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Zakład Zaopatrzenia w Wodę i Biogospodarki

Rozprawa doktorska

Analiza śladu środowiskowego miejskiej oczyszczalni ścieków w koncepcji zrównoważonego rozwoju

Analysis of the environmental footprint of an urban wastewater treatment plant in the context of sustainable development

mgr inż. Paulina Szulc-Kłosińska

Dziedzina nauk inżynieryjno-technicznych
Dyscyplina naukowa: inżynieria środowiska, górnictwo i energetyka

Promotor: dr hab. inż. Zbysław Dymaczewski, profesor Uczelni

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Spis treści

Streszczenie

W warunkach intensyfikacji działań na rzecz neutralności klimatycznej oraz wdrażania unijnych regulacji w zakresie zrównoważonego rozwoju, miejskie oczyszczalnie ścieków (OŚ) stają się istotnym przedmiotem badań w kontekście ich całkowitego oddziaływania środowiskowego. Obiekty te, będąc filarem systemów gospodarki wodno-ściekowej, generują istotne emisje gazów cieplarnianych (GHG), głównie podtlenku azotu (N_2O) oraz metanu (CH_4). Celem pracy było opracowanie spójnej metodologii oceny i kalkulacji śladu środowiskowego OŚ w warunkach polskich, ze szczególnym uwzględnieniem śladu węglowego jako głównej składowej śladu środowiskowego. Punktem wyjścia była szczegółowa analiza cyklu życia (LCA) przeprowadzona dla Centralnej Oczyszczalni Ścieków w aglomeracji poznańskiej. Analiza wykazała, że ślad węglowy, wynikający głównie z zapotrzebowania energetycznego obiektu oraz struktury miksu energetycznego, stanowi dominującą część całkowitego śladu środowiskowego. Obserwacje te potwierdziły zasadność koncentracji dalszych badań na komponentach emisji GHG i ich dokładnym odwzorowaniu w analizach środowiskowych.

Kolejnym etapem działań publikacyjnych było opracowanie autorskiego algorytmu kalkulacji śladu węglowego miejskiej oczyszczalni ścieków, zgodnego z wytycznymi GHG Protocol i unijnymi standardami raportowania (ESRS). Algorytm ten został opisany szczegółowo w cyklu dwóch publikacji. Uwzględniono w nim wszystkie zakresy emisji GHG Protocol: Zakres 1 (bezpośrednie, w tym emisje technologiczne i inne niezorganizowane emisje), Zakres 2 (emisje pośrednie związane z energią) oraz Zakres 3 (emisje pośrednie z łańcucha wartości). Zidentyfikowano również istotne luki informacyjne i zaproponowano strategię poprawy kompletności danych dla obiektów wodno-kanalizacyjnych, nadając charakter praktyczny opracowanemu algorytmowi. Jego celem było nie tylko umożliwienie spójnych obliczeń środowiskowych, ale również zapewnienie zgodności z nadchodzącymi obowiązkami raportowymi na poziomie krajowym i unijnym. Aby zweryfikować dokładność powszechnie stosowanych narzędzi kalkulacyjnych, przeprowadzono analizy porównawcze z wykorzystaniem trzech podejść: dwóch ogólnodostępnych narzędzi ECAM i CFCT (w wersjach 2014 i 2024) oraz opracowanego algorytmu. Najważniejszym elementem tego etapu była kampania pomiarowa emisji N_2O i CH_4 przeprowadzona w Centralnej Oczyszczalni Ścieków w Poznaniu w skali technicznej. Umożliwiła ona obliczenie rzeczywistych wskaźników emisji, które znacznie różniły się od wartości domyślnych używanych w narzędziach ECAM i CFCT o dwa rzędy wielkości. Narzędzia bazujące na wartościach literaturowych przeszacowywały emisje nawet o 100-200%, co potwierdza potrzebę lokalnej kalibracji i walidacji modeli emisyjnych. Opracowany algorytm, korzystający z danych rzeczywistych, oszacował emisje nawet o 80% niższe niż ECAM i znacznie dokładniejsze od CFCT.

Ostatnim komponentem rozprawy była analiza systemowa dotycząca warunków osiągnięcia stanu Zerowych emisji netto w sektorze oczyszczania ścieków, przedstawiona w artykule przeglądowym. Wskazano, że osiągnięcie neutralności węglowej wymaga synergii działań: modernizacji układów technologicznych (m.in. zastosowania technologii Anammox, nitryfikacji częściowej), poprawy efektywności energetycznej, odzysku energii z biogazu, a także rozważenia offsetowania lub wychwytywania tzw. emisji rezydualnych. Wyniki pracy potwierdzają, że precyzyjna kalkulacja śladu węglowego oczyszczalni ścieków musi opierać się na danych empirycznych i lokalnie dopasowanych wskaźnikach, szczególnie dla emisji N_2O i CH_4. Praca wnosi istotny wkład do wiedzy z zakresu inżynierii środowiska i gospodarki wodno-ściekowej, proponując spójne i operacyjne podejście do zarządzania śladem środowiskowym w sektorze infrastruktury miejskiej. Zaproponowane rozwiązania mogą stanowić fundament dla systemów decyzyjnych w zakresie dekarbonizacji obiektów komunalnych, przyczyniając się do realizacji celów Europejskiego Zielonego Ładu.

Słowa kluczowe: emisje gazów cieplarnianych, podtlenek azotu, metan, ślad węglowy, oczyszczanie ścieków

Abstract

In the context of accelerating climate neutrality commitments and the expanding scope of EU sustainability regulations, urban wastewater treatment plants (WWTPs) are becoming increasingly important objects of study due to their overall environmental footprint. These facilities, as fundamental components of urban water and sanitation systems, contribute substantially to greenhouse gas (GHG) emissions, particularly nitrous oxide (N_2O) and methane (CH_4). The aim of this study was to develop a consistent methodology for assessing and calculating the environmental footprint of WWTPs under Polish conditions, with a particular focus on the carbon footprint (CF) as the dominant contributor.

The starting point of the research was a detailed life cycle assessment (LCA) of the Central WWTP serving the Poznań metropolitan area. The analysis revealed that the CF - mainly driven by the plant's energy consumption and the composition of the local energy mix - accounted for the largest share of the total environmental impact. These observations justified a research focus on accurately quantifying GHG-related components in environmental performance analyses.

A subsequent research stage involved the development of a proprietary CF calculation algorithm tailored for urban WWTPs, fully aligned with the GHG Protocol and European Sustainability Reporting Standards (ESRS). This algorithm, described in a two-part publication series, incorporates all emission scopes: Scope 1 (direct emissions, including process and fugitive emissions), Scope 2 (indirect emissions from purchased energy), and Scope 3 (indirect emissions across the value chain). The study also identified key data gaps and proposed strategies for improving data completeness in the water and wastewater sector, rendering the algorithm not only accurate but also practically applicable. Its dual purpose was to support consistent footprint calculations and ensure compliance with upcoming national and EU-level reporting requirements. To assess the reliability of commonly used GHG calculators, a comparative analysis of three approaches was conducted: ECAM, CFCT (2014 and 2024 versions), and the proposed algorithm. A central element of this phase was a full-scale GHG monitoring campaign for N2O and CH4 emissions, carried out at the same WWTP. This enabled the derivation of site-specific empirical emission factors (EFs), which differed substantially from the default values embedded in ECAM and CFCT. Literature-based calculators were found to overestimate emissions by 100-200%, confirming the need for local calibration and validation. The algorithm developed in this study, informed by real operational data and updated global warming potentials (GWPs), yielded emission estimates up to 80% lower than ECAM and significantly more precise than CFCT.

The final component of the study involved a system-level analysis of pathways toward achieving net-zero emissions in the wastewater sector, as synthesized in a literature review. The findings emphasize that reaching carbon neutrality requires a synergistic strategy: upgrading biological treatment technologies (e.g., Anammox, partial nitrification), enhancing energy efficiency, recovering energy from biogas, and considering offsetting or capturing residual, unavoidable emissions.

The results of this study confirm that accurate CF accounting in WWTPs must be based on empirical data and locally calibrated emission factors, particularly for high-impact gases such as N_2O and CH_4. This work makes a significant contribution to the fields of environmental engineering and urban water management by offering a consistent, practical approach to managing the environmental footprint of municipal infrastructure. The proposed methodologies and findings may support decision-making processes in the decarbonization of public utilities and contribute meaningfully to the goals of the European Green Deal.

Keywords: GHG emissions, nitrous oxide, methane, carbon footprint, wastewater treatment

Wykaz publikacji stanowiących podstawę rozprawy doktorskiej

Zgodnie z Ustawą z dnia 20 lipca 2018 r. Prawo o szkolnictwie wyższym i nauce [tekst jednolity Dz. U. 2021 poz. 478 Art. 187, ust. 3]: „Rozprawę doktorską może stanowić praca pisemna, w tym monografia naukowa, **zbiór opublikowanych i powiązanych tematycznie artykułów naukowych,** praca projektowa, konstrukcyjna, technologiczna, wdrożeniowa lub artystyczna, a także samodzielna i wyodrębniona część pracy zbiorowej."

Lp.	Pozycja	Wkład własny	IF[a]	IF[b]	Cytowania[c]	Punktacja MNiSW[e] (MENiN)
A1	**Szulc, P**., Kasprzak, J., Dymaczewski, Z., Kurczewski, P. (2021). Life Cycle Assessment of Municipal Wastewater Treatment Processes Regarding Energy Production from the Sludge Line. Energies, 14(2), 356. https://doi.org/10.3390/en14020356	40%	3,252	3,100	15 (18)	140 (140)
A2	**Szulc-Kłosińska, P**., Dymaczewki Z. (2023). Algorithm of carbon footprint calculation for municipal wastewater treatment plant - part one. GAZ WODA I TECHNIKA SANITARNA, 1(10), 32-44. https://doi.org/10.15199/17.2023.10.4	75%	-	-	-	140 (20)
A3	**Szulc-Kłosińska, P**., Dymaczewski Z. (2023). Algorithm of carbon footprint calculation for municipal wastewater treatment plant - part two. GAZ WODA I TECHNIKA SANITARNA, 1(11), 32-40. https://doi.org/10.15199/17.2023.11.5	75%	-	-	-	140 (20)
A4	Comparative Analysis of Calculation Tools for Estimating Carbon Footprint of Wastewater Treatment Plants: Methodologies and Emission Factors* (2025). Archives of Environmental Protection – ***przyjęte do druku***	60%	1,300	1,300	-	100 (100)
A5	Maktabifard, M., Al-Hazmi, H. E., **Szulc, P**., Mousavizadegan, M., Xu, X., Zaborowska, E., Li, X., Mąkinia, J. (2023). Net-zero carbon condition in wastewater treatment plants: A systematic review of mitigation strategies and challenges. Renewable and Sustainable Energy Reviews, 185, 113638. https://doi.org/10.1016/j.rser.2023.113638	15%	16,300	17,500	60 (67)	200 (200)
				Indeks Hirscha	**2,0 (2,0)**	
	PODSUMOWANIE	**53% (średnia)**	**6,951**	**7,300**	**75 (89)**	**720 (480)**

a) sumaryczny Impact Factor (IF) wg bazy Web of Science (WoS) zgodny z rokiem ukazania się pracy
b) sumaryczny Impact Factor (IF) wg bazy Web of Science (WoS) 5 - letni
c) sumaryczna liczba cytowań - Web of Science (Scopus)
e) Punktacja MNiSW w roku ukazania się pracy (punktacja MENiN w roku przygotowania zestawienia)
*autor korespondencyjny Szulc-Kłosińska P.

Wykaz pozostałego dorobku doktoranta

Publikacje

1. Gielniak, W., Dymaczewski Z., **Szulc-Kłosińska P**., Jeż-Walkowiak J. (2025). Ścieki przemysłowe jako zewnętrzne źródło węgla dla procesu denitryfikacji w miejskiej oczyszczalni ścieków. GAZ, WODA I TECHNIKA SANITARNA, 1(2), 22–30. https://doi.org/10.15199/17.2025.2.4
2. **Szulc, P.**, Dymaczewski Z. (2020). Częściowa granulacja osadu czynnego w oczyszczalni ścieków pracującej w technologii SBR. Forum Eksploatatora. Nr 3/4 (108-109). S. 38-44
3. **Szulc, P.** (2018). Piasek z oczyszczania ścieków- analiza problemu, możliwości odzysku i wykorzystania na przykładzie oczyszczalni ścieków w Poznaniu. GAZ WODA I TECHNIKA SANITARNA, 1(5), 32-35. https://doi.org/10.15199/17.2018.5.9
4. **Szulc, P**., Walkowska, S., Dymaczewski Z. (2018). Stabilność składu ścieków syntetycznych zasilających laboratoryjne systemy osadu czynnego. GAZ WODA I TECHNIKA SANITARNA, 1(12), 25-30. https://doi.org/10.15199/17.2018.12.4

Konferencje

1. *Life cycle assessment of semi-centralized municipal wastewater treatment facilities - case study from Poznań, Poland - POSTER*
Współautorzy: **Szulc P**., Kasprzak J., Dymaczewski Z., Kurczewski P., Pielach M.
Prezenter: **Szulc Paulina**
Nazwa konferencji: IWA Wastewater, Water and Resource Recovery (WWRR) 2022, 10-13.04.2022

2. *Contribution of GHG emissions from bioreactors to the total carbon footprint (CF) of municipal wastewater treatment plants (WWTP) – POSTER*
Współautorzy**: Szulc Paulina**, Mąkinia Jacek, Dymaczewski Zbysław
Prezenter: **Szulc Paulina**
Nazwa konferencji: 2nd International Conference Strategies toward Green Deal Implementation - Water, Raw Materials & Energy, 8-10.12.2021, konferencja *online*

3. *Granulacja osadu czynnego w reaktorach SBR z dużym udziałem ścieków dowożonych w dopływie*
Współautorzy: **Szulc Paulina**, Dymaczewski Zbysław
Prezenter: Dymaczewski Zbysław
Nazwa konferencji: XVIII Konferencja Naukowo-Techniczna „Woda-Ścieki-Człowiek-Środowisko", Licheń, 11-12.03.2020

4. *From a floc to a granule - reconstruction of aerobic sludge structure under different SBR cycle routine and a various fed source: domestic-industrial vs. artificial wastewater*
Współautorzy: **Szulc Paulina**, Dymaczewski Zbysław
Prezenter: **Szulc Paulina**
Nazwa konferencji: 2nd IWA Polish Young Water Professionals Conference: Emerging Technologies in Water and Wastewater Sector, Warszawa, 12-14.02.2020

5. *Prowadzenie procesu biologicznego oczyszczania ścieków przy wykorzystaniu nadrzędnego systemu sterowania*
Współautorzy**: Szulc Paulina**, Pielach Monika
Prezenter: **Szulc Paulina**
Nazwa i miejsce konferencji: Innowacyjne rozwiązania w oczyszczaniu ścieków i zagospodarowaniu osadów, Słupsk - Dolina Charlotty, 10-12.10.2018

Inne wystąpienia eksperckie (wybrane):

1. Rola panelistki i wykładowczyni w cyklu "ESG - Raportowanie w praktyce" (edycja III, kwiecień-czerwiec 2023), wystąpienie online
   - Zakres: liczba kluczowych wskaźników środowiskowych, metodyka liczenia emisji GHG
2. Prezentacja podczas kursu "ESG - Raportowanie w praktyce" (13 kwietnia 2023), wystąpienie online
   - Temat: „Omówienie wskaźników i ich interpretacja", „Metody liczenia wybranych wskaźników"
3. Prelegentka na "Akademii Prawa Żywnościowego" (25-26 maja 2022), Mała Wieś k/Warszawy
   - Temat: „Ślad węglowy produkcji żywności"
   - Organizator: Polska Federacja Producentów Żywności / FoodLex
4. Panelistka "Badania Świadomości Klimatycznej Spółek 2022 (12 października 2022), występnie online
   - Temat: Standard Net-Zero, SBT-i, przygotowanie spółek do wymogów klimatycznych

Najważniejsza działalność zawodowa, naukowa i popularyzująca naukę

- Opracowanie strategii dekarbonizacji dla wielu organizacji – w tym dla AQUANET S.A. oraz Grupy AQUANET: pomoc przy zaplanowaniu i wdrożeniu ciągłego monitoringu emisji podtlenku azotu na Badawczej Centralnej Oczyszczalni Ścieków dla m/Poznania w Koziegłowach oraz Lewobrzeżnej Oczyszczalni Ścieków w Poznaniu.
- Praca na stanowisku Starszego Konsultanta ds. Zrównoważonego Rozwoju – obliczenia śladu węglowego wielu organizacji, w tym koordynacja kalkulacji śladu węglowego Grupy Kapitałowej ORLEN – długofalowa i kontynuowana współpraca w roli konsultanta 2022 r.
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- Samodzielne zaproponowanie tematu i zakresu pracy magisterskiej: Wykorzystanie granulowanego osadu czynnego do intensyfikacji pracy oczyszczalni ścieków typu SBR; organizacja i koordynacja współpracy z analizowaną w ramach pracy magisterskiej oczyszczalnią ścieków – zebranie i udostępnienie danych, a także nadzór nad pracą laboratoryjną Magistrantki, nadzór merytoryczny nad treścią pracy – wraz z Opiekunem, prof. dr. hab. inż. Z. Dymaczewskim.
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- Udział w projekcie stażowym AQUASTAŻ 2019 w roli Opiekuna Stażu; przygotowanie planu stażu (projekt badawczo-rozwojowy zgodny z bieżącymi potrzebami AQUANET SA) – zatrudnienie Studenta Politechniki Poznańskiej; staż zakończony propozycją tematu pracy inżynierskiej – zrealizowanej pod okiem prof. dr. Hab. Inż. Z. Dymaczewskiego – powstała praca otrzymała wyróżnienie w konkursie na wyróżniającą się pracę dyplomową w obszarze techniki oraz organizacji produkcji i usług organizowanym przez Poznańską Radę Federacji Stowarzyszeń Naukowo-Technicznych.
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- Zatrudnienie na stanowisku Młodszego Technologa ds. Ścieków, a następnie Specjalisty ds. Badań i Rozwoju AQUANET S.A. – aktywne wsparcie łączenia aktywności naukowej z zawodową.
- Wystąpienie w materiale promującym Wydział Inżynierii Środowiska i Energetyki Politechniki Poznańskiej pt. PUT Inżynieria Środowiska Promo 2020.
- Aktywność w Kole Naukowym Technologii Ochrony Wód: organizacja warsztatów naukowych dla uczniów szkół podstawowych oraz prowadzenie warsztatów naukowych podczas Nocy Naukowców 2016 i 2017 – warsztaty pt. „Mali-wielcy czyściciele” oraz „H_2O i tylko to!”

Udział w projektach badawczych

1. Integrated system for SImultaneous Recovery of Energy, organics and Nutrients and generation of valuable products from municipal wastewater (SIREN) 2020-2024. Granty Norweskie. Projekt **NOR/POLNOR/SIREN/0069/2019-00**:
   - Aktywny udział w przygotowaniu wniosku na obydwóch etapach,
   - Koordynacja prac badawczych z ramienia Aquanet S.A (2020-2022),
   - Zaplanowane, koordynacja i realizacja kampanii pomiarowej emisji gazów cieplarnianych na Centralnej Oczyszczalni Ścieków dla m/Poznania w Koziegłowach.

2. Shortcut nitrification in activated sludge process treating domestic wastewater - key technology for low-carbon and clean wastewater treatment (SNIT) 2020-2024. Granty Norweskie. Projekt **NOR/POLNOR/SNIT/0033/2019-00**:
   - Aktywny udział w przygotowaniu wniosku na obydwóch etapach,
   - Koordynacja prac badawczych z ramienia Aquanet S.A (2020-2022).

3. REsilient WAter Innovation for Smart Economy (REWAISE). 2020-2025. Projekt dofinansowany z programu badań i innowacji Unii Europejskiej „Horyzonty 2020” na podstawie umowy grantowej nr **869496**:
   - Aktywny udział w przygotowaniu i procesie złożenia wniosku na obydwóch etapach,
   - Koordynacja prac badawczych z ramienia Aquanet S.A (2020-2022) w roli kierownika projektu dla obszaru Europy Kontynentalnej (Polska, Czechy, Szwecja)

4. Wysokoefektywne metody oczyszczania wody i ścieków (numer projektu: **504101/0713/SBAD/0937**). Wykonawca. Grant wewnętrzny PP:
   - Realizacja prac badawczych związanych z osadem granulowanym – także z wykorzystaniem ścieków syntetycznych.

5. Wybrane zagadnienia gospodarki wodno-ściekowej (numer projektu: **01/13/DSMK//0882/18**). Wykonawca. Grant wewnętrzny PP:
   - Realizacja prac badawczych związanych ze stabilnością ścieków syntetycznych wykorzystywanych w badaniach naukowych.

Wykaz skrótów

BZT_5	Biochemiczne Zapotrzebowanie Tlenu (5-dniowe)	Parametr ścieków - jednostkowe obciążenie
CF	Carbon Footprint (ślad węglowy)	Kluczowy wskaźnik środowiskowy
CH_4	Metan	Gaz cieplarniany
COŚ	Centralna Oczyszczalnia Ścieków	
CO_2	Dwutlenek węgla	Gaz cieplarniany
CO_2e	Ekwiwalent dwutlenku węgla	Jednostka w przeliczaniu GHG
ChZT	Chemiczne Zapotrzebowanie Tlenu	Parametr jakości ścieków
ESG	Environmental, Social, Governance	Raportowanie zrównoważonego rozwoju
ESRS	European Sustainability Reporting Standards	Standardy raportowania UE
FTIR	Fourier Transform Infrared Spectroscopy	Metoda pomiarowa emisji gazów
GHG	Greenhouse Gases (gazy cieplarniane)	Kategorie emisji
GOZ	Gospodarka Obiegu Zamkniętego	Jedna z koncepcji zrównoważonego zarządzania odpadami
GWP100	Global Warming Potential (100 lat)	Potencjał cieplarniany gazów
IPCC AR6	Intergovernmental Panel on Climate Change, Sixth Assessment Report	Raport IPCC
LCA	Life Cycle Assessment (ocena cyklu życia)	Metodologia oceny środowiskowej
$N\text{-}NH_4$	Jon amonowy	Parametr ścieków
$N\text{-}NO_2$	Azotyny	Parametr ścieków
$N\text{-}NO_3$	Azotany	Parametr ścieków
N_{tot}	Azot ogólny	Parametr ścieków
N_2O	Podtlenek azotu	Gaz cieplarniany
OZE	Odnawialne źródła energii	Źródła energii
OŚ	Oczyszczalnia ścieków	Obiekt badawczy
OŚM	Oczyszczalnia ścieków miejskich	Obiekt badawczy

$P\text{-}PO_4$	Fosforany	Parametr ścieków
PIX	Siarczan żelaza(III) (koagulant)	Środek technologiczny w OŚ
PN/PN-EN/ISO	Polskie Normy / Normy Europejskie / International Organization for Standardization	Standardy normatywne
P_{tot}	Fosfor ogólny	Parametr ścieków
SCADA	Supervisory Control And Data Acquisition	System sterowania i monitoringu
SIREN	Integrated system for SImultaneous Recovery of Energy, organics and Nutrients	Projekt badawczy
SNIT	Shortcut Nitrification in activated sludge process treating domestic wastewater	Projekt badawczy
TKN	Total Kjeldahl Nitrogen (azot Kjeldahla)	Parametr ścieków
UE	Unia Europejska	Organizacja międzynarodowa
WRI	World Resources Institute	Instytucja opracowująca metodologie GHG
WWTP	Wastewater Treatment Plant	Oczyszczalnia ścieków (ang.)

Definicje

Analiza cyklu życia (LCA)	Metoda oceny wpływu systemów technicznych na środowisko w całym cyklu życia, zgodna z normami ISO 14040-14044.	Metodologia badań środowiskowych.
Environmental Footprint	Ślad środowiskowy - zbiorcza miara całkowitego wpływu systemu lub procesu na środowisko w cyklu życia.	Ocena oddziaływania środowiskowego.
Europejski Zielony Ład (European Green Deal)	Strategia UE dotycząca zrównoważonego rozwoju, dekarbonizacji i neutralności klimatycznej.	Kontekst legislacyjny.
Gazy cieplarniane (GHG)	Grupa gazów zatrzymujących ciepło w atmosferze, m.in. CO_2, CH_4, N_2O.	Podstawa kalkulacji śladu węglowego.
Odnawialne Źródła Energii (OZE)	Źródła energii niewyczerpywalne w skali ludzkiego życia, np. wiatr, słońce, biogaz.	Energetyka oczyszczalni.
One Health (Jedno Zdrowie)	Koncepcja analizująca powiązania zdrowia ludzi, zwierząt i środowiska jako jednego systemu.	Kontekst zrównoważonego rozwoju.
Zerowe emisje netto (Net Zero)	Stan równowagi, w którym emisje gazów cieplarnianych są zredukowane do określonych naukowo poziomów	Cel dekarbonizacji sektora wodno-ściekowego.
Ślad węglowy (CF)	Carbon Footprint - sumaryczna emisja gazów cieplarnianych w ekwiwalencie CO_2.	Kluczowy wskaźnik oceny środowiskowej oczyszczalni.

1. Wprowadzenie

Współczesne uwarunkowania środowiskowe, społeczne i legislacyjne wywierają silną presję na sektor gospodarki wodno-ściekowej, wymuszając nie tylko modernizację istniejących systemów technicznych, ale również redefinicję ich roli w zrównoważonym funkcjonowaniu struktur miejskich. Szczególnie istotnym ogniwem w tym kontekście są oczyszczalnie ścieków (OŚ). Mimo że przez dekady postrzegane były one niemal wyłącznie przez pryzmat ochrony zasobów wodnych oraz zdrowia publicznego - obecnie analizowane są coraz częściej jako złożone systemy oddziałujące aktywnie na środowisko w wielu wymiarach: konsumpcji energii i zasobów, minimalizacji zanieczyszczeń wodnych przy jednoczesnym generowaniu ich w innych obszarach. Zgodnie z koncepcją *Jedno Zdrowie* (ang. *One Health*), oczyszczalnie pełnią rolę nie tylko infrastruktury sanitarnej (Farkas et al., 2025), ale także środowiskowych punktów przejściowych dla patogenów, antybiotyków, mikroplastiku i innych mikrozanieczyszczeń (La Rosa et al., 2025).

W świetle aktualnych wyzwań klimatycznych i stale kurczących się źródeł zasobów, konieczna jest całościowa i ilościowa ocena funkcjonowania OŚ - nie tylko przez pryzmat ich pozytywnego wkładu w jakość środowiska wodnego, ale również z uwzględnieniem negatywnych skutków towarzyszących ich eksploatacji. Warto podkreślić, że oczyszczanie ścieków, jako proces technologicznie niezbędny, charakteryzuje się szczególnym statusem: nie może być całkowicie wyeliminowany. Dlatego też proces ten powinien być stale optymalizowany - tak, aby jego negatywny wpływ na środowisko był możliwie najmniejszy przy zachowaniu efektywności operacyjnej zgonie z postanowieniami Europejskiego Zielonego Ładu (ang. *European Green Deal*) - Rysunek 1.

Rysunek 1. Główne obszary Zielonego Ładu UE [opracowanie własne].

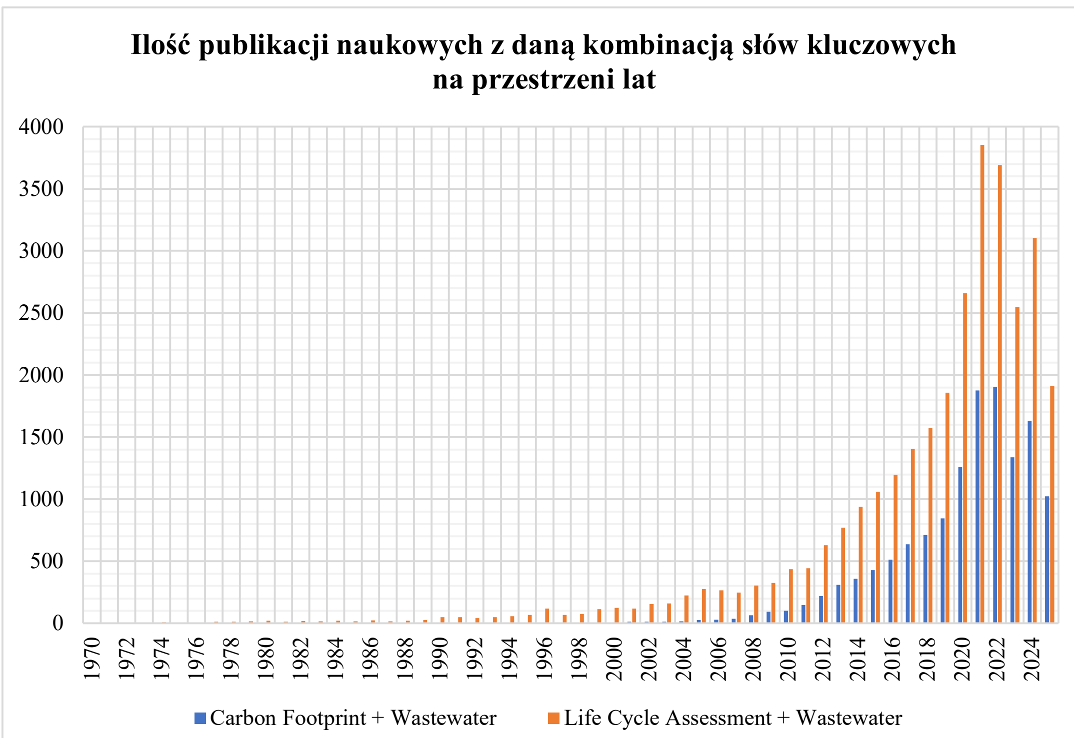
W tym kontekście kluczowym sposobem oceny staje się ślad środowiskowy (ang. *environmental footprint*), rozumiany jako zbiorcza miara całkowitego wpływu danego systemu lub procesu na środowisko w całym jego cyklu życia. W przypadku OŚ obejmuje on m.in.: zużycie zasobów (woda, paliwa, materiały), emisję gazów cieplarnianych (ang. *greenhouse gases - GHG*) z procesów technologicznych, powstawanie odpadów, uwalnianie mikrozanieczyszczeń (w tym mikroplastiku i związków farmaceutycznych), jak również emisje hałasu i uciążliwości zapachowych (Cui et al., 2025).

Narzędziem, które jest odpowiedzią na potrzebę opracowania spójnego, mierzalnego, a co za tym idzie porównywalnego sposobu oceny śladu środowiskowego miejskiej OŚ, który umożliwiałby identyfikację najbardziej obciążających etapów procesu oraz ocenę wpływu zastosowanych technologii i rozwiązań infrastrukturalnych, jest analiza cyklu życia (ang. *life cycle assessment* - LCA). Metoda ta uznawana jest na gruncie inżynierii środowiska za najbardziej kompleksowe narzędzie oceny wpływu systemów technicznych na środowisko. LCA, przeprowadzona zgodna z międzynarodowymi normami ISO 14040-14044 [1], umożliwia nie tylko ilościowe określenie śladu węglowego czy wodnego, ale także analizę

[1] ISO 14040–14044 to normy, które dostarczają wytycznych do przeprowadzenia oceny cyklu życia (Life Cycle Assessment, LCA). Ocena cyklu życia to narzędzie analityczne, które pozwala organizacjom dokładnie ocenić wpływ ich produktów, procesów lub usług na środowisko na każdym etapie - od pozyskania surowców, przez produkcję i użytkowanie, aż po utylizację

alternatywnych scenariuszy modernizacji, optymalizację efektywności energetycznej oraz weryfikację wpływu rozwiązań w kontekście gospodarki o obiegu zamkniętym (GOZ).

Dzięki możliwościom integrowania danych materiałowych, energetycznych, emisyjnych i operacyjnych, LCA pozwala na rzetelną ocenę porównawczą różnych wariantów technologicznych - np. systemów opartych na osadzie czynnym, reaktorach MBR (ang. *Membrane Bioreactor*), hybrydach biologiczno-chemicznych czy układach z odzyskiem energii z osadów (Menon & Kalyanraman, 2025). Co więcej, metodologia ta umożliwia identyfikację tzw. „gorących punktów" (ang. *hot spots*) środowiskowych, których optymalizacja może znacząco obniżyć ślad środowiskowy bez konieczności kosztownej przebudowy całych układów technologicznych. W praktyce inżynierskiej oznacza to możliwość wsparcia procesów decyzyjnych w zakresie modernizacji obiektów, planowania inwestycji i przygotowania danych na cele ujawień w ramach zrównoważonego raportowania ESG (ang. *Environmental, Social and Governance reporting*).



Rysunek 2. Ilość publikacji ze słowami kluczowymi „Carbon Footprint" oraz „Wastewater", „Life Cycle Assessment" oraz „Wastewater" [opracowanie własne na bazie Lens.org].

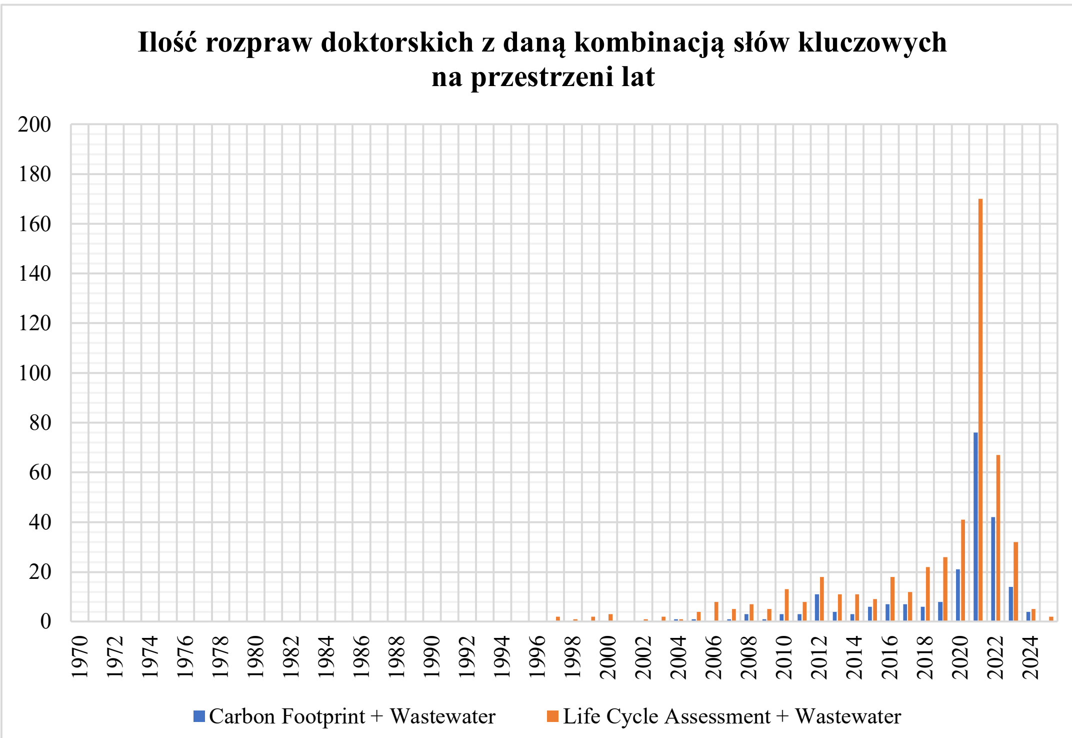
Na przestrzeni ostatnich dziesięcioleci (Rysunek 2) znaczenie analizy cyklu życia w ocenie technologii wodno-ściekowych dynamicznie wzrosło. Od roku 2000. obserwuje się coraz większą liczbę publikacji, które wykorzystują metodę LCA do badania różnych wariantów technologicznych oczyszczalni ścieków. Jak wskazują Yang i Tu (2025), LCA stała się narzędziem referencyjnym w ocenie technologii wodno-ściekowych, pozwalając na porównywanie alternatyw technologicznych oraz wspieranie decyzji inwestycyjnych. Badania Nurzhan i współautorów (2025) dowodzą, że w wymiarze globalnym LCA w sektorze wodno-kanalizacyjnym rozwija się równolegle z jego implementacją w gospodarce odpadami komunalnymi, tworząc podstawy do kompleksowych analiz systemowych dla całych aglomeracji.

W Polsce badania LCA oczyszczalni ścieków pozostają wciąż ograniczone (Burchart-Korol & Zawartka, 2019), chociaż rośnie zainteresowanie ich implementacją w związku z rosnącymi wymaganiami regulacyjnymi Unii Europejskiej (UE). Dominującym problemem w badaniach krajowych jest korzystanie z międzynarodowych baz danych, takich jak Ecoinvent, które nie pozwalają na pełne odzwierciedlenie lokalnych uwarunkowań. Wśród nich należałoby wymienić: krajowy miks energetyczny, charakterystykę ścieków czy procesy gospodarki osadami. Wskazuje się zatem na potrzebę opracowania dedykowanych krajowych baz danych oraz rozwinięcia narzędzi obiektowych do kalkulacji śladu środowiskowego (Rebello et al., 2021).

W toku realizowania niniejszej rozprawy doktorskiej, na podstawie przeglądu najnowszych publikacji naukowych oraz przeprowadzonych własnych analiz LCA obiektów w Poznaniu (**Szulc et al., 2021**), jako kluczowy element śladu środowiskowego miejskich oczyszczalni ścieków zdiagnozowano ślad węglowy (ang. *carbon footprint* - CF), wskazywany w literaturze jako główny wskaźnik obciążenia środowiskowego tego sektora (Fang al., 2025). Zhang (2025) podkreśla, że kluczowym problemem środowiskowym pozostają emisje GHG z procesów biologicznych, które w znaczącym stopniu wpływają na ślad węglowy całego systemu, co potwierdzają także inne publikacje (Srivastava et al., 2025).

Zgodnie z definicją Międzyrządowego Zespołu ds. Zmian Klimatu (ang. *Intergovermental Panel of Climate Change* - IPCC), CF obejmuje sumaryczną emisję gazów cieplarnianych przeliczoną na jednostkę ekwiwalentu dwutlenku węgla (CO_2e), powstającą w wyniku bezpośredniego i pośredniego działania danego obiektu lub systemu.

Mimo, że do roku 2021. ilość publikacji o tematyce analizy cyklu życia i śladu węglowego stale rosła (Rysunek 2 i 3), to w literaturze podkreśla się, że to właśnie emisje gazów cieplarnianych (ang. *greenhouse gases*) - w tym metanu (CH_4) i podtlenku azotu (N_2O) oraz dwutlenku węgla (CO_2) pochodzenia biogenicznego - stanowią najbardziej znaczący czynnik wpływający na całkowity bilans środowiskowy oczyszczalni, zarówno w ujęciu lokalnym, jak i globalnym (Zhang, 2025). Wspominane wyżej emisje mają charakter zarówno bezpośredni, związany z procesami biologicznymi (nitryfikacja, denitryfikacja, fermentacja osadów), jak i pośredni, na przykład wynikający z intensywnego zużycia energii elektrycznej w procesach napowietrzania, pompowania czy transportu osadów, a także zużyć chemii technologicznej wspierającej procesy (Li et al., 2025). Badania modelowe wskazują ponadto, że to właśnie CF najczęściej wyznacza ww. „gorące punkty" cyklu życia oczyszczalni, które w największym stopniu determinują końcowy wynik analizy LCA (Tang et al., 2025). Jak podkreślają Wang i in. (2025), tylko podejścia oparte na pełnym cyklu życia umożliwiają uchwycenie realnych emisji, w tym tych powiązanych z etapem eksploatacyjnym, który w infrastrukturze technicznej odpowiada za największy udział w całkowitym bilansie środowiskowym.



Rysunek 3. Ilość ze słowami kluczowymi „Carbon Footprint" oraz „Wastewater", „Life Cycle Assessment" oraz „Wastewater" [opracowanie własne na bazie Lens.org].

Brakuje obecnie narzędzia, które umożliwiałoby kompletną kalkulację CF, uwzględniając specyficzne warunki eksploatacji oczyszczalni danej OŚ w Polsce. Istniejące rozwiązania - jak pokazują badania - koncentrują się głównie na biologicznych i sezonowych aspektach pracy instalacji, pomijając potrzebę integracji danych środowiskowych i regulacyjnych w jednolitym modelu (Cui et al., 2023). Podobne wnioski wynikają z pracy Lozada-Castro i in. (2025), gdzie oceniano innowacyjne systemy oczyszczania pod kątem parametrów środowiskowych, wskazując na potrzebę nowoczesnych narzędzi obliczeniowych dla poprawy jakości raportowania i zgodności z regulacjami.

W Polsce problematyka CF OŚ dopiero zaczyna być podejmowana w literaturze naukowej, równolegle zyskując w praktyce inżynierskiej coraz większe znaczenie ze względu na konieczność raportowania emisji zgodnie z regulacjami unijnymi (Rysunek 4) takimi, jak Nowa Dyrektywa Ściekowa 2024/3019, Dyrektywa o Zrównoważonym Rozwoju 2022/2464 (ang. Corporate Sustainability Reporting Directive - CSRD) oraz zestaw standardów dot. ujawnień nt. zrównoważonego rozwoju (ang. *European Sustainability Reporting Standards* - ESRS). Dotychczasowe badania wskazują, że brak krajowych baz wskaźników oraz ograniczona dostępność danych operacyjnych stanowią kluczową barierę w dokładnym określaniu śladu węglowego w warunkach lokalnych (Rebello et al., 2021).

Rysunek 4. Wybrane, kluczowe zmiany legislacyjne w kontekście branży wodno-kanalizacyjnej [opracowanie własne].

Analiza literatury pokazuje, że w badaniach dotyczących śladu węglowego oczyszczalni ścieków powtarzają się pewne problemy. Najczęściej wymienianą trudnością jest brak pełnej standaryzacji metod obliczania emisji GHG dla obiektów OŚ, co skutkuje stosowaniem różnych granic kalkulacyjnych. W wielu przypadkach ogranicza się zakres obliczeń wyłącznie do emisji bezpośrednich, pomijając emisje pośrednie związane z transportem osadów czy produkcją reagentów chemicznych. Kolejną istotną luką jest niewystarczająca dokładność danych dotyczących procesów biologicznych - emisje CH_4 i N_2O są często trudne do uchwycenia, co prowadzi do niedoszacowania lub przeszacowania całkowitego CF (Srivastava et al., 2025).

Ponadto badania wskazują, że niewiele prac uwzględnia zmienność sezonową oraz różnorodność warunków eksploatacyjnych, które mają istotne znaczenie w emisjach GHG z procesów biologicznych (Cui et al., 2025). Wreszcie, problemem pozostaje niedostateczna integracja analiz śladu węglowego z innymi wskaźnikami środowiskowymi, co ogranicza możliwość pełnej oceny zrównoważenia technologii.

Geneza niniejszej pracy związana jest z obserwowaną luką metodologiczną pomiędzy prostymi, szacunkowymi kalkulatorami emisji GHG a potrzebą prowadzenia dokładnych, obiektowych analiz środowiskowych, wymaganych przez wyżej wspomniane, nowe regulacje UE. Aktualnie brakuje narzędzia, które umożliwiałoby kompletną i jednocześnie precyzyjną kalkulację śladu węglowego na poziomie technologicznym, z uwzględnieniem lokalnych danych operacyjnych, rzeczywistego miksu energetycznego oraz specyficznych warunków eksploatacyjnych polskich oczyszczalni ścieków.

W odpowiedzi na wyżej wspominane zdiagnozowane braki, w niniejszej rozprawie opracowano dedykowany oczyszczalniom ścieków autorski algorytm obliczania śladu węglowego, zgodny z wytycznymi i metodykami raportowania przyjętymi przez Unię Europejską. Algorytm ten został zaprojektowany tak, aby integrował szczegółowe dane technologiczne z zakresu zużycia energii, emisji procesowych i gospodarki osadami, umożliwiając przeprowadzenie wiarygodnych analiz porównawczych oraz wskazanie „gorących punktów" środowiskowych w kontekście emisji GHG. Co istotne, rozwiązanie to nie tylko wypełnia lukę między prostymi narzędziami kalkulacyjnymi a wymaganiami legislacyjnymi, ale także może stanowić praktyczne wsparcie dla operatorów oczyszczalni w procesach decyzyjnych, planowaniu modernizacji oraz w przygotowywaniu raportów ESG i dokumentów zgodnych z CSRD/ESRS.

Wspomniany algorytm został zasilony danymi empirycznymi - w tym danymi z kampanii pomiarowej przeprowadzonej w ramach niniejszej rozprawy w skali technicznej na Centralnej Oczyszczalni Ścieków w Poznaniu. Wyniki kalkulacji przeprowadzonej zgodnie z proponowanym algorytmem porównano następnie z ogólnodostępnymi kalkulatorami CF.

2. Tezy, cel i zakres pracy

Tezy

Teza 1. Ślad węglowy stanowi kluczową i dominującą składową śladu środowiskowego miejskiej oczyszczalni ścieków, co uzasadnia koncentrację analiz środowiskowych na emisjach gazów cieplarnianych jako głównym czynniku oddziaływania.

Teza 2. Kalkulacja śladu węglowego miejskiej oczyszczalni ścieków wymaga wykorzystania danych empirycznych oraz lokalnie dopasowanych wskaźników emisyjnych, szczególnie dla emisji procesowych podtlenku azotu (N_2O) i metanu (CH_4), gdyż stosowanie wartości literaturowych prowadzi do istotnych przeszacowań całkowitych emisji.

Teza 3. Opracowanie algorytmu obliczania śladu węglowego miejskich oczyszczalni ścieków umożliwiającego kalkulację i raportowanie wyników emisji zgodnie z unijnymi wymogami legislacyjnymi pozwoli na skuteczniejszą identyfikację dźwigni redukcyjnych emisji GHG dla OŚ.

Cel główny (CG)

Celem rozprawy doktorskiej pt. „Analiza śladu środowiskowego miejskiej oczyszczalni ścieków w koncepcji zrównoważonego rozwoju" było opracowanie autorskiej metody wyznaczania najważniejszej składowej śladu środowiskowego miejskich oczyszczalni ścieków, wskazanej w toku rozprawy jako ślad węglowy, z dużym naciskiem na możliwość praktycznego wykorzystania wniosków z analizy w celu opracowania dźwigni redukcyjnych śladu środowiskowego oczyszczalni wraz z szacunkiem ich efektywności.

Cele szczegółowe (CS)

Cel szczegółowy 1. Przeprowadzenie obliczeń i analizy śladu środowiskowego w oparciu o LCA dla Centralnej Oczyszczalni Ścieków dla m. Poznania. – **CS1**

Cel szczegółowy 2. Wskazanie najważniejszej składowej śladu środowiskowego dla analizowanej oczyszczalni ścieków miejskich. – **CS2**

Cel szczegółowy 3. Stworzenie wykazu metodologii obliczania śladu węglowego oczyszczalni ze wskazaniem stosowanych metod obliczeniowych wraz z analizą luk kalkulacyjnych. – **CS3**

Cel szczegółowy 4. Określenie rzeczywistych wskaźników bezpośredniej emisji gazów cieplarnianych z procesu osadu czynnego metodą empiryczną dla Centralnej Oczyszczalni Ścieków dla m. Poznania - przeprowadzenie kampanii pomiarowej w skali technicznej. – **CS4**

Cel szczegółowy 5. Stworzenie operacyjnego algorytmu wyznaczania śladu węglowego dla miejskich oczyszczalni ścieków zgodnego z wymogami legislacyjnymi UE. – **CS5**

Cel szczegółowy 6. Przeprowadzenie komparatywnych obliczeń śladu węglowego miejskiej oczyszczalni ścieków z użyciem empirycznych oraz literaturowych wskaźników emisji z procesu oczyszczania ścieków miejskich metodą osadu czynnego. – **CS6**

Zakres prac

Weryfikacja postawionych tez i osiągnięcie założonych celów zostały zrealizowane poprzez wnikliwe studia literaturowe oraz odpowiednio zaplanowane prace badawcze i analityczne. Całość prac podzielono na 13 etapów, które przedstawiono poniżej, z zaznaczeniem przyporządkowanych celów szczegółowych oraz publikacji cyklu, w których znajduje się opis wykonanych prac:

ETAP 1: Zapoznanie się z danymi literaturowymi z obszaru analiz śladu środowiskowego obiektów oczyszczania ścieków miejskich (OŚM): pozyskanie wiedzy z zakresu wykonywania analiz cyklu życia, przygotowanie wykazu danych niezbędnych do przeprowadzenia ww. analizy dla wybranej oczyszczalni ścieków – **A1**,

ETAP 2: Wybór oczyszczalni do analiz i dalszych badań,

ETAP 3: Pozyskanie zestawu szczegółowych danych rzeczywistych z Centralnej Oczyszczalni Ścieków dla miasta Poznania (COŚ), wstępna analiza danych – **A1**,

ETAP 4: Przeprowadzenie kompleksowej analizy cyklu życia dla COŚ, interpretacja wyników i składowych śladu środowiskowego obiektu –**CS1**, **CS2**, **A1**,

ETAP 5: Przegląd światowych metodologii, wytycznych i przewodników dotyczących kalkulacji emisji gazów cieplarnianych w procesach oczyszczania ścieków, analiza dostępnych baz wskaźników emisji: zdobycie wiedzy dotyczącej możliwości kalkulacji śladu węglowego OŚM, weryfikacja luk metodycznych oraz ew. braków pozyskanych dla COŚ danych, zebranie danych uzupełniających – **CS3**, **A2**, **A3**,

ETAP 6: Analiza trendów legislacyjnych Unii Europejskiej dotyczących wymagań prezentacji wpływu obiektów OŚM na środowisko – **CS5**, **A2**, **A3**,

ETAP 7: Opracowanie kompleksowego algorytmu kalkulacji śladu węglowego OŚM spełniającego wymagania UE z zakresu raportowania nt. zrównoważonego rozwoju wraz ze wskazaniem możliwości doskonalenia procesu kalkulacji – **CS5**, **A2**, **A3**,

ETAP 8: Studium przypadków – analiza technicznych aspektów wykonywania kampanii pomiarowych poziomów emisji gazów cieplarnianych celem pozyskania jednostkowych wskaźników emisji podtlenku azotu oraz metanu w literaturze światowej: pozyskanie informacji nt. przeprowadzenia badań w skali technicznej, przygotowanie wytycznych badawczych, projektu komory pomiarowej, wykonanie kosztorysu dla kampanii wraz z analizą możliwości finansowania za pomocą mechanizmów grantowych, przystąpienie do konkursów – **CS4**,

ETAP 9: Przeprowadzenie kampanii emisji gazów cieplarnianych na COŚ, kalkulacja oraz analiza otrzymanych wyników dot. wskaźników emisji – **CS4**, **A4**,

ETAP 10: Obliczenie śladu węglowego COŚ z użyciem opracowanego algorytmu (**A2**, **A3**) oraz pozyskanych wskaźników – **CS5**, **A4**,

ETAP 11: Ocena krytyczna – analiza porównawcza otrzymanych poziomów emisyjnych z wynikami uzyskanymi z ogólnodostępnych kalkulatorów śladu węglowego, wskazanie obszarów do doskonalenia – **CS5**, **A4**,

ETAP 12: Analiza możliwości zarządzania śladem węglowym OŚM celem osiągnięcia neutralności klimatycznej na podstawie przeglądu literaturowego – **A5**,

ETAP 13: Wyznaczenie dalszych kierunków badawczych z obszaru zarządzania emisjami gazów cieplarnianych z oczyszczalni ścieków**.**

3. Omówienie cyklu publikacji

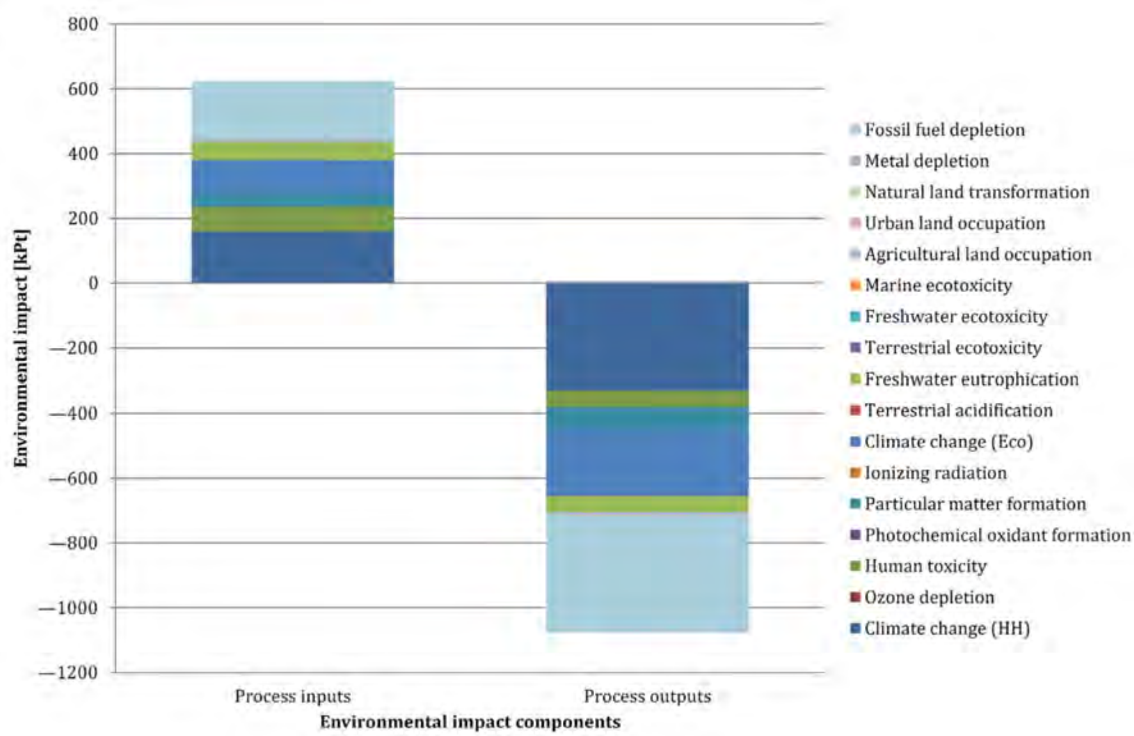
3.1. Analiza cyklu życia Centralnej Oczyszczalni Ścieków w Poznaniu

Pierwsza z publikacji wchodząca w skład cyklu badawczego (**A1**) stanowiła punkt wyjścia dla całej rozprawy. Jej celem było przeprowadzenie kompleksowej analizy cyklu życia odniesienie do celów szczegółowych, LCA, na podstawie kompletu rocznych danych rzeczywistych, dla Centralnej Oczyszczalni Ścieków (COŚ) obsługującej aglomerację poznańską. Dotychczasowy przegląd literatury wykazał brak pogłębionych analiz środowiskowych prowadzonych w skali technicznej dla oczyszczalni ścieków miejskich, które uwzględniałyby nie tylko oddziaływania środowiskowe związane z budową infrastruktury, ale także całkowity ślad środowiskowy generowany przez funkcjonowanie obiektu.

