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Life Cycle Impact Assessment of Pesticides Removal from Water Using Nanofiltration and Reverse Osmosis

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Abstract

Given the environmental burden associated with pesticides, it is necessary to quantify their potential impacts and investigate impact mitigation alongside their removal. Most existing life cycle impact assessment (LCIA) studies lack direct experimental data on residual concentrations, limiting their ability to realistically assess the toxicity-related benefits of pesticide mitigation. This study addresses that gap by linking experimentally determined pesticide removal with LCIA. Three LCIA methods (Environmental Footprint (EF), TRACI, and ReCiPe) were used to evaluate pesticide impacts, as they reflect different global regulations and conditions, resulting in different potential impact estimates. Removal of atrazine, imidacloprid, metolachlor, and tebuconazole from tap water and groundwater was carried out using nanofiltration (NF) and reverse osmosis (RO) membranes. The results showed 81–99% pesticide removal with RO membranes and 27–96% removal with NF membranes, depending on the pesticide. Freshwater ecotoxicity and human toxicity (cancer and non-cancer effects) were quantified for untreated matrices and compared to results obtained with NF and RO treatment data. The findings indicate that NF and RO treatment can reduce calculated pesticide toxicity impacts across the evaluated categories.

Keywords: life cycle impact assessment; pesticides; removal; nanofiltration; reverse osmosis

1. Introduction

The environmental impact of pesticides has been extensively evaluated using life cycle assessment (LCA), with particular attention to their contribution to ecotoxicity and human health risks. Within this framework, life cycle impact assessment (LCIA) quantifies the potential impacts of chemical emissions based on their fate, exposure pathways, and toxicological effects [1–3]. Environmental Footprint (EF), TRACI, and ReCiPe are widely used methods for evaluating freshwater ecotoxicity and human toxicity, as they incorporate substance-specific characterization factors derived from physico-chemical properties and toxicological data [4–8].

Previous LCIA studies on pesticides have primarily estimated their environmental burden using emission inventories from agricultural applications, typically relying on generic or literature-derived concentration data [9–11]. These approaches have significantly advanced the understanding of pesticide-related impacts across environmental compartments. However, they do not account for site-specific mitigation processes or experimentally measured residual concentrations after water treatment. As a result, the actual reduction in ecotoxicity and human health risk achieved through advanced treatment technologies remains insufficiently quantified.

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In parallel, pressure-driven membrane processes, particularly nanofiltration (NF) and reverse osmosis (RO), have been extensively investigated for pesticide removal from various water matrices. NF and RO membrane processes are based on the same principle, i.e., both use hydraulic pressure to force the solvent through a semipermeable membrane while retaining solutes, but they differ primarily in pore size and in the size of the retained solutes. Driven by the applied pressure, the solvent passes through the membrane, while solutes are rejected through different separation mechanisms, including size exclusion, charge exclusion, and physico-chemical interactions among solute, solvent, and membrane. The main difference between RO and NF membranes is pore size. RO membranes possess small pores (≈0.25 nm), whereas NF membranes have pores of 0.5–2 nm. This difference in pore size is reflected in the removal mechanisms of dissolved solutes. Because of their smaller pores, RO membranes reject most ions (>98%) and allow only water to permeate, indicating that the main removal mechanism is the sieve effect (based on the difference between solute size and membrane pore size). Due to their larger pore size compared with RO membranes, NF membranes have greater water permeability and higher selectivity toward divalent/polyvalent ions while allowing the passage of monovalent ions and small molecules. Therefore, the main removal mechanisms are the sieve effect and charge exclusion [12–14]. Numerous studies have demonstrated high rejection efficiencies, typically exceeding 80–95%, depending on membrane characteristics, pesticide properties, and operating conditions [12,13]. Experimental investigations have covered a wide range of matrices relevant to groundwater, tap water, and secondary-treated wastewater, confirming the effectiveness of NF and RO for micropollutant removal [15–20]. However, these studies primarily evaluate treatment performance based on concentration reduction and membrane selectivity, without assessing whether the observed removal leads to meaningful reductions in environmental impact.

Although LCA has been applied to water treatment systems, most studies rely on simplified or literature-based inventory data rather than experimentally determined pollutant concentrations [4,21–23]. Furthermore, membrane-based processes remain a comparatively limited subset of the water treatment LCA literature. To the best of our knowledge, integrating experimentally measured residual pesticide concentrations after membrane treatment into LCIA modeling remains largely unexplored. More specifically, the reviewed literature mainly addresses either agricultural emission inventories or treatment performance expressed as concentration rejection. At the same time, direct integration of experimentally measured post-treatment pesticide concentrations into LCIA-based freshwater ecotoxicity and human toxicity characterization appears to be limited. Increasingly stringent regulatory requirements, including Directive 2006/118/EC and Directive (EU) 2020/2184, set maximum allowable concentrations of 0.10 µg/L for individual pesticides and 0.50 µg/L for total pesticides in groundwater and water intended for human consumption, respectively. More recently, Directive (EU) 2024/3019 introduced requirements for quaternary treatment targeting at least 80% removal of relevant micropollutants. These regulatory drivers further emphasize the need to evaluate not only removal efficiency but also the associated reduction in environmental and human health impacts [24–26].

This study addresses the identified research gap by integrating experimentally determined pesticide rejection efficiencies into an LCIA-based assessment. The removal of four selected pesticides (atrazine, imidacloprid, metolachlor, and tebuconazole) was evaluated using NF and RO membranes in two water matrices: tap water and groundwater. By applying experimentally determined membrane rejection efficiencies to environmental pesticide concentrations derived from the literature, this study estimates potential reductions in freshwater ecotoxicity and human toxicity impacts. The assessment focuses specifically on

pesticide-related toxicity impacts associated with membrane treatment rather than on the overall environmental performance of NF and RO processes.

2. Materials and Methods

2.1. Life Cycle Impact Assessment Methods

LCIA modeling was carried out using OpenLCA software (version 2.6.1), incorporating three impact assessment methods: EF 3.1, TRACI 2.2, and ReCiPe 2016 Midpoint (H). Based on geography, different conditions, and emission baselines, these methods provide area-specific impact information. EF 3.1 is recommended by the European Commission and is based on the USEtox model, covering impact categories aligned with EU regulations; TRACI 2.2 was developed by the U.S. Environmental Protection Agency specifically for North American conditions; and ReCiPe 2016 Midpoint (H) was developed to provide more uniform global impact information [5–7]. These methods provided characterization factors (*CF*s) for quantifying contributions to the impact categories considered in this study: freshwater ecotoxicity, human cancer toxicity, and human non-cancer toxicity. In the human toxicity impact evaluation, the only exposure route considered was ingestion through drinking water.

