-site for disposal along with the solid wastes generated in the pretreatment process and will not enter the final concentrated seawater stream. Consequently, it does not contribute to the freshwater ecotoxicity footprint.

The applied $NaHSO_3$ and H_2SO_4 have their residues discharged into the sea along with the concentrated brine, which would cause water acidification. The scale inhibitor employed herein was sodium pyrophosphate. After reacting with easily precipitating ions in water, it generates phosphate, leading to water eutrophication. Therefore, the water degradation footprint of the seawater desalination process in this project includes the water acidification footprint and the water eutrophication footprint. According to Equation (3), the calculated acidification footprint is 2.74×10^{-3} kg SO_2 eq/m^3 of concentrated brine, and the eutrophication footprint is 9.81×10^{-4} kg PO_4^{3-} eq/m^3 of concentrated brine.

4.2. Interpretation of the Results

To validate the robustness of our case-study results and to contextualize their significance, we performed three levels of cross-comparison.

4.2.1. Cross-Technology Comparison

The WSF results for RO plants (0.0366 m^3/m^3 and 0.0289 m^3/m^3) and MED plants (0.1294 m^3/m^3 and 0.153 m^3/m^3) reveal a consistent and substantial difference between the two technologies, with MED exhibiting a 3- to 5-fold higher WSF. This disparity can be mechanistically attributed to two intrinsic factors. First, the thermal nature of the MED process requires a substantial external heat input (steam), which, in our case studies, was generated by dedicated coal-fired boilers: a water-intensive energy supply chain. In contrast, RO relies primarily on electrical energy for high-pressure pumps, which, although also water-intensive on the supply side, exhibits a lower specific water consumption per unit of final energy delivered. Second, the MED process typically operates at lower water recovery rates (typically 30–40%) compared to RO (40–50%), resulting in higher feedwater intake per unit of product water, which indirectly increases the associated energy demand for seawater abstraction and pretreatment. This cross-technology validation confirms that the methodology successfully captures the mechanistic differences between membrane and thermal desalination routes.

4.2.2. Literature Benchmarking

The literature we studied reveals that only a few individual studies focus on small-scale, novel seawater desalination processes. For example, Alrashidi et al. [74] examines four seawater desalination processes. These include single-stage systems and coupled hybrid systems (SWRO + SWED), and the reported water scarcity footprint values are relatively high, ranging from 2.46 to 3.92 m^3 of water consumed per m^{3}of water produced. In contrast, for large-scale seawater desalination plants with a capacity of ten thousand tons per day or more [51,52], the water consumption per cubic meter of freshwater produced ranges from 10^{-1} to 10^{-2} m^3 for thermal processes and from 10^{-2} to 10^{-3} m^3 for membrane processes. These values are fully consistent with the findings of this study.

The water degradation footprint can be compared with data from references [53–55]. The reported acidification footprints range from 10^{-2} to 10^{-3} kg SO_2 eq/m^3, and eutrophication footprints range from 10^{-3} to 10^{-4} kg PO_4^{3-} eq/m^3. The values calculated in this study (an acidification footprint of 2.74×10^{-3} kg SO_2 eq/m^3 and a eutrophication footprint of 2.99×10^{-4} kg PO_4^{3-} eq/m^3 for RO brine) fall within the ranges reported in prior literature.

This benchmarking exercise confirms that our methodological choices do not produce outlier results, thereby supporting the generalizability of our framework. More importantly, our study extends beyond these previous works by providing a fully transparent, step-by-step calculation procedure and by offering process-level attribution, which enables targeted optimization—features that earlier studies did not deliver.

4.2.3. Validation of WDF Results against Regulatory and Ecological Relevance

The WDF calculation for RO brine using the critical dilution volume method yielded a value of 0 m^3 H_2O eq, which requires careful interpretation. This result does not indicate that brine discharge has no ecological consequences; rather, it reflects the fact that the monitored pollutant concentrations were all below the thresholds stipulated in the Chinese marine industry standard HY/T 0289-2020. To validate this regulatory-based result against ecological relevance, we cross-checked the effluent quality data against ecotoxicological benchmarks reported in the literature [24,26]. The measured concentrations of heavy metals (Fe < 0.0045 mg/L, Cu 0.0027 mg/L, Cr 0.00061 mg/L, Ni 0.00163 mg/L) are all well below the acute toxicity thresholds for marine organisms documented in previous ecotoxicological studies. However, we acknowledge that the critical dilution volume method does not capture the cumulative salinity increase in semi-enclosed water bodies or the physical disturbance from discharge outfalls—limitations that we explicitly discuss in Section 4.1. Therefore, the validation of WDF results should be understood as conditional: the brine is compliant with current chemical-based regulatory standards, but a more holistic ecological validation would require complementary salinity and ecotoxicity impact assessments, which we identify as a priority for future research.