Analiza została przeprowadzona zgodnie z wytycznymi norm PN-EN ISO 14040:2009 oraz PN-EN ISO 14044:2009, a do oceny wpływu środowiskowego wykorzystano metodologię ReCiPe version 1.11, zarówno w wariancie Midpoint - koncentrującym się na poszczególnych kategoriach oddziaływania, takich jak emisje GHG, eutrofizacja, zakwaszenie czy zużycie zasobów, jak i Endpoint - umożliwiającym określenie oddziaływania na poziomie trzech głównych obszarów: zdrowia ludzkiego, oddziaływania na ekosystem oraz zasobów naturalnych. W tym celu wykorzystano oprogramowanie SimaPro zasilone bazą EcoInvent.

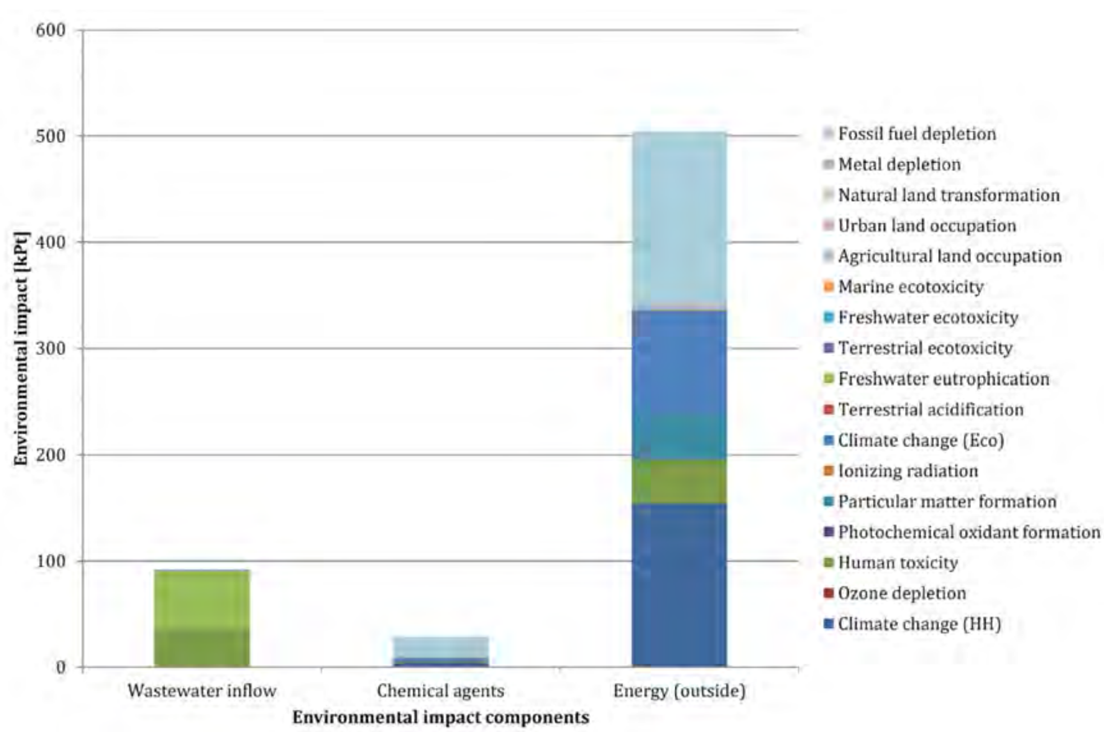
Uzyskano pełen obraz funkcjonowania oczyszczalni, a także określono, które elementy działalności obiektu odpowiadają za największe obciążenie środowiska.

Wyniki wskazały, że dominującym komponentem śladu środowiskowego Centralnej Oczyszczalni Ścieków jest oddziaływanie na klimat (Rysunek 5).



Rysunek 5. Szczegółowe profile oddziaływania COŚ - podział na wejścia i wyjścia z procesu - z wyróżnieniem kategorii oddziaływania [źródło: **A1**].

Odpowiadało ono za zdecydowaną większość całkowitego oddziaływania, a jego wielkość determinowana była głównie przez dwa czynniki: zapotrzebowanie energetyczne obiektu oraz charakterystykę krajowego miksu energetycznego (Rysunek 6).



Rysunek 6. Główne obciążenia środowiskowe COŚ - podział na kategorie wpływu [źródło: **A1**].

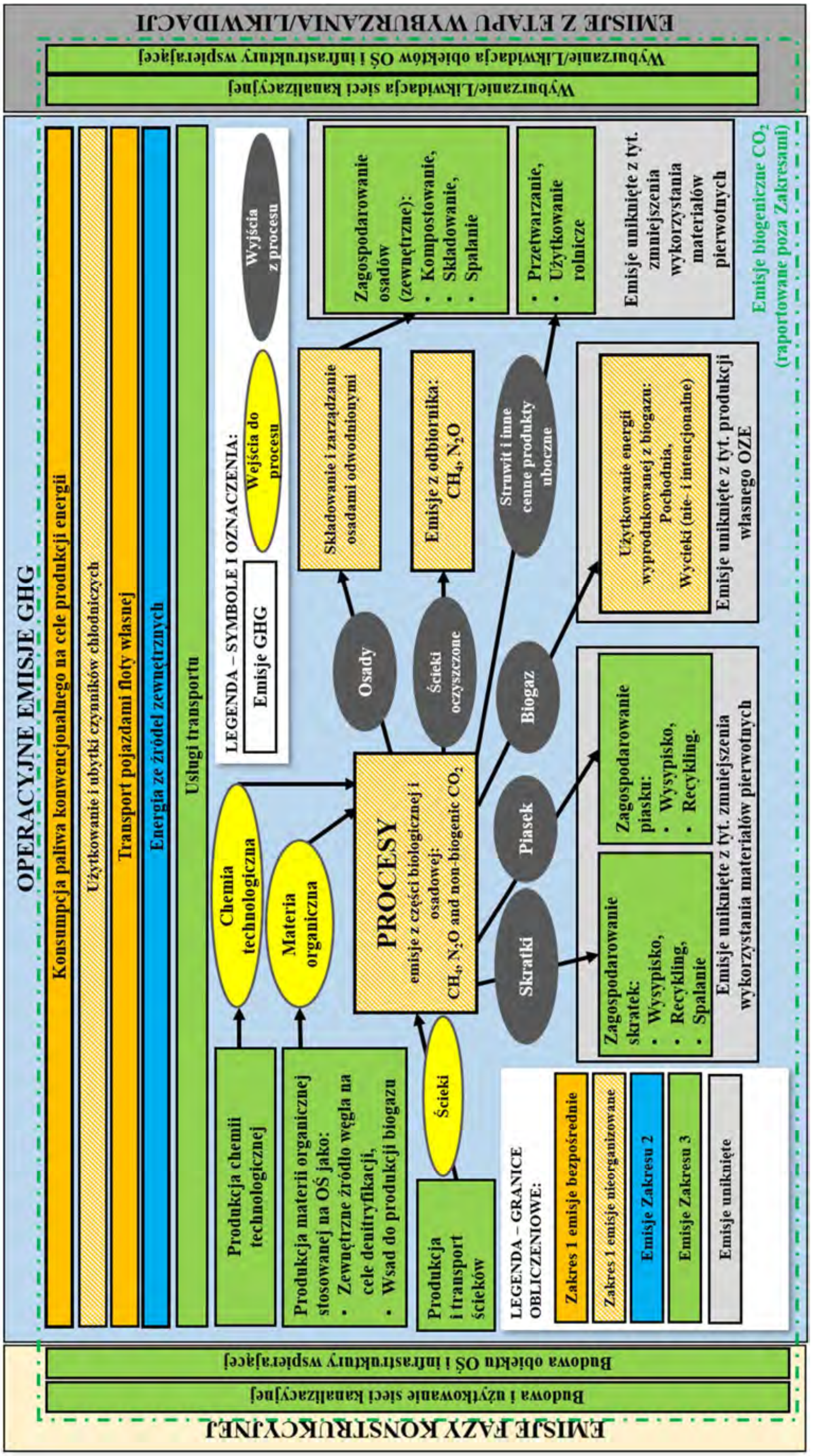
W analizowanym okresie (2019) struktura polskiego sektora energetycznego oparta była przede wszystkim na paliwach kopalnych, w szczególności na węglu kamiennym i brunatnym. Powodowało to, że każda jednostka energii elektrycznej zużytej w procesach technologicznych oczyszczalni przekładała się na znaczące emisje dwutlenku węgla i innych GHG.

Jednocześnie analiza LCA ujawniła istotne znaczenie emisji procesowych, związanych z przebiegiem procesów biologicznego oczyszczania ścieków i przeróbki osadów. Dotyczyło to zwłaszcza emisji N_2O oraz CH_4. Choć ich ilościowy udział w całkowitych emisjach był mniejszy niż emisji powiązanych z zużyciem energii, to z uwagi na bardzo wysoki potencjał cieplarniany (ang. *Global Warming Potential* – GWP) tych gazów tych miały one istotny wpływ na wynik końcowy.

W strukturze CF oczyszczalni udział emisji procesowych nie był w pełni znany i precyzyjnie oszacowany, gdyż w obiekcie nie prowadzono rutynowych pomiarów tychże gazów. W konsekwencji analizy LCA musiały opierać się na wskaźnikach emisyjnych pochodzących z literatury międzynarodowej.

Analiza LCA (**A1**) wykazała również, że mimo stosowania technologii odzysku energii z biogazu, COŚ wciąż pozostawał odbiorcą energii elektrycznej z sieci. Produkcja energii z biogazu nie była w stanie zrekompensować całkowitego zapotrzebowania energetycznego obiektu. Wskazywało to na ograniczony potencjał samowystarczalności energetycznej obiektów tego typu w analizowanych warunkach technologicznych i strukturalnych.

Dominującą składową śladu środowiskowego oczyszczalni stanowi CF, co uzasadniało koncentrację dalszych badań na tym aspekcie. Tradycyjne podejście do oceny funkcjonowania oczyszczalni, skupiające się na parametrach jakości ścieków oczyszczonych, okazało się nie być wystarczające do pełnej analizy oddziaływania na środowisko. Brak precyzyjnych danych dotyczących emisji procesowych N_2O i CH_4 ogranicza wiarygodność analiz i wskazuje na konieczność prowadzenia badań empirycznych oraz tworzenia własnych wskaźników emisyjnych dostosowanych do lokalnych warunków (Butterbah-Bahl et al., 2021). Wnioski te pozwoliły na zmapowanie źródeł bezpośrednich i pośrednich GHG (Rysunek 7) zgodnie z metodologią kalkulacji śladu węglowego GHG Protocol (WRI, 2014).



Rysunek 7. Struktura śladu węglowego OŚ zgonie z metodologią GHG Protocol [opracowanie własne].

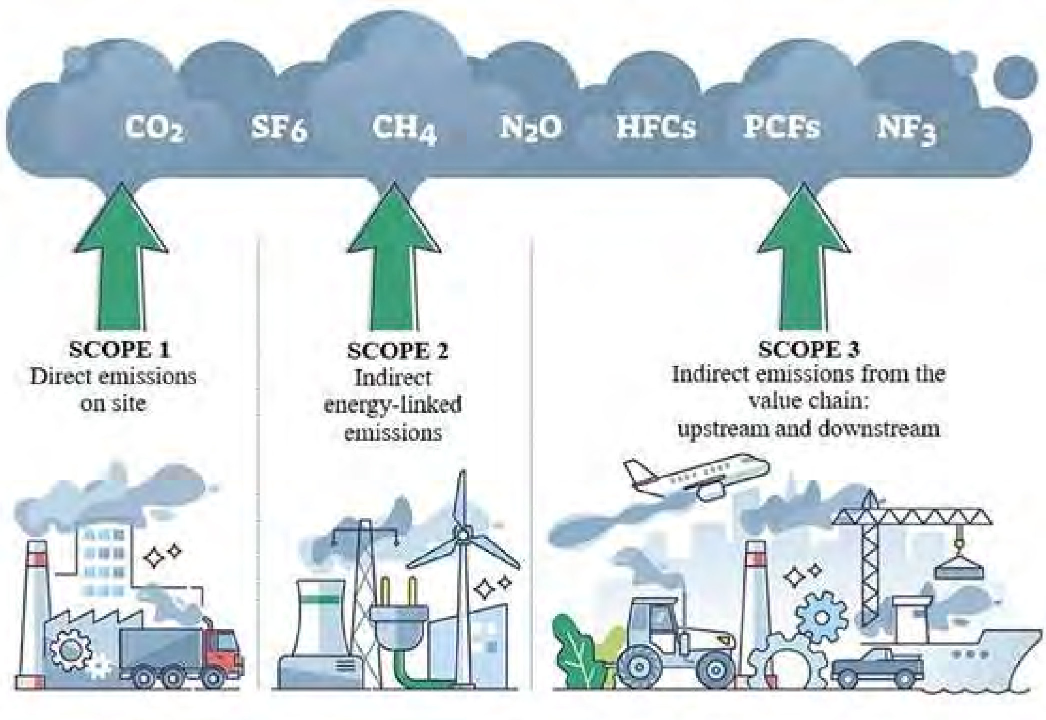
3.2. Opracowanie algorytmu kalkulacji śladu węglowego – Zakres 1

Opracowanie uniwersalnego algorytmu kalkulacji śladu węglowego dowolnej wybranej miejskiej oczyszczalni ścieków zostało przedstawione w dwuczęściowym cyklu publikacji: część pierwsza dedykowana emisjom bezpośrednim (**A2**), część 2 – emisjom pośrednim (**A3**). Tematyka, dotycząca Zakresu 1 (GHG Protocol) odpowiada na potrzebę zidentyfikowaną w analizie LCA poznańskiej oczyszczalni - konieczność stworzenia wytycznych pozwalających na spójną, powtarzalną i zgodną z międzynarodowymi wymogami legislacyjnymi ocenę emisyjności obiektów gospodarki wodno-ściekowej.

Opracowany algorytm został zbudowany po szczegółowym przeglądzie dostępnych metodologii obliczeniowych z całego świata (krajów Europy, Stanów Zjednoczonych czy Australii). W oparciu o wytyczne IPCC 2019: *Guidelines for wastewater treatment and discharge* uspójnione z wymaganiami metodologii GHG Protocol - najbardziej rozpowszechnionego na świecie standardu dotyczącego inwentaryzacji emisji gazów cieplarnianych - dostosowano go do wymogów Europejskich Standardów Raportowania nt. Zrównoważonego Rozwoju (ESRS), które w mają stać się obowiązkowe dla wybranych przedsiębiorstw sektora komunalnego w ramach implementacji dyrektywy CSRD i „stop the clock” (UE 2025/794). Dzięki temu od samego początku uwzględniono nie tylko aspekt naukowy, ale również praktyczny wymiar proponowanego rozwiązania.

Algorytm obejmował w pierwszej kolejności Zakres 1 (Z1). Dotyczy on emisji bezpośrednich, obejmujących przede wszystkim emisje procesowe N_2O i CH_4, które powstają w trakcie oczyszczania ścieków, począwszy od oczyszczania mechanicznego, a także biologicznego – np. metodą osadu czynnego, zarządzania gospodarką osadową oraz podczas fermentacji i magazynowania osadów. Uwzględniono również emisje niezorganizowane, takie jak planowe uwalnianie się CH_4 w wyniku przeglądów technicznych, jak i niekontrolowane emisje GHG z układów transportu i magazynowania biogazu.

Następnie przygotowano algorytm dla Zakresu 2 (Z2) oraz Zakresu 3 (Z3) (**A3**). Strukturę Zakresów zgodną z GHG Protocol przedstawiono na schemacie (Rysunek 8).



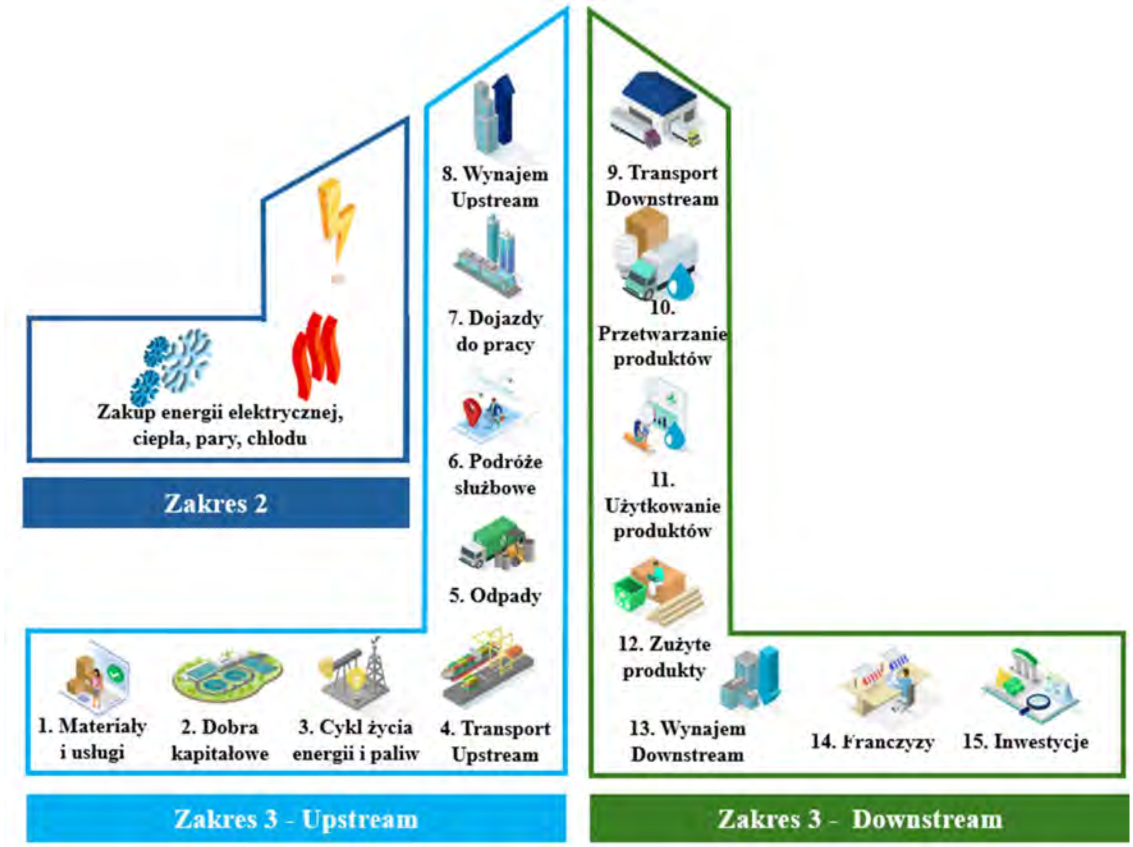
Rysunek 8. Zakresy śladu węglowego organizacji według GHG Protocol [źródło: **A4**].

Zastosowanie algorytmu pozwala na dokonanie pełnej inwentaryzacji emisji bezpośrednich oraz na ich agregację w postaci CF wyrażonego w ekwiwalencie dwutlenku węgla (CO_2e). Szczególną uwagę zwrócono na kwestię kompletności danych. W toku analiz zidentyfikowano bowiem, że w wielu przedsiębiorstwach zarządzających oczyszczalniami ścieków nie rejestruje się w sposób systematyczny części emisji niezorganizowanych ani danych, które mogłyby posłużyć do ich oszacowania – takich jak frakcje azotowe na etapie oczyszczania biologicznego. W konsekwencji wiele dostępnych dotychczas analiz środowiskowych opiera się na wskaźnikach literaturowych lub uproszczonych modelach, które nie odzwierciedlały specyfiki lokalnych obiektów.

Algorytm pozwala również na wprowadzenie elastyczności w doborze współczynników emisyjnych, co umożliwić ma zastosowanie danych empirycznych, jeśli takie były dostępne. W przypadku ich braku istniała możliwość zastosowania wartości literaturowych, jednakże z wyraźnym wskazaniem na poziom niepewności takich oszacowań. Takie podejście czyni narzędzie przydatnym zarówno w warunkach akademickich, jak i praktycznych, gdyż umożliwia jego zastosowanie także w tych oczyszczalniach, które nie prowadzą zaawansowanych pomiarów emisji, pozostawiając przestrzeń do systematycznego polepszana jakości danych i przeprowadzenia coraz to rzetelniejszych obliczeń CF. Dodatkowo algorytm obejmuje także analizę niepewności dla zastosowanych w kalkulacjach wskaźników emisji.

3.3. Rozszerzenie algorytmu kalkulacji śladu węglowego o Zakresy 2 i 3

Z2 obejmuje emisje pośrednie wynikające ze zużycia energii elektrycznej i cieplnej kupowanej z zewnętrznych źródeł, a Z3 podzielony jest zgodnie z GHG Protocol na określone Kategorie (Rysunek 9).



Rysunek 9. Poszczególne Kategorie Zakresu 3 według GHG Protocol [opracowanie własne].

Obejmują one pośrednie emisje GHG związane z działalnością oczyszczalni, ale powstające poza jej granicami. W ramach algorytmu (**A3**) przygotowano wytyczne do przeprowadzenia analizy istotności służącej do wskazania najbardziej istotnych kategorii Zakresu 3 dla danego przypadku. Do głównych kategorii zaliczono emisje wynikające z produkcji i transportu chemikaliów stosowanych w procesie oczyszczania ścieków (np. koagulantów, polielektrolitów, środków dezynfekcyjnych), emisje związane z gospodarką osadami (w tym transportem i procesami końcowego unieszkodliwiania), a także emisje generowane przez zużycie materiałów eksploatacyjnych i pomocniczych.

Podkreślono, że dane potrzebne na cele kalkulacji emisji z Zakresu 1 i 2, które, mimo zdarzających się problemów w pozyskaniu kompletnego ich zestawu, nadal są informacjami istotnie łatwiejszymi do zebrania niż w przypadku tych dotyczących łańcucha wartości organizacji. Zidentyfikowano luki w raportowaniu, w tym brak ujednoliconych wskaźników emisyjnych dla produkcji chemikaliów czy transportu i zagospodarowania osadów. W związku z tym autorzy zaproponowali strategię poprawy kompletności danych. Obejmowała ona prowadzenie lokalnych inwentaryzacji w przedsiębiorstwach, ujednolicenie metod raportowania oraz wykorzystywanie międzynarodowych baz danych, takich jak EcoInvent czy DEFRA, jako źródła wartości zastępczych.

Znaczenie włączenia Zakresu 3 do algorytmu kalkulacji śladu węglowego (**A3**) należy rozpatrywać w kilku wymiarach. Po pierwsze, z naukowego punktu widzenia pozwoliło to na stworzenie kompleksowej metodyki obejmującej wszystkie trzy Zakresy emisji, zgodnie z zasadami GHG Protocol. Po drugie, z punktu widzenia praktycznego dało przedsiębiorstwom zarządzającym OŚ narzędzie do bardziej rzetelnej oceny ich całkowitego oddziaływania środowiskowego, co jest istotne w świetle nadchodzących obowiązków raportowych wynikających z dyrektywy CSRD i standardów ESRS. Po trzecie, umożliwiło identyfikację obszarów, w których działania redukcyjne mogą przynieść największe korzyści - wskazało, że oprócz modernizacji procesów technologicznych, istotne znaczenie ma również optymalizacja logistyki dotyczącej zagospodarowania osadów czy dobór mniej emisyjnych środków chemicznych o analogicznym działaniu.

3.4. Pozyskanie wskaźników empirycznych i wykorzystanie opracowanego algorytmu

Aby pozyskać rzeczywiste wskaźniki emisji GHG, przeprowadzono kampanię pomiarową emisji podtlenku azotu (N_2O) i metanu (CH_4) w COŚ w Poznaniu (w skali technicznej) z zamiarem wykorzystania ich do kalkulacji śladu węglowego obiektu za pomocą opracowanego algorytmu (**A2**, **A3**, **A4**). Dotychczas stosowane metody kalkulacji CF OŚ opierały się na wartościach literaturowych i wskaźnikach uśrednionych. Choć takie podejście ma charakter uniwersalny i pozwala na szybkie oszacowanie emisji, to nie uwzględnia lokalnych uwarunkowań technologicznych ani warunków pracy poszczególnych obiektów.

W celu opracowania ww. empirycznych wskaźników emisji CH_4 oraz N_2O przeprowadzono kompleksową kampanię pomiarową w warunkach rzeczywistych, w skali technicznej (**A4**).

Badana w niniejszej pracy Centralna Oczyszczalnia Ścieków zlokalizowana jest w Koziegłowach, na prawym brzegu rzeki Warty, w bezpośrednim sąsiedztwie Poznania - głównego ośrodka aglomeracji wielkopolskiej. Oczyszczalnia przetwarza średnio około 100 000 m^3 ścieków na dobę, pochodzących z Poznania oraz okolicznych miejscowości, co odpowiada równoważnej liczbie 1 000 000 mieszkańców w przeliczeniu na jednostkowe obciążenie pięciodniowym biochemicznym zapotrzebowaniem na tlen (BZT_5). Technologia zastosowana w obiekcie umożliwia uzyskanie jakości ścieków oczyszczonych spełniającej wymagania dla zrzutu do rzeki Warty i opiera się na procesach biologicznego usuwania związków biogennych (ang. *Biological Nutrient Removal* - BNR) realizowanych metodą osadu czynnego w bioreaktorach skonstruowanych w systemie zmodyfikowanego Bardenpho. Komory te mają łączną objętość 6 × 25 500 m^3 i współpracują z wtórnymi osadnikami radialnymi o pojemności 6 × 9 500 m^3. Przed etapem biologicznym ścieki poddawane są wstępnemu oczyszczaniu mechanicznemu, obejmującemu zatrzymywanie skratek i separację piasku oraz osadu wstępnego. Zatrzymane frakcje stałe kierowane są najpierw do napowietrzanych zhermetyzowanych piaskowników poziomych, a następnie do osadników wstępnych o łącznej objętości 4 × 65,00 m^3, które także zostały zhermetyzowane.

Produktem ubocznym procesów oczyszczania są osady ściekowe: osad wstępny oraz osad nadmierny. Ze względu na wysoki potencjał energetyczny, oba rodzaje osadów są łączone w proporcji 2:1 i kierowane do sześciu komór fermentacyjnych beztlenowych o objętości jednostkowej 4 960 m^3. Proces prowadzony jest w warunkach mezofilowych, przy temperaturze około 35 °C i średnim czasie retencji osadu wynoszącym 26 dni, co pozwala na stabilną, dobową produkcję biogazu. Uzyskany surowy biometan, o średnim stężeniu 64% zawartości w biogazie poddawany jest wstępnej obróbce, a następnie wykorzystywany do produkcji energii elektrycznej i cieplnej w agregatach prądotwórczych.

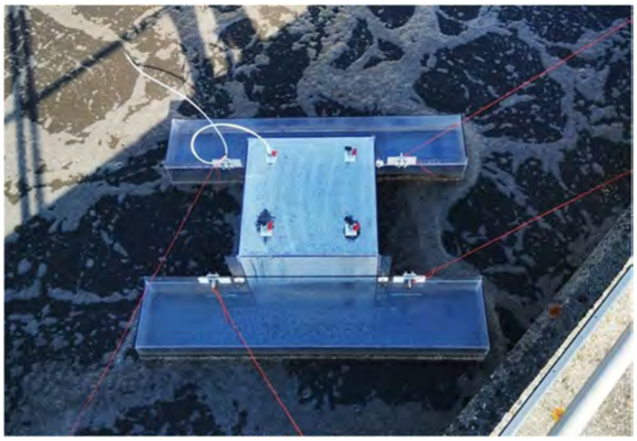
Przefermentowany osad, o średniej zawartości suchej masy na poziomie około 3%, kierowany jest do instalacji odgazowywania w zbiornikach o łącznej pojemności 6 500 m³, a następnie odwadniany w wirówkach, co pozwala na uzyskanie suchej masy na poziomie 23%. Ostateczna utylizacja odwodnionego osadu, wraz z innymi produktami ubocznymi oczyszczania (takimi jak piasek z piaskowników oraz skratki poddane kompresji), realizowana jest poza terenem oczyszczalni. Dodatkowo, aby zapewnić stabilność procesu usuwania fosforu w okresie zimowym, do ścieków wprowadza się okresowo roztwór siarczanu żelaza(III) (PIX113), w ilości gwarantującej utrzymanie wymaganego poziomu strącania fosforanów. W Tabeli 1 zestawiono uśrednione parametry charakteryzujące prace oczyszczalni podczas kampanii pomiarowej.

Tabela 1. Charakterystyka COŚ - wartości średnie roczne (2021 - rok kampanii pomiarowej).

Przepływ dobowy	*m^3/d*	92,291 ± 5,636
DOPŁYW		
ChZT	*gO_2/m^3*	1,045±55
Azot ogólny	*gN/m^3*	107±7
Fosfor całkowity	*gP/m^3*	15±1
Zawiesina ogólna	*g/m^3*	495±46
ODPŁYW		
ChZT	gO_2/m^3	46.5±2.9
Azot ogólny	gN/m^3	8.6±0.2
Fosfor całkowity	gP/m^3	0.3±0.1
Zawiesina ogólna	g/m^3	2.6±0.1

Należy wspomnieć, że prace związane z częścią badawczą niniejszej rozprawy, zostały sfinansowane w ramach projektu grantowego Integrated system for SImultaneous Recovery of Energy, organics and Nutrients and generation of valuable products from municipal wastewater - SIREN, realizowanego z mechanizmu finansowego tzw. Grantów Norweskich.

Już na etapie przygotowywania wniosku projektowego opracowano szczegółowe wytyczne dotyczące pomiarów, obejmujące zarówno merytoryczne podstawy zastosowanej metody badawczej - spektroskopii FTIR - wraz z wymaganym zakresem pomiarowym, jak i kompletny plan badań, harmonogram poborów oraz rozwiązania techniczne umożliwiające przechwytywanie mieszaniny gazów z powierzchni bioreaktora COŚ. Na potrzeby kampanii zaprojektowano i skonstruowano specjalistyczną komorę pomiarową (Rysunek 10a i b). Została ona umieszczona na powierzchni mieszaniny osadu czynnego i ścieków oczyszczonych w komorze tlenowej wybranego reaktora biologicznego. Całość instalacji unieruchomiono za pomocą lin, a gazy wydzielające się z powierzchni bioreaktora kierowano wężem wykonanym z materiału obojętnego chemicznie do analizatora FTIR.



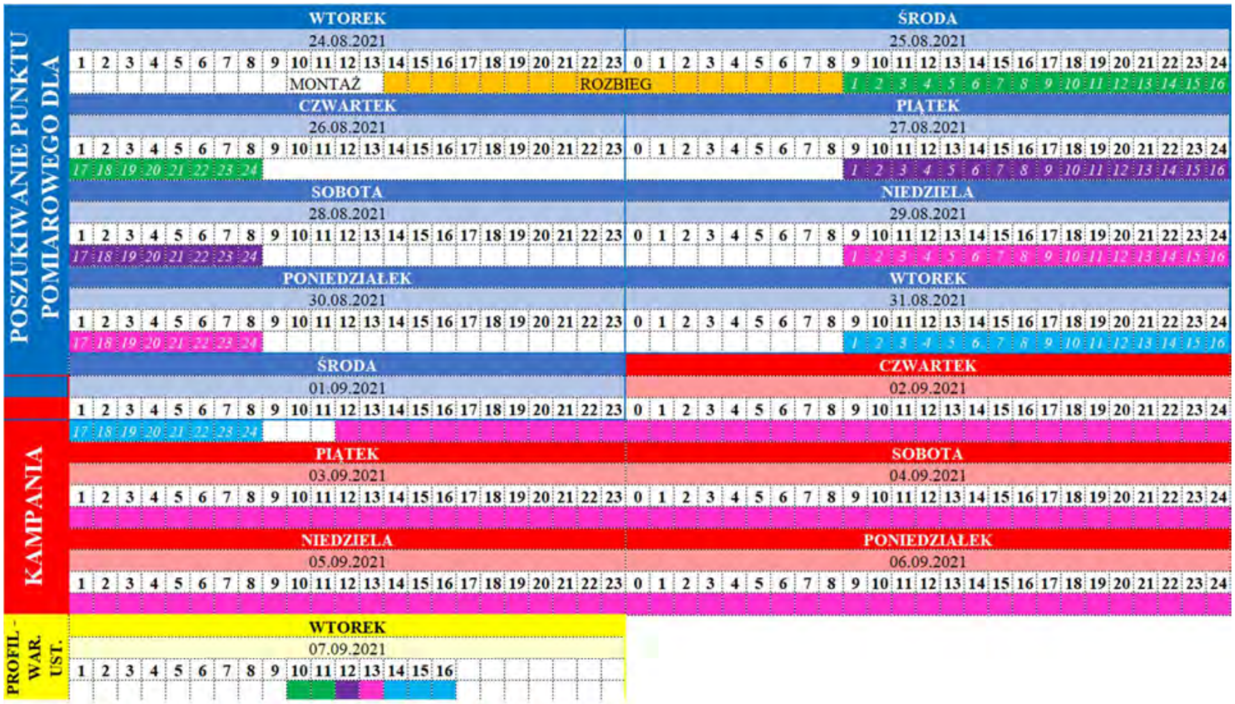
Rysunek 10a, 10b. Zdjęcie komory pomiarowej (ang. *flux chamber*). Komora o powierzchni roboczej 50 x 50 [cm] została przymocowana do dwóch pływaków. Cała konstrukcja została wykonana z poliwęglanu [zbiory własne].

Pomiary prowadzono przy użyciu aparatu GASMET DX4000 z oprogramowaniem CALCMET (Rysunek 11). System umożliwiał rejestrację wyników w czasie rzeczywistym. Kampania została zorganizowana w trybie ciągłym (24 h/dobę) w wyznaczonych punktach pomiarowych na powierzchni bioreaktora COŚ (Rysunek 12), zgodnie z wcześniej opracowanym harmonogramem (Rysunek 13). Analizy rozpoczęto od montażu aparatury i etapu tzw. rozbiegu, trwającego około 17 godzin, w trakcie którego zespół badawczy zapoznawał się z obsługą i stabilizacją aparatury (Rysunek 13 - etap: rozbieg). Następnie zrealizowano pierwszą fazę badań trwającą nieco ponad 7 dób, której celem była ocena poziomów emisji CH_4 i N_2O w różnych strefach komory nitryfikacji, a w konsekwencji identyfikacja punktu docelowego dla dalszych pomiarów (Rysunek 13 - dni 25.08÷01.09.2021) oraz okres walidacyjny w warunkach ustalonych (Rysunek 13 - 07.09.2021).

Rysunek 11. Zdjęcie aparatury pomiarowej FTIR: GASMET z kondycjonerem i laptopem z oprogramowaniem CALCMET poliwęglanu [zbiory własne].

Właściwa kampania pomiarowa, trwająca cztery dni, została przeprowadzona w punkcie pomiarowym P3 (Rysunek 12), który wytypowano na podstawie wyników analiz wstępnych.

Rysunek 12. Umiejscowienie punktów pomiarowych, w których montowano komorę - jeden z bioreaktorów (BR) (obiekt 10.5) COŚ [opracowanie własne].



Rysunek 13. Harmonogram badań - kolory tabeli odpowiadają oznaczeniem kolejnych punktów pomiarowych (Rys. 12). Zaznaczono kolejne godziny doby, w których komora znajdowała się w określonej punktem lokalizacji [opracowanie własne].

Oprócz danych dotyczących emisji GHG, w trakcie kampanii gromadzono także dodatkowe informacje ilościowe i jakościowe obejmujące cały obiekt COŚ. Zawierały one m.in. parametry technologiczne pracy poszczególnych etapów oczyszczania ścieków takie, jak dane dotyczące ładunku dopływającego ścieku oraz wyniki równoległych analiz laboratoryjnych. Na Rysunkach 8 i 9 przedstawiono kolejne punkty pomiarowe oraz lokalizację tymczasowych laboratoriów polowych, zorganizowanych na potrzeby kampanii badawczej.

W ramach projektu SIREN przygotowywano także model COŚ za pomocą oprogramowania GPS-X. W celu odpowiedniej kalibracji ww. modelu kolekcjonowano także dane z gospodarki osadowej (Rysunek 14 - punkty 11÷15).

Rysunek 14. Umiejscowienie poszczególnych punktów pomiarowych i poborowych próbek - z lotu ptaka [opracowanie – na cele projektu SIREN].

Informacje dotyczące reżimu poboru prób oraz wykonywanych analiz zestawiono w Tabeli 2. Zawarto w niej szczegółowe dane dotyczące harmonogramu i częstotliwości poborów, a także wykaz zastosowanych testów kuwetowych wykorzystujących metodę spektrofotometrii jonowej do oznaczeń jakościowych. Ponadto w tabeli przedstawiono zestawienie norm i procedur analitycznych stosowanych w laboratorium, które stanowiły podstawę walidacji uzyskanych wyników. Analizy walidacyjne wykonywano w akredytowanym laboratorium.

Tabela 2. Wykaz dodatkowych badań wykonywanych oraz danych agregowanych na cele opracowania wskaźników emisji GHG dla COŚ.

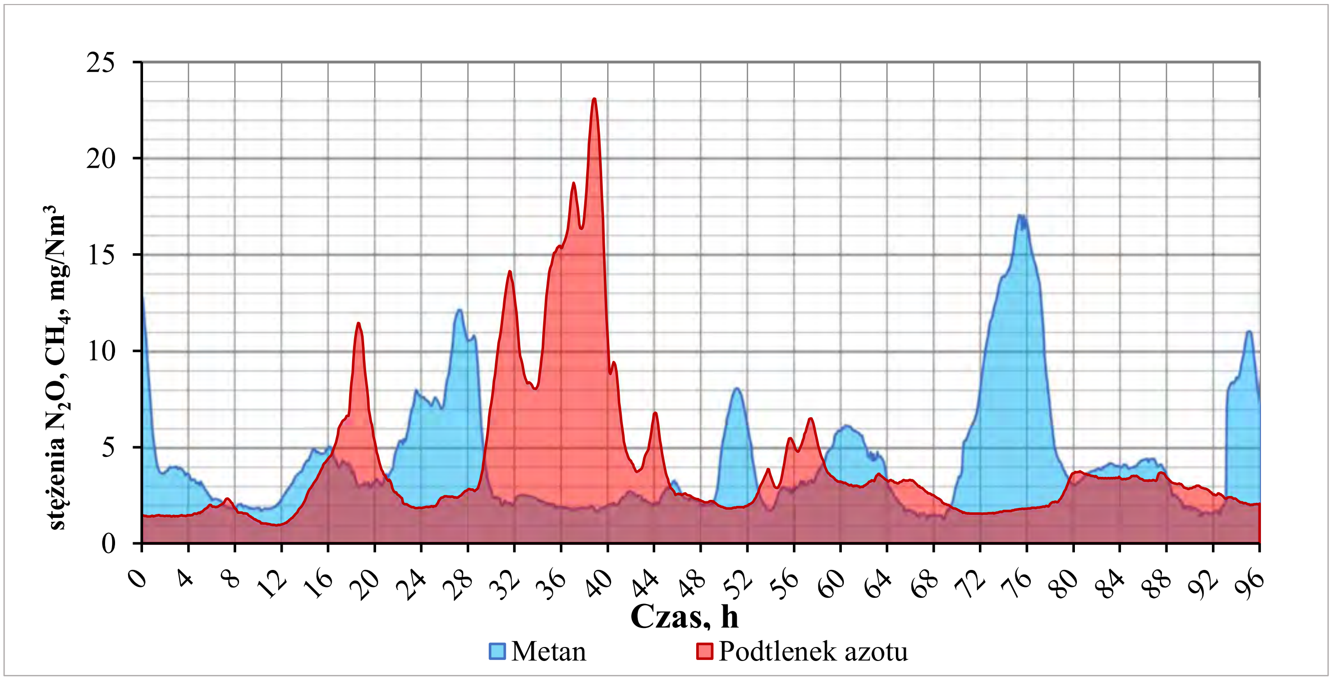
PARAMETR	JEDNOSTKA	METODA POMIAROWA	CZĘSTOTLIWOŚĆ POMIAROWA
DOPŁYW DO BIOREAKTORA, DOPŁYW DO COŚ, ODPŁYW Z COŚ			
Frakcje azotowe (TKN - azot Kiejdahla, N-NH_4, N-NO_2, N-NO_3)	*mg/L*	testy kuwetowe Hach (LCK304, 338, 340, 341), testy laboratoryjne (PN-C-04576-4:1994; PN-82/C-04576.08; PN-EN 26777:1999; PN-EN 25663:2001)	*średnia dobowa - 24 próbki uśrednione względem przepływu*
ChZT całkowite i rozpuszczone rChZT	*mg/L*	testy kuwetowe Hach (LCK314, 514, 914), testy laboratoryjne ISO 15705	*średnia dobowa - 24 próbki uśrednione względem przepływu*
Zawiesina ogólna, zawiesina organiczna	*mg/L*	testy laboratoryjne - metoda wagowa	*średnia dobowa - 24 próbki uśrednione względem przepływu*
Fosfor ogólny i fosforany P_{tot} and P-PO_4	*mg/L*	testy kuwetowe Hach (LCS349, LCK348, 350), testy laboratoryjne (PN-EN ISO 6878:2006 pkt 7 +Ap1:2010+ AP2:2010; PN-EN ISO 6878:2006 pkt 4 +Ap1:2010+ AP2:2010; PN-C-04537-7:1998)	*średnia dobowa - 24 próbki uśrednione względem przepływu*
Przepływ	*m^3/h*	SCADA	*pomiar ciągły - zapis co 5 minut*
CHARAKTERYSTYKA BIOREAKTORA			
Frakcje azotowe (TKN, N-NH_4, N-NO_2, N-NO_3)	*mg/L*	SCADA oraz testy kuwetowe Hach (LCK238, 339, 340, 341,1054), testy laboratoryjne (PN-C-04576-4:1994; PN-82/C-04576.08; PN-EN 26777:1999; PN-EN 25663:2001)	*ciągły (SCADA) oraz analiza laboratoryjna co 2 godziny*
ChZT całkowite i rozpuszczone rChZT	*mg/L*	testy kuwetowe Hach (LCK314, 914), testy laboratoryjne ISO 15705	*analiza laboratoryjna co 2 godziny*
Zawiesina ogólna, zawiesina organiczna	*mg/L*	testy laboratoryjne	*analiza laboratoryjna raz na dobę*
Fosforany P-PO_4	*mg/L*	testy kuwetowe Hach (LCS349, LCK350), testy laboratoryjne (PN-EN ISO 6878:2006 pkt 4 +Ap1:2010+ AP2:2010)	*analiza laboratoryjna co 2 godziny*
Poziomy recyrkulacji wewnętrznej i zewnętrznej	*% of Q, m^3/h*	SCADA	*pomiar ciągły - zapis co 5 minut*
Tlen rozpuszczony (osobno w każdym punkcie ustawienia komory)	*mg/L*	SCADA	*pomiar ciągły - zapis co 5 minut*
Temperatura	*°C*	SCADA	*pomiar ciągły - zapis co 5 minut*
Przepływ powietrza	*m^3/h*	SCADA	*pomiar ciągły - zapis co 5 minut*

Wyniki kampanii pomiarowej przeprowadzonej na COŚ w postaci zależności stężenia (mg gazu w przeliczeniu na Nm^3) GHG od czasu przedstawiono na wykresie – Rysunek 15. Wyniki pomiarów wskazują na wyraźną przeciwstawną korelację między wartościami CH_4 i N_2O: w okresach, gdy emisje metanu osiągały maksimum, poziomy podtlenku azotu spadały do wartości minimalnych i odwrotnie. Zjawisko to odzwierciedla dominację odmiennych mechanizmów odpowiedzialnych za emisje obu gazów. Metan jest generowany głównie w procesach beztlenowych, podczas gdy N_2O powstaje w wyniku intensywnych przemian azotowych (nitryfikacji i denitryfikacji), które przebiegają w środowisku tlenowym lub przy zmiennych warunkach dostępności tlenu.

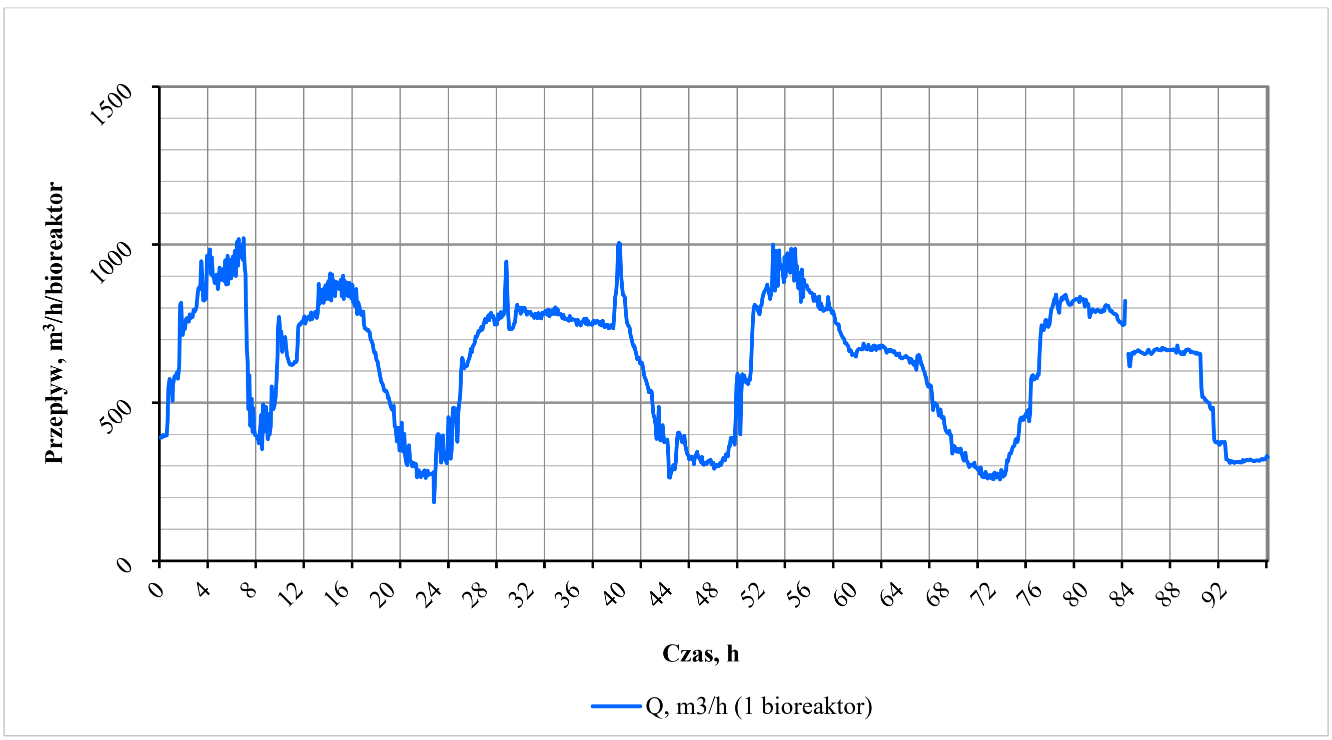
Zaobserwowano również okresy względnej stabilizacji stężeń obu gazów, trwające od 6. do 12. godziny kampanii, następnie pomiędzy 64. a 70. godziną oraz w końcowej fazie pomiarów - od 80. do 92. godziny. Stabilność ta może być interpretowana jako rezultat ustabilizowanych warunków technologicznych, w tym równomiernego dopływu ścieków i utrzymania odpowiednich parametrów tlenowych w reaktorach biologicznych.

Maksymalne stężenia CH4 występowały cyklicznie, średnio co 24 ± 2 godziny, najczęściej około godziny 13:00. Zjawisko to wskazuje na powiązanie emisji metanu z dobowym rytmem pracy oczyszczalni oraz zmianami obciążenia hydraulicznego (wykres – Rysunek 16). Interesującym aspektem jest obserwacja, że szczytom emisji CH_4 towarzyszył wcześniejszy spadek przepływu, a następnie - ze średnim opóźnieniem od 1 do 3 godzin - gwałtowny wzrost emisji metanu. Może to sugerować, że niższy przepływ sprzyja krótkotrwałemu rozwojowi warunków beztlenowych, które intensyfikują procesy metanogenezy.

W przypadku N_2O zależność była odmienna: emisje tego gazu były ściśle skorelowane z przepływem ścieków, a co za tym idzie i ładunków azotowych, bez zauważalnych opóźnień czasowych. Wynik ten wskazuje, że procesy generujące N_2O (przede wszystkim nitryfikacja i częściowa denitryfikacja) reagują bezpośrednio na zmiany dopływu ładunku azotowego i warunki operacyjne, takie jak stężenie tlenu rozpuszczonego. Tym samym N_2O okazuje się gazem bardziej wrażliwym na krótkotrwałe wahania parametrów technologicznych niż CH_4, którego emisje charakteryzują się większą inercją i cyklicznością.



Rysunek 15. Wyniki stężeń gazów - CH_4 i N_2O [mg/Nm3] podczas 96-godzinnej kampanii pomiarowej COŚ.



Rysunek 16. Zmienność napływu ścieków [m^3/h] na pojedynczy bioreaktor podczas 96-godzinnej kampanii pomiarowej COŚ (przerwa w 85. h kampanii spowodowana chwilową awarią przepływomierza).

W Tabeli 3 zestawiono wyniki badań dla GHG uzyskane podczas właściwej, 96-godzinnej kampanii pomiarowej (Rysunek 13). Średnie stężenie CH_4 w ujmowanej znad powierzchni miksu ścieków i osadu czynnego mieszaninie gazów wynosiło 4,447 mg/Nm^3, a wartości graniczne mieściły się w przedziale 1,303 (minimum) oraz 17,071 mg/Nm^3 (maksimum). W przypadku N_2O średnia koncentracja osiągnęła 8,664 mg/Nm^3, przy wartościach minimalnych 1,826 mg/Nm^3 i maksymalnych: 45,387 mg/Nm^3. Wartości te wskazują, że choć metan odgrywa rolę w strukturze emisji z komór biologicznych, podtlenek azotu stanowi dominujący gaz cieplarniany w badanym obiekcie.

Tabela 3. Podsumowanie wyników pomiarów stężeń - emisje CH_4 oraz N_2O [mg/Nm^3 gazu ujmowanego] podczas 96-godzinnej kampanii pomiarowej.