The *CF*s are model-derived conversion factors that translate elementary flows into contributions to a given environmental impact category indicator, allowing comparisons across categories. They represent the environmental cause–effect chain of events, linking emissions and resource use to environmental impacts. Emission into the environment consists of the steps described as emission, fate, exposure, and resulting effects [27]. The CFs for toxicity impacts are calculated according to Equation (1):

$$CF = EfF \times XF \times FF \quad (1)$$

where *EfF* represents effect factor for freshwater ecotoxicity, corresponding to the potentially affected fraction (PAF) of species due to substance exposure, integrated over time (PAF m3 day/kg). For human toxicity, *EfF*s relate the quantity of a substance taken in by the population to the probability of cancer or non-cancer effects in humans, expressed in units of case/kg. The exposure factor (*XF*) relates the concentration of a substance to actual human intake per day, or to ecosystem exposure, while the fate factor *FF* describes the residence time of the substance in each compartment, expressed in days. Both *XF* and *FF* depend on the physico-chemical characteristics of the chemical. In the methods EF 3.1 and TRACI 2.2, freshwater ecotoxicity is expressed as comparative toxic units (CTUe) and human toxicity as comparative toxic units (CTUh) [28,29]. ReCiPe 2016 Midpoint (H) uses a different comparative unit, expressed as kg 1,4-DB-equation (1,4-dichlorobenzene equivalent) for both human toxicity and freshwater ecotoxicity categories. 1,4-DB is used as a reference chemical in the midpoint calculations: the calculated potential impact of the substance is divided by the potential impact of 1,4-DB emitted to fresh water for freshwater ecotoxicity, and to urban air for human toxicity [6]. The LCIA system boundary included only pesticide emissions represented by pesticide concentrations entering and leaving the treatment system. Impact calculations were performed by comparing untreated water with water after pesticide removal by NF and RO membranes. Operational inputs and outputs of the membrane processes, including electricity consumption, membrane production and replacement, cleaning chemicals, concentrate treatment, and infrastructure requirements, were not included in the LCIA model. Consequently, the results represent pesticide-related impact mitigation rather than the overall environmental performance of the treatment technologies.

The CFs are used to calculate impact scores (IS) by multiplying the emission of substance by their CF, as shown in Equation (2):

$$IS_x = CF_x \times m_x \quad (2)$$

where *IS* represents impact scores such as freshwater ecotoxicity or human toxicity, *CF* is the characterization of substance *x* released into a specific compartment, and *m* is the emission of substance *x* into a specific compartment [28,29]. Three LCIA methods were used in this study to compare differences in the potential environmental impacts of pesticides based on different regulations and regional conditions worldwide.

The assessment was limited to pesticide-related toxicity impacts, using a reference water volume of 42,500 m^3/day, corresponding to the assumed minimum daily water supply for 850,000 population equivalents. Environmental burdens associated with membrane production and replacement, electricity consumption, cleaning chemicals, and concentrate management were excluded from the system boundaries.

2.2. Removal of Pesticides by NF and RO Membranes

The experiments were conducted using a laboratory-scale RO/NF system, described in detail by Dolar et al. [30]. First, the membranes were washed with approximately 2 L of demineralized water without pressure to remove conserving agent. Afterward, the membranes were preconditioned at 12 bar until the flux stabilized. After one hour under operating conditions, the flux of demineralized water was measured. The operating pressure was 10 bar, and the membrane area was 11 cm^2. The commercially available membranes used included two NF membranes, HL and DK (Suez, Paris, France), and two RO membranes, ESPA4-LD (Hydranautics, Osaka, Japan) and XLE (Dow-Filmtec, Edina, MN, USA). The characteristics provided by the manufacturers are shown in Table 1. The experiments were run for 3 h in batch circulation mode, with both permeate and concentrate recirculated to the feed to keep the feed concentration constant. Samples of the feed and permeate (approximately 1.5 mL) were collected every hour, and removal efficiencies were calculated as mean values.

Table 1. Membrane characteristics given by manufacturers.

Type	Membrane Commercial Name	Maximum Operating Pressure (bar)	pH Range	R [a] (%)
NF	DK	41	3–9	98.0 ($MgSO_4$)
	HL	41	3–9	95.0 ($MgSO_4$)
RO	ESPA4-LD	41	2–10	99.0 (NaCl)
	XLE	41	2–11	97.0 (NaCl)

[a] The concentrations of $MgSO_4$ were 2000 mg/L, NaCl was 500 mg/L for ESPA4-LD, while 2000 mg/L NaCl was used for XLE.

The pesticides used in the experiments were atrazine and tebuconazole from Thermo Fisher Scientific (Waltham, MA, USA), and imidacloprid and metolachlor from Sigma Aldrich (Burlington, MA, USA), all of analytical standard purity grade. Their physico-chemical properties are summarized in Table 2. Stock solutions of the pesticides (1 g/L) were initially prepared in methanol because of their low water solubility. Binary solutions of the individual pesticides were then prepared in tap water and groundwater at concentrations of 100 µg/L. The physico-chemical characteristics of groundwater and tap water are shown in Table 3. Since tap water and groundwater, which are the main sources of drinking water in Zagreb, were used in the investigation, Table 3 presents the parameters regulated by the

Croatian Regulations on the Health Safety of Drinking Water under regular testing of the health safety of water. These solutions are considered aqueous, as the methanol content is less than 5%. Pesticide quantification was performed using an ACQUITY UPLC H-Class System (Waters, Milford, MA, USA) coupled with a Xevo TQ-S micro triple quadrupole mass spectrometer (Waters, Milford, MA, USA), operating in positive electrospray ionization source (ESI) mode. Chromatographic separation was carried out at 60 °C on an XBridge Premier BEH C18 column (100 × 2.1 mm i.d., 2.5 µm; Waters, USA).

Table 2. Physico-chemical characteristics of pesticides.

Pesticide	Formula	CAS no.	Molecular Weight (g/mol)	Water Solubility (mg/L)	log $K_{O/W}$
Atrazine	$C_8H_{14}ClN_5$	1912-24-9	215.69	34.7	2.16
Imidacloprid	$C_9H_{10}ClN_5O_2$	138261-41-3	255.66	7172.0	0.57
Metolachlor	$C_{15}H_{22}ClNO_2$	87392-12-9	283.80	50.9	2.90
Tebuconazole	$C_{15}H_{22}ClNO_2$	107534-96-3	307.82	24.8	3.70

Table 3. Physico-chemical characteristics of tap water and groundwater.