4.3. Analysis of Influencing Factors for WSF in Seawater Desalination

To compare the effects of different processes on WSF, this study incorporated two additional large-scale desalination plant cases alongside the two previously analyzed. The projects introduced above are referred to as RO project 1, while the three newly introduced cases are RO project 2, MED project 1, and MED project 2. The calculated WSF for seawater desalination was 0.0366 m^3 per m^3 and 0.0289 m^3 per m^3 for RO project 1 and RO project 2, respectively. In contrast, the MED project 1 and MED project 2 plants exhibited significantly higher WSFs of 0.1294 m^3 per m^3 and 0.153 m^3 per m^3, respectively. This indicates that the WSF of the MED process was approximately 3 to 5 times higher than that of the RO process in these cases. It should be noted that the steam supply for both MED plants analyzed in this study was generated by coal heating water specifically for the desalination process. If the MED plant were co-located with a power plant and utilized waste heat from the power generation process as the steam produced source, the results would likely differ substantially.

Figure 3 illustrates the composition analysis of WSF for the four seawater desalination plants. In the RO Project 1 and Project 2, electricity consumption dominated the total WSF, accounting for 97.26% and 96.7%, respectively, followed by contributions from chemical agents and membranes. In the MED Project 1 and Project 2, steam consumption was the primary contributor (83.84% and 84.23%, respectively), with electricity being the secondary factor (15.61% and 14.04%). In summary, electricity consumption is the dominant factor influencing the WSF of RO seawater desalination, steam consumption is the dominant factor influencing the WSF of MED seawater desalination. Accordingly, reducing electricity and steam demand is the most effective strategy for minimizing the operational WSF of desalination plants. For RO plants, the key determinant of electricity consumption is the feed pressure prior to the RO membrane modules. Therefore, implementing ERDs to harness the hydraulic pressure of the concentrate stream, performing regular membrane cleaning, and optimizing membrane element configuration can be an efficient measure to cut down on electricity consumption. For MED plants, integrating with power plants to utilize waste steam provides a highly effective WSF reduction strategy.

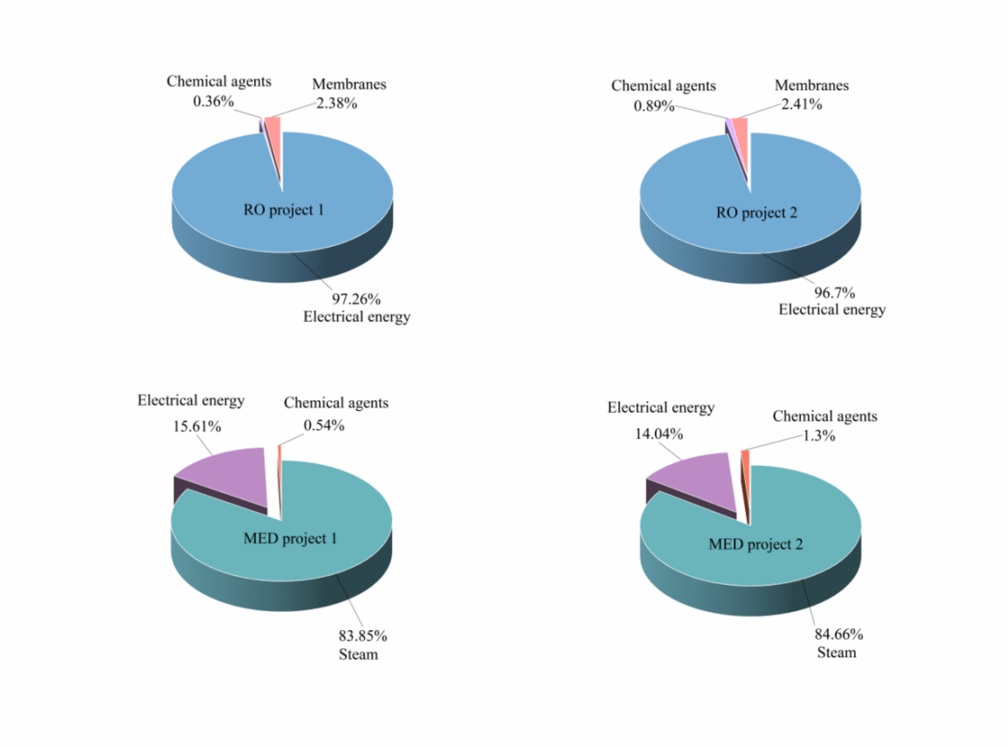

Figure 3. Composition analysis diagram of WSF for RO and MED seawater desalination plants.

As shown in Figure 4, the WSF of seawater desalination systems exhibits slight seasonal variations. The WSF of RO desalinated water marginally increases with decreasing temperature, with the highest value observed in winter, followed by spring and autumn, and the lowest in summer. The variation range is maximally 1.8×10^{-3} m^3 per m^3 of product water. This is because the decrease in temperature leads to a reduction in the permeability of the RO membrane, thereby increasing the energy consumption per unit water production.