	Stężenia GHG	
	CH_4	N_2O
	mg/Nm^3	
Minimalne	1,303	1,826
Maksymalne	17,071	45,387
Średnia	**4,447**	**8,664**
Mediana	3,688	5,931
Odchylenie standardowe	3,079	7,719

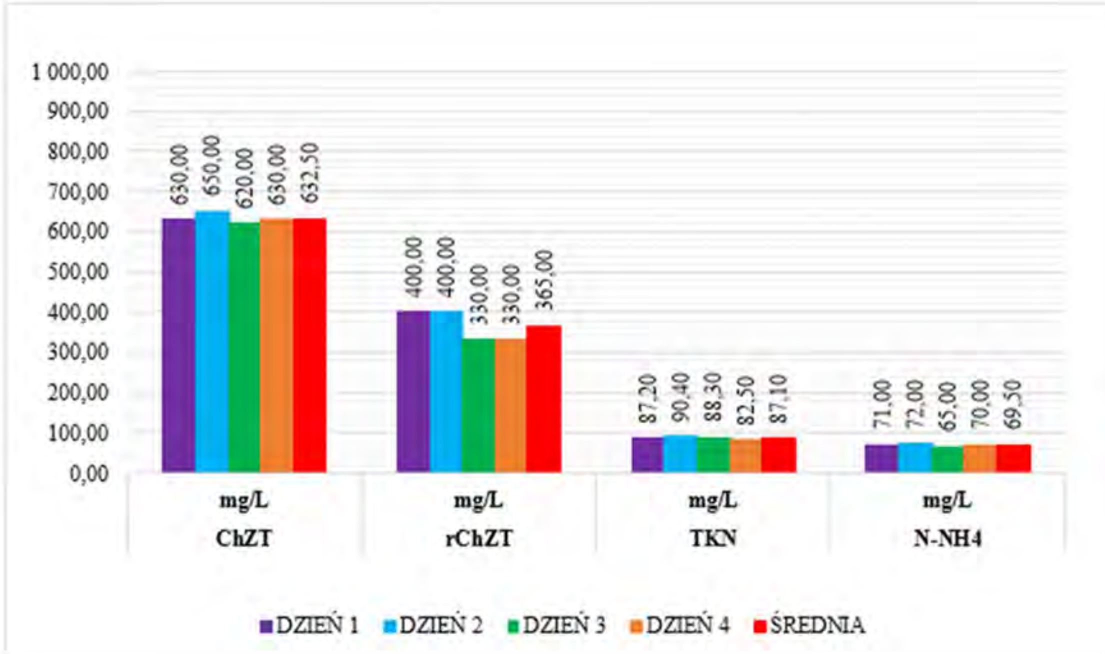
Interpretacja wyników kampanii pomiarowej wskazuje, że emisje GHG w dużej mierze zależą od parametrów eksploatacyjnych obiektu. Kluczową rolę odgrywała zmienność przepływu ścieków, poziomu tlenu rozpuszczonego (ang. *Dissolved Oxygen* - DO) oraz stabilność recyrkulacji: zarówno wewnętrznej, jak i zewnętrznej. Analiza zestawiona w Tabeli 4 potwierdza, że w okresach stabilnego dopływu i utrzymywania DO na poziomie zadanym (1,0 mg O_2/L), obserwowano spadek krótkotrwałych fluktuacji N_2O. Z kolei wszelkie zakłócenia w dopływie ścieków - szczególnie nagłe wzrosty obciążenia ładunkiem azotowym - prowadziły do gwałtownego wzrostu emisji tego gazu.

Warto podkreślić, że emisje metanu nie wykazywały bezpośredniej zależności od DO, co sugeruje, że ich źródłem były przede wszystkim procesy mikrobiologiczne związane z mikrostrefami beztlenowymi w osadzie czynnym. Wysoka cykliczność emisji CH_4, obserwowana niezależnie od stabilizacji parametrów eksploatacyjnych, wskazuje, że proces metanogenezy w warunkach komór tlenowych przebiega na stosunkowo niskim, lecz powtarzalnym poziomie. To z kolei dowodzi, że konstrukcja reaktora i sposób montażu rusztów napowietrzających gwarantuje dobre napowietrzenie, minimalizując ilość niepożądanych w strefie nitryfikacji obszarów niedotlenionych.

Tabela 4. Średnie dobowe stężenia ścieków oczyszczonych mechanicznie dopływających do bioreaktora.

	ChZT	rChZT	TKN	$N\text{-}NH_4$
	mg/L	*mg/L*	*mg/L*	*mg/L*
DZIEŃ 1	630,00	400,00	87,20	71,00
DZIEŃ 2	650,00	400,00	90,40	72,00
DZIEŃ 3	620,00	330,00	88,30	65,00
DZIEŃ 4	630,00	330,00	82,50	70,00
ŚREDNIA	**632,50**	**365,00**	**87,10**	**69,50**

Dane dotyczące jakości ścieków zestawiono w Tabeli 5 oraz, graficznie, na wykresie – Rysunek 17. Średnie stężenie ChZT w okresie kampanii dopływie wynosiło 632,5 mg/L, a w odpływie 46,5 mg/L, co odpowiadało redukcji na poziomie ponad 92%. Podobnie, skuteczność usuwania jonów amonowych ($N\text{-}NH_4$) sięgnęła 98%. Jednak całkowita redukcja azotu ogólnego wyniosła jedynie 71,2%, co oznacza, że w odpływie dominującą frakcją azotu były azotany ($N\text{-}NO_3$), stanowiące blisko połowę całkowitej zawartości związków azotu.
Wyniki te są istotne z punktu widzenia emisji GHG. Niedokończona denitryfikacja sprzyja powstawaniu podtlenku azotu jako produktu pośredniego. To właśnie w okresach zwiększonego udziału azotanów w odpływie notowano największe wahania emisji N_2O. Potwierdza to, że gospodarka azotowa oczyszczalni - mimo wysokiej skuteczności w redukcji amoniaku - pozostaje głównym źródłem emisji tego gazu.
Warto dodać, że obserwowana wysoka efektywność usuwania zanieczyszczeń organicznych i azotu amonowego nie przekładała się wprost na niskie emisje GHG. Wręcz przeciwnie, okresy intensywnego usuwania biogenów wiązały się z większym udziałem procesów mikrobiologicznych odpowiedzialnych za emisje N_2O. Zjawisko to dowodzi, że poprawa jakości ścieków w odpływie nie zawsze jest jednoznaczna z redukcją oddziaływania klimatycznego.

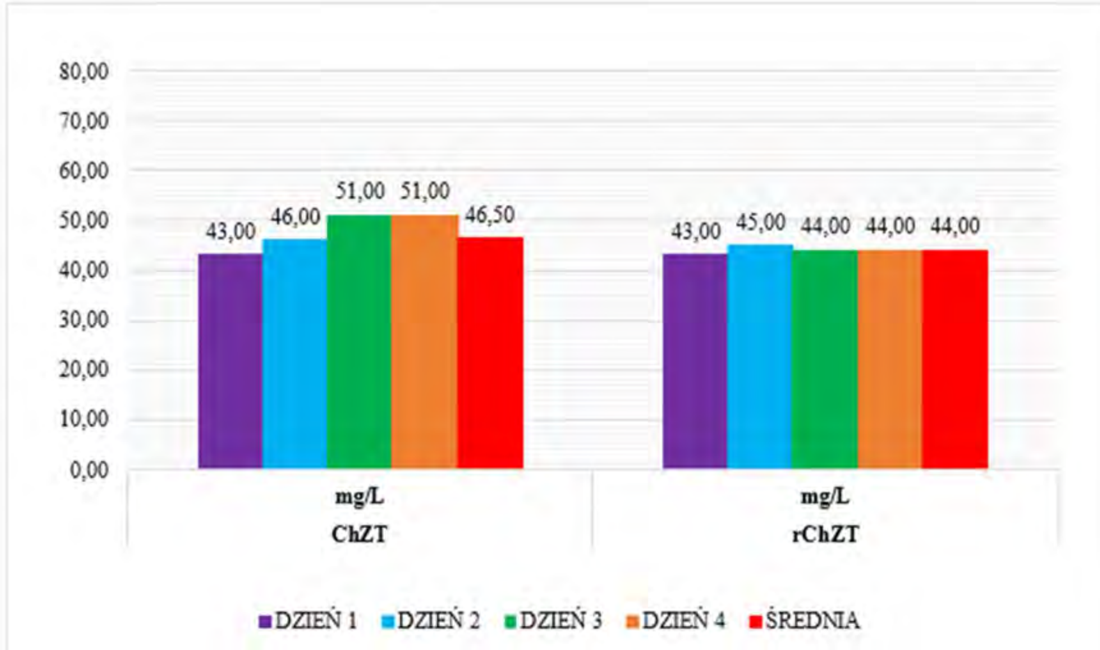


Rysunek 17. Średnie dobowe stężenia ścieków oczyszczonych mechanicznie dopływających do bioreaktora.

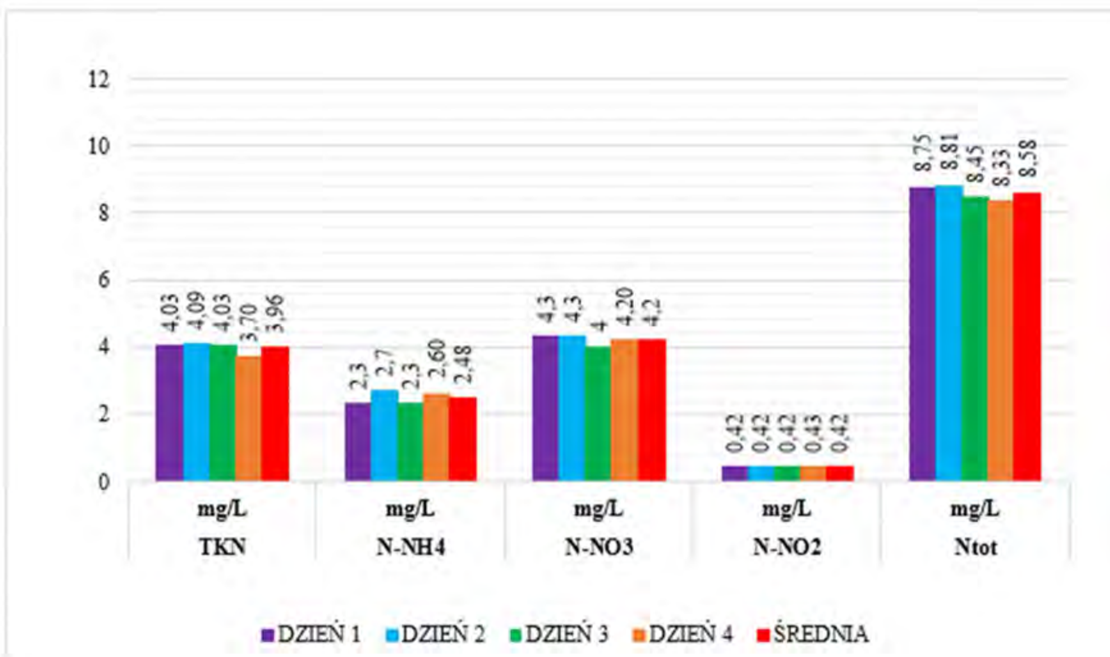
Wyniki zestawione w Tabeli 6 i wykresach – Rysunki 18 oraz 19 - przedstawiają parametry jakości ścieków oczyszczonych odprowadzanych z COŚ do odbiornika. Uzyskane wartości wskazują na bardzo wysoką skuteczność procesów usuwania zanieczyszczeń organicznych oraz azotu amonowego - redukcja ChZT przekraczała 92%, a redukcja N-NH_4 osiągała blisko 98%. Jednocześnie całkowita redukcja azotu ogólnego wyniosła 71,2%. Oznacza to, że w odpływie dominującą formą azotu pozostawały azotany (N-NO_3), stanowiące blisko połowę całkowitej zawartości azotu. Z punktu widzenia emisji GHG obserwowany udział azotanów potwierdza hipotezę, że procesy niedokończonej denitryfikacji sprzyjają generowaniu podtlenku azotu.

Tabela 6. Średnie dobowe stężenia ścieków oczyszczonych dla form azotu - odpływ z OŚ.

	ChZT	rChZT	TKN	N-NH_4	N-NO_3	N-NO_2
	mg/L	*mg/L*	*mg/L*	*mg/L*	*mg/L*	*mg/L*
DZIEŃ 1	43,00	43,00	4,03	2,30	4,30	0,42
DZIEŃ 2	46,00	45,00	4,09	2,70	4,30	0,42
DZIEŃ 3	51,00	44,00	4,03	2,30	4,00	0,42
DZIEŃ 4	46,00	44,00	3,70	2,60	4,20	0,43
ŚREDNIA	**46,50**	**44,00**	**3,96**	**2,48**	**4,20**	**0,42**



Rysunek 18. Średnie dobowe stężenia ścieków oczyszczonych dla frakcji ChZT - odpływ z OŚ.



Rysunek 19. Średnie dobowe stężenia ścieków oczyszczonych dla form azotu - odpływ z OŚ.

Wyniki pomiarów FTIR oraz parametry analiz laboratoryjnych przeliczono następnie na wskaźniki emisji. W kalkulacjach uwzględniono także wyniki otrzymane podczas ostatniego etapu kampanii – pomiarów w warunkach ustalonych - przy stałym napływie ścieków oczyszczonych mechanicznie na bioreaktor, utrzymywanym na poziomie 600 m^3/h oraz przy zadanych parametrach dotyczących stopnia recyrkulacji wewnętrznej i zewnętrznej oraz stałym stężeniu DO 1,0 mg/L. Ostatecznie, opracowania ostatecznych wartości wskaźników emisji dokonano z uwzględnieniem przepływu powietrza w komorze nitryfikacji oraz powierzchni wszystkich bioreaktorów. Szczegółowe dane z okresu kampanii wykorzystano do budowy modelu GPSX COŚ - za pomocą którego, dokonano czynności kalibracyjnych, a następnie walidacyjnych, dla opracowanych wskaźników emisji. Wyniki porównano z wartościami wytycznych IPCC (Tabela 7). Otrzymane wskaźniki empiryczne okazały się ok. 12x niższe (dla N_2O) oraz ok. 100x niższe (dla CH_4) niż dane literaturowe.

Tabela 7. Porównanie otrzymanych wskaźników emisji dla podtlenku azotu i metanu: kampania pomiarowa vs. wytyczne IPCC (wskaźniki literaturowe) – dokonano niezbędnych przeliczeń jednostek.

Opis wskaźnika	Wartość wskaźnika	Jednostka
IPCC	0,01600	*kg N_2O-N/kg Ntot*
	0,02514	*kg N_2O/kg Ntot*
Kampania	0,00212	*kg N_2O/kg Ntot*
IPCC	0,007500	*kg CH_4/kgBZT$_5$ dopływ*
Kampania	0,000074	*kg CH_4/kg ChZT dopływ*

Tak uzyskanie wyniki zastosowano do kalkulacji CF w oparciu o wcześniej opracowany algorytm. Porównanie wyników wykazało bardzo duże różnice między narzędziami. Narzędzie CFCT w wersji z 2014 roku oszacowało całkowity ślad węglowy oczyszczalni na poziomie 29 310 tCO2e rocznie, a wersja zaktualizowana z 2024 roku - na 28 730 tCO2e rocznie. Narzędzie ECAM wskazało natomiast wartość aż 54 370 tCO2e rocznie, czyli blisko dwukrotnie wyższą od wyniku CFCT. Z kolei opracowany algorytm, oparty na danych empirycznych z kampanii pomiarowej, oszacował emisje na poziomie około 10 000 tCO2e rocznie. Oznacza to, że narzędzie ECAM przeszacowało emisje o około 65% w stosunku do wartości uzyskanych na podstawie danych rzeczywistych, a w odniesieniu do emisji procesowych różnice sięgały nawet 200%.

Tak duże rozbieżności ujawniły kluczowy problem stosowania wskaźników literaturowych w analizach środowiskowych. Emisje procesowe, takie jak emisje N_2O w procesach nitryfikacji i denitryfikacji czy emisje CH_4 w procesach beztlenowej stabilizacji osadów, są bowiem silnie zależne od lokalnych warunków eksploatacyjnych, składu ścieków oraz stosowanych technologii. Stosowanie uśrednionych wartości prowadzi zatem do przeszacowania emisji w jednych obiektach, a do niedoszacowania w innych, co obniża wiarygodność całych analiz.

Wyniki tego etapu badań potwierdziły zasadność podejścia przyjętego w opracowanym algorytmie, zakładającego elastyczność w doborze wskaźników emisyjnych i preferencję dla danych empirycznych, jeśli są dostępne. Pokazały także, że w przypadku braku pomiarów konieczna jest lokalna kalibracja wartości literaturowych, aby uwzględniały specyfikę danego obiektu. Wskazano również, że rozwój metod monitoringu emisji gazów cieplarnianych w oczyszczalniach ścieków jest jednym z kluczowych wyzwań dla dalszego rozwoju sektora.

Praktyczne konsekwencje uzyskanych wyników mają istotne znaczenie dla przedsiębiorstw i decydentów. Po pierwsze, stosowanie niewłaściwie skalibrowanych narzędzi kalkulacyjnych może prowadzić do błędnych wniosków na temat efektywności środowiskowej obiektów oraz nieoptymalnych decyzji inwestycyjnych. Po drugie, w świetle nadchodzących obowiązków raportowych związanych z dyrektywą CSRD i standardami ESRS, przedsiębiorstwa muszą być przygotowane na przedstawianie wiarygodnych danych dotyczących emisji, a takie mogą być zapewnione tylko dzięki pomiarom empirycznym lub lokalnie dostosowanym wskaźnikom. Po trzecie, uzyskane wyniki otwierają nowe możliwości badawcze, związane z dalszym rozwojem technologii pomiarowych i integracją ich z systemami monitoringu procesowego w oczyszczalniach.

3.5. Droga do neutralności klimatycznej

Przegląd literatury potwierdził wcześniejsze obserwacje z badań empirycznych - emisje procesowe stanowią główne wyzwanie w bilansie klimatycznym OŚ w kontekście zarządczym, redukcyjnym – jako nieodzowny proces obiektu (**A5**). W wielu analizach wykazano, że mogą one odpowiadać za ponad 60% całkowitego śladu węglowego, podczas gdy emisje związane z energią elektryczną i cieplną sięgają około 30%. Oznacza to, że działania dekarbonizacyjne muszą obejmować zarówno obszar procesów technologicznych, jak i gospodarkę energetyczną, optymalizację połączeń transportowych czy sposoby zagospodarowania osadów.

W literaturze podkreśla się rosnące znaczenie nowych technologii biologicznych, które pozwalają na ograniczenie emisji N_2O, takich jak proces Anammox czy nitryfikacja częściowa (ang. *Partial Nitryfication* - PN). Technologie te redukują ilość azotu przekształcanego w trakcie nitryfikacji i denitryfikacji, co przekłada się na niższą emisję podtlenku azotu, a jednocześnie zmniejszają zapotrzebowanie na tlen, redukując zużycie energii elektrycznej. Z perspektywy systemowej mają one więc podwójną korzyść: ograniczają emisje procesowe i zmniejszają emisje pośrednie z energii. Dodatkową możliwością jest zastosowanie zaawansowanych, ale i kosztownych, metod usuwania azotu na cele produkcji nawozów – takich, jak techniki membranowe, albo bardziej klasycznych, opartych na procesach krystalizacji rozwiązań – produkcji struwitu.

Drugim istotnym obszarem możliwości redukcyjnych jest poprawa efektywności energetycznej. Oczyszczalnie ścieków należą do obiektów o wysokim zużyciu energii, dlatego każde działanie zmniejszające energochłonność procesów technologicznych - optymalizacja napowietrzania, zastosowanie zaawansowanych systemów sterowania, czy modernizacja urządzeń - przekłada się na realną redukcję śladu węglowego. Istotną rolę odgrywa także odzysk energii z biogazu powstającego w procesach fermentacji osadów. Choć w wielu przypadkach nie pozwala on na osiągnięcie pełnej samowystarczalności energetycznej, to znacząco obniża zapotrzebowanie na energię z sieci, a tym samym emisje w Z2.

W przeglądzie wskazano również na znaczenie strategii gospodarki o obiegu zamkniętym (GOZ). Separacja źródłowa ścieków i odzysk składników odżywczych, takich jak azot i fosfor, mogą przyczyniać się do zmniejszenia zapotrzebowania na energochłonną produkcję nawozów mineralnych, o czym wspomniano powyżej, co z kolei ogranicza emisje pośrednie w łańcuchu wartości. Takie podejście poszerza tradycyjne granice analizy i pozwala dostrzec powiązania między sektorem wodno-ściekowym a innymi gałęziami gospodarki, co ma duże znaczenie w perspektywie neutralności klimatycznej.

W literaturze pojawiają się także propozycje innowacyjnych technologii, które w przyszłości mogą zrewolucjonizować podejście do emisji gazów cieplarnianych w oczyszczalniach. Przykładem jest wychwytywanie i wykorzystanie N_2O jako źródła energii. Choć rozwiązanie to znajduje się na wczesnym etapie badań, pierwsze analizy wskazują, że mogłoby ono pozwolić na jednoczesną redukcję emisji i uzyskanie dodatkowych korzyści energetycznych. Podobne nadzieje wiąże się z rozwojem technologii wychwytu i sekwestracji dwutlenku węgla (CCS/CCU), które mogłyby znaleźć zastosowanie także w sektorze wodno-ściekowym.

Przegląd literatury wskazał również na znaczenie synergii działań. Żadna z opisanych technologii ani strategii nie jest w stanie samodzielnie doprowadzić do neutralności klimatycznej obiektu OŚ czy, tym bardziej, sektora. Dopiero ich łączne wdrożenie - obejmujące modernizację technologii oczyszczania, poprawę efektywności energetycznej, odzysk zasobów oraz zastosowanie mechanizmów kompensacyjnych - stwarza realną szansę na osiągnięcie zakładanego celu.

4. Podsumowanie i wnioski

Wykonanie przeglądu literaturowego i metodologicznego (**A1, A5**) wraz z przeprowadzoną kampanią pomiarową emisji gazów cieplarnianych w skali technicznej (**A4**) pozwoliły na wskazanie i opracowanie wytycznych do kalkulacji najważniejszego składnika śladu środowiskowego OŚM - algorytmu obliczeniowego CF zasilonego empirycznymi wskaźnikami emisji (**A2, A3**). Eliminuje on luki – zarówno w zakresie samej metodyki, jak i w obszarze zbierania danych. Algorytm spełnia wymagania legislacyjne UE w kontekście raportowania danych nt. zrównoważonego rozwoju dotyczących wpływu organizacji oraz procesów przez nią zarządzanych na klimat. Stanowi tym samym punkt wyjściowy do analizy struktury emisyjnej obiektów OŚM w celu opracowania ustrukturyzowanego, zaplanowanego w czasie podejścia do systematycznej redukcji emisji GHG – tzw. strategii dekarbonizacyjnej.

Wnioski, będące wynikiem prac, można podzielić na dwa rodzaje – wnioski naukowe (WN) oraz wnioski praktyczne (WP) realizujące poprzez publikacje (**A1-5**) wspólnie cel główny (CG) oraz cele szczegółowe (CS) rozprawy:

WN1: Na ślad środowiskowy obiektu OŚM największy wpływ ma miks energetyczny konsumowany przez obiekt oraz emisje gazów cieplarnianych (ślad węglowy) (**A1**) – CS1, CS2,

WN2: Poziomy emisyjności procesów oczyszczania ścieków mogą znacząco różnić się między obiektami OŚM, co wskazuję na potrzebę rozwoju technik pomiarowych dot. produkcji i uwalniania się GHG (**A4, A5**),

WN3: Redukcja śladu węglowego procesów oczyszczania ścieków podkreśla potrzebę kontynuacji prac badawczych nad alternatywnymi do procesów osadu czynnego metodami oczyszczania z naciskiem na ponowne wykorzystanie zasobów oraz wykorzystanie potencjału energetycznego odpadów w zgodzie z koncepcją GOZ (**A4, A5**),

WP1: Mimo popularności tematyki śladu węglowego, analiza obecnie dostępnych metodologii kalkulacji poziomów emisji z OŚM wykazała, że brakuje ujednoliconego podejścia światowego, które pozwoliłoby porównywać wyniki między obiektami (**A2, A3**) – CS3,

WP2: Przeprowadzona kampania pomiarowa dla COŚ potwierdziła, że wskaźniki literaturowe emisji N_2O i CH_4 są wyższe odpowiednio ok. 12- i 100-krotnie niż literaturowe (**A4**) – CS4,

WP3: Wskaźniki emisji N_2O i CH_4 z procesów należy dobierać indywidualnie dla danego obiektu OŚM, uwzględniając zastosowaną technologię oczyszczania oraz parametry jakościowe charakterystyczne dla procesu celem minimalizacji ryzyka przeszacowania poziomów emisji GHG (**A4**) – CS4, CS6,

WP4: Kalkulacja CF we wszystkich trzech Zakresach GHG Protocol wymaga przygotowania szczegółowych danych – ilościowych i jakościowych – nie tylko dla procesów głównych, ale także dla procesów wspierających realizację głównego zadania OŚM – oczyszczania ścieków do wymaganych poziomów redukcji zanieczyszczeń kierowanych do odbiornika (**A2, A3**), co wskazuje na potrzebę budowania kompetencji i zasobów ludzkich dedykowanych regularnemu przeprowadzaniu obliczeń CF w przedsiębiorstwach wodno-kanalizacyjnych – CS5,

WP5: Korzystanie z ogólnodostępnych narzędzi do kalkulacji CF bez uprzedniego zapoznania się ze stosowanymi metodologią i zestawem wskaźników emisji może prowadzić do znaczących przeszacowań (**A4**) – CS6,

WP6: Analiza struktury śladu węglowego OŚM pozwala zdiagnozować głównego źródła emisji GHG - tak, aby następnie zaplanować praktyczną i wykonalną strategię działań nakierowanych na ich redukcję (**A5**) – CS5,

WP7: Przygotowanie strategii dekarbonizacji z pominięciem etapu pomiarów poziomów emisyjnych z dużym prawdopodobieństwem może przyczynić się do błędnych założeń redukcyjnych, a nawet niewywiązania się z zadeklarowanych celów (A4).

5. Dalsze kierunki działań, plany badawcze

Opracowany algorytm został zastosowany do komercyjnego obliczania śladu węglowego organizacji wodno-kanalizacyjnej w trzech Zakresach. Otrzymane wyniki stanowiły dane wsadowe do opracowania strategii dekarbonizacji ww. organizacji.

Zarówno same działania związane z realizacją rozprawy, jak i działania praktyczne, wykazały, że konieczne są dalsze badania empiryczne dotyczące emisji procesowych podtlenku azotu i metanu. W kampanii pomiarowej wykazano, że wartości uzyskiwane w warunkach technicznych znacząco odbiegają od wskaźników literaturowych, co podważa zasadność stosowania uśrednionych wartości w analizach. Jednakże pojedyncze badania prowadzone w jednej oczyszczalni nie mogą być podstawą do tworzenia ogólnych wniosków. Konieczne jest rozszerzenie pomiarów na większą liczbę obiektów, zróżnicowanych pod względem technologii, wielkości i uwarunkowań lokalnych. Pozwoli to na stworzenie bazy danych empirycznych wskaźników emisyjnych, która mogłaby stanowić podstawę dla przyszłych standardów krajowych i europejskich.

Wskazane jest rozwinięcie analiz związanych emisjami z Zakresu 3, obejmującymi łańcuch wartości. Wyniki pokazały, że emisje związane z produkcją i transportem chemikaliów czy gospodarką osadami mogą stanowić istotny komponent całkowitego śladu węglowego. Jednakże brak jest jednolitych wskaźników i ustandaryzowanych metod ich kalkulacji. Dalsze badania powinny koncentrować się na opracowaniu szczegółowych baz danych dotyczących emisyjności produktów i procesów wykorzystywanych w gospodarce ściekowej, w tym chemikaliów, materiałów eksploatacyjnych oraz technologii zagospodarowania osadów - najlepiej pozyskiwanych bezpośrednio od kolejnych podmiotów stanowiących ogniwa łańcucha wartości danego obiektu OŚ.

Dalszego rozwoju wymagają metody pomiaru i monitorowania emisji w czasie rzeczywistym. Wykorzystane w badaniach spektrometry i komory pomiarowe dostarczyły cennych danych, jednakże ich wdrożenie na szeroką skalę napotyka ograniczenia techniczne i finansowe. Rozwój bardziej dostępnych, zautomatyzowanych systemów monitoringu pozwoliłby na bieżące śledzenie emisji i szybsze reagowanie na ich wahania, a także na tworzenie dynamicznych modeli emisyjnych dla oczyszczalni.

Podjęto już działania praktyczne w celu instalacji infrastruktury ciągłego monitoringu emisji podtlenku azotu na COŚ.

Kolejnym krokiem rozwoju podjętej tematyki badawczej - jest opracowanie kolejnego algorytmu postępowania - dotyczącego opracowania strategii dekarbonizacji dla obiektu OŚ we wszystkich Zakresach emisji zgodnie z GHG Protocol.

Interesującym kierunkiem dalszych badań jest integracja oceny śladu węglowego z innymi kategoriami środowiskowymi, takimi jak emisje zanieczyszczeń do wód i gleby, wykorzystanie zasobów czy generowanie odpadów. Kompleksowe podejście typu *multi-impact* pozwoliłoby na uniknięcie sytuacji, w której redukcja emisji gazów cieplarnianych prowadzi do wzrostu innych oddziaływań środowiskowych.

W perspektywie strategicznej warto rozwijać badania nad rolą oczyszczalni ścieków w gospodarce o obiegu zamkniętym. Coraz częściej traktuje się je nie tylko jako obiekty sanitarne, ale jako zakłady odzysku zasobów i energii. Badania mogłyby dotyczyć oceny potencjału w zakresie odzysku biogenów, produkcji biogazu i biometanu, a także wykorzystania innowacyjnych technologii, takich jak wychwyt i wykorzystanie N_2O czy wdrażanie procesów zintegrowanych z produkcją energii odnawialnej.

Dalsze badania powinny uwzględniać aspekt społeczny i ekonomiczny. Kalkulacja śladu węglowego, choć kluczowa dla oceny środowiskowej, musi być powiązana z analizą kosztów inwestycyjnych i operacyjnych oraz z oceną akceptacji społecznej dla proponowanych rozwiązań. Interdyscyplinarne podejście, łączące inżynierię środowiska, ekonomię i nauki społeczne, pozwoli na stworzenie kompleksowych strategii dekarbonizacji sektora wodno-ściekowego.

Propozycje dalszych badań obejmują zarówno aspekt praktyczny - związany z rozszerzeniem pomiarów i kalibracją wskaźników emisyjnych - jak i aspekt metodyczny i systemowy, związany z opracowaniem narzędzi integrujących różne kategorie oddziaływań i analizą roli oczyszczalni w gospodarce o obiegu zamkniętym. Kierunki te są nie tylko naturalnym rozwinięciem dotychczasowych badań, ale również odpowiedzią na wyzwania praktyki inżynierskiej i polityki klimatycznej, które w najbliższych latach mogą determinować rozwój sektora gospodarki wodno-ściekowej.

Załączniki:

1. Oświadczenia o współautorstwie wraz ze wskazaniem wkładu własnego doktoranta – publikacja **A1**.
2. Publikacja **A1**.
3. Oświadczenia o współautorstwie wraz ze wskazaniem wkładu własnego doktoranta – publikacja **A2**.
4. Publikacja **A2**.
5. Oświadczenia o współautorstwie wraz ze wskazaniem wkładu własnego doktoranta – publikacja **A3**.
6. Publikacja **A3**.
7. Pismo o akceptacji publikacji do druku – **A4**
8. Oświadczenia o współautorstwie wraz ze wskazaniem wkładu własnego doktoranta – publikacja **A4**.
9. Publikacja **A5**.
10. Oświadczenia o współautorstwie wraz ze wskazaniem wkładu własnego doktoranta – publikacja **A5**.
11. Publikacja **A5**.

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Life Cycle Assessment of Municipal Wastewater Treatment Processes Regarding Energy Production from the Sludge Line

Paulina Szulc [1,2], Jędrzej Kasprzak [3], Zbysław Dymaczewski [2,*] and Przemysław Kurczewski [3]

1 Aquanet S.A., 61-491 Poznań, Poland; paulina.szulc@aquanet.pl
2 Institute of Environmental Engineering and Building Installations, Faculty of Environmental Engineering and Energy, Poznan University of Technology, 60-965 Poznań, Poland
3 Institute of Transport, Faculty of Civil and Transport Engineering, Poznan University of Technology, 60-965 Poznań, Poland; jedrzej.kasprzak@put.poznan.pl (J.K.); przemyslaw.kurczewski@put.poznan.pl (P.K.)
* Correspondence: zbyslaw.dymaczewski@put.poznan.pl

Abstract: The efficient and timely removal of organic matter and nutrients from water used in normal municipal functions is considered to be the main task of wastewater treatment plants (WWTPs). Therefore, these facilities are considered to be essential units that are required to avoid pollution of the water environment and decrease the possibility of triggering eutrophication. Even though these benefits are undeniable, they remain at odds with the high energy demand of wastewater treatment and sludge processes. As a consequence, WWTPs have various environmental impacts, which can be estimated and categorized using Life Cycle Assessment (LCA) analysis. In this study, a municipal WWTP based in Poznań, Poland, was examined using the method defined in ISO 14040. ReCiPe Endpoint and Midpoint (v1.11), in a hierarchical approach, were used to evaluate the environmental impacts regarding 18 different categories. All calculations were conducted using a detailed database from 2019, which describes each chosen facility. It was found that the energy component, related to the wastewater treatment process demand and electricity production, is the main determinant of the sum of the environmental impact indicators in light of the modelled energy mix. Therefore, it determines the entire process as an environmentally friendly activity.

Keywords: wastewater; municipal wastewater treatment plant; environmental footprint; life cycle assessment; environmental effectiveness

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1. Introduction

A conventional wastewater treatment (WWT) process, based on the activated sludge method, has been considered to be an environmentally friendly action since the beginning of the twentieth century. WWT technology development, driven by unrestrained urban areas expansion, has successively broadened the municipal wastewater treatment plant's (MWWTP's) area of operation, from achieving a better quality of treated effluents to more advanced solutions where the used water stream is seen as a source of valuable materials and energy. With growing awareness of the multidimensionality of today's large WWT facility profiles, based on the generation of various bioproducts (i.e., biosolids, biopolimers), as well as energy production that can be levelled even higher than the plant's energy consumption, there are many alternatives available to improve their ecological and economic capability [1].

Balance between resource consumption and recovery, which will ensure the maximum possible limitation of an MWWTP's environmental impact, is undoubtedly an indication of the ultimate effectiveness of the facility.

Therefore, the overall efficiency of the WWT facility should be examined as a multifactor problem in a complex way in order to ensure that the holistic perspective is achieved [2]. Life Cycle Assessment (LCA) is a suitable tool that can be applied in this field for the evaluation of environmental aspects [3]. LCA allows us to compare different systems, products, and processes regarding the production and self-usage of energy and the extraction of the raw materials included in various treatment unit combinations [4] to find the best process scenario available [5]. Since the 1990s, different WWT topics have been examined in several previous LCA studies [6]: plant modifications and operations [7–12], modeling [13–19], sludge processes [11,20–22], greenhouse gas (GHG) emissions [4,23,24], rainfall impact [6,25], and uncertainty [5,26–28].

Hence, the main goal of this paper is to estimate the overall environmental impacts of a large WWTP based in Poznań, Poland, by the development of a data inventory. The different approach, in comparison to previous work published [24,29–31], lies in the provision of a detailed and widely operational WWTP database that allowed us to describe the WWT process. In this publication, for the very first time, just such a complex LCA analysis was conducted on a facility based in Poland. Some attempts and analyses for Polish conditions have already been presented (see [29–31], mentioned above), but these usually concern small installations that are utilized for domestic purposes or are mainly based on simulation data. Moreover, in the findings described in [32], as in most of the studies, a volume unit (m^3, for example) is acquired as the functional unit, which makes a direct comparison of the results of individual studies difficult.

2. Materials and Methods

2.1. WWTP Description

The municipal WWTP in this study is located in Kozieglowy, on the right bank of the Warta river, nearby Poznań, the capital city of the Greater Poland region. It is called the Central WWTP (CWWTP). The facility treats wastewater originating from Poznań and the neighboring settlements at a volume of ca. 100,000 m^3 and 1,000,000 population equivalent (PE) based on Biochemical Oxygen Demand (BOD) per day (2019 data). The processing method used to achieve the required standards of wastewater effluent, discharged to the Warta river, is based on biological nutrient removal (BNR) via activated sludge held in modified Bardenpho system bioreactor chambers ($6 \times 25{,}500$ m^3) with secondary clarifiers (6×9500 m^3), preceded by screenings and mechanical separation of solids suspended in the raw used water stream, maintained in aerated horizontal grit traps followed by circular preliminary settling tanks (4×6500 m^3).

Gravitationally thickened primary sludge ((PS); 5% dry solids (DS)) and waste activated sludge ((WAS); 6% DS), conditioned via belt compactor application as a side-products of the WWT process, both with high bio-potential, are mixed together in the proportion 2:1 and delivered to feed anaerobic digesters (ADs). This contributes to an organic loading rate of 1.7 kg vs. $m^{-3}d^{-1}$ (i.e., added per reactor volume; 6×4960 m^3). There, in mesophilic conditions of ca. 35 °C with 26 days of average sludge retention time, biogas is constantly produced in the volume of 21,250 m^3, daily. While pre-treated bio-methane, captured within the process at the level of 64% concentration in the raw stream, is transferred via generating sets into electricity, the digestate (3% DS), after being buffered for gas stripping in the tanks (6×500 m^3), is dewatered in centrifuges (23% DS) for final disposal, which is carried out outside the plant system boundaries alongside other WWT by-products such as separated grit and compacted screenings. Two of the buffer tanks mentioned are dedicated to the digested sludge pumped to the CWWTP from another facility, the left river bank WWTP (LWWTP). The main wastewater and sludge characterizations of the CWWTP in Poznań, Koziegłowy, is included in Tables 1 and 2.

In order to guarantee the stability of phosphorus removal during the winter season, an aqueous solution of iron (III) sulphate (PIX113) is dosed periodically up to the level of P-precipitation need. The PIX113 used in the biological part of the WWTP in Koziegłowy is also used to prevent struvite crystallization in digestate transportation lines in combination with other antiscalant agents. To maintain the best environment for flocculation during thickening and dewatering, different chosen polyelectrolyte solution complexes are used. Facilitation of the final sludge, collected from centrifuges, is provided by using a polymer solution. During intensive fermentation periods, antifoaming supplements are dosed. All these chemical agents were taken into consideration during the analysis (see Section 2.2.3).

Table 1. Operation parameters of the Central Wastewater Treatment Plant (CWWT) in Poznań, Koziegłowy, 2019—wastewater characterization.

Parameter	Value	Units
Capacity		
Population (served in 2019)	920,471	PE [(1)]
Population (maximum designed)	958,000	PE
wastewater treatment plant (WWTP) Inflow		
Mean daily (2019, average)	100,832	$m^3\ day^{-1}$
WWTP flow (maximum designed)	200,000	$m^3\ day^{-1}$
Peak flow (2019)	149,792	$m^3\ day^{-1}$
Peak flow (maximum designed)	260,000	$m^3\ day^{-1}$
Organic Pollution Loading and Concentration (2019—average)		
BOD_5 [(2)]—influent	594	$g\ m^{-3}$
	60,000	$kg\ day^{-1}$
(designed)	57,500	$kg\ day^{-1}$
BOD_5—effluent	3.7	$g\ m^{-3}$
BOD_5—maximum required limit	15.0	$g\ m^{-3}$
COD [(3)]	1239	$g\ m^{-3}$
	138,000	$kg\ day^{-1}$
(designed)	125,000	$kg\ day^{-1}$
COD—effluent	49.9	$g\ m^{-3}$
COD—maximum required limit	125.0	$g\ m^{-3}$
Total Suspended Solids (TSS)	571	$g\ m^{-3}$
	57,670	$kg\ day^{-1}$
(designed)	60,000	$kg\ day^{-1}$
TSS—effluent	6.4	$g\ m^{-3}$
TSS—maximum required limit	35.0	$g\ m^{-3}$
Total Nitrogen (TN)	103	$g\ m^{-3}$
	10,400	$kg\ day^{-1}$
(designed)	11,000	$kg\ day^{-1}$
TN—effluent	9.4	$g\ m^{-3}$
TN—maximum required limit	10	$g\ m^{-3}$
Total Phosphorus (TP)	7	$g\ m^{-3}$
	1515	$kg\ day^{-1}$
(designed)	1400	$kg\ day^{-1}$
TP—effluent	0.6	$g\ m^{-3}$
TP—maximum required limit	1.0	$g\ m^{-3}$

[(1)] PE—population equivalents based on daily BOD_5 load; 1 PE = $60gO_2d^{-1}$; [(2)] BOD_5—Biochemical Oxygen Demand [$gO_2\ m^{-3}$]; [(3)] COD—Chemical Oxygen Demand [$gO_2\ m^{-3}$].

Table 2. Operation parameters of the CWWTP in Poznań, Koziegłowy, 2019—sludge characterization.

Parameter	Value	Units
Thickened PS (2019—average)		
Volume	960	m^3 day^{-1}
Dry solids content	5.2	%DS
Organic matter as % of dry solids content	76.6	%
Thickened WAS (2019—average)		
Volume	480	m^3 day^{-1}
Dry solids content	5.7	%DS
Organic matter as % of dry solids content	78.6	%
Digestate (2019—average)		
Volume	1540	m^3 day^{-1}
Dry solids content	3.0	%DS
Organic matter as % of dry solids content	63.3	%
Dewatered Sludge (2019—average)		
Volume (calculated)	190	m^3 day^{-1}
Dry solids content	22.8	%DS
Organic matter as % of dry solids content	69.2	%

Configuration of the main wastewater treatment technology practiced in the CWWTP, Poznań, Koziegłowy, is presented in Figure 1.

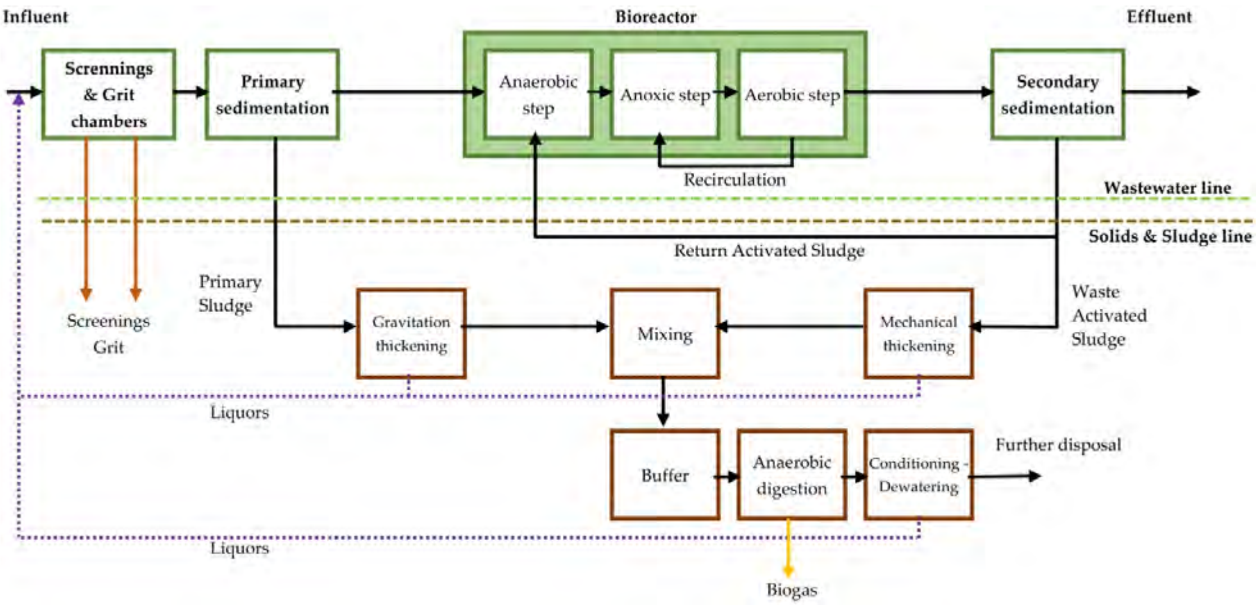

Figure 1. The CWWTP in Koziegłowy—configuration of technology.

2.2. Methods

2.2.1. Background

Analysis of the environmental impacts carried out in accordance with the Life Cycle Assessment (LCA) methodology is the main part of the research material. This method is defined in detail in ISO 14040 as the process of evaluating the effects that a product (or, in this case, a process) exerts on the environment throughout its entire life, by increasing the efficient use of resources and reducing the burden on the environment (liabilities). LCA is regarded as a "from cradle to grave" analysis. This method is more widely defined in many source publications. J. Fava [33] describes it as a technique aimed at assessing the environmental hazards associated with a product system or operation, both by identifying and quantifying the materials and energy used and the waste released into the environment, as well as assessment of the environmental impact of these materials, energy, and waste. It covers the entire life cycle of a product or activity, from the extraction and processing of mineral resources, product manufacturing processes, distribution, use, re-use, maintenance, recycling, and end-use, along with transportation. The purpose of the LCA is the study of the environmental impacts of manufacturing systems on the ecosystem, environmental hazards associated with a product system or operation, both by identifying and quantifying the materials and energy used and the waste released into the environment, as well as assessment of the environmental impact of these materials, energy, and waste. It covers the entire life cycle of a product or activity, from the extraction and processing of mineral resources, product manufacturing processes, distribution, use, re-use, maintenance, recycling, and end-use, along with transportation. The purpose of the LCA is the study of the environmental impacts of manufacturing systems on the ecosystem, human health, and used resources. W. Klöpfer [34] states that the basic idea of the LCA is that the environmental burden associated with a product or service is assessed in terms of the cycle of material acquisition to final disposal. In this sense, the term "LCA life cycle assessment" is more precise than the German "Ökobilanz" or the French "ecobilan", meaning eco-balance. The main idea is undoubtedly correct, and the LCA is the only environmental assessment tool that avoids (false) positive assessment results due to migration/displacement of various types of impacts. The LCA methodology is consistent with the principles and guidelines of ISO 14040: 2006 [35] and ISO 14044: 2006 [36], and it consists of four main steps: defining goal and scope, inventory analysis, impact assessment, and interpretation. The internal modelling involves the following steps [37]:

- Selection and definition of impact categories: The purpose of this step is to select the impact categories that will be considered as part of the overall LCA. It should be completed as part of the initial goal and scope definition phase to guide the LCI (Life Cycle Inventory) data collection process and requires reconsideration following the data collection phase. The items identified in the LCI have potential human health and environmental impacts;
- Classification: Its purpose is to organize and possibly combine the LCI results into impact categories. For LCI items that contribute to only one impact category, the procedure is a straightforward assignment. For example, carbon dioxide emissions can be classified into the global warming category;
- Characterization: This step uses science-based conversion factors, called characterization factors, to convert and combine the LCI results into representative indicators of impacts on human and ecological health. Characterization factors are also commonly referred to as equivalency factors. Characterization provides a way to directly compare the LCI results within each impact category. In other words, characterization factors translate different inventory inputs into directly comparable impact indicators;
- Normalization: This is an LCIA (Life Cycle Impact Assessment) tool used to express impact indicator data in a way that can be compared between impact categories. This procedure normalizes the indicator results by dividing by a selected reference value;
- Grouping: This step assigns impact categories into one or more sets to better facilitate the interpretation of the results into specific areas of concern;
- Weighting: This is sometimes also referred to as valuation of an LCIA, and it assigns weights or relative values to the different impact categories based on their perceived importance or relevance. Weighting is important because the impact categories should also reflect study goals and stakeholder values.

ReCiPe Endpoint and Midpoint in version 1.11, in a hierarchical approach (H) were used in this analysis for the calculation procedures in order to estimate the environmental impacts. The Endpoint version was chosen to determine the general trends related to the proportions of the size of the environmental impacts of individual treatment process components listed in Section 2.1. The Midpoint version was used to determine the absolute size of the environmental loads for the individual components of the treatment process, within the impact categories specified in the method. The ReCiPe procedure is one of the latest and most widely used life cycle assessment practices. It has been defined with a detailed breakdown into the impact categories at work [37], and its last methodological modifications are described in the paper [38].

A complete list of impact categories includes:

- Climate change—unit: kg CO_2 eq;
- Ozone depletion—unit: kg CFC-11 eq;
- Terrestrial acidification—unit: kg SO_2 eq;
- Freshwater eutrophication—unit: kg P eq;
- Marine eutrophication—unit: kg N eq;
- Human toxicity—unit: kg 1,4-DB eq;
- Photochemical oxidant formation—unit: kg NMVOC eq;
- Particulate matter formation—unit: kg PM_{10} eq;
- Terrestrial ecotoxicity—unit: kg 1,4-DB eq;
- Freshwater ecotoxicity—unit: kg 1,4-DB eq;
- Marine ecotoxicity—unit: kg 1,4-DB eq;
- Ionizing radiation—unit: kBq U235 eq;
- Agricultural land occupation—unit: m^2a;
- Urban land occupation—unit: m^2a;
- Natural land transformation—unit: m^2;
- Water depletion—unit: m^3;
- Metal depletion—unit: kg Fe eq;
- Fossil fuel depletion—unit: kg oil eq.

In the description of the analytical procedure, zero values of standardization factors were found for the impact category "water depletion"; therefore, it is not included in the final evaluation at the weighting stage (ReCiPe Endpoint).

2.2.2. Goal and Scope of Analysis

Definition of the Analysis Goal

The aim of the analysis was to assess the environmental impact of the process of wastewater stream management in the period under consideration (January–December 2019) compared to the "baseline" process, as implemented at the CWWTP in the period under consideration.

Functional Unit

The volume of wastewater managed during the reference year (2019), broken down into individual months, was assumed to be the functional unit. This volume was 38,325,867 m^3. Changes in the volume of the managed wastewater in particular months are shown in Figure 2.

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- Terrestrial ecotoxicity—unit: kg 1,4-DB eq;
- Freshwater ecotoxicity—unit: kg 1,4-DB eq;
- Marine ecotoxicity—unit: kg 1,4-DB eq;
- Ionizing radiation—unit: kBq U235 eq;
- Agricultural land occupation—unit: m^2a;
- Urban land occupation—unit: m^2a;
- Natural land transformation—unit: m^2;
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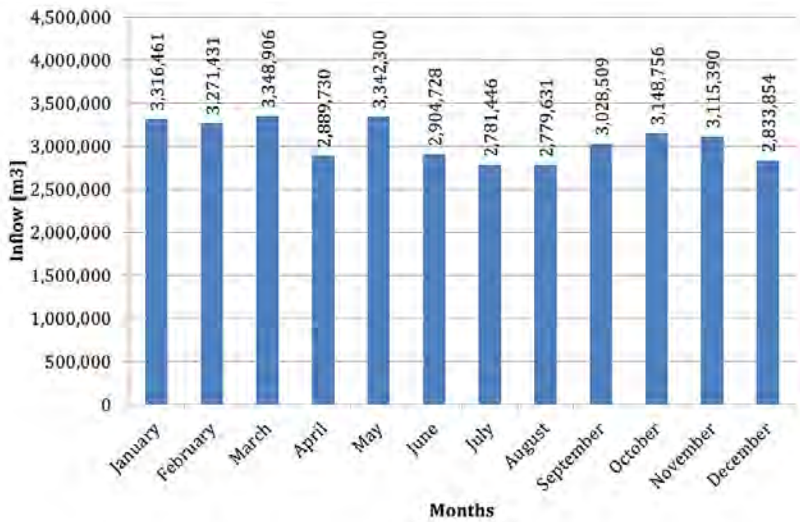

Figure 2. Actual wastewater flows in 2019 at the CWWTP, broken down by months.

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System Boundaries, Assumptions and Limitations

Based on the data provided (limited to 2019), a model of the wastewater treatment system was proposed. It is assumed that a stream of raw sewage is directed to the treatment system and, after passing through all stages of the technological process, is directed to a receiver (environment) as a stream of treated wastewater. The treatment process itself was assumed to be indivisible due to the fact that it is not possible to include division into

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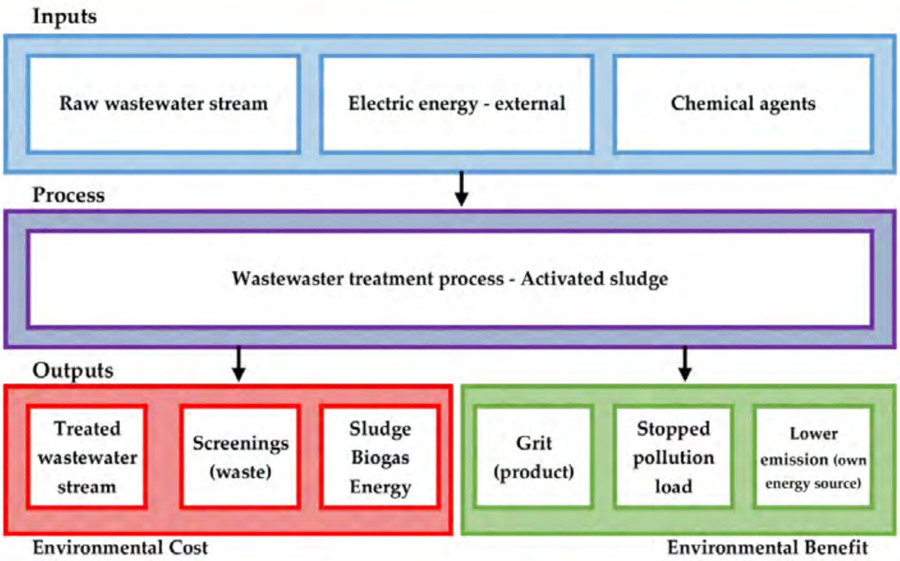

Figure 3. Model boundaries.