Physico-Chemical Characteristics	Tap Water	Groundwater
Turbidity (NTU)	0.66	0.74
pH	7.33	7.50
Electrical conductivity (µS/cm at 20 °C)	586	600
Consumption of $KMnO_4$ (mg/L)	4.10	0.82
Chlorides (mg/L)	18.5	9.8
Ammonia (mg/L (NH^{4+}))	0	0
Nitrite (mg/L (NO^{2-}))	0	0
Nitrate (mg/L (NO^{3-}))	13.9	32.1
Sulfate (mg/L (SO_4^{3-}))	20.1	35.7
Fluoride (mg/L (F^-))	0.06	0.43
Iron (µg/L)	1.9	25
Total hardness (mg/L $CaCO_3$)	327	275
Chlorites (µg/L ClO_2^-)	0	0
Chlorates (µg/L ClO_3^-)	0	0

3. Results

3.1. LCIA of Pesticides in Tap Water and Groundwater

Potential impacts of pesticides were assessed relative to the minimum daily water use defined by the United Nations (UN) and World Health Organization (WHO) as 0.05 m^3 per person per day, which is considered sufficient to meet consumption and hygiene needs [31,32]. Daily requirements per person include 0.002–0.004 m^3 of water for drinking, with the remainder allocated to cooking, hygiene, and sanitation. Zagreb County, where the water matrices were sampled, relies on a groundwater-based water supply system serving 850,000 population equivalents (PE), requiring at least 42,500 m^3 of water per day according to UN and WHO criteria [33]. Because water supplies cannot be dedicated

solely to drinking, LCIA modeling used this minimum per-person usage volume. Pesticide concentrations in the water were taken as the mean of literature-reported concentrations of atrazine, imidacloprid, metolachlor, and tebuconazole in groundwater and tap water during the period 2019–2026, as shown in Table 4.

Table 4. Mean concentrations of pesticides found in groundwater and tap water and their estimated daily intake for 850,000 PE.

Pesticide	Mean Concentration [a] (µg/L)	Daily Intake (kg/day)
Atrazine [33–37]	0.7510 ± 1.1802	0.0319 ± 0.0502
Imidacloprid [38–40]	0.2369 ± 0.3752	0.0101 ± 0.0159
Metolachlor [37,41–43]	0.9036 ± 0.8209	0.0384 ± 0.0349
Tebuconazole [37,44–47]	0.5128 ± 0.3130	0.0218 ± 0.0133

[a] $N = 5$ for atrazine and metolachlor, $N = 3$ for imidacloprid, and $N = 4$ tebuconazole.

Considerable variability was observed among pesticide concentrations reported in the literature. Consequently, the resulting LCIA values should be interpreted as indicative estimates of potential toxicity impacts rather than precise environmental predictions. The objective of the analysis was to compare relative changes in impacts resulting from membrane treatment rather than to provide statistically rigorous estimates of absolute environmental burdens. Initial pesticide impacts in feed water matrices, estimated through LCIA modeling using literature-derived mean concentrations, are shown in Figure 1. Both tap water and groundwater feeds have the same initial impacts, as both were calculated from the same literature-derived concentrations. When the obtained results are compared with a study reporting freshwater ecotoxicity impact scores for 12 pesticides ranging from 10^{-12} to 10^{7} CTUe [48], the modeled values for atrazine, imidacloprid, metolachlor, and tebuconazole fall toward the upper end of this range, indicating their higher freshwater ecotoxicity potential. As seen in the results, the impact is much more significant in the freshwater ecotoxicity category than in the human toxicity category. The observed differences in impact values among the evaluated pesticides for the same categories across different methods result from regulatory differences, regional differences, and the residence times considered in the methods. These differences affect *CF* calculations and therefore lead to different *IS* values for the same pesticide [5–7,49]. *CF*s are influenced by different calculations of *FF* and *XF* across methods and depend on the physico-chemical characteristics of the pesticides [29]. Among the evaluated pesticides, atrazine shows the highest potential impact values across all three methods in freshwater ecotoxicity, as well as in human toxicity non-cancer effects. Atrazine was the only pesticide for which human cancer effects could be quantified, as carcinogenicity-related *CF*s can be derived for only a limited number of pesticides due to the limited availability of dose–response data [10]. Consequently, atrazine was the only contributor to the calculated human cancer toxicity impacts. The absence of cancer toxicity *CF*s for imidacloprid, metolachlor, and tebuconazole may lead to underestimation of carcinogenic effects. Therefore, the calculated cancer toxicity impacts should be interpreted as representing only the quantifiable part of the carcinogenic burden associated with the pesticides investigated.

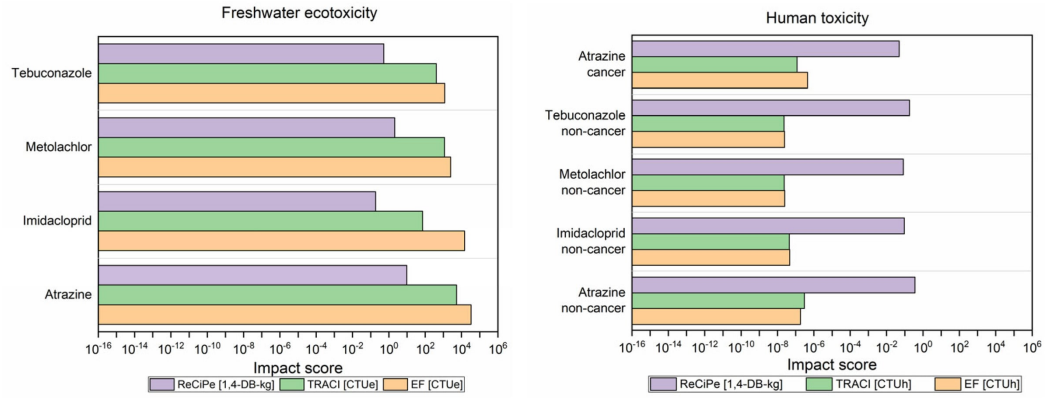


Figure 1. Impact scores of pesticides across different categories, with identical impacts in tap and groundwater feed water matrices.

When examining the values for freshwater ecotoxicity and human toxicity, the highest potential impacts are not necessarily associated with the highest concentrations. Metolachlor, although present at the highest concentration, does not show the highest potential impacts across all categories, whereas imidacloprid, with the lowest concentration, shows the second-highest potential impact in non-cancer human toxicity across all methods. These values are consistent with the WHO classification of pesticides by hazard, which classifies atrazine and metolachlor as slightly hazardous and imidacloprid and tebuconazole as moderately hazardous pesticides [50]. As noted above, impacts are proportional not only to the amount of a substance but also to its characterization factor, which can have a greater influence on potential impacts. This indicates that reducing only the pesticide concentration in water is not sufficient; it is also necessary to consider and mitigate the resulting toxicity impacts of the pesticide.

3.2. Removal of Pesticides from Different Water Matrices

Because groundwater is used for the tap water supply in Zagreb, NF and RO removal was evaluated in both tap water and groundwater matrices. Since sampling was performed hourly, the pesticide rejection factors (R) are presented as mean values in Figure 2. The differences in pesticide rejection between tap water and groundwater suggest that water matrix characteristics may influence membrane separation performance, although the individual contributions of specific water quality parameters were not evaluated in this study.