Similarly, the variation of the WSF for the MED system with seasonal temperature is less obvious, showing a change of maximally 9×10^{-4} m^3 per m^3 of product water. The highest WSF for MED occurs during summer and winter. This is primarily due to the increased cooling water demand and associated electricity consumption at higher ambient temperatures during summer. In winter, the electricity consumption required for heating steam needs to be increased when the intake seawater temperature is lower.

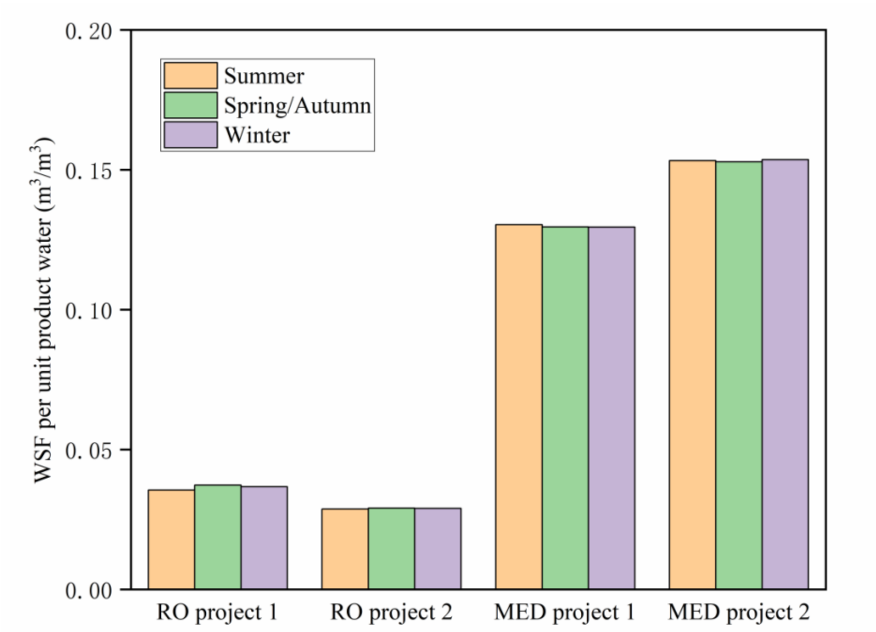

Figure 4. Impacts analysis diagram of seasons variations on the WSF of RO and MED seawater desalination plants.

4.4. Uncertainty and Variability Analysis: Regional Differences, Scale Effects, and Data Sensitivity

The case studies verify the practical applicability of the water footprint assessment method established in this paper. Nevertheless, it should be clarified that the calculation results are restricted by application scenarios, and regional discrepancies as well as fluctuations in operating conditions must be fully taken into account. Detailed analyses of several major sources of uncertainty and variability are presented in the following sections.

4.4.1. Regional Differences in Electricity Mix

As highlighted in the composition analysis (Section 4.4), electricity consumption is the dominant contributor to the WSF of RO desalination, accounting for over 96% of the total WSF in the studied RO plants. Consequently, the indirect WSF is highly sensitive to the regional electricity generation mix. The case studies presented herein are based on the operational data of plants located in China, where the national grid relies heavily on coal-fired power generation. If the same RO plant were instead located in a region with a low-carbon electricity mix, such as Iceland (geothermal/hydro), Norway (hydro), or France (nuclear), the indirect WSF associated with electricity consumption would be substantially lower. Conversely, in regions heavily dependent on lignite or heavy fuel oil, the WSF could be even higher than the values reported in this study.

4.4.2. Plant Scale and Process Parameters

The scale of the desalination plant also introduces considerable variability. The case study plants have a production capacity of 100,000 m^3/d, which represents a large-scale facility. Smaller plants (e.g., 1000–10,000 m^3/d) typically exhibit higher specific energy consumption (SEC) and higher chemical dosage rates per unit of product water due to the lack of economies of scale, leading to a higher WSF per m^3 of product water. Additionally, key process parameters such as feed water salinity (which varies with geographical location, e.g., Red Sea vs. Baltic Sea), water recovery rate, and membrane type directly influence the energy demand and brine volume, thereby affecting both WSF and WDF. For instance, a plant operating in the high-salinity Red Sea would require higher feed pressure in RO systems compared to a plant in the low-salinity Baltic Sea, resulting in a higher WSF for the same production capacity.