In Figure 3, the process inputs are the raw wastewater stream at the plant's input, electrical power fed to the treatment system from external sources, and additional (chemical) substances used in the used water treatment process. Process outputs are divided into two groups:

1. As sources of environmental loads adopted: (a) A treated wastewater stream that, although characterized by significantly lower values of the analyzed parameters determining the harmfulness of the wastewater (concentrations), still contains substances that have a negative impact on the environment; (b) The so-called "screenings", i.e., macro-pollutants separated by means of mechanical separators (screens), which were not taken into account at all in the preliminary analysis and at this stage were treated as waste to be treated; and (c) Sewage sludge, which is then directed to anaerobic digesters and serves to produce the methane-rich biogas, which is next used to produce the energy used in the process of wastewater treatment; this way of generating energy and its use in the process of wastewater treatment also has a negative environmental impact (e.g., emissions during biogas combustion, etc.).
2. As sources of the indirect environmental benefits: (a) Grit (mineral by-products that consists mostly of sand collected from the streets), recovered from the mechanical separation processes and treated as a product for later release and use; (b) The avoided impacts resulting from differences in the values of standardized parameters characterizing wastewater streams, raw and treated; and (c) the amount of avoided environmental impacts resulting from the production of energy from the plant's own sources, which in turn allows for the avoidance of consumption of an equivalent amount of energy from external sources.

Thus, such a structure of the inputs and outputs was conditioned by the structure of the data; no intermediate data from successive steps in the wastewater treatment process is available.

Wastewater streams (at process input and output) were modelled based on a set of standardized parameters characterizing wastewater streams, including:

- COD;
- BOD_5;

Wastewater streams (at process input and output) were modelled based on a set of standardized parameters characterizing wastewater streams, including:

- COD;
- BOD_5;
- total suspended solids (TTS);
- total nitrogen (TN);
- total phosphorus (TP);
- arsenic;
- chromium;
- zinc;
- cadmium;
- copper;
- nickel;
- lead;
- mercury;
- silver;
- vanadium;
- total organic carbon;
- anionic surfactants;
- substances extracted with petroleum ether;
- phenolic index;
- mineral oil index (C10-C40);
- chlorides;
- sulfates;
- silicon.

Thickening agents and flocculants, dehydrating and anti-foaming agents, and antiscalants (substances preventing struvite formation) have been adopted as additional (chemical) substances, as presented in the breakdown of trade names and quantities of consumed chemical substances. For the purposes of modelling environmental loads related to energy consumption, a profile of electricity production (consumed from the power grid for the purposes of purification processes) was also adopted based on data on the Polish electricity mix for 2019, according to data from PSE (Polish Power Grids). The share of individual energy sources in the mix is shown in Figure 4. This distribution represents the national average.

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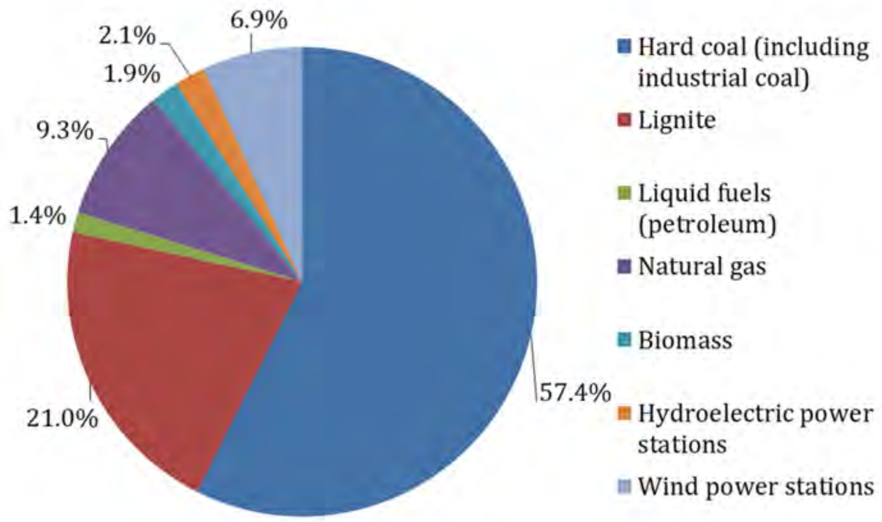

Figure 4. Adopted electricity mix [34,39].

Figure 4. Adopted electricity mix [34,39].

The data provided show that most of the electrical power used in the wastewater treatment processes came from the company's own sources. A quantitative breakdown of the energy consumed for these purposes in 2019 is presented in Table 3.

Table 3. Electricity consumption at the CWWTP in 2019, broken down by sources.

Month	Energy Consumption—TOTAL	Energy Consumption—from External Source	Energy Consumption—Produced from Biogas	Covered from Own Source
	kWh	kWh	kWh	%
January	1,991,072	709,481	1,281,591	64
February	1,965,577	755,487	1,210,090	62
March	2,091,587	755,744	1,335,843	64
April	1,735,742	396,503	1,339,239	77
May	1,770,691	441,140	1,329,551	75
June	1,618,724	405,659	1,213,065	75
July	1,585,674	342,897	1,242,777	78
August	1,497,973	158,677	1,339,296	89
September	1,515,407	259,723	1,255,684	83
October	1,720,319	416,944	1,303,375	76
November	1,719,799	503,319	1,216,480	71
December	1,723,640	599,645	1,123,995	65
Average	1,744,684	478,768	1,265,916	73
TOTAL	**20,936,205**	**5,745,219**	**15,190,986**	

2.2.3. Life Cycle Inventory (LCI)

Process Inputs

In the period under analysis, the total wastewater flow through the plant was, as already mentioned in Section 2.2.2 of this study, 38,325,867 m^3, broken down into individual months, as shown in Figure 2, and all consumption figures for individual utilities and materials refer to them as reference figures.

As already mentioned, wastewater streams are described with a set of standardized parameters. Their values for raw wastewater streams at the treatment process input are presented in Table 4.

Table 4. Actual concentration values of standardized parameters in raw wastewater in 2019.

Month	Pollutant Concentration				
	$g\ m^{-3}$				
	COD	BOD_5	TSS	TN	TP
January	1326.9	468.7	688.7	98.2	14.6
February	1258.8	548.4	537.1	111.9	17.1
March	1288.6	567.5	682.7	98.9	13.5
April	1254.0	505.5	631.2	113.4	15.4
May	971.1	458.8	371.0	96.9	13.4
June	1144.0	477.2	581.7	99.3	14.3
July	1213.7	531.5	608.9	104.7	14.0
August	1428.1	681.5	667.5	109.8	17.4
September	1231.3	579.6	518.8	104.2	16.1
October	1245.9	557.5	443.7	99.2	14.2
November	1241.0	621.4	531.4	92.0	13.9
December	1263.5	595.0	588.3	104.3	19.0
Average	1238.9	549.4	570.9	102.7	15.2

Table 5 presents the values of concentrations of additional substances analyzed in raw wastewater.

Table 5. Actual concentrations of additional substances parameters in raw wastewater in 2019.

Parameter	Pollutant Concentration											
	g m^{-3}											
	I	II	III	IV	V	VI	VII	VIII	IX	X	XI	XII
Arsenic	0.0068	0.0019	0.0017	0.0030	0.0016	0.0022	0.0021	0.0022	0.0016	0.0013	0.0020	0.0002
Chrome	0.0160	0.0120	0.0190	0.0210	0.0130	0.0120	0.0170	0.0330	0.0260	0.0490	0.0170	0.0060
Zinc	0.2800	0.1800	0.0350	0.2800	0.2000	0.2700	0.2300	0.3900	0.2000	0.3700	0.3700	0.0270
Cadmium	0.0005	0.0000	0.0059	0.0002	0.0002	0.0002	0.0001	0.0002	0.0000	0.0002	0.0002	0.0000
Copper	0.1700	0.0900	0.0960	0.0850	0.0830	0.1100	0.1100	0.1500	0.0690	0.1000	0.0840	0.0180
Nickel	0.0440	0.0250	0.0500	0.0330	0.0240	0.0170	0.0150	0.0370	0.1600	0.0580	0.0000	0.0043
Lead	0.0046	0.0030	0.0130	0.3700	0.2700	0.0390	0.0550	0.0600	0.0100	0.0410	0.0260	0.0011
Mercury	0.0005	0.0006	0.0003	0.0009	0.0004	0.0002	0.0005	0.0008	0.0003	0.0002	0.0001	0.0002
Silver	0.0016	0.0007	0.0007	0.0016	0.0013	0.0020	0.0006	0.0022	0.0004	0.0055	0.0029	0.0002
Vanadium	0.0013	0.0008	0.006	0.0016	0.0007	0.0020	0.0014	0.0027	0.0017	0.0013	0.0013	0.0001
Total organic carbon—TOC *	245.000	x	x	174.000	x	x	156.000	x	x	88.800	x	x
Anionic surfactants *	2.5600	x	x	1.6200	x	x	2.0500	x	x	0.0700	x	x
Substances that extract with petroleum ether *	76.0000	x	x	69.2000	x	x	58.0000	x	x	27.6000	x	x
Phenol index	0.1800	0.1900	0.1600	0.2600	0.1700	0.1600	0.4400	0.1000	0.1100	0.1200	0.1200	0.1200
Mineral oil index (C10-C40) *	0.3200	x	x	0.3100	x	x	0.2200	x	x	0.2300	x	x
Chlorides	245.00	216.00	225.00	263.00	321.00	227.00	413.00	173.00	235.00	215.00	211.00	232.00
Sulphates	126.00	109.00	121.00	126.00	121.00	144.00	93.00	88.00	122.00	129.00	139.00	120.00
Silicon *	8.5000	x	x	10.0000	x	x	8.1000	x	x	9.3000	x	x

* Parameters tested on a quarterly basis.

The structure of electricity consumption in individual months by source is given in Table 3. It shows that, for the total amount of almost 21 GWh of electricity consumed in 2019, the production from the plant's own sources (combustion of biogas generated from waste sludge) covered, on average, almost $\frac{3}{4}$ of the demand, fluctuating at around 62–89%. As a result of adopting the assumption described in Section 2.2.2, the amount of electricity drawn from the municipal grid was assumed to be the process input.

The implementation of treatment processes requires the supply of an appropriate amount of chemical substances that boost the technological processes. Their consumption in the reference period is presented in Table 6.

Table 6. Consumption of chemical agents for the needs of wastewater treatment in 2019.

Month	Chemical Agents							
	Thickening Agent	Dewatering Agent I—CWWTP	Dewatering Agent I—LWWTP	Anti-Foam Agent	PIX113 (P-Percipitation)	PIX113 (Antiscalant)	Anti-Scalant I	Anti-Scalant II
	kg mth^{-1}	dm^3 mth^{-1}	dm^3 mth^{-1}	dm^3 mth^{-1}	dm^3 mth^{-1}	dm^3 mth^{-1}	cm^3 mth^{-1}	dm^3 mth^{-1}
January	5937.6	7157.9	22,789.3	682.0	18,400.0	29.2	762.0	0.0
February	3653.8	5140.8	17,105.2	0.0	74,957.0	22.3	551.0	550.0
March	4824.4	4318.3	18,600.0	0.0	88,650.0	24.4	621.0	0.0
April	5290.8	6360.0	18,348.0	0.0	90,425.0	24.2	617.0	100.0
May	6596.0	6398.4	20,280.2	0.0	2587.0	25.4	637.0	200.0
June	4219.0	6375.0	19,038.0	0.0	0.0	22.3	543.0	325.0
July	5281.3	5598.6	23,197.3	0.0	4995.0	25.2	726.0	0.0
August	6179.0	6014.0	21,669.0	0.0	8150.0	26.0	901.0	0.0
September	5012.6	9984.0	18,300.0	0.0	3449.0	26.4	1057.0	0.0
October	3869.7	7257.1	17,756.8	0.0	8306.0	22.4	931.0	0.0
November	5905.9	7431.0	18,924.0	600.0	18,685.0	41.7	993.0	0.0
December	6736.0	5908.6	25,482.0	620.0	25,322.0	24.6	1285.0	0.0
Average	5292.2	6495.3	20,124.2	158.5	28,660.5	26.2	802.0	97.9
TOTAL	63,506.1	77,943.7	241,489.8	1902.0	343,926.0	314.2	9624.0	1175.0

Process Outputs

The output wastewater stream was characterized by base parameters with values presented in Table 7.

Table 7. Actual concentration values of standardized parameters in treated wastewater in 2019.

Month	**Pollutant Concentration**				
	g m^{-3}				
	COD	**BOD$_5$**	**TSS**	**TN**	**TP**
January	51.0	4.1	5.6	8.0	0.9
February	70.7	5.4	13.2	9.6	1.5
March	72.5	6.6	14.2	9.9	1.3
April	69.4	4.7	13.8	10.4	0.6
May	43.3	3.3	4.3	10.4	0.2
June	52.0	3.0	4.2	8.9	0.3
July	40.4	2.7	3.8	9.0	0.3
August	40.5	3.3	3.9	10.4	0.4
September	42.0	3.1	3.6	10.4	0.3
October	41.1	3.1	4.7	9.3	0.4
November	36.4	2.2	2.6	8.7	0.3
December	39.3	2.7	3.1	7.7	0.3
Average	49.9	3.7	6.4	9.4	0.6

Table 8 presents the values of concentrations of additional substances analyzed in treated wastewater.

As already mentioned in Section 2.2.2 of this study, in the analysis, the indirect environmental benefit of reducing the level of environmental loads as a result of the reduction of basic values and additional parameters analyzed for wastewater streams, and modelled as input, was assumed. These impacts were included in the model as "avoided" and were treated as an environmental benefit. Their magnitude results directly from the difference between the levels of standardized parameters characterizing the input wastewater streams (raw wastewater) and the output (treated wastewater discharged to a receiver) of the wastewater treatment process.

Moreover, grit and screenings (residues from mechanical separation) are also included as outputs from the treatment processes, and their quantities in the individual months of 2019 are given in Table 9.

Table 8. Actual concentration values of additional substances in treated wastewater in 2019.

Parameter	**Pollutant Concentration**											
	mg dm^{-3}											
	I	**II**	**III**	**IV**	**V**	**VI**	**VII**	**VIII**	**IX**	**X**	**XI**	**XII**
Arsenic	0.0027	0.0011	0.0012	0.0022	0.0008	0.0010	0.0009	0.0006	0.0011	0.0009	0.0010	0.0001
Chrome	0.0018	0.0031	0.0022	0.0030	0.0012	0.0017	0.0017	0.0016	0.0016	0.0011	0.0003	0.0003
Zinc	0.0420	0.0350	0.0480	0.0310	0.0150	0.0760	0.0310	0.0540	0.0370	0.0300	0.0250	0.0093
Cadmium	0.0001	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Copper	0.0040	0.0047	0.0023	0.0170	0.0130	0.0130	0.0230	0.0120	0.0180	0.0062	0.0058	0.0008
Nickel	0.0450	0.0210	0.0200	0.0160	0.0120	0.0130	0.0120	0.0230	0.1200	0.0150	0.0000	0.0028
Lead	0.0007	0.0002	0.0010	0.0096	0.0120	0.0120	0.0033	0.0020	0.0000	0.0000	0.0000	0.0001
Mercury	0.0000	0.0003	0.0001	0.0001	0.0001	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Silver	0.0002	0.0001	0.0060	0.0000	0.0005	0.0013	0.0000	0.0002	0.0000	0.0055	0.0025	0.0002
Vanadium	0.0004	0.0005	0.0002	0.0002	0.0003	0.0005	0.0005	0.0002	0.0007	0.0006	0.0005	0.0000
Total organic carbon—TOC *	13.500	x	x	14.300	x	x		x	x		x	x
Anionic surfactants *	0.2100	x	x	0.3200	x	x	0.2100	x	x	0.0100	x	x
Substances that extract with petroleum ether *	1.0000	x	x	1.2000	x	x	1.4000	x	x	1.2000	x	x
Phenol index	0.0096	0.0152	0.0044	0.0088	0.0324	0.0324	0.1360	0.0072	0.0044	0.0100	0.0080	0.0084
Mineral oil index (C10-C40) *	0.1800	x	x	0.1000	x	x	0.1300	x	x	0.1500	x	x
Chlorides	165.00	199.00	212.00	208.00	192.00	179.00	214.00	228.00	222.00	208.00	207.00	193.00
Sulphates	120.00	139.00	126.00	168.00	146.00	148.00	133.00	152.00	147.00	144.00	138.00	122.00
Silicon *	7.1000	x	x	7.9000	x	x		x	x		x	x

* Parameters tested on a quarterly basis.

Table 9. Amounts of products recovered after mechanical separation in 2019.

Month	Grit	Screenings
	Mg mth^{-1}	Mg mth^{-1}
January	98.9	57.8
February	92.4	41.9
March	105.9	48.2
April	154.1	42.5
May	195.6	56.8
June	169.7	42.8
July	173.3	31.4
August	145.4	33.5
September	191.3	38.8
October	179.1	35.6
November	248.2	57.2
December	129.6	56.7
Average	156.9	45.3
TOTAL	1883.4	543.1

Sludge is also treated as an output from the processes. Its total quantity, resulting from dewatering, in 2019 was 562,310 m^3. The breakdown into individual months is shown in Table 10.

This sludge is provided from the anaerobic digesters (ADs), where biogas is obtained and used as fuel in energy processes. In 2019, about 7,755,000 m^3 of biogas was produced, out of which just over 15 GWh of electricity was generated. This energy input is then fed into the wastewater treatment processes and, in our model, it has been treated as:

1. Electricity produced from ADs taking into account the environmental costs (emissions during production and combustion);
2. Electricity, whose consumption from the grid has been avoided, together with the environmental loads associated with its generation in accordance with the Polish electricity mix, as shown in Figure 4.

No other sewage sludge treatment options have been considered in this modelling phase.

As has been previously mentioned, the main limitations of the analysis are related to the time constraints (2019), the process model adopted (no intermediate data available), generalization of the sources of energy supported from the grid (Polish energy mix), and exclusion of sludge (mixture of dewatered PS and WAS) digestation, other than for use in internal energy production processes.

Table 10. Amount of wastewater sludge processed in 2019.

Month	Digestate for Dewatering—CWWTP	Digestate for Dewatering—LWWTP	TOTAL
	m^3 mth^{-1}	m^3 mth^{-1}	m^3 mth^{-1}
January	46,291.0	9126.0	55,417
February	33,937.0	4998.0	38,935
March	39,868.0	7289.0	47,157
April	41,167.0	9697.0	50,864
May	40,948.0	8131.0	49,079
June	33,285.0	8787.0	42,072
July	37,963.0	7144.0	45,107
August	39,649.0	7245.0	46,894
September	40,673.0	8100.0	48,773
October	38,783.0	7570.0	46,353
November	34,969.0	7254.0	42,223
December	40,669.0	8767.0	49,436
Average	39,016.8	7842.3	46,859.2
TOTAL	468,202.0	94,108.0	562,310.0

3. Results—Life Cycle Impact Assessment (LCIA)

3.1. Overall Results—Comparison of Environmental Impacts for Process Inputs and Outputs

The results of environmental calculations for the wastewater treatment process at the CWWTP were first analyzed using a point scale (ReCiPe Endpoint (H) V.1.11/Europe ReCiPe H/A/Single score) to determine:

(a) Aggregate values of total environmental impact indicators for the process under consideration in terms of (1) inputs (supplied resources) into the process and (2) outputs (received resources and environmental benefits) from the process;
(b) Distribution of environmental impacts in individual months of the analyzed period (separately for inputs and outputs);
(c) Detailed determinant(s) of environmental costs and benefits.

Figure 5 shows the values of the total environmental impact indicators for general inputs and outputs of the wastewater treatment process. The list shows that the impacts of the inputs to the wastewater treatment process are actually loads (a positive value of the environmental impact indicator is recorded), while the impacts of the outputs from the wastewater treatment process are environmental benefits (negative value of the environmental impact indicator). Moreover, the figure shows the general environmental impact profile of the process inputs and outputs. It should be noted that the overall profile is dominated by impacts related to human health (almost 45% of the total for process inputs and just over 41% of the total for process outputs). Impacts related to resource depletion account for 29% of the total for process inputs and 33% of the total for process outputs.

When breaking down the damage categories into individual impact categories (detailed profile, Figure 6), it can be seen that a significant part of the impacts (almost 42% for process inputs and more than half for process outputs) is related to climate change (these impacts, according to the adopted calculation procedure, ReCiPe, are broken down into damage to human health and ecosystems). A significant share of impacts is associated with the depletion of fossil fuel resources (about 30% in the case of inputs and outputs) as well as burdens associated with human toxicity and the formation of particulate matter. Such a structure of both profiles is clearly indicative of the dominance of the energy-intensive processes based on an energy mix typical for the combustion of fossil solid fuels.

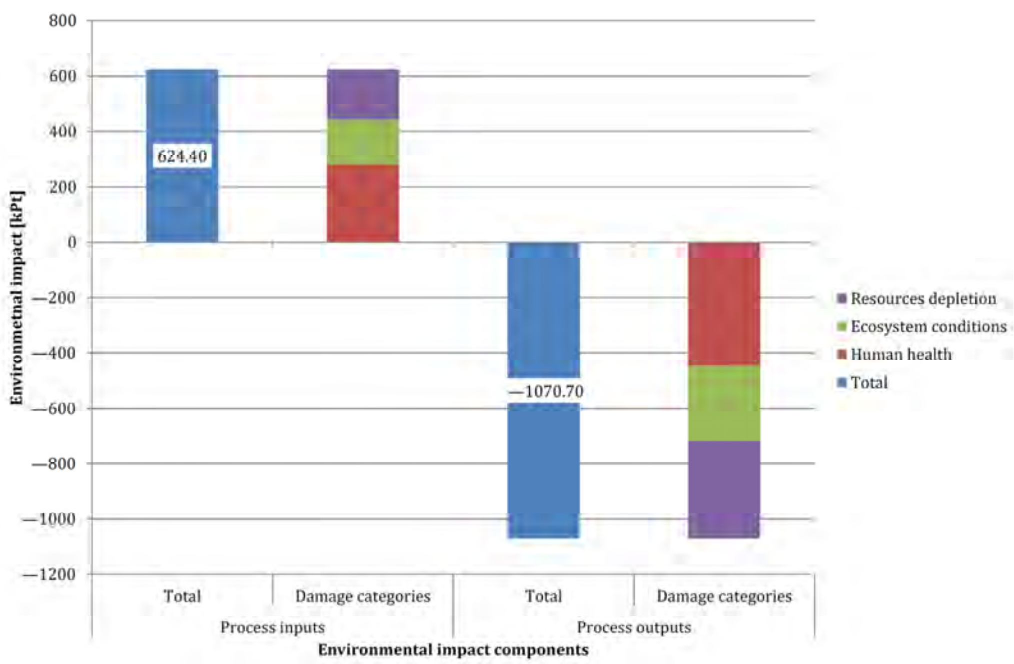

Figure 5. Comparison of inputs and outputs to the wastewater treatment process at the CWWTP in 2019—Total environmental impact indicators and general environmental impact profiles broken down into damage categories.

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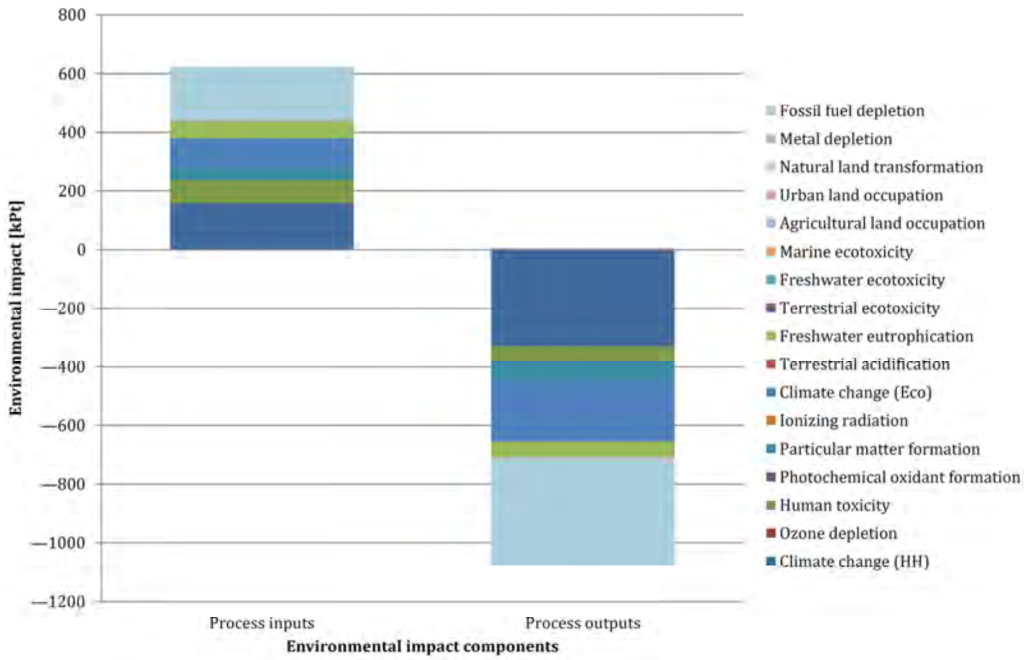

Figure 6. Comparison of inputs and outputs to the wastewater treatment process at the CWWTP in 2019—Detailed environmental impact profiles broken down into impact categories.

Figure 7 presents the monthly distribution of total environmental impact indicators related to inputs to the wastewater treatment process. The breakdown shows that a significantly higher level of environmental loads is generated in the winter months and early spring (December–January). As the temperature rises in the following months, the level of loads connected to the process inputs decreases to reach the lowest values in the summer months (July–September). The general environmental impact profile for the breakdown of environmental impacts associated with inputs to the WWT process in individual month is also shown. Constant levels in the relationships between the different categories in the profiles can be observed. The general profile is dominated by impacts related to influence on the human health (41–49%); impacts within the other damage categories are lower: resource depletion 25–41% and ecosystem quality 24–33%.

The detailed profile of environmental impacts associated with wastewater treatment process inputs in individual months (Figure 8) is dominated by the burdens related to climate change (in terms of emissions of substances harmful to human health and ecosystems), depletion of fossil fuel resources and human toxicity. As already mentioned, this distribution of impact profiles is indicative of the decisive influence of the energy-intensity component.

Figure 9 shows the monthly distribution of total impact indicators related to outputs from the wastewater treatment process. Above all, please note that the indicators take negative values, which is equivalent to the potential environmental benefits from the implementation of this process. Besides, the behavior of indicator values in individual months is much more similar than for input entrances (the differences merely achieve 17%, while, for inputs, a three-fold difference was observed). What is more, the differences in the indicator values are not correlated to the season of the year (total indicators take the lowest values in December, February, and June, and the highest values are in April, May, and August). Additionally, for the general and detailed environmental profiles of wastewater outputs shown in Figures 9 and 10, a behavior analogous to that described in Figures 7 and 8 can be observed in individual months with the predominant mix-based energy consumption component, in which fossil fuels play a significant role.

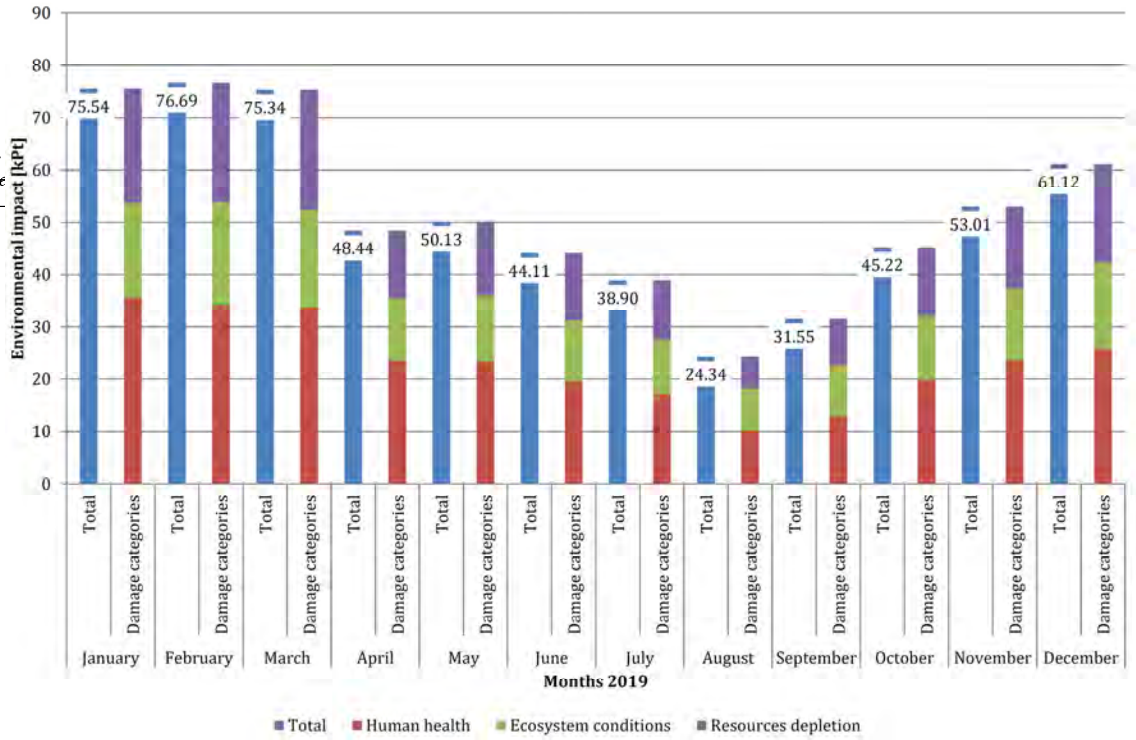

Figure 7. Monthly distribution of environmental impacts for inputs to the wastewater management processes at the CWWTP in 2019—Total environmental impact indicators and general environmental impact profiles broken down into damage categories.

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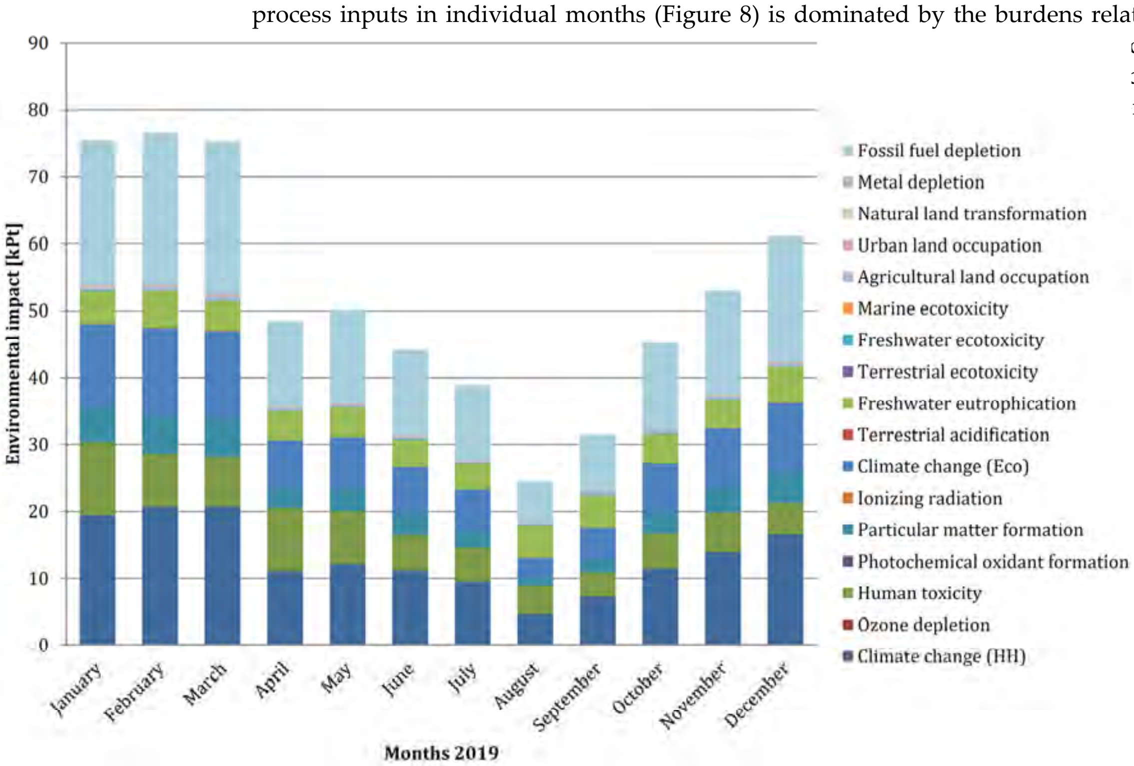

Figure 8. Monthly distribution of environmental impacts for inputs to the wastewater management processes at the CWWTP in 2019—Detailed environmental impact profiles broken down into impact categories.

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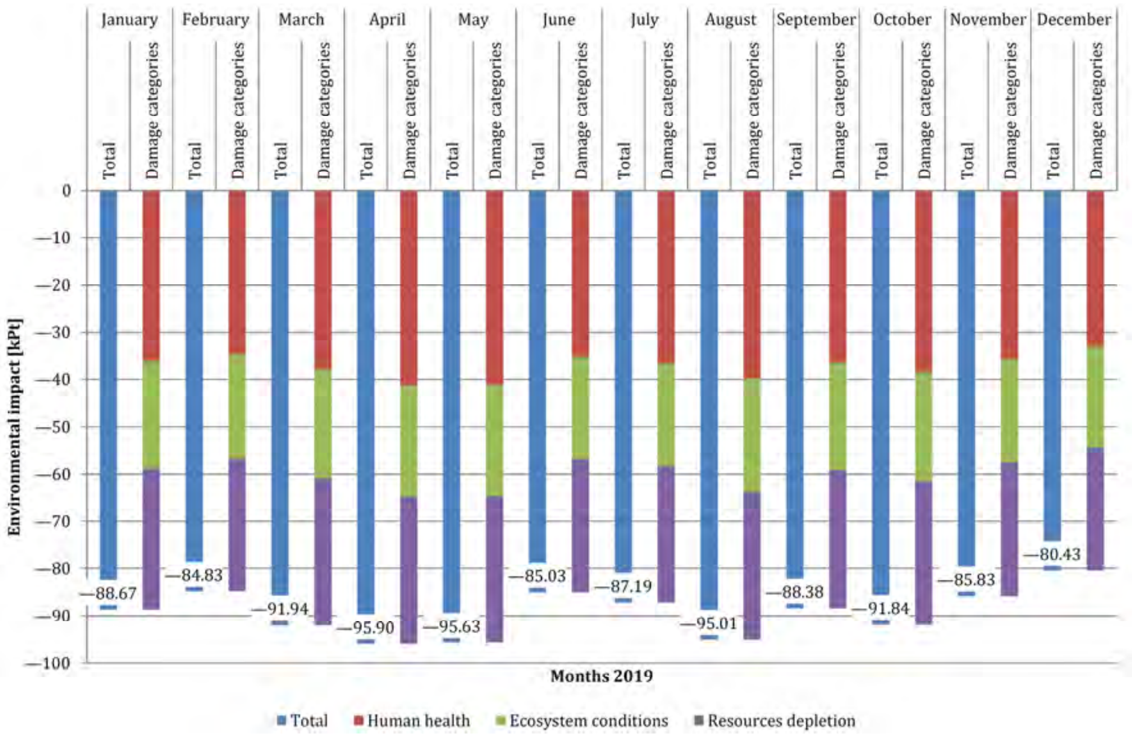

Figure 9. Monthly distribution of environmental impacts for outputs from the wastewater management processes at the CWWTP in 2019—Total environmental impact indicators and general environmental impact profiles broken down into damage categories.

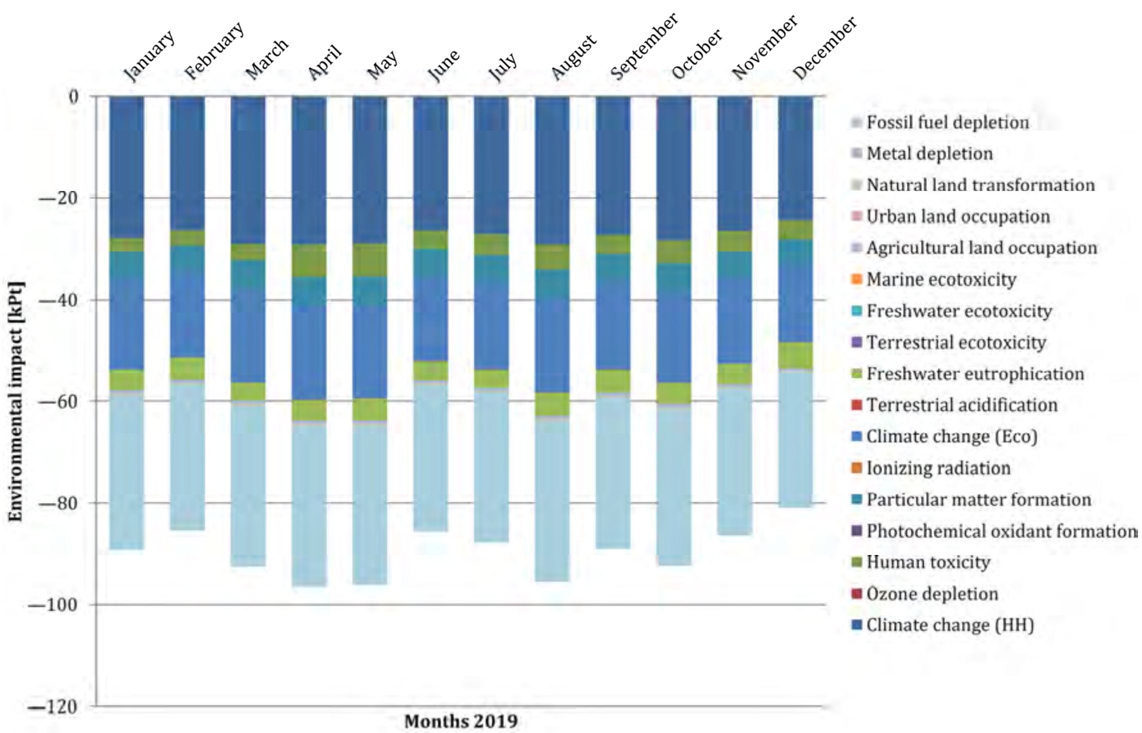

Figure 10. Monthly distribution of environmental impacts for outputs from the wastewater management processes at the CWWTP in 2019—Detailed environmental impact profiles broken down into impact categories.

3.2. Components of Environmental Impacts for Inputs and Outputs of the Wastewater Treatment Process

Characteristics of environmental impacts for individual components of the wastewater treatment process at the CWWTP (data for 2019) are presented below, grouped by inputs and outputs of the wastewater treatment process, in general, without a breakdown into data from individual months. It can be seen that the environmental impacts for the process inputs take positive values (they are thus environmental burdens),

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3.2.1. Components of Environmental Impacts for Inputs to Treatment Processes

Figure 11 shows the total values of environmental impact indicators for individual inputs to the wastewater treatment process. As already described in Section 2.2.2, it is assumed that the process inputs include the used water stream at the inflow to the process along with its characteristic values of loads, concentrations, quantities of the relevant additional analyzed substances, chemical agents supplied to the process to ensure its correct implementation, and electricity supplied to the process from external sources (from the municipal grid, generated based on the energy mix given in Figure 4).

From the general data presented in Figure 11, it can be seen that the consumption of energy from external sources is characterized by the highest values of environmental impact indicators. As mentioned in Section 2.2.2, consumption of electricity from external sources for the process needs ranged from 11 to 38% (depending on the month, being the highest in February and the lowest in August), which translated into consumption of about 5.75 GWh from the grid. With the assumptions resulting from the structure of the energy mix, such an amount translates into an environmental impact indicator of just over 500 kPt. The remaining input components have indicators with much lower values (pollutant loads in wastewater at the inlet with slightly over 90 kPt and total chemicals with less than 30 kPt). A decomposition of the general environmental impact indicator values for WWT inputs broken into individual damage categories (general profile) and into impact categories (detailed profile, Figure 12) is also shown. Significant differences in the behavior of profiles for the main process input components can be observed. The shape of the environmental impact profiles characterizing the electricity generated on the basis of the mix presented in Figure 4 is described in Section 3.1. Burdens associated with the impact on climate change dominate (in terms of human health and ecosystems). Moreover, the contribution of burdens associated with the depletion of fossil fuel resources is also noticeable.

A different environmental impact profile pattern can be observed for the remaining process inputs. For the wastewater stream at process input (characterized by the parameters described in Section 2.2.3), some of the environmental burdens relevant for the ReCiPe procedure were not observed at all (e.g., no impacts were found under the resources depletion-related damage category and for several other impact categories). The dominant burdens are those related to impacts on the (quality of) ecosystems (especially in terms of aquatic eutrophication) and impacts on human health (notably by emissions of toxic substances).

What can be observed in the behavior of the environmental impact profiles related to the use of chemicals in wastewater treatment; there is a very strong dominance of burdens resulting from the depletion of fossil fuel resources (almost 2/3 of the total impact), climate change directly affecting human health (about 15%), and ecosystems (about 9%). Such behavior of environmental impact profiles is characteristic for the dominant share of substances originating from the chemical industry (especially the petrochemical branch).

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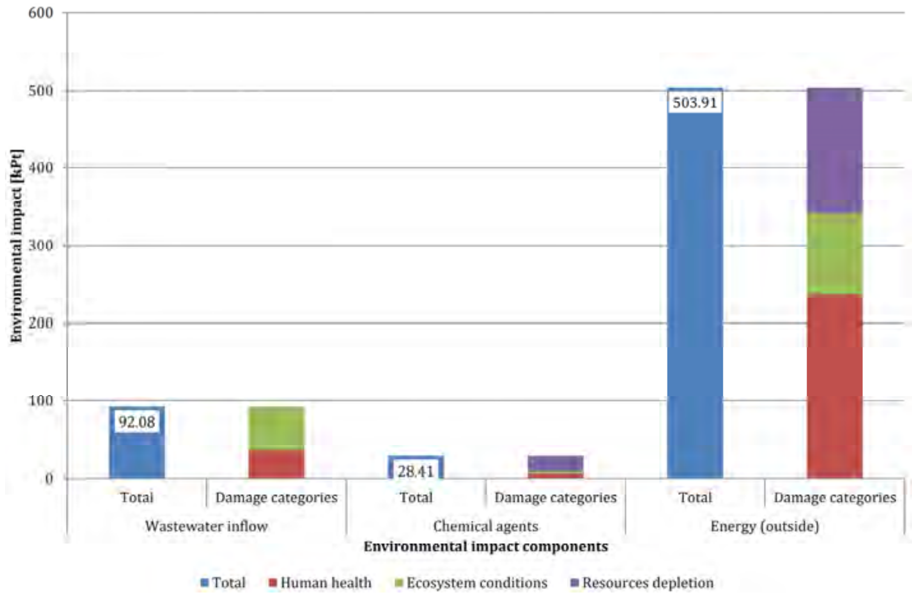

Figure 11. Breakdown of environmental impact components for inputs to the wastewater management processes at the CWWTP in 2019—Total environmental impact indicators and general environmental impact profiles broken down into damage categories.

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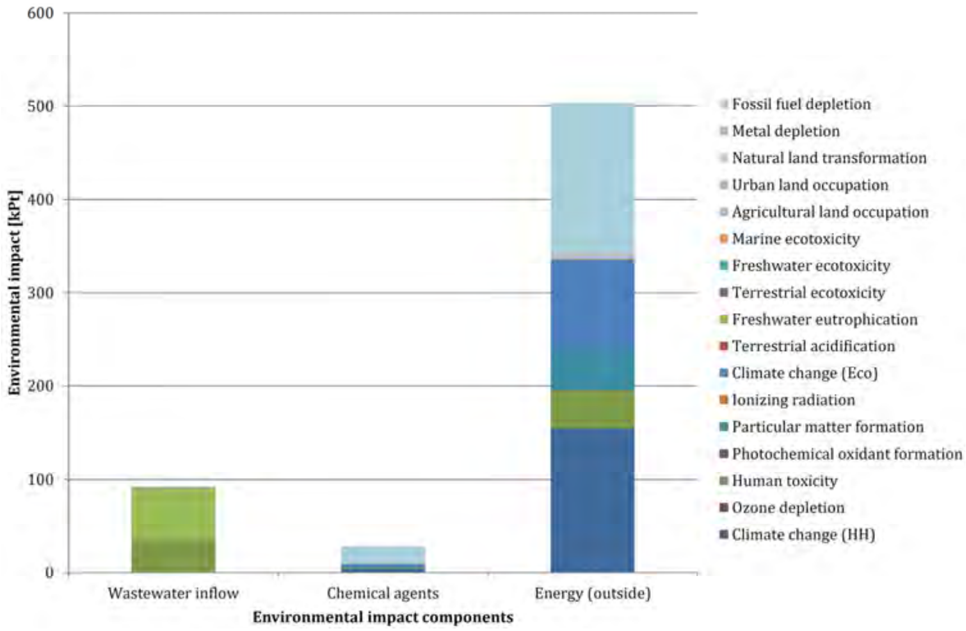

Figure 12. Breakdown of environmental impact components for inputs to the wastewater management processes at the CWWTP in 2019—Detailed environmental impact profiles broken down into impact categories.

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3.2.2. Components of Environmental Impacts for Outputs from Treatment Processes

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Similarly, as for the information presented above, Figure 13 presents the general values of environmental impact indicators and the general environmental impact profiles for wastewater treatment outputs. As presented in Section 2.2.3, the stream of treated wastewater directed to the receiver (environment), grit and screenings as derivatives of mechanical and physical separation, and energy produced on the basis of the plant's own sources (combustion of biogas produced in the anaerobic digesters (ADs) during sewage sludge digestion) were assumed to be direct process outputs. "Avoided" environmental impacts have also been assumed to be indirect outputs: reduction of concentrations (characteristic parameters) of substances (the difference between the concentrations at the process input and output) and emissions avoided as a result of energy production from own sources, and consequently avoidance of the corresponding energy consumption from external sources (produced on the basis of the mix shown in Figure 4).

Similarly, as for the information presented above, Figure 13 presents the general values of environmental impact indicators and the general environmental impact profiles for wastewater treatment outputs. As presented in Section 2.2.3, the stream of treated wastewater directed to the receiver (environment), grit and screenings as derivatives of mechanical and physical separation, and energy produced on the basis of the plant's own sources (combustion of biogas produced in the anaerobic digesters (ADs) during sewage sludge digestion) were assumed to be direct process outputs. "Avoided" environmental impacts

It can be seen that all direct outputs (except grit) are characterized by positive (and therefore harmful from the environmental point of view) values of impact indicators. This state of affairs is commented fully in Section 4, with an emphasis on the reasons and consequences.

have also been assumed to be indirect outputs: reduction of concentrations (characteristic parameters) of substances (the difference between the concentrations at the process input and output) and emissions avoided as a result of energy production from own sources, and consequently avoidance of the corresponding energy consumption from external sources (produced on the basis of the mix shown in Figure 4).

It can be seen that all direct outputs (except grit) are characterized by positive (and therefore harmful from the environmental point of view) values of impact indicators. This state of affairs is commented fully in Section 4, with an emphasis on the reasons and consequences.

Negative values of environmental impact indicators (reflecting environmental benefits) mainly result from "avoided" emissions (directly by the reduction of concentrations and indirectly by avoiding electricity consumption).

Regarding the general and detailed (Figure 14) environmental impact profiles of WWT outputs, some of the profile components have already been discussed (e.g., wastewater concentrations and electricity produced based on the mix shown in Figure 4). It should be noted that the shape of the environmental impact profiles for energy produced based on sewage sludge fermentation in the SFC differs to some extent from the shape of impact profiles for energy generated based on the Polish energy mix. Compared to this, the environmental impact profiles for energy produced based on the plant's own sources exhibits a significantly lower share of impacts related to climate change (in both aspects) and depletion of fossil fuel resources, while the share of impacts related to the emission of substances toxic to humans (from 8 to 19% of the total) and formation of particulate matter (from 8 to 14% of the total) increases.

The environmental impact profiles for mechanically separated solid waste (screenings, grit) are dominated by impacts related to greenhouse gas emissions (in both aspects), depletion of fossil fuel resources, and emissions of particulate matter, which is indicative of the energy intensity of the separation processes. Additionally, impacts associated with the transformation of natural land is clearly visible and also account for a significant portion of the general impact (about 10%), because grit resulting from mechanical separation can be managed whole (e.g., used for construction purposes), which in turn makes it unnecessary to source a similar amount of this raw material directly from the environment.

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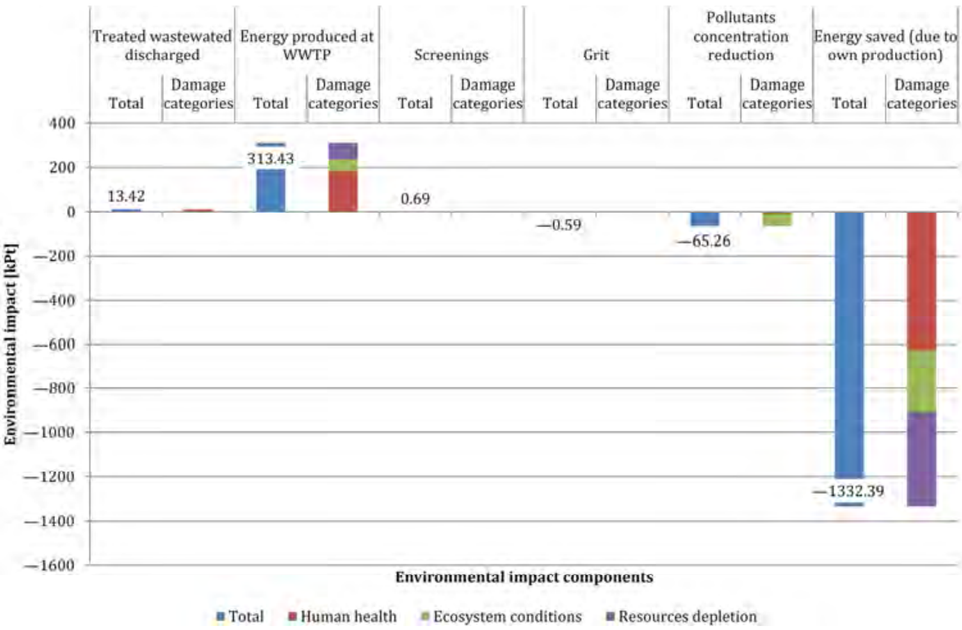

Figure 13. Breakdown of environmental impact components for outputs from the wastewater management processes at the CWWTP in 2019—Total environmental impact indicators and general environmental impact profiles broken down into damage categories.

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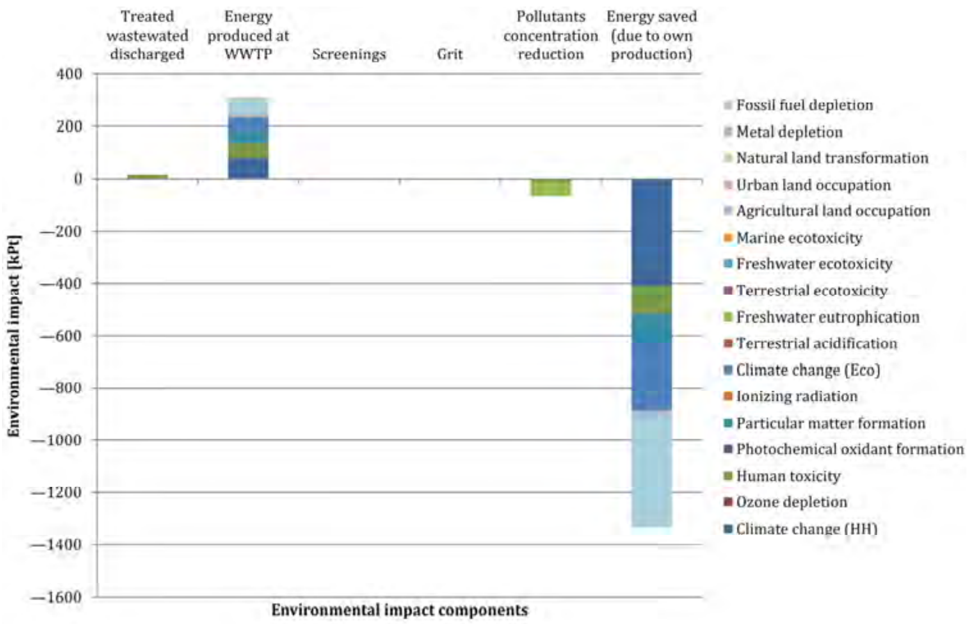

Figure 14. Breakdown of environmental impact components for outputs from the wastewater management processes at the CWWTP in 2019—Detailed environmental impact profiles broken down into impact categories.