Based on the results obtained, RO membranes had rejection factors higher than 80% for all four pesticides in both tap water and groundwater, while the two NF membranes were less efficient, with rejection factors ranging from 27% to 96%, depending on the pesticide. Atrazine and imidacloprid, which have lower molecular weights (MW), showed lower rejection factors, from 27% to 64% in tap water and from 33% to 75% in groundwater. These results are correlated with the molecular weight cut-off (MWCO) of the NF membranes, whose larger pore sizes result in higher MWCO values (150–300 Da). RO membranes, with their denser structure, have an MWCO of 100 Da, leading to higher rejection factor values. In addition, when comparing the removal efficiencies of NF membranes, the obtained rejection factors are not all proportional to the MW of the pesticides. Tebuconazole, which has the highest MW, also has the highest log $K_{O/W}$ value, making it the most hydrophobic (log $K_{O/W} > 2$) of the four pesticides, with a tendency to adsorb onto the membranes. Lower rejection rates may result from adsorbed molecules diffusing through the membrane matrix over time, leading to higher concentrations in the permeate [51]. The slight improvement

in removal efficiency in groundwater can be attributed to differences in ion content between matrices and the presence of natural organic matter in groundwater. As shown in Table 3, groundwater had higher electrical conductivity and elevated ion concentrations compared with tap water. Variations in ionic strength can influence membrane surface charge and solute-membrane interactions, potentially affecting pesticide rejection [52]. It is also worth noting that, according to our study based on rejection factors and feed concentrations reported in the literature, all RO permeates would have pesticide concentrations lower than 0.10 µg/L, as required by the EU, while the total pesticide concentration would remain below the maximum allowable level of 0.50 µg/L. To provide a more comprehensive understanding of the potential environmental impacts of using NF and RO for pesticide removal, rather than relying on a best-case scenario, all results obtained were used in LCIA modeling. Table 5 shows that the reduction in pesticide intake is directly dependent on membrane rejection performance. RO membranes produced the lowest estimated pesticide intake values for all investigated compounds, while NF membranes showed greater variability depending on pesticide properties. These estimated intakes were subsequently used as the emission inventory for LCIA modeling.

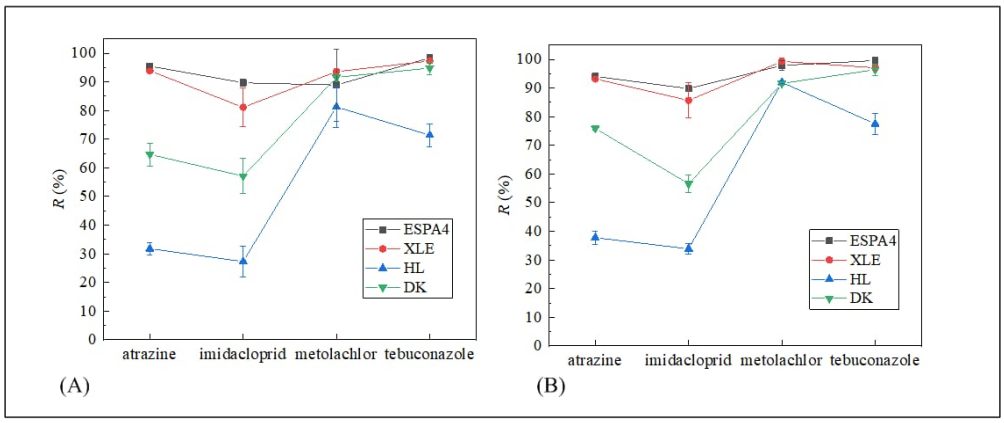


Figure 2. Removal of pesticides from tap water (**A**) and groundwater (**B**).

Table 5. Estimated daily pesticide intake after NF/RO treatment for 850,000 PE.

	Groundwater Permeates Daily Intake (kg/day)				**Tap Water Permeates Daily Intake (kg/day)**			
Pesticide	ESPA4	XLE	HL	DK	ESPA4	XLE	HL	DK
Atrazine	0.0019	0.0021	0.0199	0.0077	0.0014	0.0019	0.0218	0.0113
Imidaclo-prid	0.0010	0.0014	0.0067	0.0054	0.0010	0.0019	0.0073	0.0043
Meto-lachlor	0.0008	0.0003	0.0031	0.0032	0.0043	0.0025	0.0072	0.0032
Tebucona-zole	0.0001	0.0006	0.0049	0.0008	0.0003	0.0006	0.0062	0.0011

3.3. LCIA of NF and RO Treatment Pesticide Removal

The LCIA results provide insight into the environmental performance of NF and RO in pesticide removal, with the objective of evaluating the potential environmental impacts of removing pesticides from groundwater and tap water compared with a no-treatment scenario. Residual pesticide concentrations were calculated using our experimental removal data, assuming that NF and RO perform equally well at lower feed concentrations, as

reported in the literature [17,53,54]. The environmental impact contributions of NF and RO treatments for the selected categories are shown in Figures 3–5. The impact results correspond to a treatment capacity of 42,500 m^3/day of water intended for 850,000 PE.

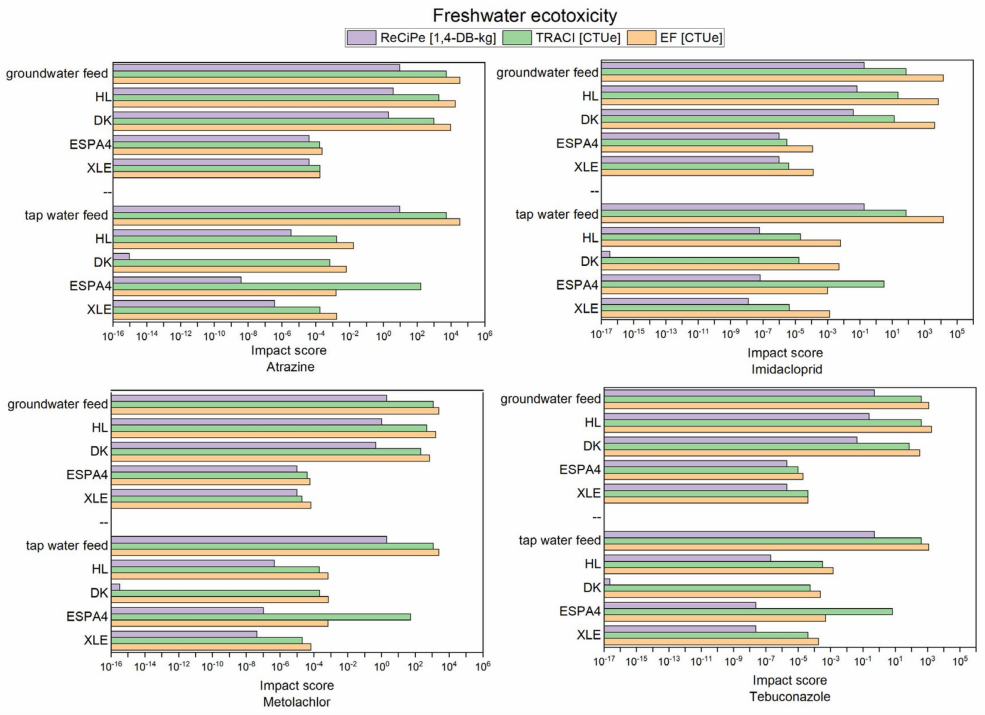


Figure 3. Impact scores for freshwater ecotoxicity of pesticides in permeates from different NF and RO membranes compared to the impact scores of the feed water matrices.