4.4.3. Data Uncertainty in Chemical and Material Characterization Factors

Beyond energy, the characterization factors for chemical agents and membrane materials also introduce uncertainty. In this study, the WF coefficients for UF and RO membranes were obtained through manufacturer investigations and process modeling. However, different membrane manufacturers employ distinct production processes (e.g., phase inversion vs. interfacial polymerization routes, solvent recovery efficiencies), resulting in widely varying water consumption footprints. Similarly, the production of coagulants (e.g., $FeCl_3$) and antiscalants involves different raw material sources and synthesis pathways across regions. Since the Ecoinvent and CLCD databases used in this study reflect average technology levels, the actual water footprint of these chemicals in a specific regional context may deviate from the values applied herein.

5. Conclusions

To evaluate the environmental impacts of the seawater desalination industry and quantify its freshwater usage and concentrated brine discharge, this study proposes a WF assessment methodology tailored for desalination processes. The methodology was demonstrated through comprehensive case studies evaluating both RO and MED plants. The results demonstrate that seawater desalination can generate a net increase in freshwater supply. They also indicate that RO technology exhibits substantially lower feed water consumption of freshwater produced compared to MED plants operating without access to waste heat steam sources.

Compositional analysis of the WSF identified energy consumption as the dominant contributing factor for both technologies. Consequently, reducing electricity demand is key to minimizing the operational WSF of desalination plants. RO plants can focus on measures such as lowering intake pressure via ERDs, optimizing feed water pH, regular membrane cleaning, and improved membrane configuration. MED plants can significantly reduce their WSF by coupling with a power station of co-generation to utilize waste steam. Analysis of operational data in the studied plants showed that seasonal variations exerted no significant influence on the WSF.

For the WDF assessment, this study employed two distinct methodologies based on a review of current approaches. In the present case studies of RO Project 1, discharge concentrated brine directly into marine environments. Therefore, the critical dilution volume method was applied, which resulted in a calculated WDF of 0 m^3 water equivalents. It is crucial to note that this result does not imply zero environmental impact, and it reflects the specific assumptions of this methodology regarding marine discharge. If precise quantification of specific environmental impacts associated with brine discharge, such as acidification or eutrophication potential, is required, the equivalency factor method is recommended to continue calculating the WDF.

Nevertheless, the current WDF framework, whether based on the critical dilution volume or equivalency factor approaches, does not account for several environmentally relevant stressors that are particularly pertinent to desalination brine discharge, such as the cumulative salinity increase in semi-enclosed water bodies, the thermal plume effects from thermal desalination plants, and the physical disturbance of benthic communities caused by discharge outfalls. The absence of these impact categories from the current WDF metrics represents a significant gap, as these stressors can have profound and lasting effects on local marine ecosystems, especially in ecologically sensitive or low-flushing coastal zones. Therefore, future research should prioritize the development of integrated impact assessment methods that combine chemical water quality indicators with physical and ecological parameters. Specifically, efforts are needed to: (1) establish salinity and temperature characterization factors for marine receiving waters; (2) develop spatial-explicit models that account for the hydrodynamic conditions and assimilation capacity of the receiving environment; and (3) incorporate ecotoxicological endpoints that reflect the combined effects of multiple stressors. Such advancements would enable a more holistic evaluation of brine discharge impacts and provide a more scientifically robust basis for environmental management and regulatory decision-making in the desalination industry.

The WF assessment framework developed herein enables systematic quantification of the resource and environmental impacts associated with desalination, facilitates the identification of key influencing factors, and provides a methodological foundation for advancing sustainable management practices within the desalination industry.

Author Contributions

J.X.: Conceptualization, Investigation, Visualization, Formal analysis, Writing—original draft. J.S.: Conceptualization, Visualization, Writing—review & editing. K.W.: Formal analysis, Writing—review & editing. G.X.: Writing—review & editing. K.H.: Writing—review & editing. H.S.: Conceptualization, Funding acquisition, Project administration, Writing—review & editing. C.X.: Conceptualization, Funding acquisition, Supervision, Resources, Project administration, Writing—review & editing. All authors have read and agreed to the published version of the manuscript.

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Data will be made available on request.

Conflicts of Interest

The authors declare no conflict of interest.

Use of AI and AI-Assisted Technologies

During the preparation of this work, the manuscript language has been polished by the authors with the assistance of AI. After using this service, the authors reviewed and edited the content as needed and takes full responsibility for the content of the published article.