3.3. Analysis Results—Detailed Environmental Impact Profiles

Basic results of the characterization stage for wastewater management processes at the CWWTP for the 2019 data are described below. The characterization is the first stage in life cycle assessment, in which all interactions with the life cycle environment of the analyzed facility, or process, as in this case, are classified and divided into so-called impact categories (for the ReCiPe procedure used in this analysis, the list of impact categories is presented in Section 2.2.1). Each impact category represents a group of environmental impacts with a common mechanism of interaction with the environment. A common physical impact unit (the so-called equivalent) can be established for them, such as, e.g., kg CO_2 eq. (equivalent kg of CO_2 emissions) for an impact category relating to climate change, i.e., greenhouse gas emissions (for each substance in the group of greenhouse gases, the so-called CO_2 equivalent is determined, i.e., the relevant number of kg of CO_2 emissions that cause the same environmental harm as 1 kg of the substance in question).

3.3.1. Components of the Environmental Impacts for Inputs to the Treatment Process

Figure 15 shows a summary of the characterization stage results for inputs to the treatment process, with a breakdown into individual months of 2019. It should be noted that impacts are not evenly distributed over individual months; there is significant variability depending on the category under investigation. Impacts related to water ecotoxicity and ionizing radiation are characterized by the highest variability (range, 2.2–14.7%), whereas impacts related to eutrophication of water resources are the most stable (range, 7–10%). To analyze the causes of such a variety in environmental impact values for process inputs over individual months, a correlation with monthly values of flows together with the analysis of variability of characteristic parameters, consumption of energy, and chemicals would be needed.

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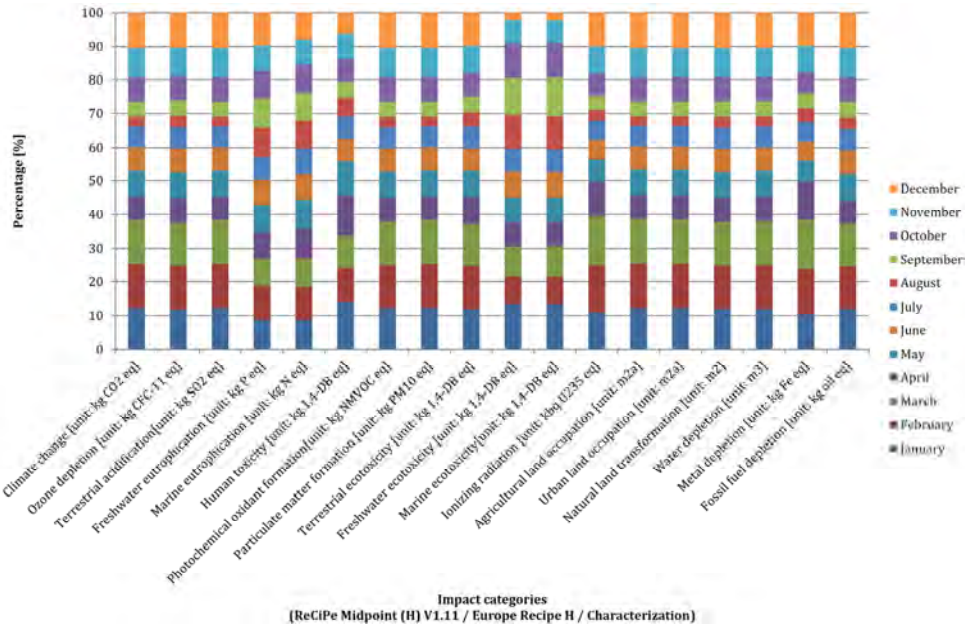

Figure 15. Results of the characterization stage for inputs to the wastewater treatment process at the CWWTP—A monthly perspective.

Figure 15. Results of the characterization stage for inputs to the wastewater treatment process at the CWWTP—A monthly perspective.

Figure 16 shows the results of characterization for inputs to the wastewater treatment process for the individual components. For most of the impact categories, the impact of the energy-intensive component is decisive. However, for four impact categories (related to eutrophication and water ecotoxicity), the impact of harmful substances contained in the wastewater stream, together with their parameters, prove particularly significant. The impact of chemicals used in the wastewater treatment processes is particularly significant for impacts related to the depletion of metal and fossil fuel resources and ionizing radiation and destruction of the ozone layer, but it never exceeds 50%.

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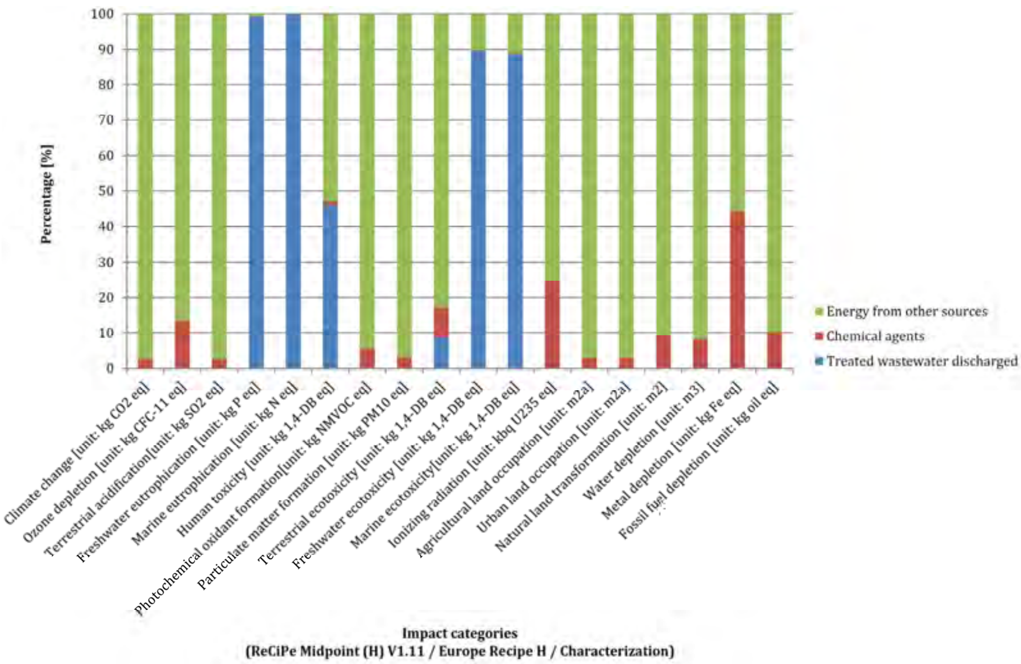

Figure 16. Results of the characterization stage for inputs to the wastewater treatment process at the CWWTP—Broken down into individual impacts.

Figure 16. Results of the characterization stage for inputs to the wastewater treatment process at the CWWTP—Broken down into individual impacts.

3.3.2. Components of Environmental Impacts for Outputs from the Treatment Process

Figure 17 shows a summary of the characterization stage results for outputs from the treatment process, with a breakdown into individual months of 2019. It is noticeable that, for most impact categories (regardless of the nature of the impact), the distribution of impacts over individual months follows a similar pattern, with the variability generally not exceeding 1% (with the exception of April and May, where variability within individual

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Only for two impact categories related to water ecotoxicity do the characterization results behave differently. Both positive and negative impacts fall within these impact categories and are the manifestations of:

- Environmental damage/burdens for the quantities characterizing the wastewater stream at the outlet to the receiver;
- Environmental benefits for quantities characterizing the reduction in the values of the characteristic parameters between the wastewater streams at the process inlet and outlet to the receiver.

For these two impact categories, the sign of the general environmental impact indicator changes over individual months; in January, February, June, August, October, and November, it takes negative values whereas, in the remaining months, it takes positive values. This state of affairs is related to the different volume structures of the individual impact components, particularly the values related to the stream of treated wastewater released to the receiver characterized by high variability (annual value: 13.4 kPt, monthly average value: 1.11 kPt, with a spread of 2.12 kPt).

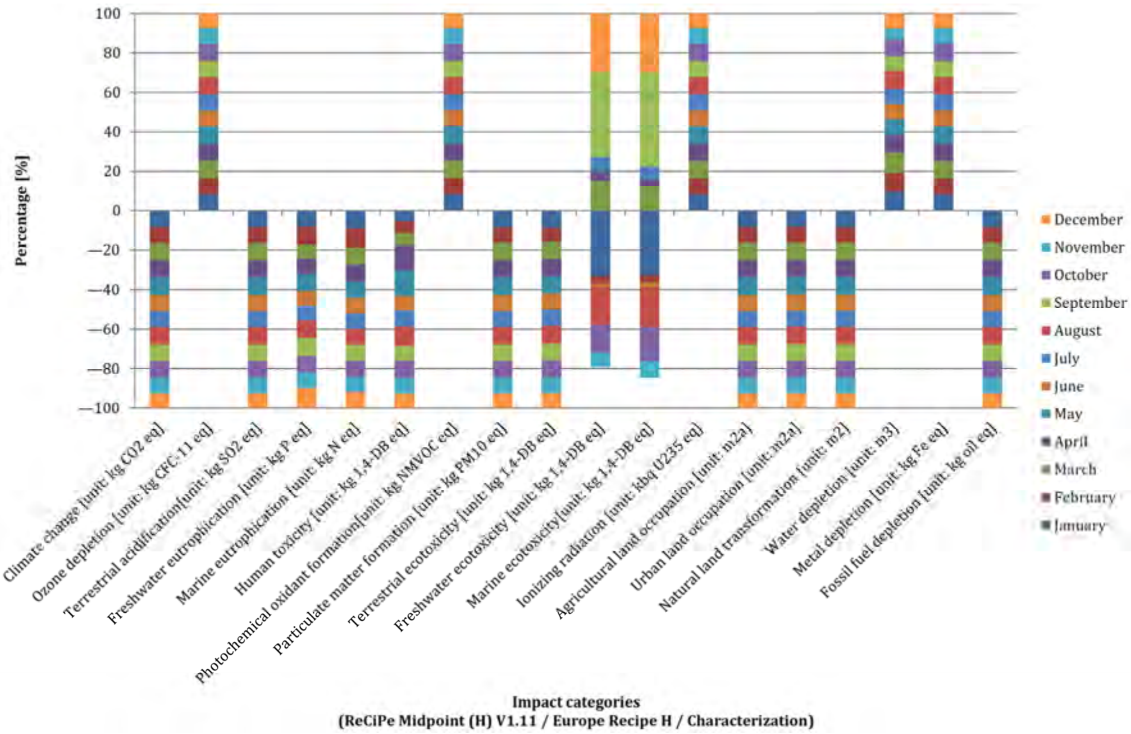

Figure 17. Results of the characterization stage for environmental benefits from the wastewater treatment process at the CWWTP—A monthly perspective.

Figure 17. Results of the characterization stage for environmental benefits from the wastewater treatment process at the CWWTP—A monthly perspective.

Figure 18 shows the results of characterization for outputs from the wastewater treatment process for individual components. Please note that, for most impact categories, it is the share of the energy intensity component that is decisive:

Figure 18 shows the results of characterization for outputs from the wastewater treatment process for individual components. Please note that, for most impact categories, it is the share of the energy intensity component that is decisive:

- In terms of direct environmental nuisance, the energy produced from the plant's own sources and the combustion of biogas generated during sewage sludge fermentation in bioreactors;
- In terms of potential environmental benefits, the avoided emissions associated with electrical power not consumed from external sources.

Only for the impact categories related to water eutrophication and ecotoxicity (fresh and marine water) could an indirect positive impact on the environment related to the

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- In terms of potential environmental benefits, the avoided emissions associated with electrical power not consumed from external sources.

Only for the impact categories related to water eutrophication and ecotoxicity (fresh and marine water) could an indirect positive impact on the environment related to the reduction in the burdens of the environmentally harmful substances contained in wastewater streams be observed.

For most impact categories, potential indirectly beneficial environmental impacts prevail. In three cases (ozone depletion, emission of ionizing radiation, and depletion of metal resources), the impact is clearly negative, while for four (formation of photochemical oxidants, ecotoxicity, and depletion of water resources), burdens and benefits are balanced.

3.4. Uncertainty Analysis—Elements

It is obvious in the analytical works involving the live cycle models, that all the data are characterized by different levels of uncertainty, mainly resulting from variation/instability of the data and possible incompleteness or representativeness of the model acquired. In order to avoid the possible ambiguity resulting from the data variability and uncertainty, Monte Carlo analyses have been applied. The small volume of this paper does not allow us to present the full results of the uncertainty analysis; however, an example for process inputs is shown in Figure 19. For the round of 1000 runs and a confidence interval of 95%, it was noticed that the calculated impact was 624,400 Pt (624.4 kPt—please see Figure 5), the mean was 625,000 Pt (dashed red line), the median was 621,000 Pt (solid red line), the standard deviation (SD) value was 58,700 Pt, the coefficient of variation (CV) value was 9.37%, and the standard error of mean was 0.00297.

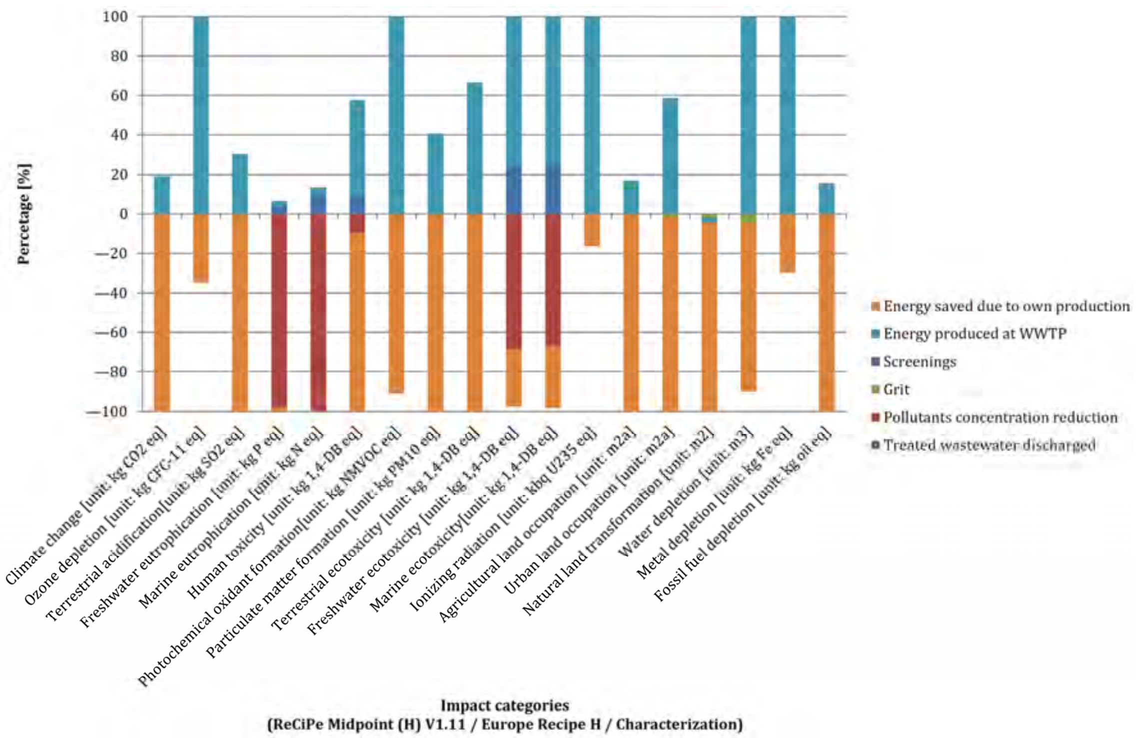

Figure 18. Results of the characterization stage for environmental benefits from the wastewater treatment process at the CWWTP—Broken down into individual impacts.

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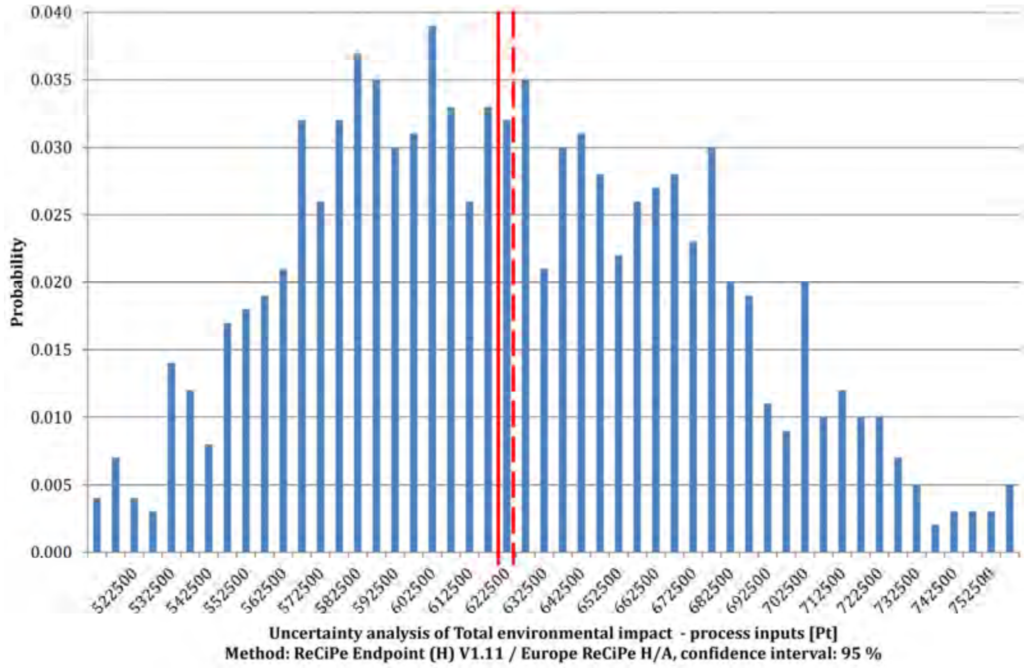

Figure 19. An example of the uncertainty analysis results for process inputs for the wastewater management at the CWWTP in 2019.

4. Discussion and Major Conclusions

The subject of the analysis included the environmental impacts arising from the implementation of the wastewater treatment process at the CWWTP (based on data from the period January–December, 2019). The analyzed data included inputs and outputs from the process under consideration, including:

- Levels of relevant parameter values characteristic of the wastewater streams: raw and treated before release to the environment;
- The chemical additives (chemical agents and reagents) needed for the process;
- Electricity supplied to the wastewater treatment process from the outside, modelled on the Polish electrical power generation mix (according to the structure shown in Figure 4);
- By-products of a wastewater treatment process seen as waste (screenings) or as a regular replacement for a product directly sourced from the environment (grit as sand), which can then be used in construction;
- Energy generated from the plant's own sources (combustion of biogas produced during sludge fermentation in bioreactors).

Moreover, two types of indirect environmental benefits associated with the implementation of the wastewater treatment process at the CWWTP were also taken into account. These were the impacts "avoided" as a result of:

- Reduction of burdens from the wastewater as a result of the treatment process (the difference between the values of characteristic parameters at process input and output);
- The absence of environmental burdens as a result of preventing the need for supplying (based on the above-mentioned energy mix) electrical power in an amount equal to that produced from the plant's own sources.

As mentioned previously, at the stage of the analysis presented in the study, only the method of sewage sludge management by processing it in bioreactors and producing biogas for energy purposes is taken into account; this process is simplified, and the amount of electricity produced was assumed to be its result.

The environmental impact levels (on a relative point scale and as results of characterization as physical units—see Section 3.3) were estimated in total (annual perspective), and monthly distribution was calculated for:

- Inputs to the wastewater treatment process;
- Outputs from the wastewater treatment process.

The analysis of the results obtained allows us to conclude that:

1. Estimated environmental impacts for the inputs to the wastewater treatment process are environmental loads. They take positive values for all components of the process inputs: the raw wastewater stream that fed into the process, the chemical reagents used, and the electricity supplied from external sources.
2. All direct outputs (except grit) are characterized by positive (and therefore harmful, environmentally) values of impact indicators. This is due to the following:
    - The wastewater stream at the process outlet leading to the receiving body (environment) contains certain quantities of characteristic, environmentally harmful, substances, and it is required by the relevant standards to determine their levels; these quantities have been significantly reduced (at least to the level required by law), but these substances are present in the treated wastewater, which, once directed to the environment, must be re-treated to be suitable for use;
    - Screenings (solid residues from mechanical separation processes) were modelled only as waste; the reason for adopting such a modelling approach was the impossibility of standardizing the composition of the screenings and the preliminary lack, at this stage of modelling, of options for unambiguous determination of the usefulness of these screenings, which would allow us to treat them as a fully-fledged by-product of the treatment process;
    - Energy from bioreactors is produced as a result of a physico-chemical process of sewage sludge fermentation in digesters, where a certain amount of biogas is released; this biogas is then incinerated and the energy obtained from this process is used in wastewater treatment processes. The share of energy from the plant's own sources varied, depending on the month, between 62 and 89% of the total energy used.
3. The estimated environmental impacts for the remaining wastewater treatment outputs and grit and intermediate outputs are environmental benefits (negative values assumed). For all the environmental benefits, this is due to the fact that there are so-called "avoided" impacts, i.e., indirect environmental benefits compared to obtaining an equivalent effect using primary environmental resources. This finding fully corresponds to the overall tendencies in the results obtained from other Polish studies [28–30]. For environmental benefits arising from the implementation of the wastewater management process in question, the "avoided" impacts concerned are:
    - Grit—All grit shown in Table 9 can be treated as a fully valuable product ready for use in the relevant industries (e.g., as a raw material in the construction industry), which in turn eliminates the need to obtain the same amount of sand from primary sources, i.e., from the environment;
    - Diminution of pollutant concentrations—The environmental benefit results from the fact that the measured values of parameters describing the content of harmful substances in wastewater are reduced from their top levels, characterizing the stream of raw wastewater at the process inflow to a lower level, and characterizing the stream of treated wastewater ready to be released to the receiver (environment); this parameter illustrates the amount of "avoided" environmental loads resulting from the wastewater with the original concentration of harmful substances not being released to the environment;
    - "Saved" energy—In this case, the environmental benefit results from the fact that the energy required for the wastewater treatment process, in an amount

equal to the amount of energy that was produced in the plant from its own resources, was not taken from external sources.

4. It was found, in the case of both process inputs and outputs, that the energy component, related to the energy intensity of the processes themselves, as well as the "avoided" environmental loads, is the main determinant of the magnitude of the generated impacts (the value of total environmental impact indicators). This finding supports the point of view of the case studies presented in [32]. Moreover, it can be noted that the amount of "avoided" emissions from electricity generated based on the modelled energy mix determines the negative value of the overall environmental impact indicator of the whole process (as the main factor, it determines the environmental friendliness of the process); the calculated overall environmental impact indicator of the process, including process inputs and outputs modelled according to the assumptions described in Section 2.2.2, is −446 kPt.
5. Analysis of the structure of environmental impacts included in the detailed impact profiles, characterization, and weighting results allows us to conclude that, for the energy-intensive component and the chemical agent used, loads connected with the following phenomena account for the largest share:
   - Depletion of fossil fuels;
   - Climate change (in terms of harm to humans and ecosystems);
   - Human toxicity;
   - The formation of particulate matter in the atmosphere.

   Furthermore, for the component related to the use of chemicals in wastewater treatment, depletion of metal resources also constitutes a significant burden.
6. With regard to the quantities of harmful substances tested (according to the set of parameters given in Tables 4–8) in raw and treated wastewater streams, a different distribution of environmental impact profiles was found. Only some impact categories are present there, of which the most significant ones are:
   - Eutrophication of freshwater: about 55% of the total impact for the raw wastewater stream and about 16% of the total impact for the treated wastewater stream;
   - Human toxicity: about 39% of the total impact for the raw wastewater stream and about 82% of the total impact for the treated wastewater stream,
   - Freshwater ecotoxicity: about 1–2% of the total impact.

   Besides these, yet another, though much smaller, impact was discovered, associated with marine and soil ecotoxicity.
7. With regard to the variability of the environmental impacts across individual months, it should be noted that:
   - Environmental loads from wastewater treatment inputs are subject to considerable variability over individual months. The discrepancies reach 300%; the top load level was recorded in February (76.7 kPt) and the lowest in August (24.3 kPt);
   - Environmental impacts from wastewater treatment outputs are characterized by much lower variability (differences between individual months do not exceed 20%).
8. Due to the fact that, in previous publications, available system boundaries and analysis assumptions, as well as the choice of functional units, were done arbitrarily, straightforward comparisons between the results are possible; nevertheless, tendency evaluation still can be done:
   - The major inference based on the statement that the energy component, related to the balance between WWT process demand and electricity production, is the main contributor to the environmental impact indicator sum remains, along with conclusions made by other authors [1,3,6,30];

- ○ Moreover, the statement made in this paper that the energy mix for the so-called conventional electricity source has the most significant contribution to the overall environmental impact indicator of the whole process is valid, and has been noticed, not only in the area of municipal WWTs, but also in industrial facilities [40],
- ○ Interestingly, it has been confirmed that the LCA results reveal concise areas of environmental optimization for large WWTPs (>50,000 PE), which was also noticed in the results of a case study done on 113 treatment facilities across Spain [9]; bigger facilities, although maintaining high treatment efficiency standards, present poorer environmental profiles than the plants that have a lack of careful continuous monitoring infrastructure. Therefore, LCA appears to be an even more crucial evaluation tool for scheduling future and holistic-approach optimization scenarios for WWTP significant stream volumes.

9. For further analysis stages, we postulate that this study's limitations must be considered. Thus, the following is necessary:
    - ○ Clarifying the wastewater treatment model and modelling the flow of environmental impacts between individual process stages;
    - ○ Taking into account inputs and outputs omitted at the described stage, output modelling of sewage sludge, and perhaps a separate environmental analysis of sludge use.

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Paulina Szulc-Kłosińska
Imię i nazwisko doktoranta

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Wykaz autorów	Wkład procentowy	Podpis autora
Paulina Szulc-Kłosińska	**40%**	Szulc Kłosińska
Jędrzej Kasprzak	30%	
Zbysław Dymaczewski	15%	
Przemysław Kurczewski	15%	Kurczewski

Wkład doktoranta:

Przygotowanie koncepcji i pierwszej wersji artykułu, opracowanie metodyki, walidacja i analiza wyników, korekty i opracowanie finalnego tekstu

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Algorithm of carbon footprint calculation for municipal wastewater treatment plant – part one

Algorytm obliczania śladu węglowego miejskiej oczyszczalni ścieków – część pierwsza

Paulina Szulc-Kłosińska, Zbysław Dymaczewski*)

Keywords: *GHG emissions, nitrous oxide, methane, carbon footprint, wastewater treatment, CSRD*

Abstract
In the forthcoming years, urban wastewater management utilities will be required by the European Union to perform CF calculations in accordance with the Corporate Sustainability Reporting Directive (CSRD) and European Sustainability Reporting Standards (ESRS) indicators. Yet, no standardized approach that expressly addresses the rules for WWTPs in respect to GHG emissions, giving the water bodies a clear instruction to calculate their CF is given. This paper provides an in-depth examination of the present approaches for calculating GHG emissions. An algorithm for calculating the carbon footprint of a wastewater treatment facility is developed and described in detail by the authors. Furthermore, the research evaluates the extent to which facility data is complete and suggests remedies to any detected information gaps. A data enhancement strategy is also offered. The primary goal of this research is to bridge a knowledge gap in the understanding of the carbon footprint associated with WWTPs and their organisational framework. The analysis also included a thorough investigation into the significance and sources of GHG Protocol Scope 1 (part one arcticle), 2, and 3 emissions (part two article) within the larger framework of carbon footprint, particularly in relation to the legislative goals of CSRD reporting with its upcoming obligations imposed on waterworks organizations.

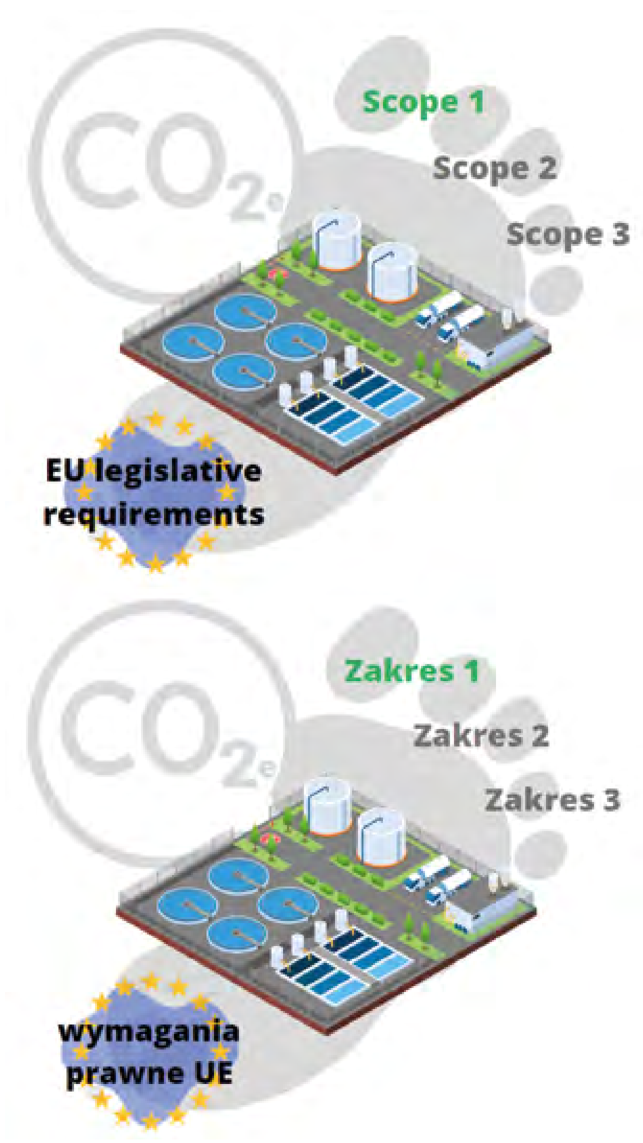
Słowa kluczowe: *Emisje gazów cieplarnianych, podtlenek azotu, metan, ślad węglowy, oczyszczanie ścieków, CSRD*

Streszczenie
W nadchodzących latach Unia Europejska będzie wymagać od miejskich zakładów oczyszczania ścieków wykonywania obliczeń śladu węglowego (CF) zgodnie z Dyrektywą w sprawie sprawozdawczości dotyczącej zrównoważonego rozwoju przedsiębiorstw (CSRD) i wskaźnikami Europejskich Standardów Sprawozdawczości dotyczącymi Zrównoważonego Rozwoju (ESRS). Nie istnieje jednak żadne ustandaryzowane podejście, które wyraźnie odnosiłoby się do zasad dotyczących oczyszczalni ścieków w odniesieniu do emisji gazów cieplarnianych (GHG), dając organom wodnym jasne instrukcje dotyczące obliczania ich CF. Niniejszy artykuł zawiera dogłębną analizę obecnych podejść do obliczania emisji GHG. Algorytm obliczania śladu węglowego miejskiej oczyszczalni ścieków został opracowany i szczegółowo opisany przez autorów. Ponadto, w analizie oceniają zakres, w jakim dane dotyczące oczyszczalni ścieków są kompletne i sugerują środki zaradcze dla wszelkich wykrytych luk informacyjnych. Zaproponowano również strategię ulepszania danych. Głównym celem tego badania jest wypełnienie luki w wiedzy na temat śladu węglowego związanego z oczyszczalniami ścieków i ich ramami organizacyjnymi. Analiza obejmowała również dokładne zbadanie znaczenia i źródeł emisji z wg podziału na zakresy GHG Protocol 1 (część pierwsza), 2 i 3 (część druga) w szerszych ramach śladu węglowego, szczególnie w odniesieniu do celów legislacyjnych raportowania CSRD z nadchodzącymi obowiązkami nałożonymi na przedsiębiorstwa wodociągowe.

1. INTRODUCTION

In light of the escalating apprehension surrounding the phenomenon of global warming, numerous nations have made a steadfast commitment to attaining net-zero emission status by the year 2050 [44], ful lling Intergovernmental Panel on Climate Change (IPCC) guidelines of not surpassing the 1.5°C threshold [38]. The wastewater treatment plants (WWTPs) have been identi ed as a signi cant source of direct greenhouse gas (GHG) emissions, especially N_2O and CH_4, accounting for approximately 3% of GHG emissions on a global [41, 56]. In addition, when considering indirect emissions, it is evident from an energy standpoint that WWTPs continue to consume a signi cant amount of energy, still primarily derived from fossil fuel sources. Given the current circumstances of the prevailing focus on climate change, escalating electricity costs, and the imperative to reduce carbon footprint (CF) emissions within an organisation or activity, the wastewater treatment facility is regarded as a dual prospect, presenting both challenges and opportunities for enhancement. As a nal result, water utilities are at the forefront of e orts to mitigate GHG emissions, with a number of waterworks organisations setting their sights on achieving net-zero emissions in their operations within the upcoming decades [6, 7, 37, 47]. The very

*) **Paulina Szulc-Kłosińska,** Mgr inż. email: paulina.szulc-klosinska@bureauveritas.com Sustainability Consultant at Sustainability Department, Bureau Veritas Polska Sp. z o.o. ul. Migdałowa 4, 02-796 Warszawa; PhD student at Faculty of Environmental Engineering and Energy, Poznan University of Technology, Berdychowo 4, 60-965 Poznan, Poland, **Zbysław Dymaczewski**, Dr hab. inż., prof. PUT, Faculty of Environmental Engineering and Energy, Poznan University of Technology, Berdychowo 4, 60-965 Poznan, Poland, e-mai zbyslaw.dymaczewski@put.poznan.pl



GRAPHICAL ABSTRACT

first step of building a GHG reduction strategy is the annual CF calculation enabling the transparency of comparison of the results between the facilities and other economic areas.

In the forthcoming year, the primary urban water management utilities will be required to perform CF calculations in accordance with the Corporate Sustainability Reporting Directive (CSRD) and European Sustainability Reporting Standards (ESRS) indicators by European Financial Reporting Advisory Group (EFRAG). However, it is worth noting that there is currently no established methodology that specifically addresses the guidelines for WWTPs in relation to GHG emissions which would give the water bodies a clear instruction to calculate their CF. The obligations of CSRD require adherence to the reporting practise of the Greenhouse Gas Protocol (GHG Protocol) regime, as mandated by the European Union (EU). Consequently, water utilities are obligated to comply with these requirements.

Several studies have been conducted to analyse the GHG emissions associated with different wastewater and sludge treatment technologies. However, these studies have primarily focused on specific aspects of wastewater treatment configurations rather than examining the entire facilities with their technological configuration [34, 35, 57] in order to deliver a simplified CF calculation method. In many cases evaluation of the WWTP's CF was done as the final step of in-situ off-gas measurements to cover direct GHG emissions from BNR treatment stage, sometimes combined with model-based approach [39]. Prior research has indicated that the utilisation of distinct wastewater treatment methodologies may lead to varying levels of direct GHG emissions, as well as differing degrees of energy consumption and sludge generation [10, 24, 41, 42, 51, 58]. Piao et al. (2016) [41] found that the implementation of the Modified Ludzack-Ettinger (MLE) process in wastewater treatment facilities resulted in a 20% increase in the carbon footprint, as compared to facilities utilising the Anaerobic-Anoxic-Oxic (A2/O) process. In addition, it has been found that the implementation of aerobic wastewater treatment methods may result in a significant increase of 105% in direct GHG emissions, as compared to the utilisation of anaerobic wastewater treatment techniques [10].

Nevertheless, there remain uncertainties regarding the comprehension of carbon footprints associated with various configurations of WWTPs. Previous analyses did not take into account several potentially significant sources of GHG emissions, particularly in the context of sludge handling processes focusing only on the chosen emission-related topic rather than on the entire global warming enhancing gases fluxes.

Goal and scope of the work presented in the two-part paper

The overall aim of this work is to provide a valuable resource for researchers, policymakers, and practitioners involved in the assessment and management of the carbon footprint of wastewater treatment plants. By shedding light on the role of evaluating direct GHG emissions in the carbon footprint structure of wastewater treatment facilities and providing guidance on MWWTP's CF estimation, the study can contribute to the development of more accurate and comprehensive carbon footprint assessments and support the efforts to mitigate climate change by delivering the municipal wastewater treatment plant CF calculation algorithm.

Moreover, the research investigates the extent to which facilities data is complete and proposes solutions to address any identified gaps in information regarding the understanding of the carbon footprint associated with WWTPs [52] and their organisational framework. Additionally, a data improvement plan is presented. The analysis also involved a comprehensive investigation into the importance and sources of Scope 1, 2, and 3 emissions (GHG Protocol) within the wider framework of carbon footprint, particularly in relation to the legislative goals of CSRD reporting with its obligations put on waterworks organisations in the near future.

2. CF IN WASTEWATER TREATMENT – EU LEGISLATIVE BACKGROUND

The European Climate Law (Regulation (EU) 2021/1119 of the European Parliament and of the Council of 30 June 2021 establishing the framework for achieving climate neutrality and amending Regulations (EC) No 401/2009 and (EU) 2018/1999) [43], which writes into law the goal set out in the European Green Deal, establishes the overarching structure for the European Union's involvement in the Paris Agreement. This includes both the specific objectives to be achieved and the procedural mechanisms to be employed. The legislation establishes enforceable objectives of a legally binding nature.

The main objective is to achieve a minimum reduction of 55% in GHG emissions relative to the levels recorded in 1990 by the year 2030 (so-called 'Fit for 55' package). Additionally, the aim is to attain net-zero GHG emissions by the year 2050. The achievement of the climate-neutrality objective necessitates the implementation of appropriate measures by the relevant institutions of the European Union and its Member States.

The Corporate Sustainability Reporting Directive, known as CSRD (EU Directive 2022/2464 of the European Parliament and of the Council of 14 December 2022) [23] came into effect on January 5th, 2023, as a part of Green Deal EU legislative tools. This regulation extends the scope and reporting requirements of the already existing Non-Financial Reporting Directive (NFRD; Directive 2014/95/EU of the European Parliament and of the Council of 22 October 2014) [21] – a regulatory framework that mandates sizeable public interest entities to report on their sustainability performance since 2018. The revised CSR directive aims to enhance and update the regulations pertaining to the disclosure of environmental and social data by corporations, thereby enhancing their effectiveness and robustness. A more extensive range of sizable corporations, along with publicly listed small and medium-sized enterprises (SMEs), are mandated to disclose information regarding their sustainability practises. This requirement will encompass approximately 50,000 companies in aggregate, involving waterworks organisations across EU. CSRD will be introduced in three planned phases (Scheme 1). The initial group of companies, including water bodies of the biggest urban areas, will be required to implement the newly established regulations for the inaugural occasion during the fiscal year of 2024, with the corresponding reports being made public in 2025.

Scheme 1. Summary of CSRD implementation phases.

Schemat 1. Podsumowanie faz wprowadzania CSRD w życie.

The implementation of the new regulations will guarantee that investors and other relevant parties are provided with the necessary access to information in order to evaluate investment risks that may arise from climate change, including WWTPs' CF in three scopes, and other matters related to sustainability. Additionally, they will foster a culture that promotes transparency regarding the societal and environmental effects of corporations. In the medium to long term, companies will experience a reduction in reporting costs through the harmonisation of the information required.

Organisations that fall under the jurisdiction of the CSRD will be obligated to adhere to the European Sustainability Reporting Standards (ESRS) when submitting their reports. The draught standards are formulated by the EFRAG, formerly recognised as the European Financial Reporting Advisory Group, an autonomous entity that convenes diverse stakeholders. The proposed standards will be customised to align with the policies of the European Union, while also drawing upon and making contributions to ongoing international standardisation efforts. On the 6th of June, the Commission initiated a four-week period for public feedback regarding the initial set of sustainability reporting standards intended for companies. The draught standards have incorporated technical advice provided by EFRAG in November 2022. The most recent draft of the ESRS [20] necessitates the disclosure of environmental data pertaining to five distinct domains, encompassing climate-related aspects such as adaptation and mitigation. The following examples illustrate CSRD disclosures, specifically focusing on GHG emissions. The numerical values of Scopes 1, 2, and 3, as well as the comprehensive assessment of carbon footprint emissions will be therefore required to present. Additionally, it pertains to the formulation of climate change policy and the development of a strategy aimed at reducing greenhouse gas releases. The ultimate objective of this strategy is to attain a state of net-zero emissions prior to the year 2050.

The evaluation process of carbon footprint, given its comprehensive nature, necessitates extensive data collection efforts across various contexts to identify information gaps. However, the results of CF assessments, which encompass the emission structure of direct activities of wastewater treatment plants (WWTPs), detailed analysis of energy consumption and production schemes, and examination of value chain activities, can also serve as a valuable source of additional knowledge for optimising overall facility operations. Additional European Union legislative instruments related to wastewater treatment may provide the comprehensive data necessary for compliance with CSRD.

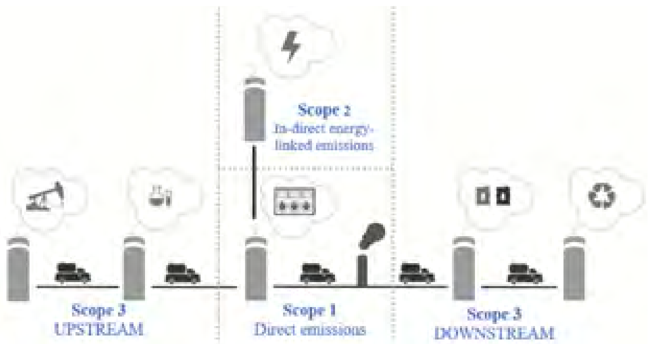
The current regulatory framework, specifically the Urban Waste Water Treatment Directive (UWWTD, 91/271/EEC) and the Sewage Sludge Directive (SSD, 86/278/EEC), does not explicitly consider the matter of GHG emissions, such as nitrous oxide and methane fluxes or, namely, WWTP carbon footprint, in relation to the management and utilisation of wastewater and sewage sludge. The successful implementation of UWWTD over a period of approximately thirty years has resulted in significant reduction of methane emissions through the utilisation of efficient centralised infrastructure for the collection and treatment of wastewater. In contrast to alternative treatment methods, these facilities demonstrate a significant decrease in the release of GHG.

The Sewage Sludge Directive (SSD), which was introduced more than three decades ago, regulates the utilisation of sewage sludge to protect the environment, specifically the soil, from the harmful effects of contaminated sludge when used in agriculture. The ongoing evaluation of the UWWTD is presently in progress. Simultaneously with the assessment of the aforementioned directive, the European Commission commenced a study in the third quarter of 2020 to facilitate the evaluation of regulations pertaining to sewage sludge. Moreover, a supplementary inquiry will be undertaken to assess the feasibility of implementing further actions pertaining to GHG emissions, specifically focusing on methane emissions originating from sewage sludge. Based on the findings of the assessment of the SSD, along with supplementary inquiries and the evaluation of the effects of the amendment to the urban UWWTD, the Commission will consider the adoption of measures aimed at limiting the emission of greenhouse gases derived from sewage sludge.

The proposed Directive of The European Parliament and of The Council, dated October 26th, 2022 (2022/0345) [22], aims to amend the existing UWWTD. According to the proposal, it is crucial for Member States to ensure the regular conduct of energy audits on urban wastewater treatment plants and collecting systems every four years. The audits have a broad range of coverage, which includes the identification of potential avenues for the cost-effective utilisation or production of renewable energy. The primary focus of the audits will be to prioritise the identification and optimal utilisation of biogas production capacity, while simultaneously addressing the reduction of methane emissions. The deadline for compliance with regulations pertaining to urban wastewater treatment plants and their associated collecting systems is contingent upon the specific load of each plant. Wastewater treatment facilities with a capacity of 100,000 population equivalents (p.e.) or higher are required to meet compliance standards by December 31st, 2025. Conversely, facilities treating a load ranging from 10,000 p.e. to 100,000 p.e. are granted an extended compliance deadline until December 31st, 2030.

3. CORPORATE CARBON FOOTPRINT – GHG PROTOCOL GUIDELINESS

The Greenhouse Gas Protocol (GHG Protocol) developed by the World Resources Institute (WRI) and the World Business Council for Sustainable Development (WBCSD) was responsible for the initial step that involves the categorization of GHG emissions according to the three different scopes in 2004 (Scheme 2).



Scheme 2. GHG Protocol emission Scopes 1, 2, 3.
Schemat 2. Zakresy emisji wg GHG Protocol.

Scope 1 encompasses the direct GHG releases originating from facilities that are directly managed by the reporting entity. Therefore, the first scope, is recognised as crucial in the case of the WWT-specific emissions. The compounds generated during the treatment of wastewater and sludge are specifically N_2O and CH_4. Nitrous oxide primarily originates from the BNR procedures employed in the context of wastewater treatment. In contrast to N_2O, CH_4 typically originates within the sewer system and specific areas of the WWTP where anaerobic conditions prevail, such as anaerobic wastewater and sludge treatment processes [16]. According to the IPCC AR6 Report (IPCC, 2021), the greenhouse gases N_2O and CH_4 possess global warming potentials (GWPs) of 273 and up to 29.8 carbon dioxide equivalents (CO_2e), respectively. As a result, even minimal emissions of N_2O and CH_4 from a wastewater treatment process make a substantial contribution to its carbon footprint. According to De Haas and Hartley (2004) [29], it was approximated that a conversion of 1% of nitrogen loading to N_2O could result in a 30% increase in the overall carbon footprint of wastewater treatment plants. Scope 2 covers the emissions linked to the externally purchased streams of energy such as: electricity heat, steam and cooling [4]. Third scope indicates all indirect emissions that can be linked to the reporting entity activities that arise from assets that are not under the ownership or control of the organisation responsible for reporting in their value chains. Scopes 2 and 3 will be further described in the following part two article.

The GHG Protocol methodology is applicable to various industries and is widely acknowledged as a set of principles for calculating the carbon footprint of organisations, particularly for the purpose of external reporting. However, as forementioned, it does not provide specific guidelines for WWTPs. The GHG Protocol acknowledges that there are limitations in its methodology when it comes to reporting entities that own WWT facilities. As a result, it recommends referring to the IPCC (2006 and 2019) as a supplementary resource to address these gaps [30]. The GHG Protocol Scope 3 Calculation Guidance (2013) [55] suggests that for the purpose of emissions calculation, it is advisable for the reporting entity to gather emission data from waste treatment, including wastewater. The Protocol places a significant emphasis on the fact that greenhouse gas emissions originating from wastewater exhibit considerable variability, which is contingent upon the extent of processing required for water treatment. This variability is determined by the levels of nitrogen loads and, as well, biochemical oxygen demand (BOD) and/or chemical oxygen demand (COD) characterising the facility's compresence. In summary, GHG Protocol should be used as an overall CF calculation guidelines package accompanied by detailed methodology.

4. CURRENT STATUS OF CARBON FOOTPRINT METHODOLOGIES IN WWT

Intergovernmental Panel on Climate Change (2006 with 2019 Refinement) [30]

As previously mentioned in the Introduction, the methodology employed by the IPCC for calculating methane emissions is widely recognised as one of the most popular approaches within the field of wastewater treatment. Established originally in 1996, actualised in 2006, then updated in 2019 – mainly with the novel guidelines both for N_2O and CH_4 emissions included based mainly on the full-scale measurements.

The methodology encompasses three distinct tiers, which are utilised for the estimation the greenhouse gas effect, gas fluxes resulting from the treatment process, as well as the discharge of treated wastewater. These tiers are outlined in Table 1. The application of Tier 1 is appropriate in cases where there is a lack of available activity data. Tier 2 is employed when there are some country-specific factors that are present. Finally, Tier 3 is selected when all country-specific factors have been collected. The calculations for all these Tiers are derived from specific methane and nitrous oxide emission rates that are associated with biological oxygen demand (BOD), chemical oxygen demand (COD) or specific N-fraction loads respectively. These often are adjusted using specific corrective factors tailored to each particular situation.

Table 1. Presentation of IPCC (2019) methodology Tiers for assessment of WWT GHG emissions.
Tabela 1. Prezentacja metodologii IPCC (2019) do oceny emisji gazów cieplarnianych z OŚ.

IPCC Tier GHG emissions calculation methodology (2019)	Treatment process	Discharge
Tier 1	Estimate emissions from treatment using default emission factors and methodology.	Estimate emissions from discharge to all aquatic environments using default emission factor and methodology.
Tier 2	Estimate emissions using country-specific emission factors and/or activity data and default methodology.	Estimate emissions from discharge to aquatic environments using default emission factors dedicated to the type of the recipient and methodology
Tier 3	Estimate emissions from treatment using country-specific method based on measured emissions data from facilities.	Estimate emissions from discharge to specific types of aquatic environments using country-specific emissions data and methodology.

The utilisation of default emission factors (Tier 1) is recommended by the IPCC for estimating CH_4 and N_2O emissions from wastewater treatment plants in GHG inventories, particularly in cases where data is scarce, as observed in many developing countries. Nevertheless, the accuracy of these estimations may be signi cantly imprecise due to the limited availability of dependable data regarding the functioning of the treatment technology itself and the speci c environmental factors in the area. There is lack of locally measured methane and nitrous oxide emission factors from MWWTPs still observed, resulting in a low accuracy, especially in the case of the CH_4 emission estimates. Currently, the veracity of the national emission estimations derived from the Tier 1, IPCC Guidelines for wastewater treatment and discharge (Vol. 5, Chapter 6; IPCC, 2019) [30] is constrained by various factors: the exclusion of CH_4 emissions from closed sewer systems and dissolved CH_4 in the in uent from WWTP is not accounted for. Furthermore, the appropriate approach for addressing aerobic treatment systems supplemented with anaerobic sludge digesters under Tier 1 is unclear. Additionally, it is imperative to acknowledge the anaerobic-aerobic process for nutrient removal as a novel addition to the repertoire of treatment systems.

Australian National Greenhouse Accounts/ National Greenhouse and Energy Reporting (2023 update) [14, 15]

Australia has implemented two primary protocols, namely the National Greenhouse and Energy Reporting (NGER) and the National Greenhouse Accounts (NGA) guidelines. The NGER framework encompasses corporations that surpass speci c emission thresholds to meet Australia's overall international reporting obligations. The emission factors and instructions provided by the NGA are designed speci cally for use by corporations and individuals (NGA, 2013). As the result, the methodologies employed by the NGA exhibit a higher level of precision in their de nition, as they are designed to be applicable to a narrower scope being tailored to encompass a wider range of emissions and sources. Due to the increased level of sophistication and the corresponding abundance of wastewater-speci c methodologies, the NGA and its methodologies are generally more pertinent compared to the NGER standards. Australian methodologies provide three di erent methods of emission calculations depending on the data access (Table 2).

Table 2. Presentation of NGA/NGER (2023) methodology: Methods for assessment of WWT GHG emissions.

Tabela 2. Prezentacja metodologii NGA/NGER (2023): Metody oceny emisji gazów cieplarnianych z OŚ.

NGER/NGA Calculation methods (2023)	WWT emissions
Method 1	Estimate emissions from using default emission factors – based on national average estimates.
Method 2	Estimate emissions using facility specific method based on industry practices for sampling and Australian or equivalent standards for analysis.
Method 3	Estimate emissions using facility specific method based Australian or equivalent standards for both sampling and analysis.

The topic of methane emissions from wastewater collection systems is not addressed in either the NGER or the NGA. However, it is worth noting that the estimation of methane uxes from the initial stages of treatment, speci cally the mechanical stage, is based on levels of chemical oxygen demand (COD) reduction. There is a lack of guidelines speci cally addressing the release of CH4, as the existing methodology assumes that well-managed treatment systems do not emit methane.

The emissions of CH_4 associated with sludge management are extensively debated within the eld of biogas production through AD. The biogas production stream can be classi ed into three distinct categories: captured, ared, and transferred. The accounting of AD fugitive emissions is currently lacking. Default emissions factors are provided for the combustion of biogas on-site.

The calculation of N_2O emissions is conducted using a population-based approach that incorporates the average protein intake and assumes a certain percentage of nitrogen composition. Hence, the emissions originating from the BNR stage exhibit a correlation with the N-removal efficiency. Additionally, in the context of N_2O emissions associated with the effluent, the N-concentration in the treated wastewater stream plays a role, contingent upon the recipient type, as determined b xed emission factors [54].

California Air Resources Board's Local Government Operations Protocol (2010) [11]

The Local Government Operations Protocol (LGOP) owned by California Air Resources Board (CARB) is designed to establish a standardised protocol for local governments, such as cities and counties, within USA, to accurately report emissions resulting from their own operational activities. This endeavour speci cally omits the emissions produced by the general population residing in those identical jurisdictions that are not subject to direct governmental authority.

The protocol developed by the LGOP exclusively addresses the topic of methane emissions in relation to the decomposition of sludge biosolids within land ll management practises. In the context of AD, the methodology consistently operates under the assumption that the entire biogas generated is captured and subsequently combusted with a 99% efficiency rate, thereby preventing any methane emissions.

N_2O emissions, as determined by the LGOP methodology, originate from BNR systems and solids decomposition processes. The calculations are derived from established population-based plant emission factors, which have been categorised separately for nitri cation/denitri cation and compartments without nitrogen removal.

U.S. Community Protocol for Accounting and Reporting of Greenhouse Gas Emissions by International Council for Local Environmental Initiatives – Local Governments for Sustainability USA (2013) [48]

The International Council for Local Environmental Initiatives (ICLEI) standards pertain to local governments operating within the United States. The protocol's stated objectives encompass informing climate action planning and fostering community engagement, monitoring GHG emissions trends, facilitating comparisons among similar communities, enabling the aggregation of regional GHG emissions data, demonstrating compliance with regulations, agreements, and standards, as well as showcasing accountability and leadership. The U.S. Protocol encompasses government operations such as the LGOP, as well as emissions originating from the general public.