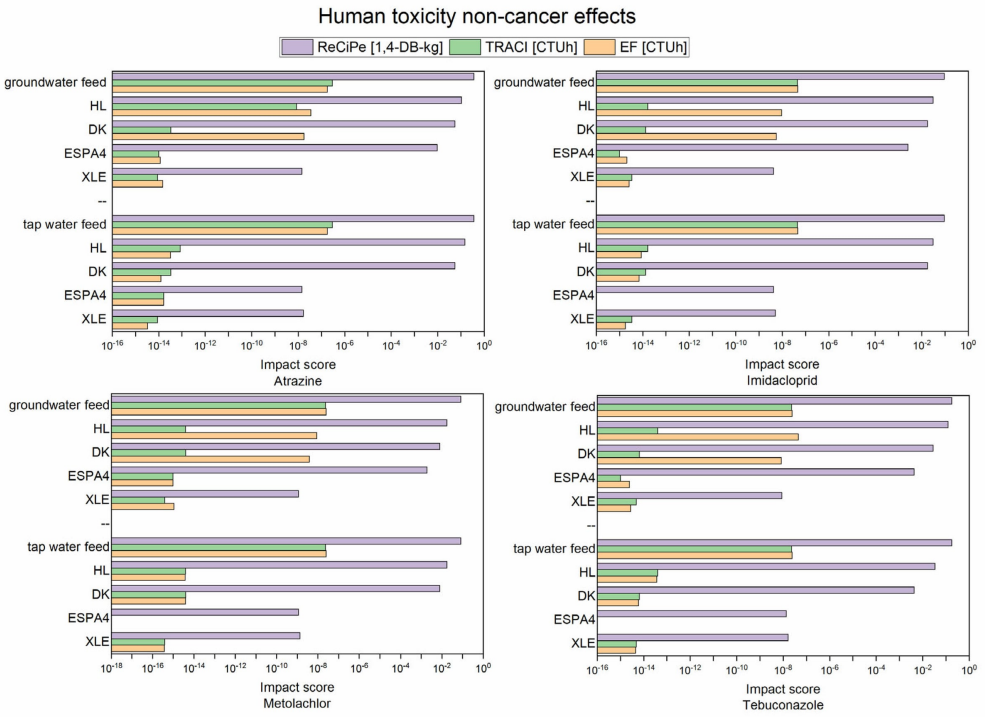


Figure 4. Impact scores for human toxicity non-cancer effects of pesticides in permeates from different NF and RO membranes compared to impact scores in the feed water matrices.

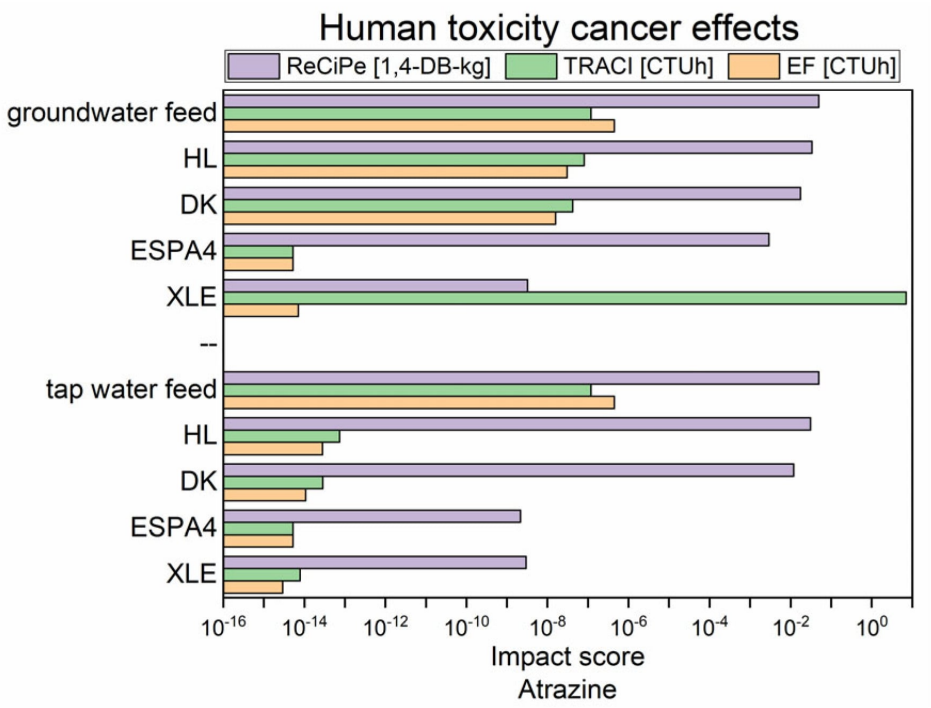


Figure 5. Impact scores for human toxicity cancer effects of atrazine in permeates from different NF and RO membranes compared to impact scores in the feed water matrices.

By implementing NF and RO treatment, the mass of pesticides used in *IS* calculations was reduced. Although the absolute impact scores differ among EF, TRACI, and ReCiPe because of method-specific characterization factors, all three methods indicate reductions in pesticide toxicity impacts following membrane treatment. These reductions result from the lower pesticide masses considered in the LCIA calculations, while the characterization factors remain unchanged for each pesticide within a given method. For imidacloprid and tebuconazole, despite their lower concentrations and removal efficiencies similar to those of atrazine and metolachlor, their impact mitigation is lower. When comparing these pesticides, their differing physico-chemical characteristics affect their *CF*s and, consequently, their *IS*. Considering their hazard classifications, this explains why imidacloprid and tebuconazole, despite their lower concentrations, exhibit lower *IS* mitigation. The estimated impact mitigation results depend on several factors, including toxicity, data quality, removal efficiency, and the methods used; therefore, the results should be interpreted with caution. Because only the impacts of pesticides were assessed in LCIA modeling, *IS* values between the two matrices cannot be compared in relation to matrix influence. The only difference between matrices is the higher removal efficiency for groundwater, explained in Section 3.2, which contributes to *IS* mitigation. For an individual pesticide, the relative reduction in the calculated impact score corresponds to its membrane rejection efficiency, as the characterization factor remains constant within each LCIA method. However, the contribution of each pesticide to the overall toxicity impact depends on its concentration and pesticide-specific characterization factor. Nevertheless, when comparing membrane efficiency, RO membranes showed more pronounced mitigation across all evaluated impact categories than NF membranes. Since RO membranes have higher removal efficiency than NF membranes, the impacts of the resulting RO permeates are consequently lower. Because there are few LCIA studies on pesticide removal using membrane technologies, meaningful comparisons among them are difficult. The literature on LCIA of pesticide removal in general is scarce, and to the best of our knowledge, no studies have specifically examined only the impacts of NF and RO pesticide removal. Existing studies generally focus on the