References

1. Zhang, X.M.; He, J.J.; Shan, T.; et al. A comprehensive review on the characteristics and kinetics of freshwater separation by hydrate-based method: Current progress, challenges and perspectives. *Desalination* **2024**, 576, 117279. https://doi.org/10.1016/j.desal.2023.117279.
2. Gopu, A.P.; Sureshkumar, C.; Samundeeswari, R.; et al. Strategic siting of desalination plants and design of an optimal water distribution network. *Desalination Water Treat.* **2026**, *325*, 101620. https://doi.org/10.1016/j.dwt.2025.101620.
3. Ali, A.S.; Bounahmidi, T. Photovoltaic thermal desalination review. *Desalination Water Treat.* **2026**, *325*, 101669. https://doi.org/10.1016/j.dwt.2026.101669.
4. Ma, M.L.; Feng, T.; Li, Z.Y.; et al. Synergistic optimization of reverse osmosis and disinfection processes in seawater desalination systems: Operational experience from a tropical desalination plant. *Desalination Water Treat.* **2025**, *323*, 101392. https://doi.org/10.1016/j.dwt.2025.101392.
5. Tayeh, Y.A. A comprehensive review of reverse osmosis desalination: Technology, water sources, membrane processes, fouling, and cleaning. *Desalination Water Treat.* **2024**, *320*, 100882. https://doi.org/10.1016/j.dwt.2024.100882.
6. Ghazi, Z.M.; Rizvi, S.W.F.; Shahid, W.M.; et al. An overview of water desalination systems integrated with renewable energy sources. *Desalination* **2022**, *542*, 116063. https://doi.org/10.1016/j.desal.2022.116063.
7. Aboulkacem, C.; Sigue, S.; Lachheb, A.; et al. Seawater desalination and water reuse using FO-RO hybrid system: Short review. *Desalination Water Treat.* **2025**, *324*, 101487. https://doi.org/10.1016/j.dwt.2025.101487.
8. Ihm, S.; Al-Waznani, E.S.B.; Li, S. The performance of nanofiltration membranes in seawater and desalination brine applications: Saudi Water Authority experience. *Desalination Water Treat.* **2024**, *320*, 100793. https://doi.org/10.1016/j.dwt.2024.100793.
9. Katibi, K.K.; Azimov, A.M.; Nawi, N.M.; et al. An insight into real-time monitoring and predictive modelling of ultrafiltration pretreatment for flow rate and fouling mitigation in seawater desalination. *Desalination Water Treat.* **2025**, *323*, 101386. https://doi.org/10.1016/j.dwt.2025.101386.
10. Asrar, N.; Mubashir, M.; AlSahary, A.B.; et al. Non-Metallic Materials for Sustainable Desalination Systems: Performance, Durability, and Industrial Validation. *Desalination Water Treat.* **2026**, *326*, 101765. https://doi.org/10.1016/j.dwt.2026.101765.
11. Ayaz, M.; Namazi, M.A.; Din, M.; et al. Sustainable seawater desalination: Current status, environmental implications and future expectations. *Desalination* **2022**, *540*, 116022. https://doi.org/10.1016/j.desal.2022.116022.
12. Zolghadr-Asli, B.; McIntyre, N.; Djordjevic, S.; et al. The sustainability of desalination as a remedy to the water crisis in the agriculture sector: An analysis from the climate-water-energy-food nexus perspective. *Agric. Water Manag.* **2023**, *286*, 108407. https://doi.org/10.1016/j.agwat.2023.108407.
13. He, Z.F.; Lai, C.Y.; Huang, H.Y.; et al. Empowering remote communities: A portable, self-powered integrated desalination system. *Desalination* **2025**, *614*, 119179. https://doi.org/10.1016/j.desal.2025.119179.
14. International Desalination and Reuse Association (IDRA). *Desalination & Reuse Handbook 2024–2025*; International Desalination and Reuse Association (IDRA): Danvers, MA, USA, 2025.
15. Prakash, N.; Chaudhuri, A.; Das, S.P. Multi-stage roto-dynamic RO system for desalination. *Desalination Water Treat.* **2025**, *324*, 101499. https://doi.org/10.1016/j.dwt.2025.101499.
16. Malatji, M.; Mashifana, T.P.; Siwela, J.; et al. An update on the use of forward osmosis desalination technology for water treatment: Current trends and future research outlooks. *Desalination Water Treat.* **2026**, *326*, 101744. https://doi.org/10.1016/j.dwt.2026.101744.
17. Bibi, W.; Asif, M.; Iqbal, F.; et al. Hybrid vacuum membrane distillation-multi effect distillation (VMD-MED) system for reducing specific energy consumption in desalination. *Desalination Water Treat.* **2024**, *317*, 100064. https://doi.org/10.1016/j.dwt.2024.100064.
18. Adityawarman, D.; Lugito, G.; Kawi, S.; et al. Advancements and future trends in nanostructured membrane technologies for seawater desalination. *Desalination* **2025**, *597*, 118390. https://doi.org/10.1016/j.desal.2025.118390.
19. Tayyeban, E.; Deymi-Dashtebayaz, M.; Dadpour, D. Multi objective optimization of MSF and MSF-TVC desalination systems with using the surplus low-pressure steam (an energy, exergy and economic analysis). *Comput. Chem. Eng.* **2022**, *160*, 107708. https://doi.org/10.1016/j.compchemeng.2022.107708.
20. Tu, W.H.; Zhao, Y.; Chan, W.P.; et al. Reclaimed seawater discharge—Desalination brine treatment and resource recovery system. *Water Res.* **2024**, *251*, 121096. https://doi.org/10.1016/j.watres.2024.121096.
21. Rouane, D.B.; Errouhi, A.A.; Mghaiouini, R. Seawater desalination: A review of technologies, environmental impacts, and future perspectives. *Desalination Water Treat.* **2025**, *324*, 101578. https://doi.org/10.1016/j.dwt.2025.101578.
22. Abelsson, C.; Fuglesang, A. Subsea desalination—significant energy savings and greatly reduced environmental impact. *Desalination Water Treat.* **2023**, *309*, 43–48. https://doi.org/10.5004/dwt.2023.29877.