There are 14 basic and 6 alternative methods of both CH_4 and N_2O calculation in the area of wastewater treatment given by the U.S. Protocol. Each dedicated to the di erent GHG source complied with the treatment technology, type of the gas, emission type and available data. Table 3 summarizes methods dedicated to CH_4 emissions while Table 4 presents N_2O emissions calculation methods. As presented in the Tables, U.S. Protocol considers CH_4 and N_2O emission in the entire value chain of WWT, giving methods to calculate methan uxes in the lifecycle.

Table 3. Presentation of U.S. Protocol (2013) methodology: Methods for assessment of WWT CH_4 emissions.

Tabela 3. Prezentacja metodologii protokołu amerykańskiego (2013): Metody oceny emisji CH_4 z OŚ.

U.S. Protocol CH4 Calculation methods (2013)	Emission type	CH4 source with data requirements
Method WW.1.a Method WW.1.b	Stationary emissions	Combustion of digester gas at a centralized WWTP with anaerobic digestion of biosolids when process data known
Method WW.1.(alt)	Stationary emissions	Combustion of digester gas at a centralized WWTP with anaerobic digestion of biosolids based on the population served by the facility
Method WW.4	Stationary emissions	Emissions from residuals combustion based on the mass composition of the material sent to the combustion device
Method WW.6	Process emissions	Emissions from anaerobic or facultative lagoons based on the BOD_5 daily load and BOD_5 fraction removal performance
Method WW.6.(alt)	Process emissions	Emissions from anaerobic or facultative lagoons based on population served by the facility
Method WW.11	Fugitive emissions	Emissions from septic systems based on the BOD_5 daily load
Method WW.11.(alt)	Fugitive emissions	Emissions from septic systems based on population served by the facility
Method $WW.13_{CH4}$	Attributed emissions	Process emissions from cluster package systems based on population served by the facility
Method WW.14	Lifecycle emissions associated with water acquisition, distribution and treatment	Upstream indirect emissions resulting from the energy consumption based on the amount of energy purchased and consumed or on the national average energy consumption per unit of water and wastewater and the total annual volume of the water and wastewater passing through the analysed system(s)
Method WW.15	Lifecycle emissions associated with water acquisition, distribution and treatment	Upstream indirect emissions resulting from the energy consumption based on the amount of energy purchased and consumed or on the national average energy consumption per unit of water and wastewater and the population served by the system(s) divided into the groups linked to the type of the wastewater treatment process (e.g., centralized and other modalities)

The protocol does not address the CH_4 emissions originating from sewer collection systems, as well as those resulting from preliminary and primary treatment processes. The omission of methane emissions from BNR technologies is due to their perceived lack of significance and, consequently, they are not addressed. Furthermore, the absence of guidance pertaining to the management of AD or digestate handling, specifically in relation to thickening and dewatering processes, raises concerns regarding the control of CH_4 fugitive emissions. In the context of sludge combustion, specific emission factors have been established to account for methane releases.

N_2O emissions, as quantified using the U.S. Protocol methodology, are generated from two primary sources: biological nutrient removal (BNR) systems and the decomposition of organic waste materials.

Clean Development Mechanism Treatment of wastewater methodology (2014) [12]

The Clean Development Mechanism is a carbon offset initiative administered by the United Nations (UN). It enables countries to finance projects aimed at reducing greenhouse gas emissions in other nations, while attributing the resulting emission reductions to their own endeavours towards meeting global emissions objectives. CDM was designed with two primary aims: firstly, to support primarily developing nations, in their pursuit of sustainable development and the mitigation of their carbon emissions; and secondly, predominantly industrialised nations, in meeting their obligations to reduce GHG and achieve compliance with their emission reduction targets. In order for a proposed CDM project to undergo validation, approval, and registration, it is imperative that it adheres to an authorised baseline and monitoring methodology, which is also available in the area of wastewater treatment.

The CDM methodology primarily centres around AD processes that involve the capture of biogas for the purpose of generating electricity and/or heat. Additionally, it provides guidelines for calculating the dewatering of sludge and its subsequent application to land. CDM does not differentiate between distinct methods for calculating GHG emissions. Instead, it provides equations that must be adhered to when determining the CF components. These equations rely on data obtained from the facility monitoring system, including

Table 4. Presentation of U.S. Protocol (2013) methodology Methods for assessment of WWT N_2O emissions.

Tabela 4. Prezentacja metodologii protokołu amerykańskiego (2013): Metody oceny emisji N_2O z OŚ.

U.S. Protocol Calculation methods (2013)	Emission type	N_2O source with data requirements
Method WW.2.a Method WW.2.b	Stationary emissions	Combustion of digester gas at a centralized WWTP with anaerobic digestion of biosolids when process data known
Method WW.2.(alt)	Stationary emissions	Combustion of digester gas at a centralized WWTP with anaerobic digestion of biosolids based on the population served by the facility
Method WW.5	Stationary emissions	Emissions from residuals combustion based on the mass composition of the material sent to the combustion device
Method WW.7	Process emissions	Centralized WWTP with nitrification/denitrification or aeration basin based on the population served
Method WW.8	Process emissions	Centralized WWTP without nitrification/denitrification or aeration basin based on the population served
Method WW.10	Process emissions	Emissions from cluster package system based on the population served
Method WW.12	Fugitive emissions	Effluent discharge to receiving aquatic environments based on the daily N-load
Method WW.12.(alt)	Fugitive emissions	Effluent discharge to receiving aquatic environments based on the population served
Method $WW.13_{N2O}$	Attributed emissions	Process emissions from cluster package systems based on population served by the facility
Method WW.14	Lifecycle emissions associated with water acquisition, distribution and treatment	Upstream indirect emissions resulting from the energy consumption based on the amount of energy purchased and consumed or on the national average energy consumption per unit of water and wastewater and the total annual volume of the water and wastewater passing through the analysed system(s)
Method WW.15	Lifecycle emissions associated with water acquisition, distribution and treatment	Upstream indirect emissions resulting from the energy consumption based on the amount of energy purchased and consumed or on the national average energy consumption per unit of water and wastewater and the population served by the system(s) divided into the groups linked to the type of the wastewater treatment process (e.g., centralized and other modalities)

parameters such as the volume and COD of the treated wastewater or sludge, the amount of biogas collected and the methane concentration within it, The methodology employed does not take into account the emissions originating from wastewater collection systems, primary or secondary clarifiers, and the biological treatment stage.

5. DIRECT GHG EMISSIONS IN WASTEWATER TREATMENT

According to estimates, the direct emissions originating from the wastewater sector, specifically CH_4 and N_2O, contribute to approximately 5% of global non-CO_2 GHG emissions. Projections indicate that these emissions are expected to increase by 22% by the year 2030. From a CF perspective of WWTPs, the significance of direct gaseous emissions is noteworthy, particularly for facilities that receive influent with elevated concentrations of pollutants.

The literature exhibits inconsistencies in the accounting of direct CO_2 emissions. The IPCC and CDM excluded biogenic carbon dioxide emissions from their analysis, while LGOP and U.S. Protocol are not discussing them at all. L. Li et al. (2022) [33] contend that while it is commonly believed that CO_2 emissions primarily originate from biogenic organic matter found in human excreta or food waste, this perspective overlooks the potential contribution of fossil carbon sources, including pharmaceuticals and personal care products or even external carbon sources for denitrification enhancement, which could potentially augment the overall quantity of direct carbon emissions. The topic of fossil CO_2 emissions from wastewater has garnered increased attention, prompting the IPCC (2019) to revise its 2006 guidelines to include a comprehensive discussion on this matter. However, the majority of the studies that contribute to the advancement of net-zero carbon wastewater treatment primarily focus on N_2O and CH_4, as detailed in this paper and summarized in a simplified Scheme 3.

Methane

During wastewater collection, methane generation conditions reach their optimal levels. Pumping systems can anaerobicize ascending sewers [5, 60]. These circumstances and high organic matter concentrations encourage methanogenesis [28]. At a discharge well, pressure differentials cause CH_4 emissions and an olfactory disturbance that draws nearby residents [44]. Gravitational systems can also measure CH_4 fluxes [59]. Air depletion stratification beneath the free surface can develop oxygen-deprived underground layers, where CH_4 generation occurs. In this scenario, wastewater flow disturbances reveal emission locations.

The mechanical stage of WWTPs emits CH_4 from compensation tanks (aerobic and anaerobic), preliminary oxidation tanks for slurry wastewater, aerated grit chambers, and sludge concentration tanks, in descending order [32, 33].

GHG emissions, including methane fluxes, are observed at several points in bioreactor compartments during BNR operations. Oxygen deficit in anaerobic and anoxic zones promotes methane production, which is released in aerobic chambers during nitrification [3, 26, 33].

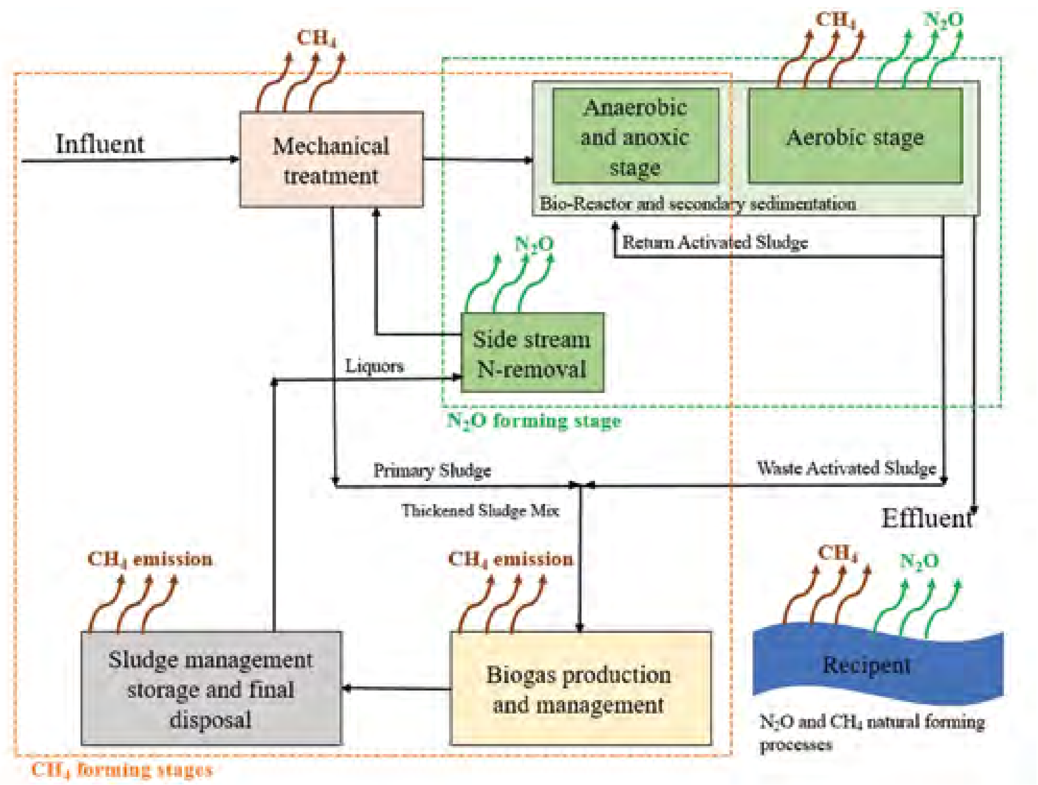
According to Khabiri et al. (2021) [32], the facility emits 75% of its CH_4 via anaerobic digestion of main and secondary sludge. The reported CH_4 emissions mostly come from stripping incoming sewage and anaerobic digesters (Foley et al., 2009). In the event of planned or accidental mechanical disturbances that interrupt pipeline continuity, biogas production, collecting, and transportation systems may emit CH_4 from AD process.

Further digestate or sludge management and disposal is also linked to methane releases, e.g., via long-term storage, in-land application or incineration [25, 59, 62].

Lakes and rivers that receive treated wastewater streams are major methane emitters [61]. When oxygen is scarce, processed wastewater inflows provide organic materials for methanogenesis in aquatic habitats.

Nitrous oxide

While CH_4 emissions can be spotted at different stages of wastewater treatment, N_2O fluxes requires more sophisticated conditions to be released which is particularly observed in BNR compartments. The production of N_2O in the bioreactors encompasses a series of microbiological reactions that necessitate either aerobic or anoxic conditions. Researchers have identified three



Scheme 3. N_2O and CH_4 formation and emission stages/points in a conventional WWTP.

Schemat 3. Tworzenie N_2O i CH_4 oraz etapy/punkty emisji w konwencjonalnej oczyszczalni ścieków.

primary biological pathways in WWT systems with nutrients removal. These involve the N_2O production: (1) as an intermediate product during the oxidation of hydroxylamine (NH_2OH) by ammonia-oxidizing bacteria (AOB), (2) the final product of nitrifier denitrification by AOB, and (2) the intermediate product of denitrifying heterotrophic bacteria (DHET) denitrification.

Drawing upon the existing body of knowledge pertaining to N_2O production pathways, Vasilaki et al. (2019) [50] identified several key operational variables that are responsible for the generation of N_2O, such as the accumulation of low dissolved oxygen (DO), nitrite (NO_2^-), or free nitrous acid (HNO_2), as well as changes in the ammonium (NH_4^+) concentration in the nitrifying compartments or the situation when denitrifying compartments exhibit a low chemical oxygen demand to nitrogen ratio, as well as an accumulation of nitrite ions (NO_2^-) and the alternation of anoxic and aerobic conditions in switching compartments.

The bioreactors emit most of the WWTPs' N_2O, but other elements of the plant release a small amounts of this gas as well, such as partial nitrogen conversion produces N_2O in the effluent receiver [46]. Sludge storage N_2O emissions are estimated based on 12-month uncovered storage. On the other hand, sewer N_2O emission case studies are scarce.

WWT area CF calculation can evaluate GHG emissions from a single facility (centralised WWTP) or develop CF level for many cooperating plants. The reporting organisation shall follow GHG Protocol consolidation requirements in each circumstance. Scheme 4 in the upcoming chapter shows how each criterion differs. Consolidation procedure and organisational boundary requirements guarantee correct CF results.

Hence, it is important to acknowledge that a multitude of diverse factors contribute to and interact within bioreactors in full-scale wastewater treatment plants, and these processes can occur simultaneously in a dynamic manner, often beyond the direct control of operators

6. WWTPs' CARBON FOOTPRINT SCOPE 1 EVALUATION

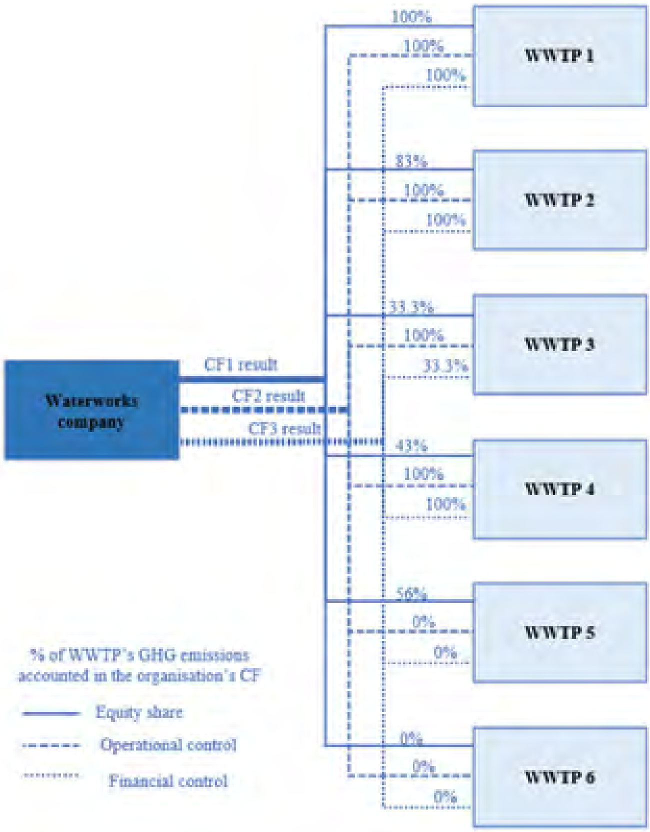
6.1 ESTABLISHING CALCULATION BOUNDARIES – ORGANISATIONAL MATERIALITY ANALYSIS

As per the guidelines provided by the GHG Protocol, the reporting of GHG emissions at the organisational level of CF commences with the fulfilment of the two-stage materiality analysis. Stage one is dedicated to convey the organizational CF calculation boundaries, while the second stage addresses the establishment of operational boundaries.

Organisational boundaries analysis, in a so-called consolidation process, entails determining the responsibility for emissions within the entire structure of the organisation, including subsidiaries, among other factors. The process of ascertaining accountability can be conducted based on either of two criteria: (1) equity share or (2) control: operational or financial control [53].

The concept of equity share pertains to the allocation of ownership in an organisation. Control can be categorized into two main types: operational control, which involves decision-making authority over day-to-day operations, and financial control, which encompasses the ability to influence financial decisions and strategic direction. In the case of consolidation based on financial control, emissions from joint ventures in which the partners share financial control are accounted for on the basis of equity participation. Under the operational control approach, the organisation is responsible for 100% of the emissions from operations over which the organisation or one of its subsidiaries has operational control.

There are two primary scenarios that may arise when calculating the CF in WWT area. The first scenario involves assessing GHG emissions from a single facility, specifically a centralised WWTP. The second scenario involves determining the CF level for a group of collaborating facilities. The consolidation process, as per the requirements of the GHG Protocol, should be undertaken annually in each instance from the standpoint of the reporting organisation. The Scheme 4 effectively demonstrates the distinctions observed in the application of each criterion. The accuracy of CF results can be ensured by providing clear consolidation processes and well-defined criteria for organisational boundaries.



Scheme 4. Defining the organizational boundary of complex waterworks company and 6 WWT facilities.

Schemat 4. Określenie granicy organizacyjnej złożonego przedsiębiorstwa wodociągowego i 6 obiektów OŚ.

Establishing the boundaries for calculating carbon footprint within a specific plant may present challenges. In the scientific investigations, researchers typically direct their attention towards the GHG emission domain of the selected WWTP aspects in order to provide a comprehensive overview of GHG emissions stemming from specific facets of the process, encompassing both direct and fugitive emissions [1, 2, 8, 10, 17, 27, 40, 41, 42, 49, 50, 58] and/or chosen phases of the organisational CF value chain, such as externally sourced energy, transportation, chemical agents usage and waste disposal [19, 24, 34, 35, 36, 39, 56, 57]. As stated before, the GHG Protocol Standard includes the establishment of operational boundaries as the second part of the materiality analysis. This phase is specifically focused on the assessment of indirect GHG emissions, particularly those falling under the purview of Scope 3, as all direct emissions (Scope 1) and energy-related indirect emissions (Scope 2) should be covered by CF calculation. Each Scope 3 category will be discussed further in the article part two.

6.2 SCOPE 1 EMISSIONS

Scope 1 pertains to the direct emissions of greenhouse gases that take place at sites that are either owned or under the control of the organisation, as determined through the analysis of materiality and the process of consolidation [54]. There are two primary categories of direct emissions that can be identi ed: emissions originating from supporting activities (Table 5), and emissions resulting from process and technological factors (Table 6). These two categories can be further delineated as direct GHG emissions that arise from the combustion of fuels in stationary and mobile combustion sources, as well as from physical or chemical processing and fugitive emissions. The various types mentioned are illustrated with speci c examples

Table 5. Summary of direct GHG emission types from a WWTP's supportive activities.

Tabela 5. Podsumowanie bezpośrednich rodzajów emisji gazów cieplarnianych z działań pomocniczych OŚ.

Scope 1 direct emission type	Emission source	Explanation	WWTP example	Minimum data required for calculation
Emissions from supportive activities				
Combustion	Stationary combustion	GHG emissions from combustion of conventional fuel e.g., coal, oil, natural gas in stationary sources owned or controlled by the organisation in order to produce energy	Boilers, turbines, aggregates	Annual usage of fuels collected within the types
Combustion	Mobile combustion	GHG emissions from combustion of conventional fuel e.g., petrol, diesel in the car fleet owned or controlled by the organisation (including leased vehicles)	Passenger vehicle fleet, heavy vehicle fleet, transportation fleet	Annual usage of fuels collected within the types
Fugitive	Mobile	GHG emissions from AdBlue usage in the diesel fleet owned or controlled by the organisation (including leased vehicles)	Diesel vehicle fleet	Annual usage of AdBlue
Fugitive	Refrigerants leaks	GHG emissions from refrigerants leaks (air-conditioning, AC, units/installations)	AC units in administrative buildings and samples storage rooms, probes rooms	Annual amount of the refrigerants refilled (usually in kg) collected within the types (e.g., R407c, R410A)
Fugitive	Welding	GHG emissions from welding procedure done by the organisation itself – usage of GHG-containing shielding gases mixtures	Gases used while the repair process via welding	Annual amount of the welding gases mixtures with its safety data sheets (including weight mass composition)

Table 6. Summary of a WWTP's direct GHG process and technological emission types.

Tabela 6. Podsumowanie bezpośrednich rodzajów emisji gazów cieplarnianych z procesów technologicznych OŚ.

Scope 1 direct emission type	Emission source	Main GHG type	Authors' proposed CF calculation algorithm statement with justification	Emissions included in the algorithm as in methodology
Process and technological emissions				
Fugitive	Collection system	CH_4	Not considered – due to the lack of data, research results and no methodology given. Authors recommend following the updates in the area as fugitive emissions from sewer systems may play crucial role in WWTP's CF structure [18, 31, 45].	IPCC, NGA/NGER, U.S. Protocol, LGOP, CDM
Fugitive	Preliminary and primary treatment	CH_4	Considered – emissions calculated on the basis of EFs (IPCC, 2019 and NGA/NGER, 2023 as complimentary data sources).	IPCC, NGA/NGER
Fugitive	BNR treatment	N_2O, CH_4	Considered – emissions calculated on the basis of EFs (IPCC, 2019, U.S. Protocol, 2013 NGA/NGER, 2023 and LGOP, 2010 as complimentary data sources) depending on the technological process involved.	IPCC, NGA/NGER, U.S. Protocol, LGOP
Fugitive	Denitrification supported by external C-source dosage	Fossil (non-biogenic) CO_2	Considered – emissions calculated on the basis of EFs (IPCC, 2019 – Table 6AP.1) depending on the C-source type	IPCC
Fugitive	Secondary clarification	N_2O	Not considered – authors follow discussed methodologies approach and, on this basis, assume that N_2O emission from secondary clarification process are covered by BNR emissions. Authors recommend following the updates in the area for the probable future supplementary calculation methods.	IPCC, NGA/NGER, U.S. Protocol, LGOP, CDM
Fugitive	Side stream N-removal	N_2O	Considered – emissions calculated on the basis of EFs (IPCC, 2019 and U.S. Protocol, 2013 as complimentary data sources) depending on the technological process involved.	IPCC, U.S. Protocol
Fugitive	Effluent discharge	N_2O, CH_4	Considered – emissions calculated on the basis of EFs (IPCC, 2019 and U.S. Protocol, 2013 as complimentary data sources) depending on the recipient type.	IPCC, NGA/NGER, U.S. Protocol, LGOP, CDM
Fugitive	Anaerobic Digestion	CH_4	Considered – calculated for 4 types of emissions: (1) intended and unintended biogas leaks from digesters, (2) biogas flared, (3) linear biogas transportation leaks, (4) captured biogas combustion (e.g., cogeneration process)	IPCC, NGA/NGER, CDM
Fugitive	Thickening and dewatering	CH_4	Not considered – authors follow discussed methodologies approach delivering no discussion on GHG emissions from thickening or dewatering procedures. Authors recommend following the updates in the area for the probable future supplementary calculation methods.	IPCC, NGA/NGER, U.S. Protocol, LGOP, CDM
Fugitive	Dewatered sludge and digestate management and usage	CH_4, N_2O	Considered depending on materiality analysis and consolidation process results	IPCC, NGA/NGER, U.S. Protocol, LGOP, CDM
Fugitive	Process waste management	CH_4, N_2O	Considered depending on materiality analysis and consolidation process results	IPCC, NGA/NGER, U.S. Protocol, LGOP, CDM

in the Tables with the indication of minimum data required for CF Scope 1 supportive activities' emission calculation.

As stated in the aforementioned article, the primary objective of this study involves the development of comprehensive guidelines for calculating CF in WWTPs. These guidelines aim to address the existing gaps in accounting for both direct process emissions and technological emissions. The Table 6 provided below presents a summary of the authors' work in developing fundamental aspects to enhance the GHG emissions evaluation algorithm, which can be utilised by professionals in the eld. Moreover, comprehensive guidelines are provided in subsequent sections of this document.

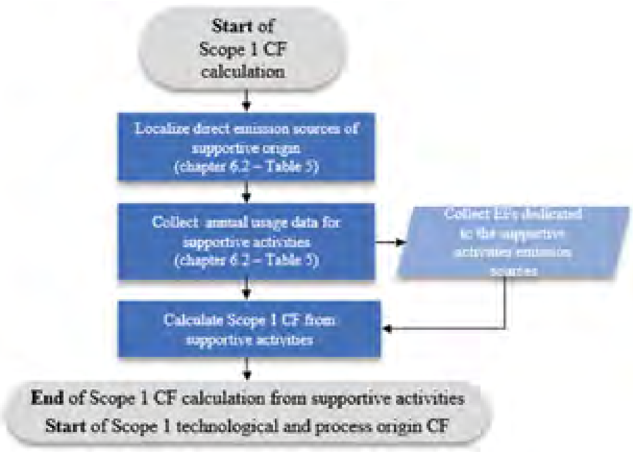
7. CARBON FOOTPRINT ALGORITHM OF MWWTP: SCOPE 1 – STEP BY STEP INSTRUCTION

The algorithm for calculating the carbon Footprint of MWWTP, presented in this study as a two-part article, encompasses seven distinct stages that adhere to the guidelines set forth by the GHG Protocol and meet the requirements of the CSRD. The following steps are:

1. Consolidation process evaluation.
2. Scope 1 emission calculation:
   a. GHG emissions from the supportive activities,
   b. N_2O direct fugitive emissions,
   c. CH_4 direct fugitive emissions from wastewater treatment path,
   d. CH_4 direct fugitive emissions from sludge management, biogas production and utilisation.
3. Scope 2 (location – and market-based methods) calculation.
4. Scope 3 calculation.
5. Results summary, uncertainty discussion and report preparation.
6. Conclusions in the area of data aggregation.
7. Carbon footprint results analysis and GHG emission reduction planning.

In this article, there are guidelines for Scope 1 in the light of steps 1, 2, 5, 6. To enhance the clarity of the proposed calculation approach for CF, we have included seven supplementary decision trees (algorithms) along with corresponding instructions. This part one article includes guidelines, as well as analyses of various GHG calculation methodologies used in WWTP, namely:

- IPCC – 2006 methodology with 2019 Guidelines for wastewater treatment and discharge (Vol. 5, Chapter 6) and its default value summary tables,
- NGA/NGER – 2023 methodology and its default value summary tables,
- U.S. Protocol – 2013 methodology and its default value summary tables from Appendix F.

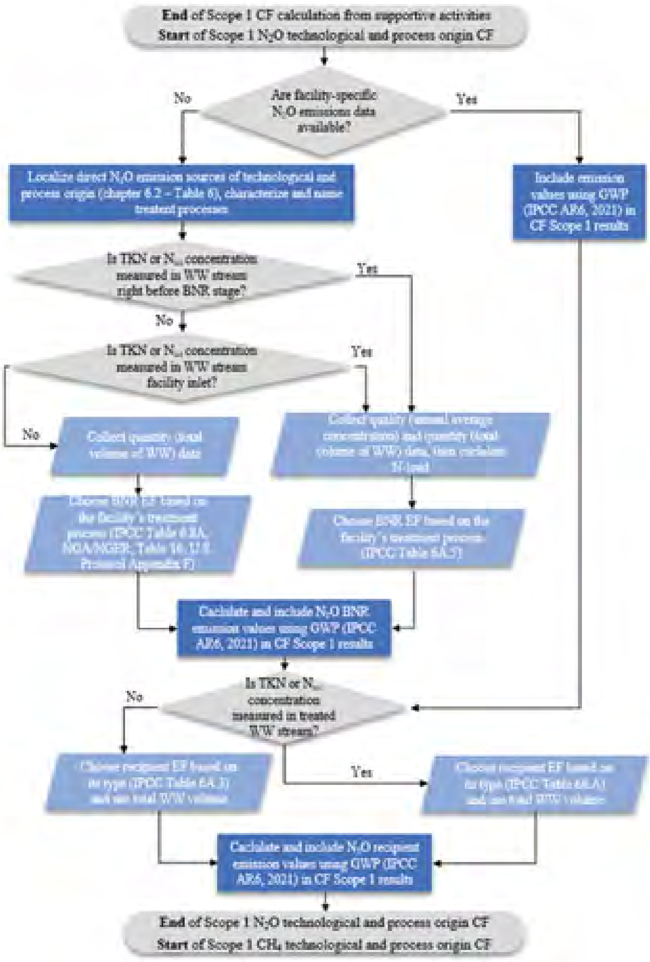


Algorithm 1. Instruction for WTTP Scope 1 carbon footprint calculation.
Algorytm 1. Instrukcja obliczania śladu węglowego dla OŚ w Zakresie 1.

The proposal of the WWTP's CF calculation steps suggests commencing the evaluation of Scope 1 carbon footprint by assessing the greenhouse gas emissions associated with supportive activities, as outlined in Algorithm 2.

Algorithm 2 initiates the phase of fugitive emissions calculations, speci cally focusing on the quanti cation of N_2O xes. When implementing side stream N-removal in a WWTP, it is recommended to follow the same steps as previously presented.

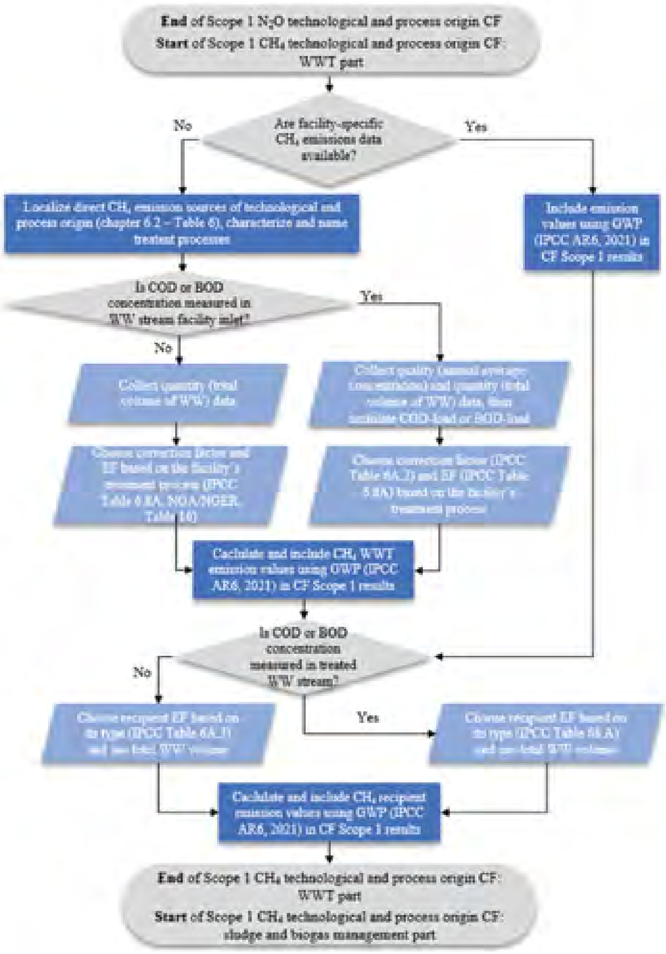
Algorithms 3 and 4 are speci cally designed for the calculation of direct fugitive emissions of CH_4. Given the absence of comprehensive biogas quality data pertaining to methane concentration, it is advisable to rely on average information derived from scholarly literature sources, such as reports published by IEA.



Algorithm 2. Decision tree and instructions for N_2O direct fugitive emissions from WWTP.
Algorytm 2. Drzewo decyzyjne i instrukcje dotyczące bezpośrednich emisji niezorganizowanych N_2O z OŚ.

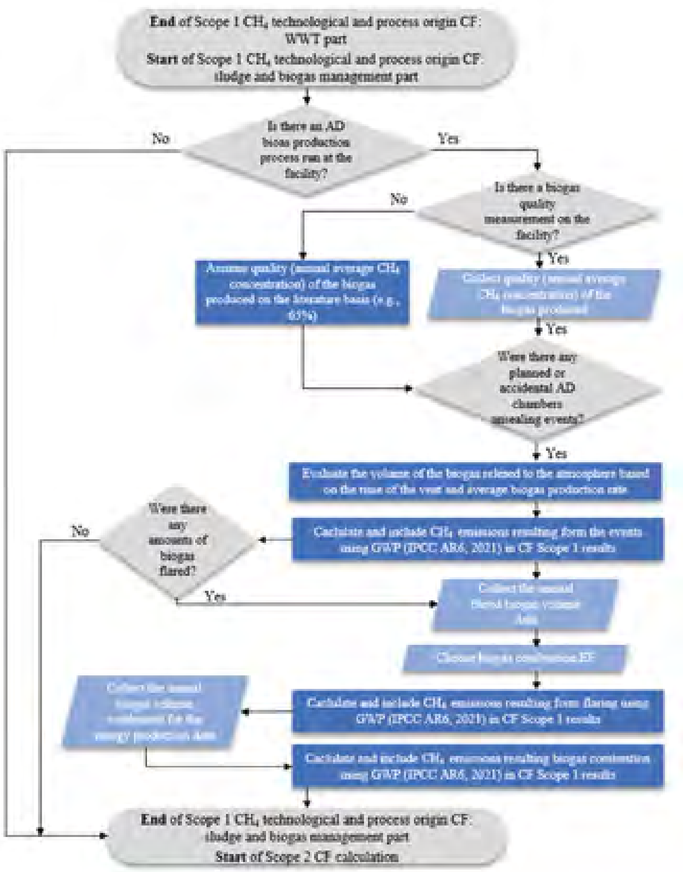
8. UNCERTAINTY DISCUSSION

As per the guidelines outlined by the GHG Protocol, it is imperative to include an analysis of uncertainty or, at the very least, engage in a thorough discussion of it within the calculation procedure. Uncertainty in the context of direct emissions in WTTPs arises from two sources: measurement uncertainties, which pertain to the quality of the data collected, and uncertainties associated with the calculation of emission factors, which relate to the quality of the factors used in estimating emissions. The authors put forth a methodology for ascertaining the level of uncertainty associated with emission factors, denoted as Scheme 5. Further discussion on the uncertainty topic within the authors' recommendation will be presented in the article part two.



Algorithm 3. Decision tree and instructions for CH_4 direct fugitive emissions from WWTP: wastewater treatment path.

Algorytm 3. Drzewo decyzyjne i instrukcje dotyczące bezpośrednich emisji niezorganizowanych CH_4 z OŚ: ścieżka ściekowa.



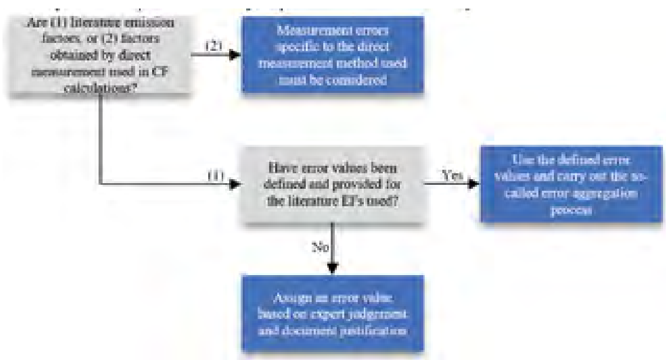
Algorithm 4. Decision tree and instructions for CH_4 direct fugitive emissions from WWTP: sludge management and AD part.

Algorytm 4. Drzewo decyzyjne i instrukcje dotyczące bezpośrednich emisji niezorganizowanych CH_4 z OŚ: część osadowa i produkcja biogazu.

9. CONSLUSIONS

This paper highlights the need for standardized approaches in calculating GHG emissions from WWTPs to meet the requirements CSRD and ESRS indicators. The developed algorithm for calculating the carbon footprint of WWTPs, provides a comprehensive framework for assessing GHG emissions. Authors address nal conclusions described below.

- The calculation of Scope 1 GHG emissions, as outlined by the GHG Protocol, presents a challenge in terms of data aggregation for wastewater utilities. As a result, it is recommended that these utilities adopt a best practise approach by establishing a dedicated team or department responsible for the calculation of carbon footprints.
- The calculation of fugitive emissions in Scope 1, which includes N_2O and CH_4 necessitates comprehensive technological data, encompassing both quantitative and qualitative aspects. Consequently, it may be necessary to periodically update the measurement schedule on an annual basis to gather data for evaluating carbon footprint.
- WWTPs which employ complete nitrogen removal processes exhibit a higher level of emission in contrast to facilities that do not utilise biological nutrient removal techniques.
- In order to conduct a comprehensive comparison of CF results, especially within Scope 1, it is essential to establish emission intensity ratios (expressed as $kgCO_2e/m^3$ of treated wastewater).



Scheme 5. Simplified procedure for emission factors uncertainty assessment.

Schemat 5. Uproszczona procedura oceny niepewności wskaźników emisji.

Relying solely on total CF levels may lead to inaccurate or incomplete conclusions.

- It is assumed that in the foreseeable future, there will be an implementation of improved on-site GHG emission evaluation campaigns at MWWTPs. The purpose of these campaigns would be to gather precise N_2O and CH_4 emission factors that are to each facility. This is necessary because relying solely on literature-based indicators may lead to an overestimation of Scope 1 CF results, as literature-based indicators may overestimate Scope 1 CF results even up to two orders of magnitude [18].

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Paulina Szulc-Kłosińska
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Zbysław Dymaczewski	25%	

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Algorithm of carbon footprint calculation for municipal wastewater treatment plant – part two

Algorytm obliczania śladu węglowego miejskiej oczyszczalni ścieków – część druga

Paulina Szulc-Kłosińska, Zbysław Dymaczewski*)

Keywords: *GHG emissions, nitrous oxide, methane, carbon footprint, wastewater treatment, CSRD*

Abstract
In the forthcoming years, urban wastewater management utilities will be required by the European Union to perform CF calculations in accordance with the Corporate Sustainability Reporting Directive (CSRD) and European Sustainability Reporting Standards (ESRS) indicators. Yet, no standardized approach that expressly addresses the rules for WWTPs in respect to GHG emissions, giving the water bodies a clear instruction to calculate their CF is given. This paper provides an in-depth examination of the present approaches for calculating GHG emissions. An algorithm for calculating the carbon footprint of a wastewater treatment facility is developed and described in detail by the authors. Furthermore, the research evaluates the extent to which facility data is complete and suggests remedies to any detected information gaps. A data enhancement strategy is also offered. The primary goal of this research is to bridge a knowledge gap in the understanding of the carbon footprint associated with WWTPs and their organisational framework. The analysis also included a thorough investigation into the significance and sources of GHG Protocol Scope 1 (part one arcticle), 2, and 3 emissions (part two article) within the larger framework of carbon footprint, particularly in relation to the legislative goals of CSRD reporting with its upcoming obligations imposed on waterworks organizations.

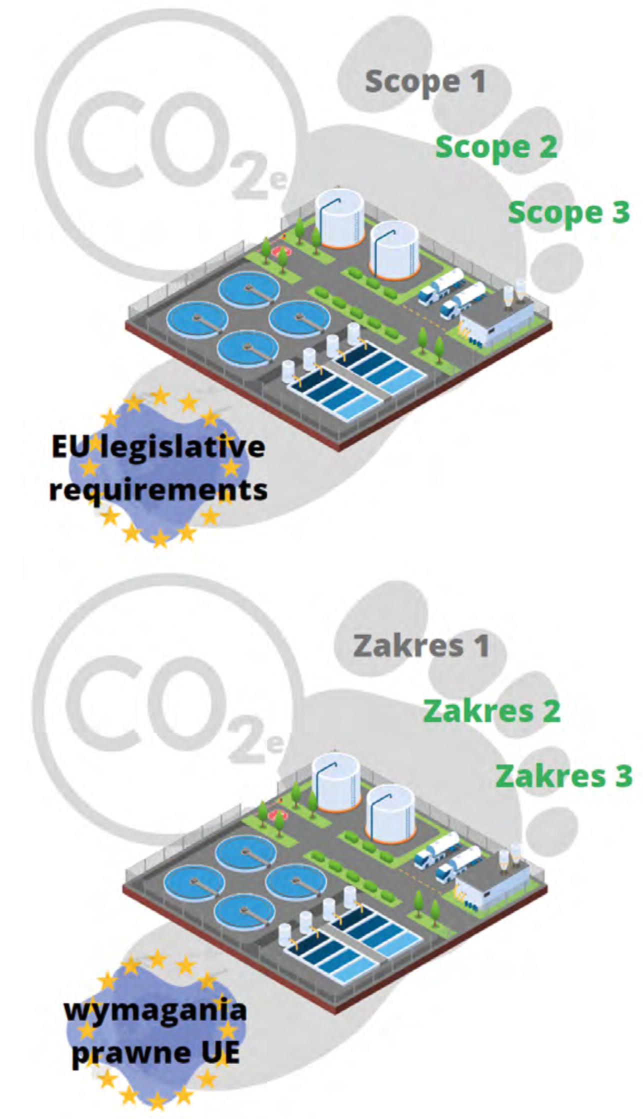
Słowa kluczowe: *Emisje gazów cieplarnianych, podtlenek azotu, metan, ślad węglowy, oczyszczanie ścieków, CSRD*

Streszczenie
W nadchodzących latach Unia Europejska będzie wymagać od miejskich zakładów oczyszczania ścieków wykonywania obliczeń śladu węglowego (CF) zgodnie z Dyrektywą w sprawie sprawozdawczości dotyczącej zrównoważonego rozwoju przedsiębiorstw (CSRD) i wskaźnikami Europejskich Standardów Sprawozdawczości dotyczącymi Zrównoważonego Rozwoju (ESRS). Nie istnieje jednak żadne ustandaryzowane podejście, które wyraźnie odnosiłoby się do zasad dotyczących oczyszczalni ścieków w odniesieniu do emisji gazów cieplarnianych (GHG), dając organom wodnym jasne instrukcje dotyczące obliczania ich CF. Niniejszy artykuł zawiera dogłębną analizę obecnych podejść do obliczania emisji GHG. Algorytm obliczania śladu węglowego miejskiej oczyszczalni ścieków został opracowany i szczegółowo opisany przez autorów. Ponadto, w analizie oceniają zakres, w jakim dane dotyczące oczyszczalni ścieków są kompletne i sugerują środki zaradcze dla wszelkich wykrytych luk informacyjnych. Zaproponowano również strategię ulepszania danych. Głównym celem tego badania jest wypełnienie luki w wiedzy na temat śladu węglowego związanego z oczyszczalniami ścieków i ich ramami organizacyjnymi. Analiza obejmowała również dokładne zbadanie znaczenia i źródeł emisji z wg podziału na zakresy GHG Protocol 1 (część pierwsza), 2 i 3 (część druga) w szerszych ramach śladu węglowego, szczególnie w odniesieniu do celów legislacyjnych raportowania CSRD z nadchodzącymi obowiązkami nałożonymi na przedsiębiorstwa wodociągowe.

1. INTRODUCTION

Given the increasing concern surrounding the issue of global warming, many countries have made a rm pledge to achieve a state of net-zero emissions by the year 2050 [13], in accordance with the guidelines set forth by the Intergovernmental Panel on Climate Change (IPCC) to prevent exceeding the 1.5°C limit [10]. Wastewater treatment plants (WWTPs) have been recognised as a notable contributor to direct greenhouse gas (GHG) emissions, particularly in the form of N_2O and CH_4. These emissions constitute roughly 3% of worldwide GHG emissions [11, 19]. Furthermore, when examining indirect emissions, it becomes apparent from an energy perspective that WWTPs persist in consuming a substantial quantity of energy, predominantly obtained from fossil fuel origins. In light of the contemporary emphasis on climate change, rising electricity expenses, and the necessity to mitigate carbon footprint emissions in organisational or operational contexts, the wastewater treatment facility is perceived as a complex situation that offers both obstacles and possibilities for improvement. Water utilities are taking a leading role in addressing GHG emissions, as evidenced by several waterworks organisations aiming to achieve net-zero emissions in their operations in the next decades [2, 3, 9]. The initial stage in developing a GHG reduction strategy involves

*) **Paulina Szulc-Kłosińska,** Mgr inż. email: paulina.szulc-klosinska@bureauveritas.com Sustainability Consultant at Sustainability Department, Bureau Veritas Polska Sp. z o.o. ul. Migdałowa 4, 02-796 Warszawa; PhD student at Faculty of Environmental Engineering and Energy, Poznan University of Technology, Berdychowo 4, 60-965 Poznan, Poland, **Zbysław Dymaczewski**, Dr hab. inż., prof. PUT, Faculty of Environmental Engineering and Energy, Poznan University of Technology, Berdychowo 4, 60-965 Poznan, Poland, e-mail zbyslaw.dymaczewski@put.poznan.pl



GRAPHICAL ABSTRACT

doing an annual carbon footprint (CF) estimate. This calculation serves to enhance transparency and facilitate the comparison of outcomes across various facilities and economic regions.

In the upcoming year, the primary urban water management utilities will be obligated to conduct CF calculations in compliance with the Corporate Sustainability Reporting Directive (CSRD) and European Sustainability Reporting Standards (ESRS) indicators as stipulated by the European Financial Reporting Advisory Group (EFRAG). Nevertheless, it is important to acknowledge that there is presently no recognised methodology that explicitly deals with the rules for WWTPs in regard to GHG emissions. This lack of a standardised approach hinders water bodies from receiving clear instructions on how to calculate their CF. The compliance with the reporting requirements of the Greenhouse Gas Protocol (GHG Protocol) regime, as legislated by the European Union (EU), is necessary in order to fulfil the duties of CSRD. As a result, water providers have a responsibility to adhere to these stipulations.

Previous studies have demonstrated that the application of different wastewater treatment methods can result in different levels of direct GHG emissions, as well as various degrees of energy usage and sludge production [4, 8, 11, 12, 14, 20]. According to the study conducted by Piao et al. (2016) [11], the introduction of the Modified Ludzack-Ettinger (MLE) process in wastewater treatment plants led to a 20% rise in the carbon footprint when compared to facilities employing the Anaerobic-Anoxic-Oxic (A2/O) process. Furthermore, research has indicated that the adoption of aerobic wastewater treatment methods might lead to a substantial rise of 105% in direct GHG emissions, in contrast to the usage of anaerobic wastewater treatment techniques [4].

However, there are still ambiguities around the understanding of carbon footprints linked to different layouts of WWTPs. Previous studies failed to consider several potentially major sources of greenhouse gas emissions, particularly in relation to sludge handling operations. These studies focused solely on specific emission-related aspects, rather than considering the overall fluxes of gases that contribute to global warming.

Goal and scope of the work presented in the two-part paper

The primary objective of this study is to offer a helpful resource for academics, policymakers, and practitioners engaged in the evaluation and control of the carbon footprint associated with wastewater treatment facilities. This study aims to enhance our understanding of the carbon footprint composition of wastewater treatment facilities by emphasising the evaluation of direct greenhouse gas emissions. Additionally, it seeks to offer guidance on estimating the carbon footprint of municipal wastewater treatment plants. By doing so, this research can contribute to the advancement of more precise and comprehensive carbon footprint assessments and aid in the mitigation of climate change.

Furthermore, this study examines the degree of completeness of facilities data and suggests potential remedies to resolve any discovered deficiencies in information pertaining to the comprehension of the carbon footprint linked to WWTPs [15] and their organisational structure. Furthermore, a plan for enhancing the quality of data is proposed. The analysis encompassed a thorough examination of the significance and origins of Scope 1, 2, and 3 emissions (as defined by the GHG Protocol) within the broader context of carbon footprint, specifically in relation to the regulatory objectives of CSRD reporting and the imminent responsibilities imposed on waterworks organisations.

2. INDIRECT EMISSIONS IN WASTEWATER TREATMENT – EU LEGISLATION

The Corporate Sustainability Reporting Directive (CSRD), officially referred to as EU Directive 2022/2464 of the European Parliament and of the Council of 14 December 2022 [7], was implemented on January 5th, 2023, as a component of the legislative measures associated with the Green Deal EU initiative. The article's first section presents the regulatory scopes and introductory phases.

Organisations who are subject to the oversight of the CSRD will have a duty to comply with the ESRS when presenting their reports. The disclosure of CSRD reports highlights the significance GHG emissions, with particular emphasis on Scope 2 and Scope 3 indirect emissions. These emissions are of utmost relevance since they arise from the value chain activities of each firm. Furthermore, this pertains to the formulation of climate change policy and the development of a strategy aimed at mitigating greenhouse gas emissions. The primary objective of this strategy is to achieve a state of net-zero emissions by the year 2050, as outlined in the second and third scope reduction goals.

The assessment of CF, due to its complete character, requires significant data collection endeavours in several settings to discover areas lacking information. This will entail the development of data collection schemes in collaboration with the organization's suppliers.

3. WWTPs' SCOPE 2 AND 3 CARBON FOOTPRINT EVALUATION

3.1. ESTABLISHING CALCULATION BOUNDARIES – OPERATIONAL MATERIALITY ANALYSIS

GHG Protocol requires annually provided materiality analysis – organisational and operational. Organisational assessment process has been described in detail in the article part one.

For the purpose of operational boundaries assessment, it is recommended to convey initial Scope 3 emission structure analysis on the basis of spend-based method [18]. The spend-based method is a technique that enables the estimation of emissions by gathering data on the economic value of goods and services acquired, and subsequently multiplying this value by appropriate secondary emission factors, such as industry averages (Equation 1). The emission factors provided herein represent the mean emissions attributed to each unit of monetary value of goods. Typically, the availability of expenditure data surpasses that of detailed mass or relevant units' information. It is imperative to gather economic aggregates for every category. In order to ensure comprehensive reporting, it is recommended that the emission factors necessary for calculating the CF of Scope 3 emissions be collected. The resulting CF values should then be presented individually for each relevant area. This presentation should encompass categories that account for over 90% of the emissions within the value [17].

$$sCF_{Scope\ 3\ category\ i,n}[tCO_2e] = Ec_{i,n}\ [currency] \cdot sEF_{i,n}\ [tCO_2e/currency] \quad (1)$$

$sCF_{Scope\ 3\ category\ i,n}$ – initial (screening) CF category *i* result of the year *n*, tCO_2e
$Ec_{i,n}$ – economic value of the purchased good or service in category *i*, spent in the year *n*, currency, e.g., PLN, EUR, USD
$sEF_{i,n}$ – initial (screening) CF category i emission factor used for the year *n*, *tCO_2e/currency, e.g., tCO_2e /PLN, tCO_2e /EUR, tCO_2e /USD*

The selection of emission factors (EFs) may pose a signi cant challenge in conveying the Scope 3 screening procedure. The utilisation of the Climatiq [5] free database is advised as the primary source of data. It is necessary to update each GHG emissions factor (EF) on an annual basis in order to accurately re ect the evaluation year. For instance, when conducting a materiality analysis for the year 2023, it is imperative to recalculate the EFs if they are outdated – dedicated to previous years (as indicated in Equation 2). For this purpose, the in ation rate is required to be used. It is important to acknowledge that the predominant currencies utilised in EF databases are USD, GBP, and EUR. Consequently, the utilisation of currency exchange rates may also be necessary.

$$sEF_{i,n}\ [tCO_2e/currency] = \frac{sEF_{i,n-x}\ [tCO_2e/currency]}{1 + IR_{n\ vs.n-x}\ [-]} \quad (2)$$

$sEF_{i,n}$ – initial (screening) CF category *i* emission factor used for the year *n*, *tCO_2e/currency, e.g., , tCO_2e /PLN, tCO_2e /EUR, tCO_2e /USD*
$sEF_{i,n-x}$ – initial (screening) CF category *i* emission factor used for the year *n-x* (outdated), *tCO_2e/currency, e.g., tCO_2e /PLN, tCO_2e /EUR, tCO_2e /USD*
$IR_{n\ vs.n-x}$ – in ation rate in the year n vs. year *n-x*, –

3.2. SCOPE 2 EMISSIONS

Scope 2 encompasses the energy indirect emissions that are associated with the acquisition and utilisation of electricity, heat, steam, or cooling by the organisation during the speci ed reporting period [16] reported in the energy consumption points falling under the calculation boundaries established through materiality analysis [17]. The inclusion of energy generated from self-owned renewable energy systems in the calculation provides a comprehensive assessment of the organization's total energy consumption. It is argued that the calculation of carbon footprints should not be applied to energy generated through self-owned renewable energy source (RES) installations.