LCIA of the entire treatment process rather than on NF/RO removal alone, and they differ in methods, functional units, assumptions, and impact assessment categories. Even the few studies that address membrane-based micropollutant removal have limited applicability to pesticide-specific assessments. For example, Zepon Tarpani and Azapagic identified NF as the most effective option among advanced wastewater treatment technologies for reducing freshwater toxicity and human toxicity impacts from pharmaceuticals and personal care products, and Krishna Manda et al. reported that NF membranes outperform granular activated carbon in these same toxicity categories for drinking water micropollutant removal. However, neither study addresses pesticide removal specifically, nor do they separate the environmental impacts attributed solely to the membrane process [55,56]. The LCIA framework applied in this study assumes that reduced pesticide concentrations in the permeate correspond to lower potential toxicity impacts. However, pesticide removal by NF and RO involves separation rather than destruction. Consequently, the removed pesticides remain concentrated in the concentrate stream and may require further treatment or appropriate disposal. If discharged without treatment, concentrate streams could constitute a secondary source of environmental contamination. Therefore, the toxicity reductions quantified in this study should be interpreted as benefits associated with permeate quality improvement, not necessarily as reductions in the total pesticide mass released to the environment. For future studies, it is recommended to conduct experiments with more realistic pesticide feed concentrations to obtain more accurate mitigation estimates. It is also recommended to perform an LCIA of the entire treatment process to determine whether the environmental benefits achieved by NF and RO outweigh their overall environmental impacts.

3.4. Study Limitations

Several limitations should be considered when interpreting the results of this study. First, the LCIA was limited to impacts associated with pesticide occurrence and removal and did not include the complete life cycle inventory of NF and RO treatment systems. Because the investigation was performed at laboratory scale, environmental burdens related to membrane manufacture, energy consumption, cleaning chemicals, concentrate management, infrastructure, and membrane replacement were not considered. Second, pesticide removal experiments were conducted with feed concentrations of 100 µg/L, which are higher than concentrations typically reported in groundwater and drinking water. If reported concentrations in groundwater and drinking water were used as feed concentrations in this study, determining concentrations in the permeate would be constrained by the limits of detection and quantification of the analytical methods used. To improve environmental relevance, literature-derived concentrations were used in the LCIA modeling, while experimentally determined rejection factors were applied to estimate post-treatment concentrations. This approach assumes similar NF and RO rejection behavior at environmentally relevant concentrations, as reported in previous studies. Third, uncertainty associated with the reported environmental concentrations was not carried through the LCIA calculations. Fourth, the experiments were conducted on a laboratory scale membrane system under controlled conditions, and the resulting removal efficiencies may not fully represent full-scale treatment performance. The reference water volume of 42,500 m^3/day was used solely to estimate pesticide toxicity impacts and does not represent a full-scale membrane system design. Therefore, full-scale operational factors, including membrane recovery rates, staging configurations, and potential variations in rejection efficiency, were not considered. Finally, characterization factors for human cancer toxicity were available only for atrazine among the investigated pesticides, limiting the applicability of conclusions related to carcinogenic effects.

4. Conclusions

The LCIA results demonstrated differences in potential freshwater ecotoxicity and human toxicity for atrazine, imidacloprid, metolachlor, and tebuconazole, depending on pesticide-specific characterization factors and the LCIA method applied. NF and RO, which have been confirmed as efficient treatments for pesticide removal, have not been investigated in terms of LCIA for pesticide impact mitigation. This study conducted an LCIA using experimental data to determine the environmental impact of implementing NF and RO for pesticide removal. The RO membranes achieved more than 80% removal of all pesticides, compared with 27–96% removal with NF membranes; based solely on concentration removal efficiency, RO membranes are more effective. The LCIA results indicate that both NF and RO treatment reduced the calculated freshwater ecotoxicity and human toxicity impact scores across the evaluated LCIA methods and impact categories. When evaluating both removal efficiency and impact mitigation, RO is more favorable for pesticide removal. Within the scope of this study and based solely on pesticide removal efficiency and resulting toxicity impact mitigation, RO membranes showed greater reductions in impact scores than NF membranes. However, no conclusions regarding the overall environmental sustainability of NF and RO systems can be drawn because operational- and infrastructure-related impacts were outside the defined system boundaries. The results should be interpreted considering several limitations, including the exclusion of operational- and infrastructure-related impacts of NF and RO systems, the use of literature derived environmental concentrations, the absence of uncertainty propagation, the laboratory scale nature of the experiments, and the limited availability of characterization factors for certain toxicity categories. Therefore, future studies should evaluate the complete life cycle impacts of full-scale NF and RO systems and incorporate uncertainty analysis and environmentally realistic pesticide concentrations.

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Abbreviations

The following abbreviations are used in this manuscript:

CFs	Characterization factors
EF	Environmental Footprint
NF	Nanofiltration
IS	Impact scores
LCA	Life cycle assessment
LCIA	Life cycle impact assessment
MW	Molecular weight
MWCO	Molecular weight cut-off
PAF	Potentially affected fraction
PE	Population Equivalents
R	Rejection factor
UN	United Nations
RO	Reverse osmosis
WHO	World Health Organization