23. Alluhaidan, A.S.; Prabu, P.; Basheer, S.; et al. Integrated hybrid membrane system for enhanced water treatment and desalination for environmental preservation. *Desalination Water Treat.* **2024**, *320*, 100722. https://doi.org/10.1016/j.dwt.2024.100722.
24. Xevgenos, D.; Marcou, M.; Louca, V.; et al. Aspects of environmental impacts of seawater desalination: Cyprus as a case study. *Desalination Water Treat.* **2021**, *211*, 15–30. https://doi.org/10.5004/dwt.2021.26916.
25. Al-Saidi, M.; Saadaoui, I.; Ben-Hamadou, R. Governing desalination, managing the brine: A review and systematization of regulatory and socio-technical issues. *Water Resour. Ind.* **2023**, *30*, 100225. https://doi.org/10.1016/j.wri.2023.100225.
26. Almasoudi, S.; Jamoussi, B. Desalination technologies and their environmental impacts: A review. *Sustain. Chem. One World* **2024**, *1*, 100002. https://doi.org/10.1016/j.scow.2024.100002.
27. Nasrollahi, M.; Motevali, A.; Banakar, A.; et al. Comparison of environmental impact on various desalination technologies. *Desalination* **2023**, *547*, 116253. https://doi.org/10.1016/j.desal.2023.116253.
28. Alhaj, M.; Tahir, F.; Al-Ghamdi, S.G. Life-cycle environmental assessment of solar-driven Multi-Effect Desalination (MED) plant. *Desalination* **2022**, *524*, 115451. https://doi.org/10.1016/j.desal.2022.115451.
29. Bai, X.; Ren, X.; Khanna, N.Z.; et al. Comprehensive water footprint assessment of the dairy industry chain based on ISO 14046: A case study in China. *Resour. Conserv. Recycl.* **2018**, *132*, 369–375. https://doi.org/10.1016/j.resconrec.2017.07.021.
30. Corominas, L.; Byrne, D.M.; Guest, J.S.; et al. The application of life cycle assessment (LCA) to wastewater treatment: A best practice guide and critical review. *Water Res.* **2020**, *184*, 116058. https://doi.org/10.1016/j.watres.2020.116058.
31. Lee, K.; Jepson, W. Environmental impact of desalination: A systematic review of Life Cycle Assessment. *Desalination* **2021**, *509*, 115066. https://doi.org/10.1016/j.desal.2021.115066.
32. Aziz, N.; Hanafiah, M.M. Application of life cycle assessment for desalination: Progress, challenges and future directions. *Environ. Pollut.* **2021**, *268*, 115948. https://doi.org/10.1016/j.envpol.2020.115948.
33. Wang, Y.; Morosuk, T.; Cao, W. Carbon Footprint of Seawater Desalination Technologies: A Review. *J. Energy Resour. Technol.* **2024**, *146*, 1–48. https://doi.org/10.1115/1.4063845.
34. Liu, S.; Song, D.; Han, M.; et al. Multi-dimensional low-carbon assessments of wind-driven seawater desalination system via hybrid life cycle modelling. *Environ. Impact Assess. Rev.* **2024**, *106*, 107475. https://doi.org/10.1016/j.eiar.2024.107475.
35. Cabrera, P.; Carta, J.A.; Matos, C.; et al. Reduced desalination CF on islands with weak electricity grids: The case of Gran Canaria. *Appl. Energy* **2024**, *358*, 122564. https://doi.org/10.1016/j.apenergy.2024.122564.
36. Pandit, R.R.; Deshmukh, S.S.; Khatod, K.J.; et al. Efficiency, cost, and carbon footprint considerations in analysing hybrid solar thermal-photovoltaic desalination systems. *Desalination Water Treat.* **2026**, *325*, 101606. https://doi.org/10.1016/j.dwt.2025.101606.
37. Leon, F.A.; Martín, A.R.; Alvarado, Y.F.; et al. Proposal to determine the carbon and ecological footprint of seawater reverse osmosis desalination in the Canary Islands plants considering the energy mix. *Desalination Water Treat.* **2021**, *230*, 9–16. https://doi.org/10.5004/dwt.2021.27353.
38. Kabiri, S.; Meratizaman, M. Energy, exergy, exergoeconomic, and environmental (4E) and carbon footprint analysis of coupling the various energy recovery devices with seawater and brackish water reverse osmosis desalination. *Desalination Water Treat.* **2021**, *210*, 103–122. https://doi.org/10.5004/dwt.2021.26548.
39. Hoekstra, A.Y.; Martinez-Aldaya, M.; Mekonnen, M. *Water Footprint Manual: State of the Art 2009*; Water Footprint Network: Enschede, The Netherlands, 2009.
40. Matuštík, J.; Kočí, V. What is a footprint? A conceptual analysis of environmental footprint indicators. *J. Clean. Prod.* **2021**, *285*, 124833. https://doi.org/10.1016/j.jclepro.2020.124833.
41. Yerli, C. Reducing grey water footprint input and Blue-Green water footprint output of silage maize in biochar applications under irrigation with different water sources. *Desalination Water Treat.* **2025**, *321*, 101051. https://doi.org/10.1016/j.dwt.2025.101051.
42. Zhang, W.L. Calculation model and management system of WRCC based on grey water footprint calculation. *Desalination Water Treat.* **2023**, *313*, 266–278. https://doi.org/10.5004/dwt.2023.30026.
43. Wang, A.L.; Wang, S.S.; Liu, T.F.; et al. Spatial-temporal evolution analysis and deep learning inversion of water-carbon-three-dimensional ecological footprint of urban agglomeration in the middle reaches of the Yangtze River. *Desalination Water Treat.* **2023**, *290*, 193–200. https://doi.org/10.5004/dwt.2023.29424.
44. Ahmad, M.; Al-Sa’ed, R.; Steen, P. Enhancing the governance of industrial wastewater using cleaner production and water footprint principles: A case study of two dairy companies in Palestine. *Desalination Water Treat.* **2022**, *275*, 339–349. https://doi.org/10.5004/dwt.2022.29136.
45. Su, F.; Shang, H.Y. Social water cycle and sustainable consumption in the perspective of water footprint—taken the low water consumption patterns of Zhangye city as case. *Desalination Water Treat.* **2018**, *122*, 170–175. https://doi.org/10.5004/dwt.2018.22713.