The GHG Protocol's Scope 2 guidance outlines two methods for calculating emissions from purchased electricity: the location-based (LB) method and the market-based (MB) method [16]. Both methods are essential as the comparison of the results they provide allows for an examination of the in uence of the organization's decision-making process on CF. The nal result for Scope 2 carbon footprint is presented in the form of market-based method outcomes.

The emissions in the LB method are determined by utilising the country average EFs obtained from the National Centre for Emissions Management (KOBiZE) for Poland, and the European Environment Agency (EEA) or International Energy Agency (IEA) for other countries worldwide. The calculation of MB CF is derived from the residual energy mixes of national EFs, as published by the Association of Issuing Bodies (AIB) in the European Residual Mixes Report on an annual basis. In the event of acquiring energy through a specialised tariff for renewable energy systems (RES) or through a Guarantee of Origin (GO or GoO) supported by an appropriate statement document issued by the national energy-balancing body, the EF for electricity under said tariff or GO is effectively reduced to zero. To account for externally procured heat, process steam, and cold, it is recommended to utilise the average emission factors speci c to the country in question.

The concept of a Guarantee of Origin refers to an energy certi cate that is explicitly outlined in Article 15 of the European Directive 2009/28/EC [6]. A Green Option programme is designed to certify the origin of electricity derived from renewable sources and furnish customers with pertinent information regarding the energy source. GOs represent the sole established mechanisms that provide evidence regarding the source of electricity derived from renewable energy sources. Therefore, in case of the externally purchased electricity GOs, according to GHG Protocol, proves zero-levelled EF for MB Scope 2 calculation.

The national residual electricity mix (residual mix) represents the composition of the electricity supply that is not accounted for by Guarantees of Origin or other reliable tracking mechanisms. In order to ensure the reliability of the tracking instrument, it is necessary to incorporate a residual mix when not all consumption is accounted for using GO certi cates. The residual mix refers to the composition of energy sources used for generation, excluding any tracked energy generation attributes. The existence of a residual mix can be understood as a logical outcome of the implementation of energy attribute tracking. This implementation serves to prevent the inadvertent disclosure of the attributes associated with GOs to multiple consumers through an implicit mix. Without a residual mix, renewable electricity sold with GOs would be double counted because the same electricity would be disclosed to consumers buying "regular" electricity. It is advisable to refrain from utilising uncorrected generation statistics for the purpose of CF disclosure in MB Scope 2 calculation procedure [1].

The inclusion of energy generated by self-owned renewable energy systems in the calculation enables a comprehensive assessment of the organization's total energy consumption. The calculation of carbon footprints should not be applied to energy generated through self-owned renewable energy systems (RES).

When obtaining heat from external sources, such as process steam and cold, it is recommended to utilise the average emission factors provided by the country.

As previously indicated, the GHG Protocol [16] restricts the consideration of Scope 2 emissions to solely encompass the energy procured and utilised by the reporting organisation. Indirect emissions associated with the procurement and subsequent resale of energy are classified under Category 3 of Scope 3 as Well-to-Tank (WTT) emissions pertaining to electricity [18].

Table 1 summarizes possible WWTP's Scope 2 emission sources and indicates minimum data required for the CF calculations.

Table 1. Summary of possible WWTP's Scope 2 emission sources with minimum data required for the CF calculations indicated.

Tabela 1. Podsumowanie możliwych źródeł emisji Zakresu 2 oczyszczalni ścieków ze wskazaniem minimalnych danych wymaganych do obliczeń CF.

Scope 2	Type of externally sourced energy	Energy consumption point	Minimum data required for calculation
LB, MB	Electricity	Pumping wastewater collection system owned or controlled by the reporting organisation	Amount of energy in kWh or MWh purchased and consumed in the reporting year
LB	Heat and cooling		Amount of energy in GJ purchased and consumed in the reporting year
LB, MB	Electricity	Offices, mechanical workshop, warehouses, data centre rooms and other supporting buildings including own electrical car fleet charging points owned or controlled by the reporting organisation	Amount of energy in kWh or MWh purchased and consumed in the reporting year
LB	Heat and cooling		Amount of energy in GJ purchased and consumed in the reporting year
LB, MB	Electricity	WWTP installations with the entire energy-consuming infrastructure	Amount of energy in kWh or MWh purchased and consumed in the reporting year
LB	Heat, steam and cooling		Amount of energy in GJ purchased and consumed in the reporting year

The overall Scope 2 emission equation (3) is given below.

$$CF_{Scope\ 2\ _{LB\ or\ MB,n}}[tCO_2e] = E_{Ext.\ n}\,[kWh\ or\ GJ] \cdot EF_{LB\ or\ MB\,n}\left[\frac{tCO_2e}{kWh}\ or\ \frac{tCO_2e}{GJ}\right] \quad (3)$$

$CF_{Scope\ 2\ LB\ or\ MB}$ – Scope 2 CF (MB or LB) result of the year n, tCO_2e

$E_{Ext,\ n}$ – energy (electricity/heat/steam/cooling) purchased and consumed from external sources in the year n, kWh (electricity) GJ (heat, steam, cooling)

$EF_{LB\ or\ MB,n}$ – EF (LB – country average energy mix, or MB – national residual mix) for the year n, tCO_2e/kWh (electricity) or tCO_2e/GJ (heat, steam, cooling)

3.3. SCOPE 3 EMISSIONS

Scope 3 encompasses indirect GHG emissions that occur indirectly within the value chain of the WWTP. These emissions include those that arise upstream in the supply chain, as well as downstream in the processes of waste disposal or the beneficial reuse of by-products, along with the associated transportation activities. The categorization of GHG emissions, as outlined by the GHG Protocol [18] encompasses a total of 15 distinct categories. In this article, the authors delineate the categories that exhibit the highest likelihood of being of paramount significance for a typical WWTP organisation. As stated, the determination of categories to be included in the calculation of CF is conducted through a materiality analysis using the Scope 3 screening procedure.

The overall Scope 3 emission equation (4) is given below.

$$CF_{Scope\ 3\ _{i,n}}[tCO_2e] = C_{i,z,n}\,[usage\ units] \cdot EF_{i,z,n}\left[\frac{tCO_2e}{unit}\right] \quad (4)$$

$CF_{Scope\ 3\ i,n}$ – Scope 3, category i CF result of the year n, tCO_2e

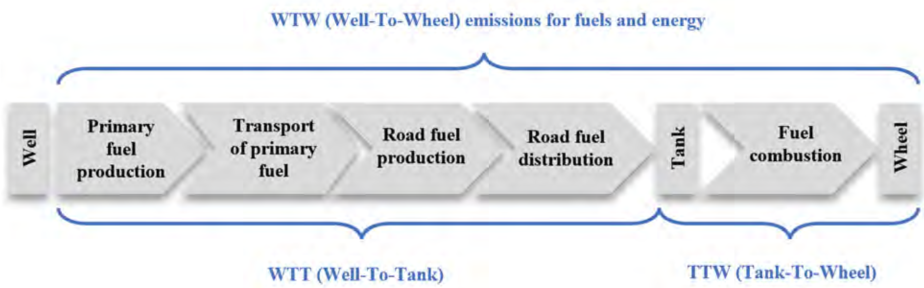
$C_{i,\ n}$ – z-type consumption in the year n, usage units

$EF_{i,z,n}$ – z-type EF for category i emissions for the year n, $tCO_2e/unit$

The Tables (2 – for Scope 3 upstream categories; 3 – for Scope 3 downstream categories) provided, contain a summary of the proposed work for evaluating emissions in the value chain. These include the rationale for selecting specific categories, recommended databases for emission factor data, and the minimum data necessary for calculating CF.

Special explanation should be given in case of the Category 3 od GHG Protocol Scope 3 – fuel – and energy-related activities, not included in Scope 1 or Scope 2. Scope 3 within Category 3 encompasses WTT (Well-To-Tank) emissions, which refers to emissions produced during the fuel production stage and the use of fuels for energy generation, including electricity, heat, steam, and cooling streams [18]. It also includes emissions associated with the energy transmission and distribution losses. The terminology employed within category 3 is derived from the fuel life cycle and pertains to distinct stages, namely extraction (well), fuel storage (tank), and utilization/combustion (wheel). Components of category 3, reflecting as said fuel life cycle, constitute a portion of Well-To-Wheel (WTW) emissions, which are presented in the life cycle phases on the Scheme 1. The residual portion the WTW emissions consists of TTW (Tank-To-Wheel) emissions, which occur when fuels are combusted within the organization's facilities (Scope 1) or during the generation of electricity procured by the organisation (Scope 2).

The provided Tables 2 and 3 present the summarised recommendations of authors regarding the calculation procedure for Scope 3 categories in a 'typical case' municipal wastewater treatment plant (MWWTP): Table 2 – upstream, Table 3 – down-



Scheme 1. Well-To-Wheel life cycle for fuels and energy with characteristic points and stages.

Schemat 1. Cykl życia Well-To-Wheel dla paliw i energii z charakterystycznymi punktami i etapami.

stream. This 'typical scenario' entails the presence of a reporting organisation that possesses both a treatment facility and a sewer collection system. The WWTP under consideration comprises several stages, including the mechanical treatment stage, the BNR removal process, and the production of biogas through anaerobic digestion, which is further utilised for energy co-generation to meet the plant's own energy needs. The organisation under consideration does not possess or exercise authority over sludge or other waste disposal sites. There are also simpli ed recommendations for each category indicated as well as minimal data required for the calculation in a 'quick-win' scenario which can be only introduced in the case of rst year calculation when data gaps are detected. The authors emphasise that the information presented in this paper should be regarded as a mere indication and must be substantiated by additional evidence of consolidation process analysis results and Scope 3 screening procedure conclusions.

Equations dedicated for the upstream indirect emission calculations:

$$CF_{Scope\ 3\ _{4,m,n}}[tCO_2e] = aL_{4,m,n}\,[t] \cdot tD_{4,m,n}\,[km] \cdot EF_{4,m,n}\,\left[\frac{tCO_2e}{tkm}\right] \quad (5)$$

$CF_{Scope\ 3\ 4,m,n}$ – Scope 3, category 4 CF result for m-mode transportation of the year n, tCO_2e

$aL_{4,m,n}$ – average vehicle load in m-mode transportation in the year n, t

$tD_{4,m,n}$ – total distance done by m-mode transportation cargo in the year n, km

$EF_{4,m,n}$ –EF for category 4 emissions and m-mode transportation for the year n, tCO_2e *per tonne-kilometre* (tkm)

Scope 3 Category 4 emissions can be also calculated on the basis of total cargo load in tonnes and average distance made via each transportation mode.

$$CF_{Scope\ 3\ _{6,m,n}}[tCO_2e] = P_{6,m,n}\,[passenger] \cdot D_{6,m,n}\,[km] \cdot EF_{4,m,n}\,\left[\frac{tCO_2e}{pkm}\right] \quad (6)$$

$CF_{Scope\ 3\ 6,m,n}$ – Scope 3, category 6 CF result for m-mode transportation of the year n, tCO_2e

$P_{6,m,n}$ – number of the employees taking par in a journey via m-mode transportation in the year n, *passenger* (p)

$D_{6,m,n}$ – total distance done by m-mode transportation during the journey (two way) in the year n, km

$EF_{6,m,n}$ – EF for category 6 emissions and m-mode transportation during for the year n, tCO_2e *per passenger-kilometre* (pkm)

Table 2. Summary of recommendations for Scope 3 calculation – upstream categories.

Tabela 2. Podsumowanie zaleceń dotyczących obliczania zakresu 3 – kategorie upstream.

Cat. No.	Scope 3 emission Category name	Authors' proposed CF calculation algorithm statement for a typical MWWTP with justification	First year CF calculation method proposed as 'quick-win' scenario	EF data base potential sources *commercial
Upstream indirect emissions				
1	Purchased goods and services	Relevant – cradle-to-gate emissions of the crucial materials purchased and used in the reporting year by the facility such as at least chemical agents should be enclosed	Purchased goods: calculation on the basis of site-specific data in kg, t or L, m^3 (Equation 3 and 4) Services: calculation via spend-based method (Equation 1 and 2) and then in the following years on the basis of actual materials and electricity usage of services	EcoInvent* Climatiq DEFRA Scientific papers
2	Capital goods	Relevant – emissions resulting from construction services (linear and cubature installations) purchased and done in the reporting year	Calculation via spend-based method (Equation 1 and 2) and then in the following years on the basis of actual materials and electricity usage of services (Equation 3 and 4)	EcoInvent* Climatiq Scientific papers
3	Fuel – and Energy-Related Activities, Not Included in Scope 1 or Scope 2	Relevant – emissions resulting from fuels generation and transportation services (WTT) as well as WTT of energy purchased and used with transmission and distribution (T&D) losses emission; all fuels (given in Scope 1) and energy (included in Scope 2) should be covered by WTT calculation	Calculation via actual energy consumption values in kWh or GJ (Equation 4)	EcoInvent* KOBiZE EEA IEA DEFRA
4	Upstream transportation services	Relevant – emissions resulting from external transportation services of the purchased goods covered in Category 1 calculation	Calculation on the annual basis of: average transportation mode type load in tonnes and total distance of each cargo in kilometres (Equation 5); transportation modes include wheel, rail, ferry, air	EcoInvent* Climatiq DEFRA
5	Waste generated in operations	Relevant – emissions resulting from process waste transportation via external services and its final disposal (excluding potential recycling procedures)	Calculation on the basis of the total mass (tonnes) of each type of waste generated with the final disposal method information provided for each waste stream (Equation 4)	EcoInvent* Climatiq DEFRA
6	Business travel	Irrelevant yet recommended for calculation on the basis of GHG Protocol good practice – emissions resulting from work-related journeys taken by the employees during the reporting year via vehicle not owned or controlled by organisation	Calculation on the annual basis of: number of employees travelling and total distance of business journey in kilometres (Equation 6) for each mode of transport (taxi, coach, train, ferry, aeroplane)	EcoInvent* Climatiq DEFRA
7	Employee commuting and remote working	Irrelevant – in comparison to other emission types, Category 7 may not play a relevant contribution to WWTP's CF structure	-	-
8	Upstream leased assets	Irrelevant – in comparison to other emission types, Category 8 may not play a relevant contribution to the WWTP's CF structure due to rare situation when the WWTP rents/leases its assets (e.g., offices, warehouses) for external entities	-	-

Table 3. Summary of recommendations for Scope 3 calculation – downstream categories.

Tabela 3. Podsumowanie zaleceń dotyczących obliczania zakresu 3 – kategorie downstream.

Cat. No.	Scope 3 emission Category name	Authors' proposed CF calculation algorithm statement for a typical MWWTP with justification	First year CF calculation method proposed as 'quick-win' scenario	EF data base potential sources *commercial
		Downstream indirect emissions		
9	Downstream transportation services	Irrelevant – Category 9 may not appear at all in the WWTP's CF structure due to rare situation when the WWTP releases final products to the market	When WWTP releases final products (e.g., struvite granulates or fertilizers, sand recovered from the grit), the transportation services (collection of goods) should be included as in Category 4	EcoInvent* Climatiq DEFRA
10	Processing of sold products	Irrelevant – Category 11 may not appear at all in the WWTP's CF structure due to rare situation when the WWTP releases intermediate products to the market	When WWTP releases intermediate products requiring further processing, emissions should be calculated via Equation 4	EcoInvent* Climatiq DEFRA Scientific papers
11	Use of sold products	Irrelevant – Category 10 may not appear at all in the WWTP's CF structure due to rare situation when the WWTP releases final products to the market	Category 11 emissions should be calculated only when Category 12 is included in the CF; usually emissions included in this category result from fertilizers usage	EcoInvent* Climatiq DEFRA Scientific papers
12	End-of-Life (EoL) of sold products (intermediate product, if relevant)	Irrelevant – Category 12 may not appear at all in the WWTP's CF structure due the fact that all possible products do not possess EoL stage; there can be a rare situation of biopolymers and bioplastic released by WWTP to the market which requires Category 12 calculation	-	-
13	Downstream leased assets	Irrelevant – in comparison to other emission types, Category 13 may not play a relevant contribution to the WWTP's CF structure due to rare situation when the WWTP rents/leases assets (e.g., offices, warehouses) from external entities	-	-
14	Franchises	Irrelevant – franchise mechanism rarely present in WWTP area	-	-
15	Investments	Irrelevant – WWTP rarely plays a Financial Market Participant (FMP) role	-	-

3.4. DOUBLE COUNTING ELIMINATION

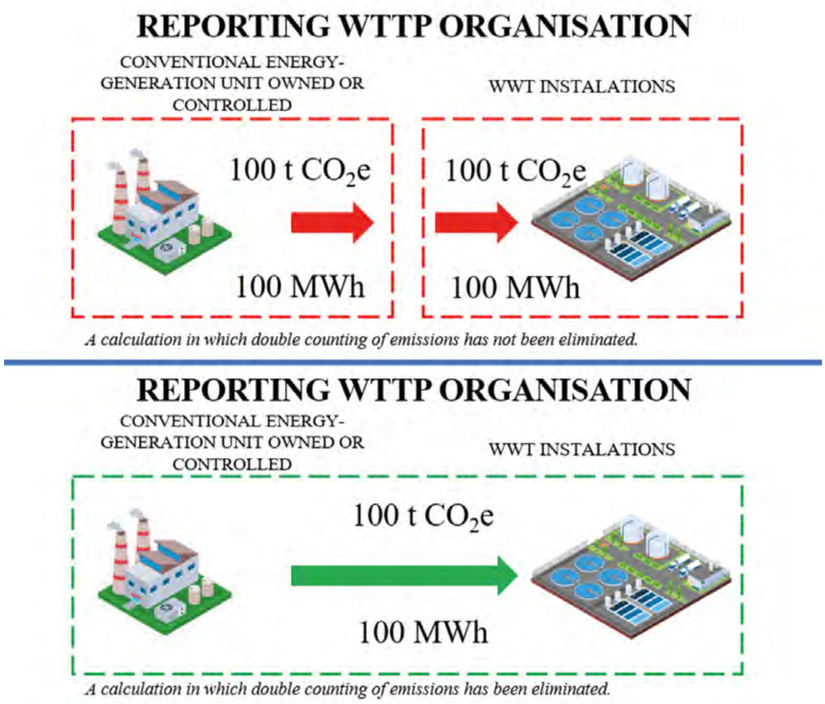
When performing carbon footprint calculations, it is crucial to exercise prudence in order to prevent the potential occurrence of duplicating greenhouse gas emissions. There are two distinct occurrences of double counting that can be discerned: (1) those that emerge as a result of organisational structure, and (2) those that manifest due to errors in operational structure.

One instance of double counting arises when an internally generated good or energy ow is erroneously considered as an external input (Scheme 2).

Second type of double counting mistakes occurs as result or wrong type or Scope emission assignment within the CF structure given by GHG Protocol guidelines. Common mistakes are summarized by the authors in the Table 4.

4. CARBON FOOTPRINT ALGORITHM OF MWWTP. SCOPES 2 AND 3 – STEP BY STEP INSTRUCTION

The methodology employed in this study to compute the carbon footprint of MWWTP is provided as a two-part article. It comprises seven distinct stages that align with the rules established by the GHG Protocol and the criteria outlined in the CSRD. The subsequent procedures are as follows:



Scheme 2. Organisational cause double counting case – CF calculation without its elimination (upper part) and with its elimination (bottom part).

Schemat 2. Przypadek podwójnego liczenia przyczyny organizacyjnej – obliczenie CF bez jej eliminacji (górna część) i z jej eliminacją (dolna część).

Table 4. Summary of common operational cause mistakes committed during WWTP's CF calculation.
Tabela 4. Podsumowanie typowych błędów operacyjnych popełnianych podczas obliczania CF dla OŚ.

Sc. No.	Scope and or Category involved in the scenario	Scenario description	Justification
1	Scope 1 vs. Scope 2	Emissions linked to energy produced in the facilities owned or controlled by the organisation calculated both in Scope 1 (stationery fuels combustion) and Scope 2 (energy purchase)	Scope 1 – direct GHG emissions from energy-production units Scope 2 – indirect GHG emissions from generation of externally purchased energy streams (CF calculation done via both LB and MB method)
2	Scope 1 vs. Scope 3, Category 4 and 9	Emissions linked to transportation of purchased goods and distribution of the products done by owned or controlled vehicles calculated both on the basis of fuel consumption (Scope 1) and tonne-kilometres (Scope 3, Category 4 or 9)	Scope 1 – direct GHG emissions from fuels combustion in vehicles owned or controlled by the organisation Scope 3 – indirect GHG emissions from transportation services acquired externally
3	Scope 1 vs. Scope 3, Category 5	Emissions linked to waste transportation done by owned or controlled vehicles calculated both on the basis of fuel consumption (Scope 1) and tonne-kilometres (Scope 3, Category 5)	Scope 1 – direct GHG emissions from fuels combustion in vehicles owned or controlled by the organisation Scope 5 – indirect GHG emissions from transportation and final waste disposal
4	Scope 1 vs. Scope 3, Category 6	Emissions linked to business travel done by owned or controlled vehicles calculated both on the basis of fuel consumption (Scope 1) and tonne-kilometres (Scope 3, Category 6)	Scope 1 – direct GHG emissions from fuels combustion in vehicles owned or controlled by the organisation Scope 6 – indirect GHG emissions from transportation services acquired externally (taxi, uber)
5	Scope 3, Category 4 vs. Scope 3, Category 3	Emissions linked to the transportation of fuels (e.g., coal, burning oil) via external services calculated in Scope 3, Category 4 instead of Scope 3, Category 3	Scope 3, Category 3 – indirect GHG emissions related to the lifecycle of fuels, including its transportation services stage Scope 3, Category 4 – indirect GHG emissions related to transportation services of purchased goods

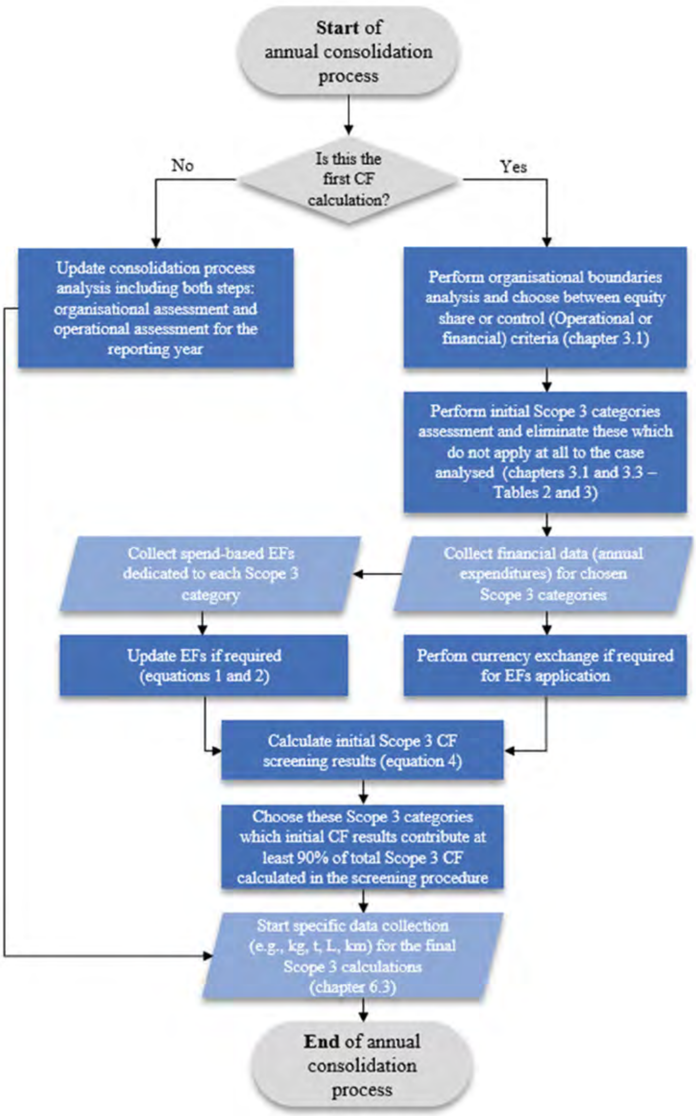
1. Consolidation process evaluation.
2. Scope 1 emission calculation:
   a. GHG emissions from the supportive activities,
   b. N_2O direct fugitive emissions,
   c. CH_4 direct fugitive emissions from wastewater treatment path,
   d. CH_4 direct fugitive emissions from sludge management, biogas production and utilisation.
3. Scope 2 (location – and market-based methods) calculation.
4. Scope 3 calculation.
5. Results summary, uncertainty discussion and report preparation.
6. Conclusions in the area of data aggregation.
7. Carbon footprint results analysis and GHG emission reduction planning.

This part two article presents rules pertaining to Scope 2 and 3, with a focus on steps 1, 3, 4, 5, 6, and 7. In order to improve the comprehensibility of the suggested computational methodology for CF, we have incorporated seven additional decision trees (referred algorithms) accompanied by appropriate guidelines. This article provides a comprehensive analysis of greenhouse gas calculation methodologies employed in wastewater treatment plants. , it references ng:

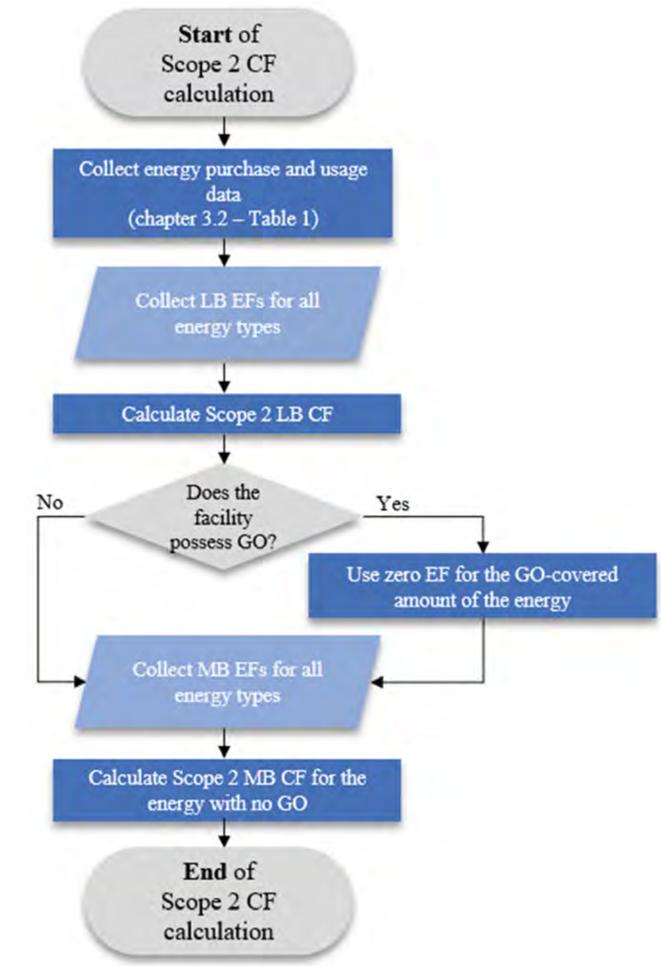
- IPCC – 2006 methodology with 2019 Re ment, Guidelines for wastewater treatment and discharge (Vol. 5, Chapter 6) and its default value summary tables,
- NGA/NGER – 2023 methodology and its default value summary tables,
- U.S. Protocol – 2013 methodology and its default value summary tables from Appendix F.

The implementation of Algorithm 1 procedure, as mandated by the GHG Protocol, is an annual requirement. Ensuring the completion of the consolidation process is of utmost importance, particularly in instances involving the acquisition of new facilities. It is advisable to initiate the collection of data for Scope 3 promptly following the acquisition of screening outcomes, as the data collection procedure is notably laborious and time intensive.

The implementation of Algorithm 1 procedure, as mandated by the GHG Protocol, is an annual requirement. Ensuring the completion of the consolidation process is of utmost importance, particularly in instances involving the acquisition of new facilities. It is advisable to initiate the collection of data for Scope 3 promptly following the acquisition of screening outcomes, as the data collection procedure is notably laborious and time intensive.



Algorithm 1. Decision tree and instruction for consolidation process procedure – WWTP case.
Algorytm 1. Drzewo decyzyjne i instrukcja postępowania w procesie konsolidacji – dla OŚ.



Algorithm 2. Instruction for WWTP Scope 2 (location-based and market-based methods) carbon footprint calculation.
Algorytm 2. Instrukcja obliczania śladu węglowego OŚ w Zakresie 2 (metody location – i market-based).

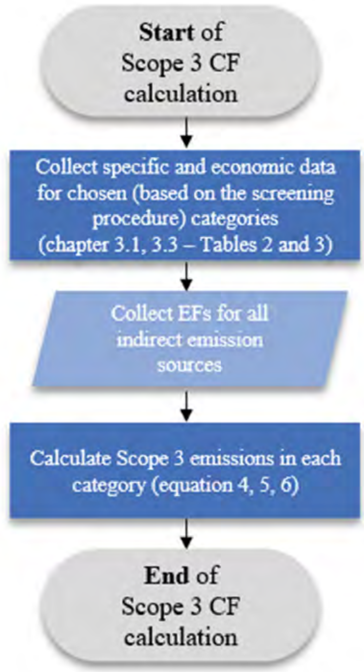
The instructions for calculating Scope 2 carbon footprint are provided in Algorithm 2, which includes both the location-based and market-based methods. This phase necessitates the preparation of the wastewater treatment plant's energy framework, encompassing both quantitative and qualitative data. In the event that the organisation procured GOs or energy through a specialised green tariff during the reporting year, it is necessary to include this information (in addition to the themselves) in order to apply a zero emission factor for the corresponding amount of energy covered in the MB method.

The data gathered during this phase is additionally utilised for the calculation of Scope 3 Category 3 emissions, for energy-related indirect emissions, such as WTT and T&D losses.

The seventh algorithm is a streamlined set of instructions for conducting Scope 3 calculations, as outlined in chapter 6.4 of this paper.

When CF calculation procedure is done and the results are obtained in each of the three Scopes, it is crucial to present the Cf structure in a proper way required by GHG Protocol guidelines. Example of the CF results submission is presented in the Table 5.

Given the intricate nature of the data collection process involved in CF calculation, it is advisable to generate an internal report within the organisation subsequent to the of CF calculation. This report should encompass data aggregation conclusions, with the aim of identifying any existing data gaps and proposing potential solutions for future data acquisition. Additionally, the report should address issues pertaining to low data quality and document pertinent information regarding the team responsible for the data collection procedures.



Algorithm 3. Instruction for WWTP Scope 3 carbon footprint calculation.
Algorytm 3. Instrukcja obliczania śladu węglowego OŚ w Zakresie 3.

Table 5. An example of the final WWTP's CF results.
Tabela 5. Przykład prezentacji finalnych wyników CF dla OŚ.

Scope	Total GHG emission [t CO_2e]	CO_2 emission [t CO_2e]	CH_4 emission [t CO_2e]	N_2O emission [t CO_2e]
Scope 1				
Scope 2 LB				
Scope 2 MB				
Scope 3 – TOTAL (MB)				
Category 1				
Category…				
Category n				
TOTAL CF				

The calculation of carbon footprint is deemed comprehensive only when it is accompanied by an analysis of the carbon footprint structure, in which the planning procedure for greenhouse gas reduction is outlined or prepared as a strategy for limiting emissions.

5. UNCERTAINTY DISCUSSION

As per the guidelines outlined by the GHG Protocol, it is imperative to include an analysis of uncertainty or, at the very least, engage in a thorough discussion of it within the calculation procedure. The selected methods for validating uncertainty are outlined in Table 6, accompanied by the authors' rationale for assessing the level of di ulty in implementing these methods. A three-tiered rating system is provided, consisting of low, medium, and high levels. Authors recommend providing at least uncertainty qualitative discus- culation.

Uncertainty in the context of WWTPs arises from two sources: measurement uncertainties, which pertain to the quality of the data collected, and uncertainties associated with the calculation of emission factors, which relate to the quality of the factors used in estimating emissions. The assessment of data quality in WWT processes can be straightforward when it comes to qualitative discussions. However, evaluating EF may present certain challenges. Hence, the authors put forth a methodology for ascertaining the level of uncertainty associated with emission factors, denoted as Scheme 5

6. CONSLUSIONS

This paper highlights the need for standardized approaches in calculating GHG emissions from WWTPs to meet the requirements CSRD and ESRS indicators. The developed algorithm for calculating the carbon footprint of WWTPs, provides a comprehensive framework for assessing GHG emissions. .

The calculation of Scope 1, 2, and 3 GHG emissions, as outlined by the GHG Protocol, presents a challenge in terms of data aggregation for wastewater utilities. As a result, it is recommended that these utilities adopt a best practise approach by establishing a dedicated team or department responsible for the calculation of carbon footprints.

The consolidation process plays a crucial role in obtaining accurate CF results by establishing calculation boundaries. Errors made during this stage

The levels of energy consumption and the energy structure of a WWTP, s cally the proportion of renewable energy sources in the mix, can pose a impact on the facility's CF results. The proposed calculation algorithm allows for the assessment of the i nce of stakeholders' decisions in this area on the indirect emissions levels in Scope 2, through a comparison of the LB and MB methods.

In order to conduct a comprehensive comparison of CF results, it is essential to establish emission intensity ratios (expressed as $kgCO_2e/m^3$ of treated wastewater). Relying solely on total CF levels may lead to inaccurate or incomplete conclusions.

The emissions associated with the value chain of rapidly expanding WWTP utilities are likely to be higher due to the dynamic development of infrastructure.

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Table 6. Summary of chosen uncertainty assessment methods.

Tabela 6. Podsumowanie wybranych metod oceny niepewności.

Uncertainty assessment method	Short description	Difficulty of implementation
Qualitative discussion	• Sources of uncertainty are listed and discussed. • The relative value of the imprecision may be given if known.	Low
Expert evaluation of data quality	• The assessment is based on expert judgement; it may be combined with a qualitative discussion. • This method is also present in IPCC (2019) methodology where uncertainty range given by the expert judgment by lead methodology authors are given	Low
Data Attribute Rating System – DARS	• The numerical values representing the relative uncertainty are assigned using objective methods.	Medium
Expert estimation method	• Experts estimate the parameters of the emission distribution (i.e., mean, standard deviation and type of distribution). • Simple analytical and graphical techniques are then used to estimate confidence limits based on the assumed distribution data. • Delphi or Pedigree Matrix methods are used.	Medium
Error propagation method	• The means and standard deviations for the emission design parameters are estimated using expert judgement, measurements or other methods. • Standard error propagation statistical techniques are used, typically based on Taylor series expansions, which are then used to estimate the compound uncertainty.	Medium
Direct simulation method	• The Monte Carlo method and other numerical methods are used to directly estimate confidence intervals of individual emission components. In the Monte Carlo method, expert judgment is used to estimate the values of the distribution parameters before the Monte Carlo simulation is performed. Other methods do not require such assumptions.	High
Direct or indirect measurement (validation) method	• Direct or indirect measurements of the Organisation's emissions are made; these are then used for direct emission calculations and uncertainty analysis. • These methods include direct measurements such as stack sampling, Fourier-Transform Infrared Spectroscopy (FTIR) analyses and indirect measurements such as tracer tests. • These methods also provide data for validation of emission estimates and emission uncertainties.	High

Paulina Szulc-Kłosińska
Imię i nazwisko doktoranta

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Comparative Analysis of Calculation Tools for Estimating Carbon Footprint of Wastewater Treatment Plants: Methodologies and Emission Factors

Dear Paulina Szulc-Kłosińska,

I am pleased to inform you that your manuscript, entitled: Comparative Analysis of Calculation Tools for Estimating Carbon Footprint of Wastewater Treatment Plants: Methodologies and Emission Factors, has been finally accepted for publication in our journal.
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Original paper

Comparative Analysis of Calculation Tools for Estimating Carbon Footprint of Wastewater Treatment Plants: Methodologies and Emission Factors

Title (in Polish)

Analiza porównawcza narzędzi obliczeniowych do szacowania śladu węglowego oczyszczalni ścieków: metodologie i współczynniki emisji

Keywords (in English)

GHG emissions, CF validation, Emission factors, Nitrous oxide, Methane, Sustainability

Abstract (in English)

Objectives

Two tools for estimating the carbon footprint (CF) of wastewater treatment plants (WWTPs): CFCT (versions 2014 and 2024) and ECAM (2024) were compared. A validation tool was developed using empirically derived emission factors (EFs) for nitrous oxide (N2O) and methane (CH4) from the activated sludge process. The study had an applied character, assessing the accuracy, consistency, and sustainable reporting utility of the tools.

Material and methods

The Poznań WWTP (100,000 m³/d; 1000,000 PE) was selected for analysis. Data from 2021 and results from a four-day measurement campaign of N2O and CH4 emissions were used. Measurements were conducted with a Fourier Transform Infrared (FTIR) spectrometer (Gasmet DX4000) and a dedicated floating hood. Obtained EFs were implemented in the validation tool. The comparison followed a specially designed procedure to assess the impact of different GWP values (2007, 2014, 2021) and EF sources (default vs. empirical).

Results

CFCT (2014) estimated 29,310 tCO2e/year; version 2024: 28,730 (-2%). ECAM reported 54,370 (+88% compared to CFCT 2024). The validation tool computed results: 10,344 (GWP 2007), 10,077 (2021), and 9,995 (2014), i.e., ~65% lower than ECAM. Scope 1 (S1) emissions dominated over Scope 2 (S2), accounting for more than 75% of total emissions.

Conclusions

EF selection is critical - default values may overestimate CF by up to 200%. Using empirical data increases accuracy and reflects actual conditions. Methodological harmonization and site-specific adaptation are essential for reliable reporting. CF tools should be empirically validated before being used in operational decision-making. This study supports the development of adaptive, sustainability-oriented tools in the WWTP sector.

Abstract (in Polish)

Cel pracy

Celem artykułu była porównawcza analiza dwóch narzędzi do szacowania śladu węglowego (CF) oczyszczalni ścieków: CFCT (wersje 2014 i 2024) oraz ECAM (wersja 2024). Opracowano również narzędzie walidacyjne z wykorzystaniem współczynników emisji (WE) dla podtlenku azotu (N2O) i metanu (CH4) z procesu osadu czynnego wyznaczonych empirycznie. Praca miała charakter aplikacyjny - oceniano dokładność, spójność i przydatność narzędzi dla potrzeb raportowania równoważonego rozwoju.

Materiał i metody
Analizie poddano oczyszczalnię ścieków (OŚ) w Poznaniu (100000 m³/d, 1000000 RLM). Wykorzystano dane z 2021 roku oraz wyniki czterodniowej kampanii pomiarowej emisji N2O i CH4, wykonanej za pomocą spektrometru FTIR (spektrometria w podczerwieni z transformatą Fouriera, urządzenie pomiarowe Gasmet DX4000) ze specjalną pływającą kopułą pomiarową. Uzyskane WE wprowadzono do autorskiego narzędzia walidacyjnego. Porównanie oparto na specjalnie zaprojektowanej procedurze, umożliwiającej ocenę wpływu różnych wartości GWP (2007, 2014, 2021) oraz wartości WE (domyślnie wartości literaturowe versus empiryczne).

Wyniki
CFCT (2014) zwrócił wynik 29 310 tCO2e/rok; 2024: 28 730 (−2%). ECAM: 54 370 (+88% względem CFCT 2024). Narzędzie walidacyjne: 10 344 (GWP 2007), 10 077 (2021), 9 995 (2014), czyli ~65% mniej niż ECAM. Zakres 1 (Z1) dominował względem Zakresu (Z2) (stanowiąc >75% sumy emisji Z1 oraz Z2).

Wnioski
Wybór WE ma kluczowe znaczenie - domyślne wartości mogą zawyżać CF nawet o 200%. Stosowanie danych empirycznych znacząco zwiększa trafność analiz i umożliwia odzwierciedlenie rzeczywistych warunków pracy. Ujednolicenie metodyk i uwzględnienie lokalnych uwarunkowań stanowi warunek rzetelnego raportowania środowiskowego. Narzędzia CF powinny być walidowane empirycznie, zanim zostaną użyte do decyzji operacyjnych. Praca wspiera rozwój nowoczesnych, adaptowalnych narzędzi zgodnych z celami zrównoważonego rozwoju w obszarze OŚ.

Comparative Analysis of Calculation Tools for Estimating Carbon Footprint of Wastewater Treatment Plants: Methodologies and Emission Factors

Keywords: GHG emissions; CF validation; Emission factors; Nitrous oxide; Methane; Sustainable Development Goals; Sustainability

Abstract

Wastewater treatment plants (WWTPs) are significant sources of greenhouse gas (GHG) emissions, particularly methane (CH_4) and nitrous oxide (N_2O). This study compares two widely used carbon footprint (CF) calculation tools, CFCT and ECAM, and validates the results with a self-developed Validation Tool. The findings reveal substantial differences in CF estimates, with ECAM reporting emissions more than twice as high as those computed by CFCT. The Validation Tool, which incorporates site-specific empirical emission factors (EFs), estimates emissions approximately 80% lower than the other tools. The analysis identifies key methodological limitations, including the oversimplification of N_2O emissions in CF models, inconsistencies in EF selection, and the lack of standardized validation methodologies. The study underscores the need for refining CF methodologies by integrating real-world operational data and establishing harmonized validation frameworks to enhance the reliability of emissions accounting.

1. Introduction

Wastewater treatment plants (WWTPs) are crucial for urban infrastructure, ensuring water quality and compliance (Zhang et al. 2024). Yet, they also contribute to environmental burdens due to high energy use and greenhouse gas (GHG) emissions, especially methane (CH_4) and nitrous oxide (N_2O), which significantly impact climate change. Reducing these two factors while meeting strict regulations is a key challenge (Asadi et al. 2024). Emissions vary due to regional differences in design, operation, technology and local conditions, complicating accurate assessment (Toivonen and Räsänen 2024). Carbon footprint (CF) is widely used to quantify and reduce WWTP emissions, aiding design and operations while supporting policy goals like the European Green Deal. CF is a widely applied method, yet it often uses default emission factors (EFs) and global warming potential (GWP) values, even within advanced scenario analysis.

Existing CF tools have key limitations (Fighir et al. 2019) as many rely on fixed EFs, causing inaccuracies under specific facility conditions. Inconsistencies with IPCC guidelines, including outdated GWPs and EFs, reduce credibility (Massara et al. 2017) and cross-tool validation. Biogenic emissions, CH_4 leakages, and N_2O from biological treatment are often over- or underreported, leading to incorrect and incomplete CF estimates.

Current studies on WWTP GHG emissions show gaps (Toivonen and Räsänen 2024). There's no standardized framework for comparing CF outputs across plant configurations (Huang et al. 2022), and little validation using real-world data or regional traits (Faragò et al. 2022). Temporal impacts of revised GWPs on CF are underexplored (Wei et al. 2024). Research often focuses on specific technologies, overlooking broader CF tool relevance (Lotfikatouli et al. 2024). Fugitive CH_4 and N_2O emissions remain underreported, and CF calculators lack empirical validation, limiting their decision-making utility. Researchers rarely justify CF tool choice or compare outputs, leading to inconsistent conclusions (Tian et al. 2022).

Despite increased CF tool use, gaps in standardization and validation persist. No unified framework exists for comparing outputs across WWTP types, and real-world data use is limited in regions with poor infrastructure. Effects of changing GWPs on long-term estimates remain unaddressed. This study tackles these issues by assessing open-access CF tools and proposing a refined validation framework. It incorporates site-specific EFs and updated GWPs to boost accuracy and adaptability. By improving CF methods, it enables reliable reporting, aids tool selection, and supports development of standardized calculators. The findings assist utility managers, policymakers, and researchers in achieving globally consistent WWTP emission assessments.

2. Materials and methods

2.1. CF calculation tools

Supported by the broad adoption, two open-access CF tools were selected: Tool 1 (CFCT – versions 2014 and 2024) from Sweden (CFCT 2014, CFCT 2024), and Tool 2 (ECAM, WaCCliM 2024) from Germany. Tool 1 (v2014) uses 2007 GWP values (Pachauri and Reisinger 2007) and allows user modifications to the calculations. Tool 1 (v2024) includes updated EFs and GWPs (Lee and Romero 2023), but its equations and fixed factors are not user editable. Tool 2 is a web-based tool with default equations, EFs, and GWP values (Pachauri and Meyer 2014); however, users can request computation model changes via its developers. Both CFCT (Nejad 2020) and ECAM (Tian et al. 2022) are widely used in research (Saidan et al. 2019). Table 1 provides a comparison of Scope 1 process emission components and EF considerations in both tools, focusing on N_2O and CH_4 to highlight methodological similarities and differences.

Table 1. Scope 1 process emissions calculation completeness – comparison of the CF tools selected for the in-depth analysis

Tool name	**Nitrogen-based calculations** N_2O emission		**Organics-based calculation** CH_4 emission		**Biogas-based calculation** CH_4 emission		
	BNR	**Recipient**	**BNR**	**Recipient**	**Combustion**	**Leakage**	**Upgrading Slip**
CFCT v2014	$N_{denetrified}$ (removed)	TN effluent load	COD influent load	Not calculated	Calculated as Scope 1	Possibility of calculation as Scope 1 if needed	Calculated as default Scope 1
CFCT v2024	$N_{denetrified}$ (removed)	TN effluent load	COD influent load	BOD effluent load	Considered as biogenic	Possibility of calculation as Scope 1 if needed	Calculated as default Scope 1

ECAM	TN influent load	TN effluent load	BOD influent load	BOD effluent load	Considered as biogenic	Calculated as default Scope 1	Possibility of calculation as Scope 1 if needed

TN – total nitrogen, COD – chemical oxygen demand, BOD – biochemical oxygen demand

2.2. CF validation tool

The CF algorithm for the municipal WWTP in this study follows GHG Protocol guidelines (WRI, WBCSD 2014) and IPCC standards (IPCC 2019), aligning with EU requirements (EU 2022/2464). The new tool, customized for the facility, includes:

1. **Scope 1 emissions:**

   a. N_2O from bioreactors and recipient,

   b. CH_4 from bioreactors and recipient,

   c. CH_4 from sludge management, biogas production, and use.

2. **Supporting activity emissions:**

   - Fossil fuel use in on-site power units
   - Scope 2: electricity from external grid.

Fugitive Scope 1 emissions are calculated using TN load for N_2O and COD load for CH_4. Emissions from biogas used for heat and power were excluded as renewable, but leakage and incomplete combustion were included.

2.3. Stepwise analysis procedure

A stepwise procedure was developed to compare the analysed CF tools. Figure 1 outlines the study structure, while Figure 2 details the multi-step evaluation approach. Six distinct CF results were assessed, comparing tool-default GHG EFs with updated and site-specific empirical EFs and GWPs. A new independent algorithm developed by the authors was used for validation. This structured comparison identified key contributors to direct emissions (N_2O and CH_4), ensuring that the most impactful elements inform the final recommendations.

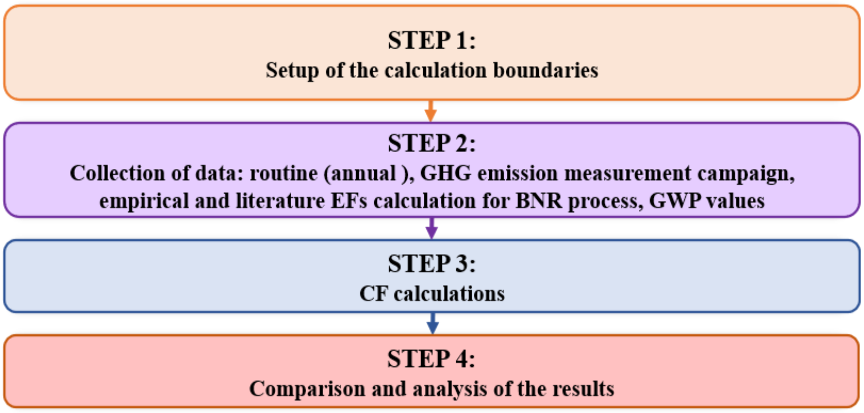

Figure 1. Study framework.

Step 1: Setup of the calculation boundaries (section 2.4)

The process begins with CF calculation boundaries setup. Each tool analysed covers the scope of the analysis.

Step 2: Collection of data (section 2.6)

The tools incorporate a primary dataset: annual qualitative and quantitative operational data from the studied WWTP, ensuring consistent full-scale input. Additionally, a special measurement campaign at the Poznań WWTP provided full-scale GHG emission data for estimating empirical EFs (N_2O and CH_4) specific to the facility's BNR process.

Step 3: CF calculations and validation (section 3)

The selected CF tools and collected data were used for calculations. The comparison applied various CH_4 and N_2O GWP updates and default EFs to evaluate their impact on CF results. Each outcome was then verified using Tool 3, which uses EFs from the GHG measurement campaign and incorporates all GWP sets.

Step 4: Comparison and analysis of the results (section 4)

The structured comparison reveals that changes in EFs and GWPs significantly affect CF results, pinpointing key areas to target for optimizing GHG mitigation.

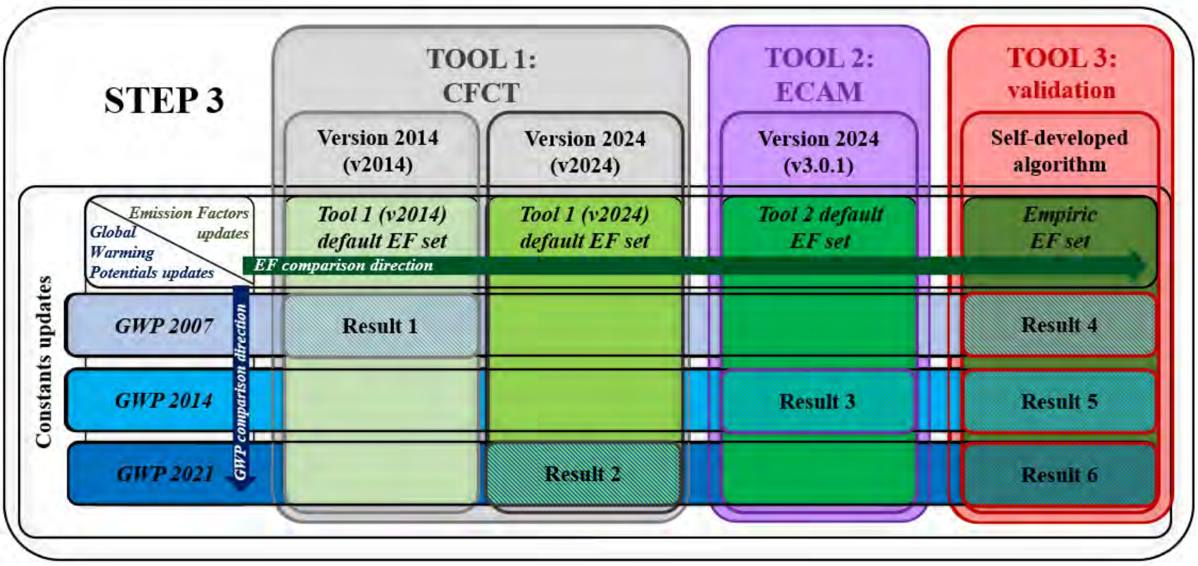

Figure 2. The concept of Step 3.

Step five, shown in Figure 2, is the study's critical phase. GWP values from IPCC reports: 2007 (Pachauri and Reisinger 2007), 2014 (Pachauri and Meyer 2014), and 2021 (Lee and Romero 2023), were combined with default EFs in the CF tools. The self-developed algorithm included all GWPs and empirical EFs. Two approaches were used: vertical comparison assessed GWP updates, while horizontal comparison showed results based on different EF values.

2.4. Setup of the calculation boundaries

Recently, CF has been used as a tool for controlling GHG impacts (Maktabifard et al. 2019). The GHG Protocol (WRI, WBCSD 2014) defines three CF areas (Figure 3). Scope 1 (S1) includes direct emissions, essential for emissions from WWT - N_2O from BNR processes, and CH_4 from anaerobic stages and sludge treatment with biogas. Scope 2 (S2) covers emissions from purchased energy, while Scope 3 (S3) involves indirect emissions, like those from chemical production or by-product disposal beyond the WWTP's value chain.

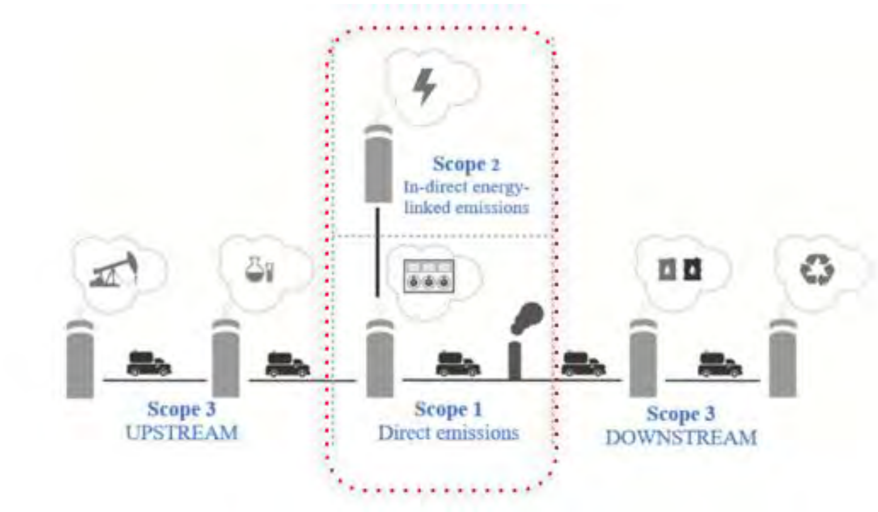

Figure 3. Overview of CF structure: scopes of GHG emission (WRI, WBCSD 2014). Red dotted line presents calculation boundaries of this study.