References

1. *ISO 14040:2006*; Environmental Management—Life Cycle Assessment—Principles and Framework. ISO: Geneva, Switzerland, 2006.
2. *ISO 14044:2006*; Environmental Management—Life Cycle Assessment—Requirements and Guidelines. ISO: Geneva, Switzerland, 2006.
3. Rosenbaum, R.; Bachmann, T.; Gold, L.; Huijbregts, M.; Jolliet, O.; Juraske, R.; Koehler, A.; Larsen, H.; MacLeod, M.; Margni, M.; et al. USEtox—The UNEP-SETAC toxicity model: Recommended characterisation factors for human toxicity and freshwater ecotoxicity in Life Cycle Impact Assessment. *Int. J. Life Cycle Assess.* **2008**, *13*, 532–546. [CrossRef]
4. Corominas, L.; Byrne, D.M.; Guest, J.S.; Hospido, A.; Roux, P.; Shaw, A.; Short, M.D. The application of life cycle assessment (LCA) to wastewater treatment: A best practice guide and critical review. *Water Res.* **2020**, *184*, 116058. [CrossRef]
5. Bare, J. TRACI 2.0: The tool for the reduction and assessment of chemical and other environmental impacts 2.0. *Clean Technol. Environ. Policy* **2011**, *13*, 687–696. [CrossRef]
6. Huijbregts, M.A.J.; Steinmann, Z.J.N.; Elshout, P.M.F.; Stam, G.; Verones, F.; Vieira, M.; Zijp, M.; Hollander, A.; Van Zelm, R. ReCiPe2016: A harmonised life cycle impact assessment method at midpoint and endpoint level. *Int. J. Life Cycle Assess.* **2017**, *22*, 138–147. [CrossRef]
7. Fazio, S.; Biganzoli, F.; De Laurentiis, V.; Zampori, L.; Sala, S.; Diaconu, E. *Supporting Information to the Characterisation Factors of Recommended EF Life Cycle Impact Assessment Methods—Version 2, from ILCD to EF 3.0*; European Commission: Ispra, Italy, 2018. [CrossRef]
8. Saouter, E.; Biganzoli, F.; Ceriani, L.; Versteeg, D.; Crenna, E.; Zampori, L.; Sala, S.; Pant, R. *Environmental Footprint: Update of Life Cycle Impact Assessment Methods-Ecotoxicity Freshwater, Human Toxicity Cancer, and Non-Cancer*; Publications Office of the European Union: Luxembourg, 2018.
9. Peña, N.; Knudsen, M.T.; Fantke, P.; Antón, A.; Hermansen, J.E. Freshwater ecotoxicity assessment of pesticide use in crop production: Testing the influence of modeling choices. *J. Clean. Prod.* **2019**, *209*, 1332–1341. [CrossRef]
10. Fantke, P.; Jolliet, O. Life cycle human health impacts of 875 pesticides. *Int. J. Life Cycle Assess.* **2016**, *21*, 722–733. [CrossRef]
11. Nemecek, T.; Antón, A.; Basset-Mens, C.; Gentil-Sergent, C.; Renaud-Gentié, C.; Melero, C.; Naviaux, P.; Peña, N.; Roux, P.; Fantke, P. Operationalising emission and toxicity modelling of pesticides in LCA: The OLCA-Pest project contribution. *Int. J. Life Cycle Assess.* **2022**, *27*, 527–542. [CrossRef]
12. Bellona, C.; Drewes, J.E.; Xu, P.; Amy, G. Factors affecting the rejection of organic solutes during NF/RO treatment—A literature review. *Water Res.* **2004**, *38*, 2795–2809. [CrossRef]
13. Verliefde, A.R.D.; Cornelissen, E.R.; Heijman, S.G.J.; Hoek, E.M.V.; Amy, G.L.; Bruggen, B.V.D.; Van Dijk, J.C. Influence of Solute−Membrane Affinity on Rejection of Uncharged Organic Solutes by Nanofiltration Membranes. *Environ. Sci. Technol.* **2009**, *43*, 2400–2406. [CrossRef]
14. Lim, Y.J.; Goh, K.; Nadzri, N.; Wang, R. Thin-film composite (TFC) membranes for sustainable desalination and water reuse: A perspective. *Desalination* **2025**, *599*, 118451. [CrossRef]
15. Zhang, Y.; Van Der Bruggen, B.; Chen, G.X.; Braeken, L.; Vandecasteele, C. Removal of pesticides by nanofiltration: Effect of the water matrix. *Sep. Purif. Technol.* **2004**, *38*, 163–172. [CrossRef]
16. Madsen, H.T.; Søgaard, E.G. Applicability and modelling of nanofiltration and reverse osmosis for remediation of groundwater polluted with pesticides and pesticide transformation products. *Sep. Purif. Technol.* **2014**, *125*, 111–119. [CrossRef]
17. Nikbakht Fini, M.; Madsen, H.T.; Muff, J. The effect of water matrix, feed concentration and recovery on the rejection of pesticides using NF/RO membranes in water treatment. *Sep. Purif. Technol.* **2019**, *215*, 521–527. [CrossRef]
18. Mukherjee, A.; Mehta, R.; Saha, S.; Bhattacharya, A.; Biswas, P.K.; Kole, R.K. Removal of multiple pesticide residues from water by low-pressure thin-film composite membrane. *Appl. Water Sci.* **2020**, *10*, 244. [CrossRef]
19. Ates, N.; Uzal, N.; Yetis, U.; Dilek, F.B. Removal of pesticides from secondary treated urban wastewater by reverse osmosis. *Environ. Sci. Pollut. Res.* **2023**, *30*, 8732–8745. [CrossRef]
20. Rodríguez-Alegre, R.; Pérez Megías, L.; Sanchis, S.; Andecochea Saiz, C.; You, X. Nanofiltration & reverse osmosis technical assessment for pesticides removal. *Discov. Environ.* **2024**, *2*, 52. [CrossRef]
21. Corominas, L.; Larsen, H.F.; Flores-Alsina, X.; Vanrolleghem, P.A. Including Life Cycle Assessment for decision-making in controlling wastewater nutrient removal systems. *J. Environ. Manag.* **2013**, *128*, 759–767. [CrossRef]
22. Razman, K.K.; Hanafiah, M.M.; Mohammad, A.W. An overview of LCA applied to various membrane technologies: Progress, challenges, and harmonization. *Environ. Technol. Innov.* **2022**, *27*, 102803. [CrossRef]
23. Roth, C.; Wünsch, R.; Dinkel, F.; Hugi, C.; Wülser, R.; Antes, R.; Thomann, M. Micropollutant abatement with UV/H2O2 oxidation or low-pressure reverse osmosis? A comparative life cycle assessment for drinking water production. *J. Clean. Prod.* **2022**, *336*, 130227. [CrossRef]