46. Goudouva, G.T.; Zorpas, A.A. Water footprint determination by quarry operation in island regions. *Desalination Water Treat.* **2017**, *86*, 271–276. https://doi.org/10.5004/dwt.2017.20814.
47. Mohammed, S.; Darwish, M. Water footprint and virtual water trade in Qatar. *Desalination Water Treat.* **2017**, *66*, 117–132. https://doi.org/10.5004/dwt.2017.20221.
48. Charchousi, D.; Tsoukala, V.K.; Papadopoulou, M.P. How evapotranspiration process may affect the estimation of water footprint indicator in agriculture. *Desalination Water Treat.* **2015**, *53*, 3234–3243. https://doi.org/10.1080/19443994.2014.934118.
49. Papadopoulou, M.P.; Charchousi, D.; Tsoukala, V.K.; et al. Water footprint assessment considering climate change effects on future agricultural production in Mediterranean region. *Desalination Water Treat.* **2016**, *57*, 2232–2242. https://doi.org/10.1080/19443994.2015.1049408.
50. Hadji, A.; Merzouk, N.K.; Merzouk, M.; et al. Study of a modified multiple effect distillation (MED) brackish water system for a minimal water footprint. *Desalination Water Treat.* **2021**, *213*, 44–52. https://doi.org/10.5004/dwt.2021.26715.
51. Ding, Y.X.; Sun, D.G.; Tao, S.Y.; et al. Comprehensive water footprint assessment of RO and MED desalination plants in China based on ISO method. *J. Environ. Chem. Eng.* **2025**, *13*, 120399. https://doi.org/10.1016/j.jece.2025.120399.
52. Zhang, X.L.; Jiang, J.J.; Yuan, F.; et al. Estimation of water footprint in seawater desalination with reverse osmosis process. *Environ. Res.* **2022**, *204*, 112374. https://doi.org/10.1016/j.envres.2021.112374.
53. Meneses, M.; Pasqualino, J.C.; Céspedes-Sánchez, R.; et al. Alternatives for Reducing the Environmental Impact of the Main Residue from a Desalination Plant. *J. Ind. Ecol.* **2010**, *14*, 512–527. https://doi.org/10.1111/j.1530-9290.2010.00225.x.
54. Zhou, J.; Chang, W.C.; Fane, A.G. Environmental life cycle assessment of reverse osmosis desalination: The influence of different life cycle impact assessment methods on the characterization results. *Desalination* **2011**, *283*, 227–236. https://doi.org/10.1016/j.desal.2011.04.066.
55. Muñoz, I.; Fernández-Alba, A.R.G. Reducing the environmental impacts of reverse osmosis desalination by using brackish groundwater resources. *Water Res.* **2008**, *42*, 801–811. https://doi.org/10.1016/j.watres.2007.08.021.
56. Bai, X.; Ren, X.; Khanna, N.Z.; et al. A comparative study of a full value-chain water footprint assessment using two international standards at a large-scale hog farm in China. *J. Clean. Prod.* **2018**, *176*, 557–565. https://doi.org/10.1016/j.jclepro.2017.12.085.
57. Ene, S.A.; Teodosiu, C.; Robu, B.; et al. Water footprint assessment in the winemaking industry: A case study for a Romanian medium size production plant. *J. Clean. Prod.* **2013**, *43*, 122–135. https://doi.org/10.1016/j.jclepro.2012.11.018.
58. *ISO 14046:2014*; Environmental Management—Water Footprint—Principles, Requirements and Guidelines. ISO: Geneva, Switzerland, 2014.