24. European Parliament; Council of the European Union. *Directive 2006/118/EC of the European Parliament and of the Council of 12 December 2006 on the Protection of Groundwater Against Pollution and Deterioration*; Publications Office of the European Union: Luxembourg, 2006.
25. European Parliament; Council of the European Union. *Directive (EU) 2020/2184 of the European Parliament and of the Council of 16 December 2020 on the Quality of Water Intended for Human Consumption (Recast)*; Publications Office of the European Union: Luxembourg, 2020.
26. European Parliament; Council of the European Union. *Directive (EU) 2024/3019 of the European Parliament and of the Council of 27 November 2024 Concerning Urban Wastewater Treatment (Recast)*; OJ L, 2024/3019; Publications Office of the European Union: Luxembourg, 2024.
27. Hauschild, M.; Rosenbaum, R.; Olsen, S. *Life Cycle Assessment: Theory and Practice*; Springer: Cham, Switzerland, 2017; pp. 1–1216.
28. Li, Y.; Zhang, S.; Zhang, W.; Xiong, W.; Ye, Q.; Hou, X.; Wang, C.; Wang, P. Life cycle assessment of advanced wastewater treatment processes: Involving 126 pharmaceuticals and personal care products in life cycle inventory. *J. Environ. Manag.* **2019**, *238*, 442–450. [CrossRef]
29. Sala, S.; Biganzoli, F.; Mengual, E.S.; Saouter, E. Toxicity impacts in the environmental footprint method: Calculation principles. *Int. J. Life Cycle Assess.* **2022**, *27*, 587–602. [CrossRef]
30. Dolar, D.; Vuković, A.; Ašperger, D.; Košutić, K. Effect of water matrices on removal of veterinary pharmaceuticals by nanofiltration and reverse osmosis membranes. *J. Environ. Sci.* **2011**, *23*, 1299–1307. [CrossRef]
31. Howard, G.; Bartram, J.; Williams, A.; Overbo, A.; Fuente, D.; Geere, J. *Domestic Water Quantity, Service Level and Health*; World Health Organization: Geneva, Switzerland, 2020.
32. United Nations. Available online: https://www.un.org/en/global-issues/water (accessed on 16 April 2026).
33. Vodoopskrba I Odvodnja. Available online: https://www.vio.hr/o-nama/vodoopskrba/1494 (accessed on 16 April 2026).
34. Institut Za Vode Josip Juraj Strossmayer. *Izvješće o Stanju Podzemnih Voda u Republici Hrvatskoj u 2024. Godini*; Institut Za Vode Josip Juraj Strossmayer: Zagreb, Croatia, 2025.
35. Hong, J.; Revel, M.; Boussetta, N.; Enderlin, G.; Nadaud, F.; Roig, C.; Nguimapi Tsague, L.; Firmin, S.; Grimi, N.; Merlier, F. Acute toxicity of atrazine degradation solutions via advanced oxidation processes: An experimental and in-silico approach based on mass spectrometry data. *Ecotoxicol. Environ. Saf.* **2025**, *302*, 118706. [CrossRef]
36. Li, S.; Huang, F.; Piao, H.; Li, W.; Liu, F.; Zhu, Q.; He, Y.; Wang, J.; Yan, M. Occurrence and distribution of atrazine in groundwater from agricultural areas in China. *Sci. Total Environ.* **2024**, *955*, 177161. [CrossRef]
37. Pires, N.L.; Araújo, E.P.D.; Muniz, D.H.D.F.; Oliveira-Filho, E.C.; Caldas, E.D. Pesticides in Ground and Surface Water from the Rio Preto Hydrographic Basin, an Important Agricultural Area in the Midwestern Region of Brazil. *Water* **2025**, *17*, 1186. [CrossRef]
38. Aparicio, V.; De Gerónimo, E. Pesticide pollution in argentine drinking water: A call to ensure safe access. *Environ. Chall.* **2024**, *14*, 100808. [CrossRef]
39. Jurado, A.; Walther, M.; Díaz-Cruz, M.S. Occurrence, fate and environmental risk assessment of the organic microcontaminants included in the Watch Lists set by EU Decisions 2015/495 and 2018/840 in the groundwater of Spain. *Sci. Total Environ.* **2019**, *663*, 285–296. [CrossRef]
40. Luo, W.; Putri, Z.S.; Sato, T.; Yamamuro, M. Monthly monitoring of neonicotinoid concentrations in tap water in Japan. *Sci. Total Environ.* **2026**, *1013*, 181250. [CrossRef]
41. Wan, Y.; Han, Q.; Wang, Y.; He, Z. Five degradates of imidacloprid in source water, treated water, and tap water in Wuhan, central China. *Sci. Total Environ.* **2020**, *741*, 140227. [CrossRef]
42. Campos, P.; Bruno, D.; Brugnera, M.; Dores, E. Pesticides in surface and groundwater in three micro-basins of the São Lourenço River, state of Mato Grosso, Brazil: Grab and passive sampling. *Int. J. Environ. Anal. Chem.* **2024**, *105*, 4993–5021. [CrossRef]
43. Herrero-Hernández, E.; Ordax, J.M.; Marín-Benito, J.M.; Nogal Sánchez, M.D.; Rodríguez-Cruz, M.S. Presence of Pesticides and Transformation Products and Associated Risk Assessment in Groundwater of a Region with an Intensive Agricultural Activity. *Environments* **2026**, *13*, 27. [CrossRef]
44. Bagagnan, R.; Abhervé, R.; Laverman, A.; Vautier, C. Groundwater controls on legacy antibiotics and pesticides in an intensive agricultural headwater catchment. *J. Hydrol.* **2026**, *669*, 135118. [CrossRef]
45. Stefano, P.H.P.; Roisenberg, A.; Santos, M.R.; Dias, M.A.; Montagner, C.C. Unraveling the occurrence of contaminants of emerging concern in groundwater from urban setting: A combined multidisciplinary approach and self-organizing maps. *Chemosphere* **2022**, *299*, 134395. [CrossRef]
46. Čelić, M.; Jaén-Gil, A.; Briceño-Guevara, S.; Rodríguez-Mozaz, S.; Gros, M.; Petrović, M. Extended suspect screening to identify contaminants of emerging concern in riverine and coastal ecosystems and assessment of environmental risks. *J. Hazard. Mater.* **2021**, *404*, 124102. [CrossRef]

47. Manjarres-López, D.P.; Andrades, M.S.; Sánchez-González, S.; Rodríguez-Cruz, M.S.; Sánchez-Martín, M.J.; Herrero-Hernández, E. Assessment of pesticide residues in waters and soils of a vineyard region and its temporal evolution. *Environ. Pollut.* **2021**, *284*, 117463. [CrossRef]
48. Xue, X.; Hawkins, T.R.; Ingwersen, W.W.; Smith, R.L. Demonstrating an approach for including pesticide use in life-cycle assessment: Estimating human and ecosystem toxicity of pesticide use in Midwest corn farming. *Int. J. Life Cycle Assess.* **2015**, *20*, 1117–1126. [CrossRef]
49. Koch, D.; Friedl, A.; Mihalyi, B. Influence of different LCIA methods on an exemplary scenario analysis from a process development LCA case study. *Environ. Dev. Sustain.* **2023**, *25*, 6269–6293. [CrossRef]
50. World Health Organization. *The WHO Recommended Classification of Pesticides by Hazard and Guidelines to Classification 2019*; World Health Organization: Geneva, Switzerland, 2020.
51. Kimura, K.; Amy, G.; Drewes, J.; Watanabe, Y. Adsorption of hydrophobic compounds onto NF/RO membranes: An artifact leading to overestimation of rejection. *J. Membr. Sci.* **2003**, *221*, 89–101. [CrossRef]
52. Tan, L.S.; Ahmad, A.L.; Shukor, S.R.A.; Yeap, S.P. Impact of Solute Properties and Water Matrix on Nanofiltration of Pesticides. *Chem. Eng. Technol.* **2019**, *42*, 1780–1787. [CrossRef]
53. Dagher, G.; Saab, G.; Martin, A.; Couturier, G.; Candido, P.; Moulin, L.; Croué, J.P.; Teychene, B. Understanding and predicting the adsorption and rejection of pesticides and metabolites by hollow fiber nanofiltration membranes. *Sep. Purif. Technol.* **2024**, *330*, 125323. [CrossRef]
54. Garcia, N.; Moreno, J.; Cartmell, E.; Rodriguez-Roda, I.; Judd, S. The application of microfiltration-reverse osmosis/nanofiltration to trace organics removal for municipal wastewater reuse. *Environ. Technol.* **2013**, *34*, 3183–3189. [CrossRef]
55. Zepon Tarpani, R.R.; Azapagic, A. Life cycle environmental impacts of advanced wastewater treatment techniques for removal of pharmaceuticals and personal care products (PPCPs). *J. Environ. Manag.* **2018**, *215*, 258–272. [CrossRef]
56. Manda, B.M.K.; Worrell, E.; Patel, M.K. Innovative membrane filtration system for micropollutant removal from drinking water—Prospective environmental LCA and its integration in business decisions. *J. Clean. Prod.* **2014**, *72*, 153–166. [CrossRef]

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