59. Li, X.; Zhu, L.; Dong, Y.; et al. Water footprint assessment of wool products with a low-water footprint baseline, Sustain. *Prod. Consum.* **2022**, *34*, 310–317. https://doi.org/10.1016/j.spc.2022.06.013.
60. *ISO 14040:2006*; Environmental Management—Life Cycle Assessment—Principles and Framework. ISO: Geneva, Switzerland, 2006.
61. *ISO 14044:2006*; Environmental Management—Life Cycle Assessment—Requirements and Guidelines. ISO: Geneva, Switzerland, 2006.
62. *ISO/TR 14073:2017*; Environmental Management—Water Footprint—Illustrative Examples on how to Apply ISO 14046. ISO: Geneva, Switzerland, 2017.
63. Mikosch, N.; Berger, M.; Finkbeiner, F. Addressing water quality in water footprinting: Current status, methods and limitations. *Int. J. Life Cycle Assess.* **2021**, *26*, 157–174. https://doi.org/10.1007/s11367-020-01812-9.
64. Ridoutt, B.G.; Pfister, S. A new water footprint calculation method integrating consumptive and degradative water use into a single stand-alone weighted indicator. *Int. J. Life Cycle Assess.* **2013**, *18*, 204–207. https://doi.org/10.1007/s11367-012-0533-8.
65. Aldaya, M.M.C.; Hoekstra, A.Y.; Mekonnen, M.M. *The Water Footprint Assessment Manual: Setting the Global Standard*; World Bank Group: Washington, D.C., DC, USA, 2011.
66. Mekonnen, M.M.; Hoekstra, A.Y. The green, blue and grey water footprint of crops and derived crop products. *Hydrol. Earth Syst. Sci.* **2011**, *15*, 1577–1600. https://doi.org/10.5194/hess-15-1577-2011.
67. Schomberg, A.C.; Bringezu, S.; Flörke, M. Extended life cycle assessment reveals the spatially-explicit water scarcity footprint of a lithium-ion battery storage. *Commun. Earth Environ.* **2021**, *2*, 11. https://doi.org/10.1038/s43247-021-00012-9.
68. Dekamin, M.; Barmaki, M.; Kanooni, A. Selecting the best environmental friendly oilseed crop by using Life Cycle Assessment, water footprint and analytic hierarchy process methods. *J. Clean. Prod.* **2018**, *198*, 1239–1250. https://doi.org/10.1016/j.jclepro.2018.07.089.
69. Huijbregts, M.A.J.; Steinmann, Z.J.N.; Elshout, P.M.F.; et al. ReCiPe2016: A harmonised life cycle impact assessment method at midpoint and endpoint level. *Int. J. Life Cycle Assess.* **2016**, *22*, 138–147. https://doi.org/10.1007/s11367-016-0164-y.

70. Hauschild, M.Z.; Stig, R.K.R.; Olsen, I. *Life Cycle Assessment: Theory and Practice*; Springer: Cham, Switzerland, 2017.
71. Deepa, R.; Anandhi, A.; Alhashim, R. Volumetric and Impact-Oriented Water Footprint of Agricultural Crops: A Review. *Ecol. Indic.* **2021**, *130*, 108093. https://doi.org/10.1016/j.ecolind.2021.108093.
72. Pradinaud, C.; Núñez, M.; Roux, P.; et al. The issue of considering water quality in life cycle assessment of water use. *Int. J. Life Cycle Assess.* **2019**, *24*, 590–603. https://doi.org/10.1007/s11367-018-1473-5.
73. Zhu, J.; Yang, Y.; Li, Y.; et al. Water footprint calculation and assessment of viscose textile. *Indust. Textila* **2020**, *71*, 33–40. https://doi.org/10.35530/IT.0034-3665.2020.004.
74. Alrashidi, A.; Aleisa, E.; Alshayji, K. Life cycle assessment of hybrid electrodialysis and reverse osmosis seawater desalination systems. *Desalination* **2024**, *578*, 117448. https://doi.org/10.1016/j.desal.2024.117448.