This research targets only S1 and S2 emissions within WWTP boundaries, as these are prioritized by regulations (EU 2022/2464, EU 2024/3019) and required for compliance (Burchart-Korol and Zawartka 2019). This boundary setting reveals how CF tool selection affects WWTP GHG emission composition and levels (Awaitey 2021). Scope 3 emissions, including chemical production and third-party transport, are excluded due to high uncertainty and limited WWTP control (Ko et al. 2024).

2.5. Study site

The Poznań WWTP (PWWTP) is among Poland's largest, treating about 100,000 m^3/day of wastewater from Poznań and nearby areas. It serves up to 1,000,000 PE (1 PE = 60 g BOD_5). The plant uses a Johannesburg (JHB) BNR configuration with six 25,500 m^3 activated sludge bioreactors. Effluent flows to the Warta River after secondary clarification. Primary and waste activated sludge are thickened and anaerobically digested at 35°C in 29,760 m^3 digesters. The digested sludge is dewatered and incinerated externally. Biogas from digesters powers on-site CHP units for heat and electricity, with energy demand supplemented by the grid and gas boilers. Figure 4 shows the PWWTP layout.

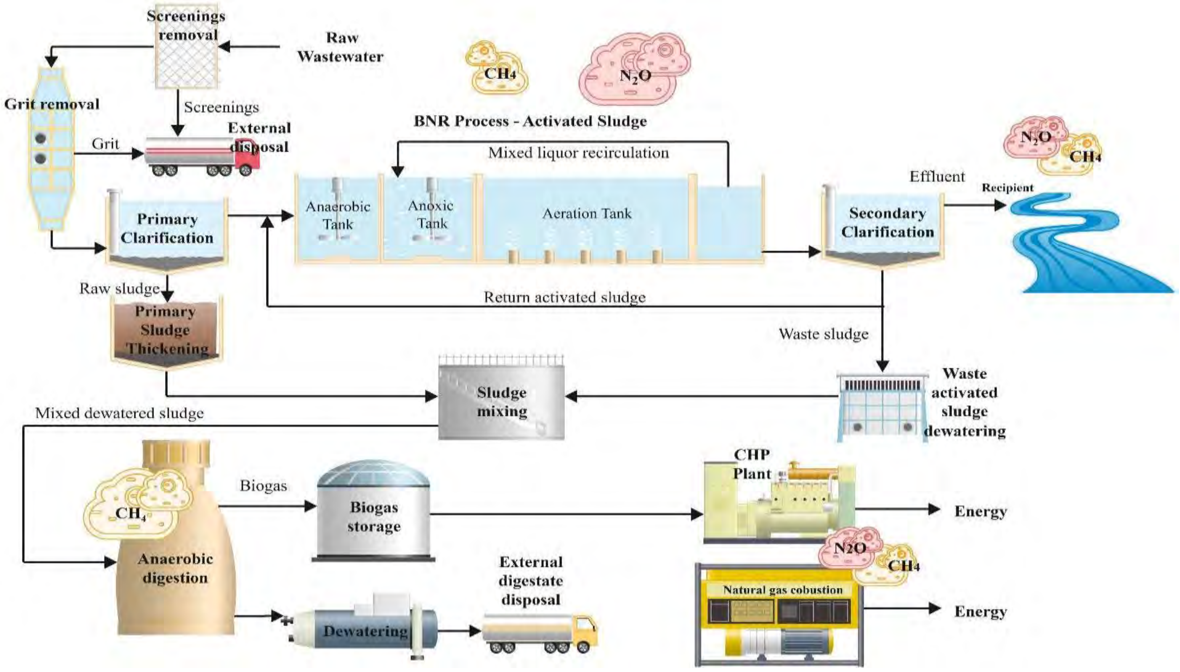

Figure 4. The idea of wastewater and sludge treatment processes employed in the investigated facility with GHG emission hotspots included in assessment.

2.6. Data collection

2.6.1. Routine data

The CF assessments use the 2021 routine operational dataset from PWWTP (Table 2). Average influent and effluent concentrations for COD, BOD_5 and TN were provided by the operator, based on biweekly sampling. Quantitative data on flow rates, fuel and biogas use, and energy production/consumption were recorded via water, gas, and electricity meters.

Table 2. Summary of the WWTP's annual average characteristics (2021).

		Influent	Effluent
Flowrate	*m^3/d*	100,000	
COD	*gO_2/m^3*	1,240	49.9
BOD_5	*gO_2/m^3*	550	3.7
TN	*gN/m^3*	100	9.4
		Biogas CHP plant	**External grid**
Electricity used	*kWh/a*	15,200,000	5,750,000
Biogas produced/consumed	*Nm^3/a*	7,760,000	-

Natural gas consumption	*m^3/a*	130,000

2.6.2. Emission factors

In September 2021, a 4-day full-scale campaign at PWWTP included routine data collection, sampling, lab tests, and off-gas measurements in biological reactors. These data were used to calculate direct N_2O and CH_4 EFs and apply them in Validation Tool 3. PWWTP operates autonomously, with DO levels adjusting via ammonia-based control. For this study, average DO was fixed at 1.0 ± 0.3 mg O_2/L and inflow at 600.0 ± 29.8 m^3/h. EFs were obtained by measuring four points in the aerobic compartment for one hour each, with 5-minute intervals, using FTIR (Gasmet DX 4000) via a floating hood. Wastewater samples were taken at each point, and the Gasmet was calibrated daily. To minimize uncertainty, the optimal GHG measurement point was selected pre-campaign. Final EFs, in kgCO_2e per kg COD (CH_4) and TN (N_2O), were weighted by compartment area and gas flux.

3. Results

Figures 5 and 6 present CF results comparing GWP and EF impacts. CFCT v2014 estimates 29,310 tCO_2e /a, slightly reduced to 28,730 tCO_2e /a in v2024. ECAM reports a significantly higher 54,370 tCO_2e /a - more than double CFCT values. The Validation Tool, adjusting for three GWP scenarios, gives much lower values: 10,340-10,080 tCO_2e /a (Figure 5a). S1 fugitive emissions dominate CF composition (Figure 5b), especially CH_4 and N_2O from processes. In CFCT v2014, these account for 22,280 tCO_2e/a (76% of total), and 21,820 tCO_2e/a in v2024. ECAM estimates 47,300 tCO_2e/a, over 87% of its total, with CH_4 from BNR and N_2O from effluent discharge showing ~211% higher values than CFCT v2024. Biogas emissions, including CH_4 leakage and slip, are also critical. CFCT v2014 and v2024 estimate 2,400 and 2,600 tCO_2e /a, respectively; ECAM shows 2,410 tCO_2e/a; the Validation Tool remains at 1,930

tCO_2e /a across GWPs (Figure 5b). Natural gas emissions are lower but vary: CFCT v2014 reports 0.321 tCO_2e /a, v2024 at 0.239; ECAM and Validation Tool are around 0.259 t CO_2e/a. Electricity-related Scope 2 emissions are consistent: 4.30 (CFCT v2014), 4.10 (v2024), and ~4.07 (ECAM, Validation Tool). GWP updates from 2007 to 2021 cause slight changes in total GHG emissions and CH_4/N_2O contributions to CF, having less impact than EF changes. Total CF aligns with IPCC's GWP trends for N_2O. For example, using the Validation Tool without EF changes, CF was 10,344 tCO_2e/year under GWP 2007, dropping to 10,077 (-2.58%) with GWP 2021 and 9,995 (-0.81%) under GWP 2014 (Figure 5a). Comparing Tool 3 results (Figure 6), CH_4's S1 share increased 2% due to an 8.8% GWP rise (from 25 to 27.2). Similarly, CFCT v2014 and v2024 show a 13% drop in total CF from 54,366 t CO_2e/year (GWP 2021) to 47,303 (GWP 2014). The horizontal comparison evaluates the impact of default vs. empirical EFs on CF estimates. Using default EFs, CFCT (v2014, v2024) and ECAM v2024 consistently show higher process GHG emissions than Tool 3, which uses empirical EFs. Tool 1 and Tool 2 report significantly higher annual CFs - by 18,192, 16,718, and 17,068 tCO_2e, respectively - than the Validation Tool (Figure 5a). Within Tool 1, the difference is small (459 t CO_2e/year), a 2.1% drop from v2014 to v2024. Figure 6 shows how EF changes affect both CF levels and emission composition. ECAM v2024 uses BOD-based EFs, different recipient assumptions, and IPCC (2019) tiers, leading to higher CH_4 emissions. Its role differs further as CFCT v2014 omits emissions from the recipient. These differences highlight varied approaches to biogenic process emissions. A clear disparity in N_2O CF levels is seen between Tool 1 and Tool 2 (Figure 7). N_2O emissions heavily influence CF due to EF selection. CFCT versions use 2010 EFs (Foley et al. 2010), while ECAM applies IPCC 2019 values, which are 37.6% higher.

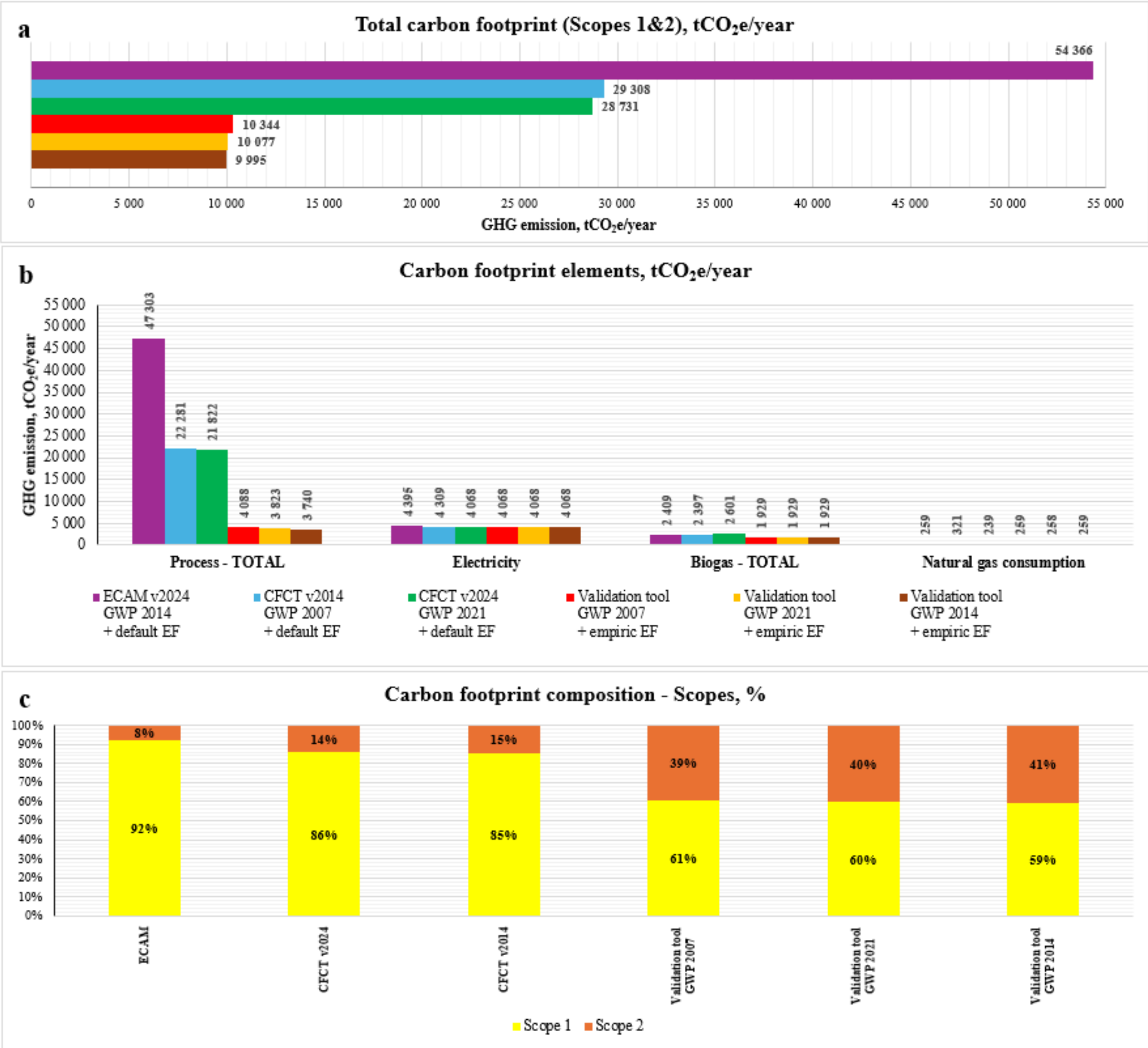

Figure 5. CF calculation results: a) Total CF (Scopes 1&2), tCO_2e/year; b) CF elements, tCO_2e/year c) Total CF composition - Scopes, %

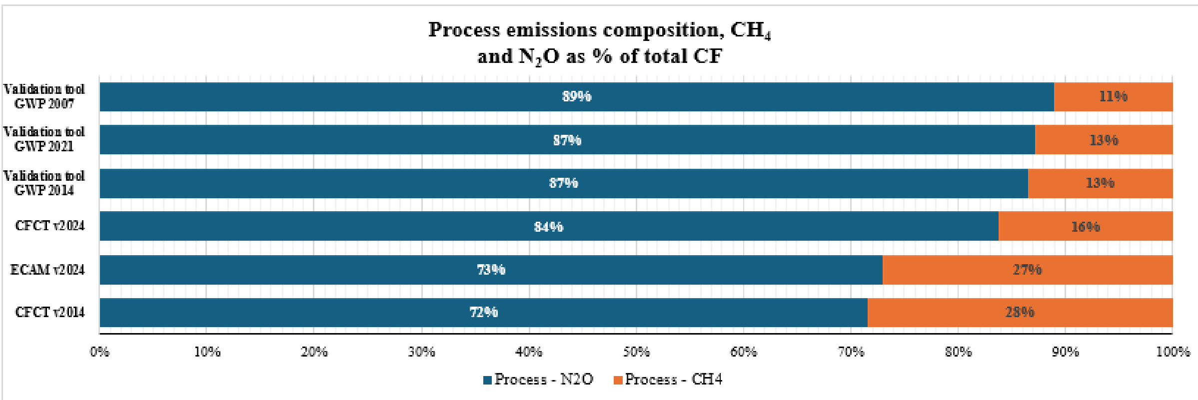

Figure 6. Process emission composition, CH_4 and N_2O as % of total (Scope1&2) CF

4. Discussion

CF estimates differ across tools due to variations in EFs, GWP constants, and calculation methods. S1 process emissions dominate GHG fluxes in all tools, particularly in CFCT (both versions) and ECAM, more than in the Validation Tool. S2 emissions, from purchased electricity, remain more consistent - with only a 7.5% variation - due to uniformly applied grid-based EFs. These results show that EF selection and methodology have a greater impact on S1 emissions than GWP updates, underscoring the need for standardized CF methods to enhance comparability and accuracy in WWTP assessments.

Heatmaps (Figures 7 and 8) visually show deviations from mean CF estimates by category [tCO_2e/a] using a blue-to-red gradient to indicate negative to positive variations. Despite differences in accounting methods, CH_4 and N_2O emissions from recipients remain relatively consistent (Figure 7). CFCT v2014 excludes CH_4 from recipient pathways, slightly lowering its process-related CF impact. ECAM's higher N_2O values suggest its BNR-specific EFs cover more emission pathways. In contrast, CFCT versions yield consistent results, implying minimal changes in methods or EFs over the past decade. Validation Tool results show GWP updates have less effect than EF, as CF values remain stable across the GWP scenarios.

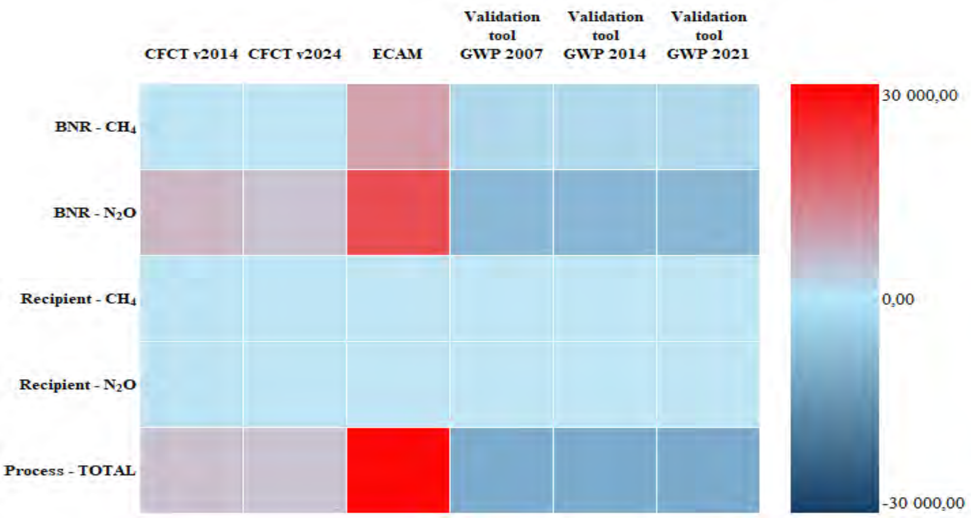

Figure 7. Heatmap of deviations from mean emissions by category - process, tCO_2e/a (2021).

Figure 8 illustrates how biogas combustion CF varies by tool. CFCT v2014, links biogenic CO_2 to direct emissions, while CFCT v2024 and other tools exclude it based on IPCC 2019. CFCT v2014 assumes high biogas capture efficiency, showing the lowest leakage, while v2024 shows moderate leakage, aligning more with operational realities. ECAM and the Validation Tool report higher leakage, stressing fugitive CH_4's impact on overall CF. Analysis of biogas slip emissions during upgrading reveals major methodological differences between CF tools. CFCT v2014 and v2024 show higher-than-average emissions, indicating alignment with Swedish WWTP practices not reflected in other tools.

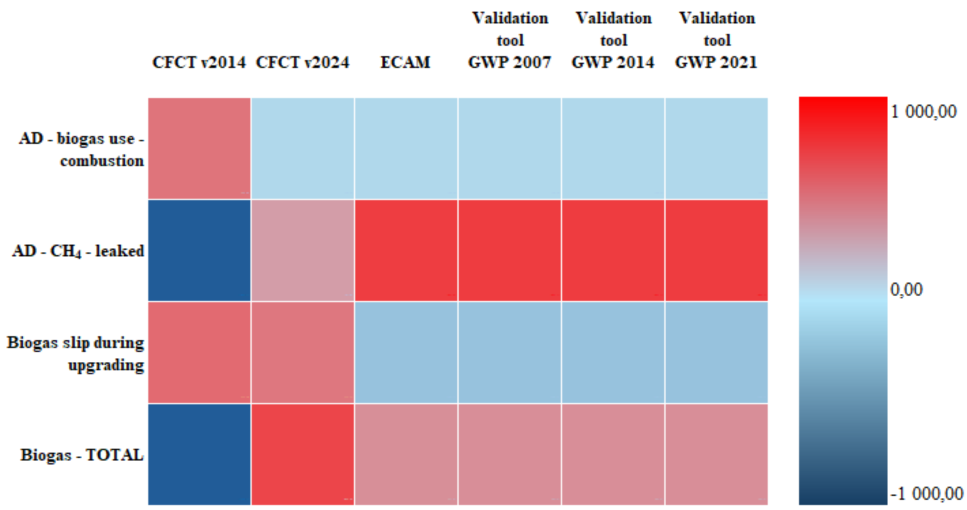

Figure 8. Heatmap of deviations from mean emissions by category - biogas, tCO_2e/a (2021).

ECAM and the Validation Tool, reflecting Central European norms, omit biogas slip due to its limited presence in the region. Despite covering more biogas-related categories, CFCT v2014 reports the lowest total emissions, showing that broader boundaries don't always yield higher CFs. In contrast, CFCT v2024 includes biogas slip, resulting in the highest emissions. ECAM and the Validation Tool consistently attribute emissions to leakage events, offering a conservative estimate. Figure 8's heatmap underscores the need for site-specific EF adjustments supported by real-world data. These variations highlight the importance of choosing CF tools that match a facility's actual biogas management scenario for informed policy and operational decisions.

This study reinforces that CFCT and ECAM tools depend heavily on EF assumptions (De Haas and Andrews 2022). Emission levels and CF composition highlight the limitations of using uniform EFs like those from IPCC, which can cause over- (De Haas and Andrews 2022) or under-estimation (Song et al. 2024) of fugitive emissions. It supports concerns about oversimplified N_2O calculations and argues that relying on generic EFs fails to reflect site-specific conditions, especially regarding nitrogen emissions (Maktabifard et al. 2021). Empirical BNR EFs in this study challenge De Haas and Andrews' (2022) suggested default of 1.1% TN, showing instead ~0.11% TN for the investigated facility. The tool comparison, showing minor S2 variation, confirms CF's relevance in sustainable energy optimization (Szaja and Bartkowska 2024). Differences in AD-related fugitive emissions support Fighir et al.'s (2019) call for harmonized biogas integration in CF tools, as biomethane recovery isn't fully captured. Findings align with prior research suggesting energy-efficient upgrades (e.g., aeration systems) can reduce S2 emissions (Maktabifard et al. 2021). The tools show limits in addressing regional and site-specific variability, causing potential CF biases. This study also confirms the dominance of N_2O in biological emissions (Maktabifard et al. 2021), with CH_4 and N_2O forming over 80% of S1 emissions (Smith et al. 2019). Inconsistencies persist in CFCT and ECAM's treatment of sludge and biogas emissions, supporting calls for tools to use flexible, region-specific EFs (Jiménez-Paute 2025) and updated GWPs for accurate policy reporting.

The findings highlight the need for CF tools to adapt to site-specific conditions and evolving regulations, ensuring emissions estimates match operational data and policy goals. Due to discrepancies across tools, the study recommends a harmonized validation framework using site-specific EFs and updated GWPs to correct for generalized assumptions. Accounting for fugitive emission variability is essential to improve CH_4 and N_2O estimates, as they significantly influence WWTP CF. Enhancing CF tools to reflect these trends will support better mitigation, reduce reporting uncertainty, and improve decision-making in WWTP operations.

This study offers valuable insights into CF estimations for WWTPs, but further research could improve its scope. Expanding to multi-year or multi-facility data would better capture GHG variability. Although S1 and S2 are assessed, future work could include S3 emissions for a fuller environmental footprint. Moreover, including tools from regions like Australia would broaden the analysis. Tool structure, input needs, and output formats differ, complicating direct comparisons and highlighting the need for methodological harmonization. Operational strategies like aeration regimes, which affect emissions, were not fully explored. Integrating such factors could clarify how operations influence CF over time, improving assessment accuracy and offering practical guidance for sustainability and emissions control.

This research underscores the potential for adaptive CF tools using machine learning to calibrate EFs in real time from operational data, improving emission estimation accuracy. Real-time data use enables long-term trend analysis, boosting predictive insights and operational planning. Site-specific EFs enhance CF comparability across regions, supporting the creation of a shared emission database reflecting diverse WWTP setups, climates, and efficiencies. Regional CF models should incorporate climate, treatment methods, and regulations, including standard biogenic emissions accounting. The effect of climate change, like extreme weather, on CFs also warrants study, as it impacts treatment efficiency and emissions. In-situ biogas leakage data is vital for more accurate benchmarks than current assumptions. Future research should assess how resource recovery - especially circular economy strategies - can reduce CF. Internationally accepted validation protocols are needed to standardize emissions reporting. WWTPs could also engage in carbon markets, using emission cuts from biogas recovery to earn credits and aid global decarbonization.

5. Conclusions

This study reveals major discrepancies in CF estimates for WWTPs across tools, driven by differences in EFs, GWP values, and methodologies. S1 emissions, especially from BNR and fugitive CH_4 and N_2O, dominate all tools, while S2 emissions from electricity are relatively stable. ECAM shows higher CF due to its BOD-based method, while the Validation Tool, using empirical EFs, reports lower values, highlighting the value of real-world data. The study confirms concerns over N_2O emission oversimplification and the limitations of default IPCC EFs, which may distort CF accuracy. Although GWP updates from 2007–2021 affect results, EF choice has greater influence. Variability in biogas emissions due to differing methods stresses the need for standardization. Overall, empirical EFs notably improve CF accuracy, especially for BNR emissions. As this study is based on a single full-scale WWTP and does not include Scope 3 emissions - due to limited data availability - the findings should be interpreted within this specific context. Nevertheless, the methodology and results offer a useful reference point for future studies across diverse wastewater treatment settings.

Acknowledgements

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Table 1. Scope 1 process emissions calculation completeness – comparison of the CF tools selected for the in-depth analysis

Tool name	Nitrogen-based calculations N_2O emission		Organics-based calculation CH_4 emission		Biogas-based calculation CH_4 emission		
	BNR	**Recipient**	**BNR**	**Recipient**	**Combustion**	**Leakage**	**Upgrading Slip**
CFCT v2014	$N_{denetrified}$ (removed)	TN effluent load	COD influent load	Not calculated	Calculated as Scope 1	Possibility of calculation as Scope 1 if needed	Calculated as default Scope 1
CFCT v2024	$N_{denetrified}$ (removed)	TN effluent load	COD influent load	BOD effluent load	Considered as biogenic	Possibility of calculation as Scope 1 if needed	Calculated as default Scope 1
ECAM	TN influent load	TN effluent load	BOD influent load	BOD effluent load	Considered as biogenic	Calculated as default Scope 1	Possibility of calculation as Scope 1 if needed

TN – total nitrogen, COD – chemical oxygen demand, BOD – biochemical oxygen demand

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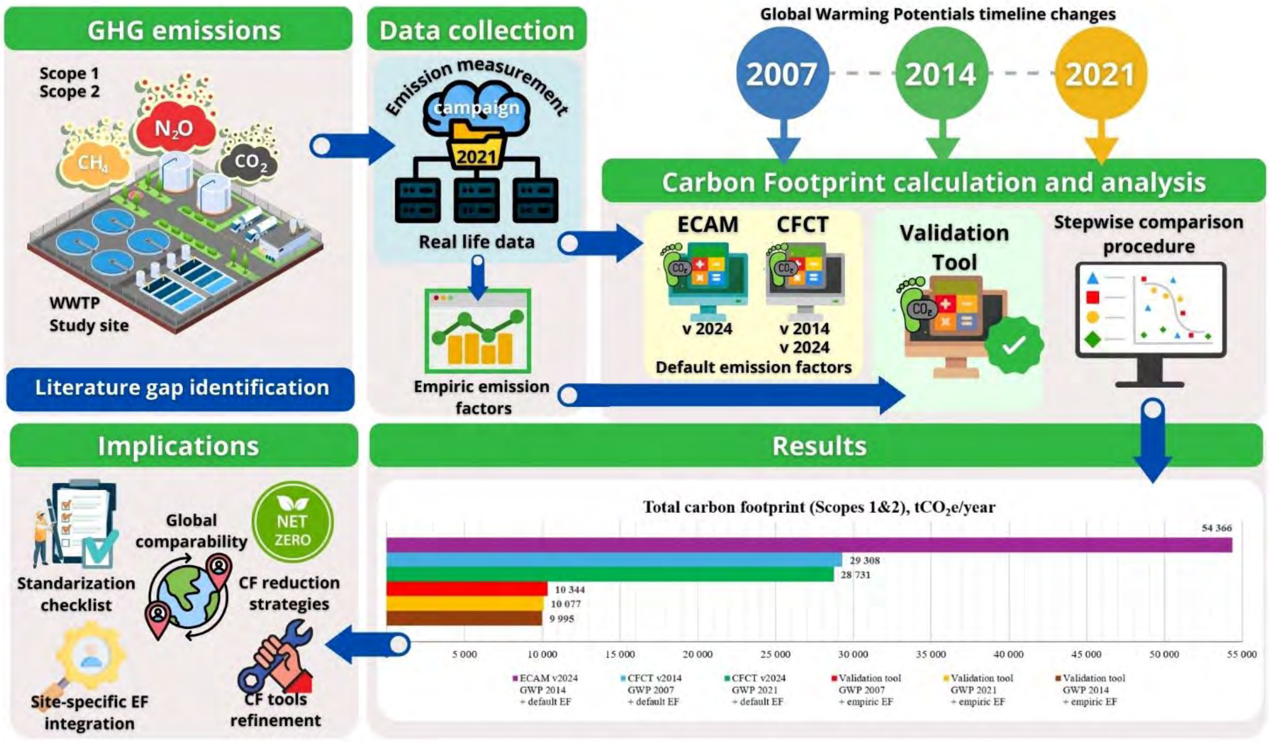
Table 2. Summary of the WWTP's annual average characteristics (2021).

	Unit	Influent	Effluent
Flowrate	m^3/d	100,000	
COD	gO_2/m^3	1,240	49.9
BOD_5	gO_2/m^3	550	3.7
TN	gN/m^3	100	9.4
		Biogas CHP plant	**External grid**
Electricity used	kWh/a	15,200,000	5,750,000
Biogas produced/consumed	Nm^3/a	7,760,000	-
Natural gas consumption	m^3/a		130,000



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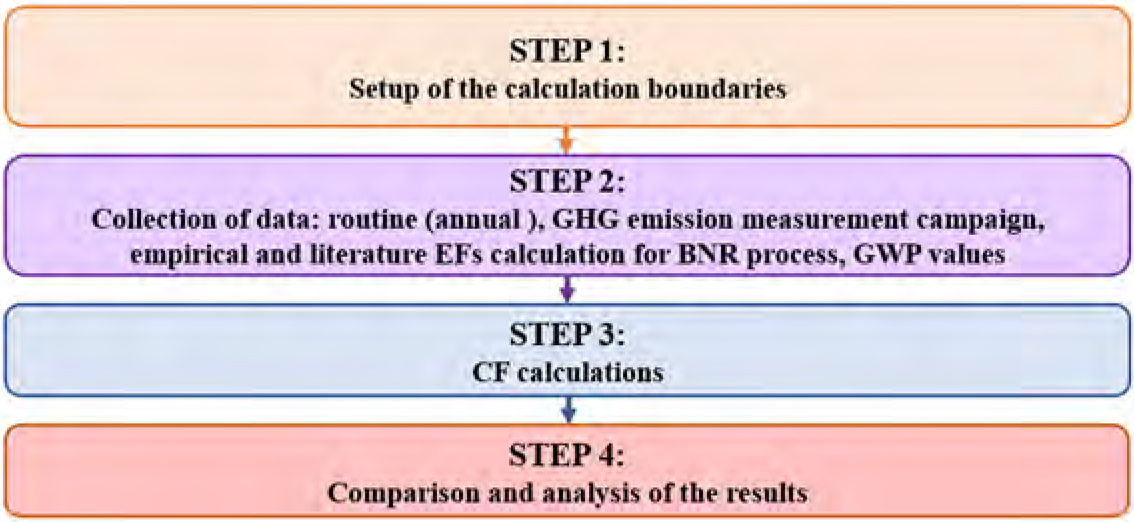

Figure 1. Study framework.

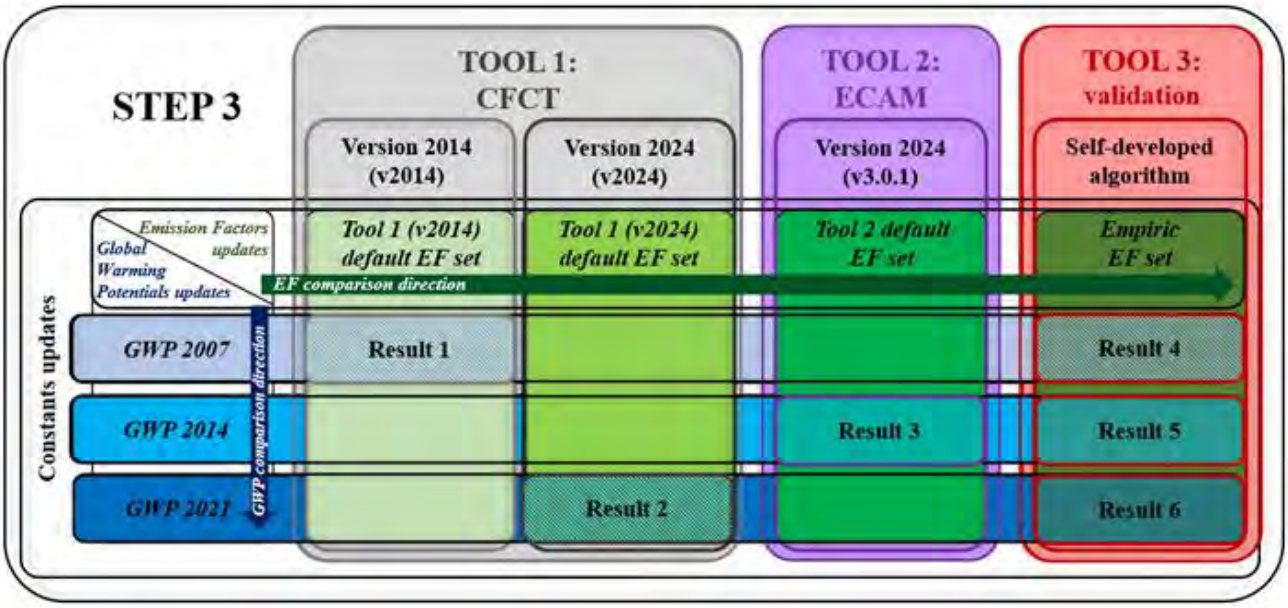

Figure 2. The concept of Step 3.

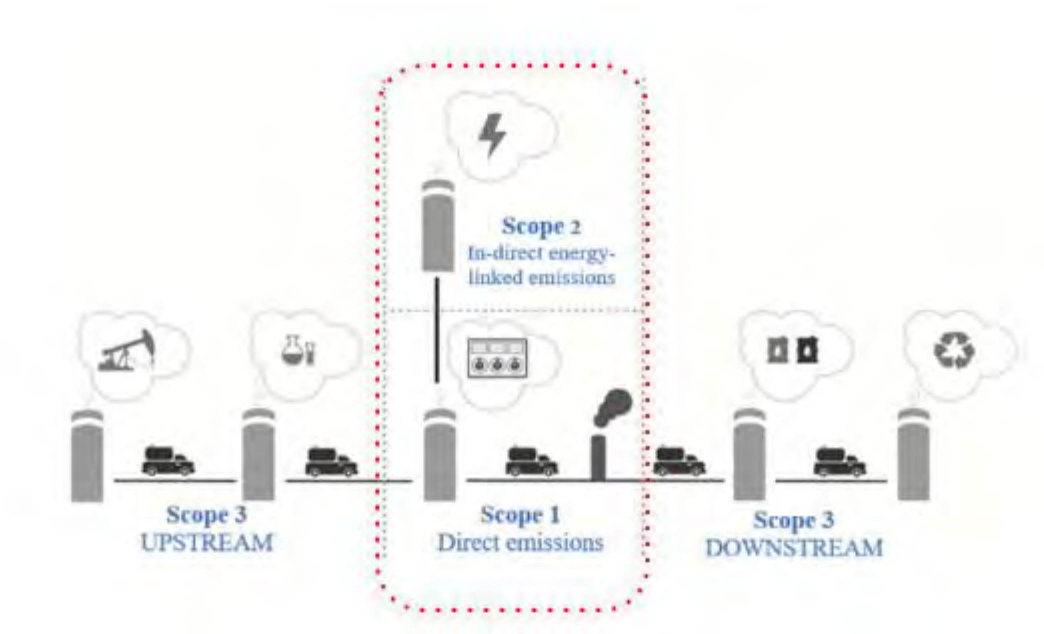

Figure 3. Overview of CF structure: scopes of GHG emission (based on [39]). Red dotted line presents calculation boundaries of this study.

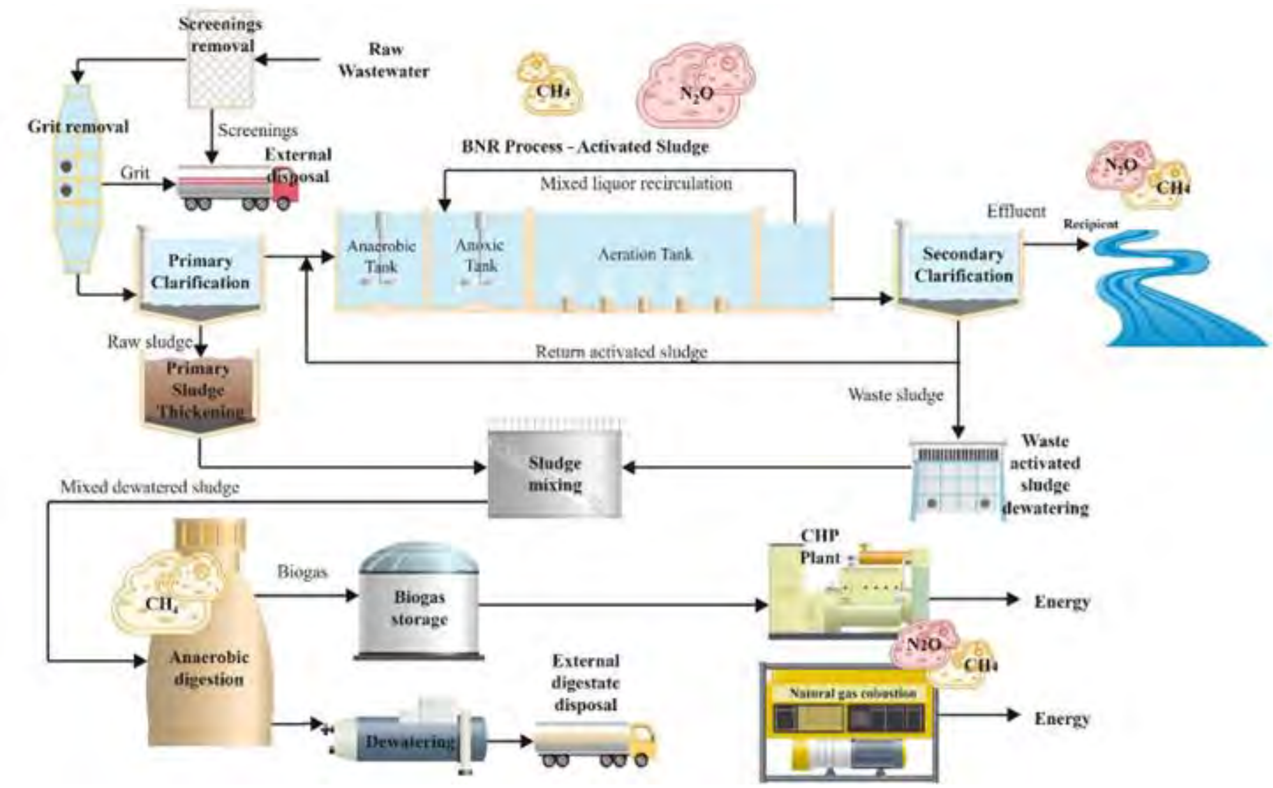

Figure 4. The idea of wastewater and sludge treatment processes employed in the investigated facility with GHG emission hotspots included in assessment.

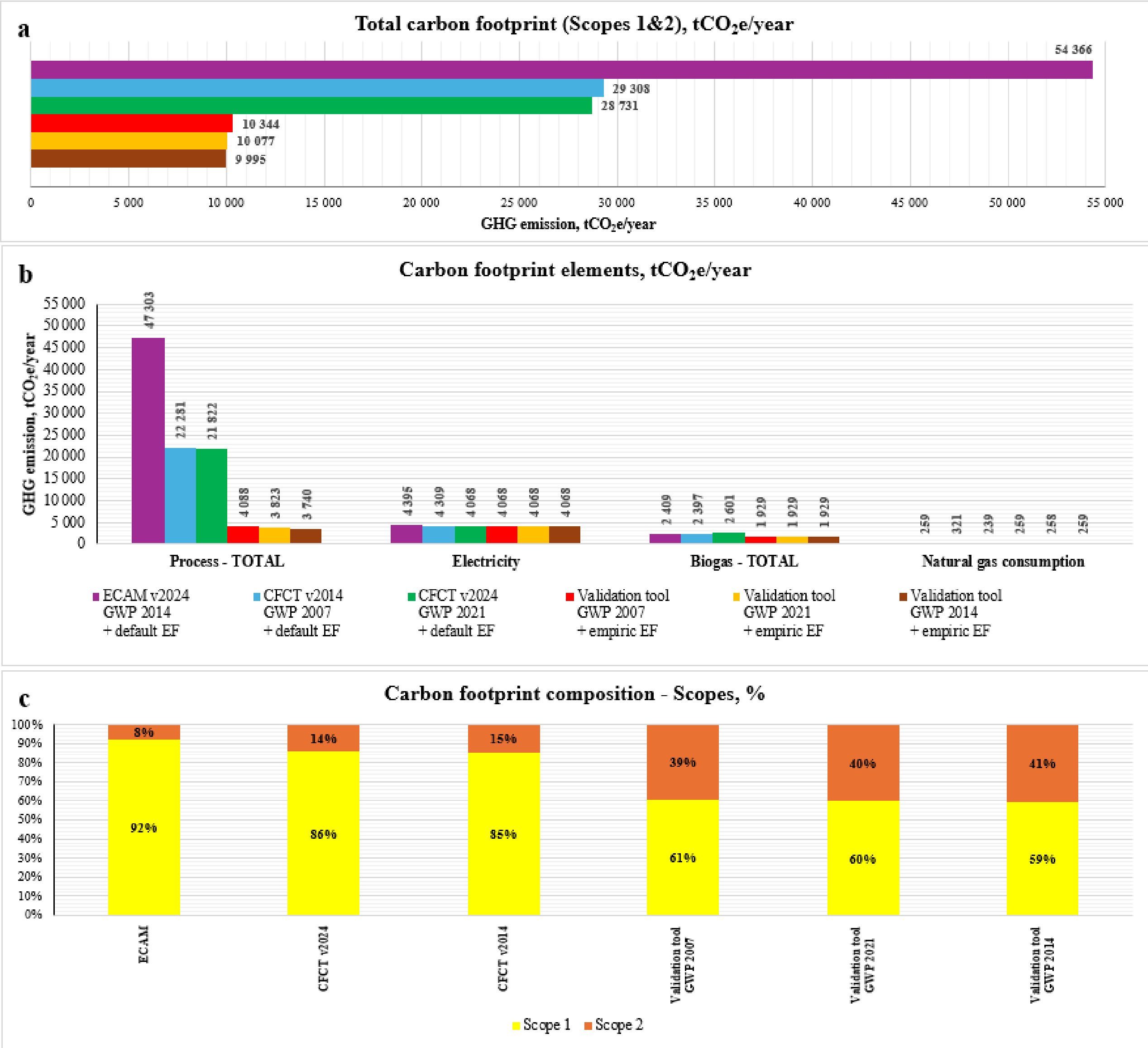

Figure 5. CF calculation results: a) Total CF (Scopes 1&2), tCO2e/year; b) CF elements, tCO2e/year c) Total CF composition - Scopes,%

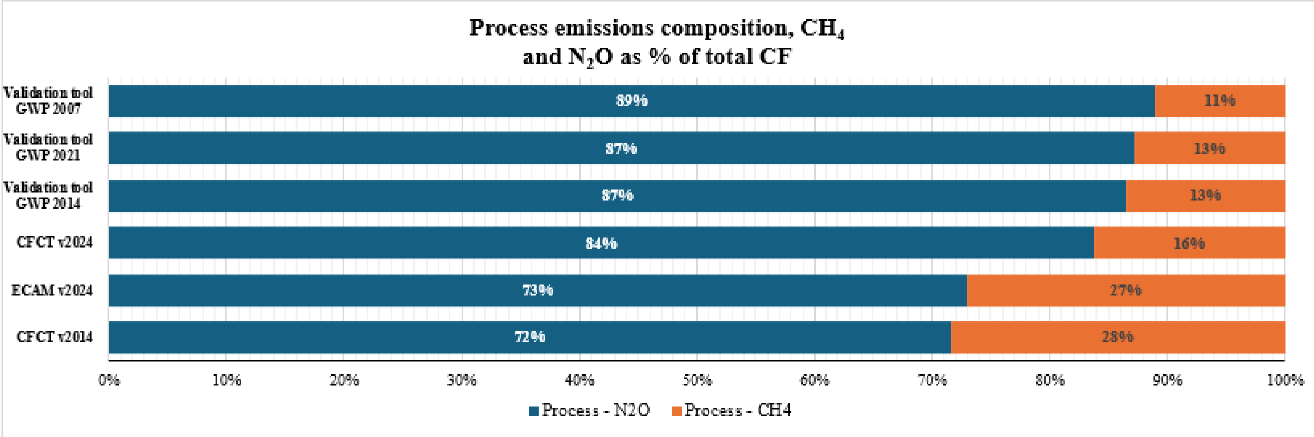

Figure 6. Process emission composition, CH_4 and N_2O as % of total (Scope1&2) CF

Figure 7. Heatmap of deviations from mean emissions by category - process, tCO_2e/a (2021).

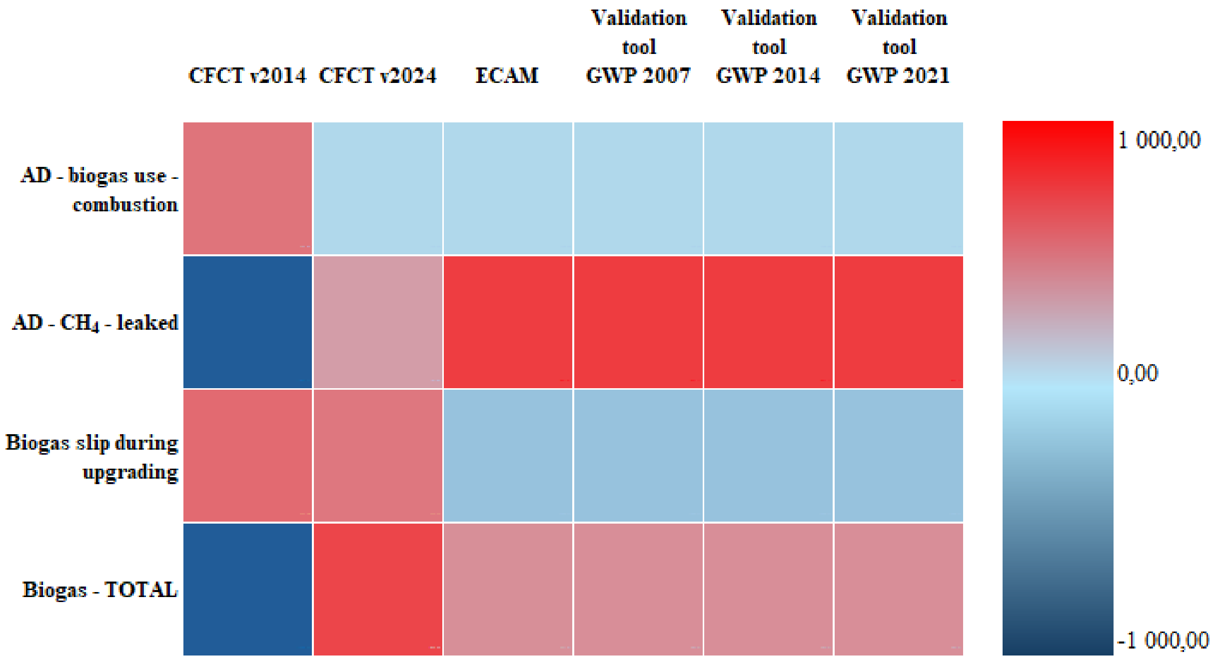

Figure 8. Heatmap of deviations from mean emissions by category - biogas, tCO_2e/a (2021).

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Net-zero carbon condition in wastewater treatment plants: A systematic review of mitigation strategies and challenges

Mojtaba Maktabifard [a,b,c,*], Hussein E. Al-Hazmi [a], Paulina Szulc [b], Mohammad Mousavizadegan [a], Xianbao Xu [d], Ewa Zaborowska [a], Xiang Li [d], Jacek Mąkinia [a]

[a] Faculty of Civil and Environmental Engineering, Gdańsk University of Technology, Narutowicza Street 11/12, 80-233, Gdańsk, Poland
[b] Faculty of Environmental and Energy Engineering, Poznań University of Technology, Berdychowo 4, 60-965, Poznań, Poland
[c] R2M Solution SAS, 3275 D2085, 06330, Roquefort-les-Pins, France
[d] College of Environmental Science and Engineering, Donghua University, 2999 North Renmin Road, Shanghai, 201620, China

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ABSTRACT

The wastewater sector accounts for up to 7 and 10% of anthropogenic CH_4 and N_2O emissions, respectively. Nowadays wastewater treatment plants are going through a paradigm shift to approach a net-zero carbon condition. Numerous ongoing measures have taken place to identify the sources of greenhouse gases and minimize the carbon footprint. This paper systematically reviews all known practices leading towards net-zero carbon wastewater treatment. The greenhouse gas emissions from the wastewater sector are identified and carbon footprint quantification tools, such as reliable models and emission factors are compared. The direct process emissions can contribute to over 60% of the carbon footprint in wastewater treatment plants, while around 30% of the carbon footprint is due to energy-related indirect emissions. Therefore, greenhouse gas mitigation via process optimization and energy usage in wastewater treatment plants are comprehensively described. The implantation of novel nitrogen removal processes can reduce both greenhouse gas emissions and energy consumption. Other techniques such as source separation systems can potentially allow mitigation of N_2O emissions by 60% while avoiding energy-intensive nitrogen fertilizer production. Nutrient recovery methods are another approach which offer negative value for the net carbon footprint. Recovering N_2O for energy production is a promising method which can lead to both direct and indirect carbon footprint reductions. Ultimately, to achieve full decarbonization any remaining emissions need to be offset, including carbon footprint of chemicals usage and transportation.

1. Introduction

The biological wastewater treatment process has historically been focused on efficient removing organic pollutants and nutrients to reduce the impact of wastewater on the aquatic environment. However, the wastewater sector is now undergoing a paradigm shift towards an integrated operation which is focused on resource recovery and net-zero carbon condition [1]. It is estimated that wastewater treatment plants (WWTPs) are responsible for nearly 5% of the global non-CO_2 greenhouse gas (GHG) emissions and this is projected to increase by 22% by 2030 [2,3].

GHG emissions from WWTPs are classified as either direct or indirect. The direct, non-biogenic GHG emissions, termed also scope 1 emissions, take place during wastewater and sludge treatment processes [4]. The direct CO_2 emissions are considered carbon neutral due to their biogenic nature, i.e., produced by the biodegradation of organic compounds in wastewater. This approach is in contrast with other sectors, such as transportation and energy, where the fossil-based CO_2 is the major contributor to GHG emissions. On the other hand, non-CO_2 direct emissions, including methane (CH_4) and nitrous oxide (N_2O), are of a significant concern due to a high global warming potential (GWP) of those gases. It is widely acknowledged that the wastewater treatment sector significantly contributes to CH_4 and N_2O emissions, estimated at 7–10% for each gas [5,6]. However, these percentages can vary depending on site-specific factors, such as treatment technologies, plant size, and regional differences in wastewater management practices. Based on the latest Intergovernmental Panel on Climate Change (IPCC) report [7], the GWP in a 100 years horizon is 27.2, 29.8 and 273 for non-fossil origin CH_4, fossil-origin CH_4 and N_2O, respectively.

* Corresponding author. Faculty of Civil and Environmental Engineering, Gdańsk University of Technology, Narutowicza Street 11/12, 80-233, Gdańsk, Poland.
E-mail address: mojmakta@pg.edu.pl (M. Maktabifard).

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Nomenclature and abbreviations

Item	Description
AD	Anaerobic Digestion
ADM	Anaerobic Digestion Model
ANAMMOX	Anaerobic Ammonia Oxidation
AnAOB	Anaerobic Ammonia Oxidizing Bacteria
AOB	Ammonia Oxidizing Bacteria
ASM	Activated Sludge Model
ASMN	Activated Sludge Model for Nitrogen
BEAM	Biosolids Emissions Assessment Model
BNR	Biological Nutrient Removal
BOD	Biochemical Oxygen Demand
BW	Black Water
CANDO	Coupled Aerobic-Anoxic Nitrous Decomposition Operation
CE	Circular Economy
CF	Carbon Footprint
CH_4	Methane
CHP	Combined Heat and Power
CO_{2e}	Carbon Dioxide Equivalent
COD	Chemical Oxygen Demand
CWA	Clean Water Act
DAMO	Denitrifying Anaerobic Methane Oxidation
DAMO/A	Denitrifying Anaerobic Methane Oxidation-Anammox
DEFRA	Department for Environment, Food and Rural Affairs
DNRA	Dissimilatory Nitrate Reduction to Ammonium
DO	Dissolved Oxygen
DW	Dry Waste
ECAM	Energy Performance and Carbon Emissions Assessment and Monitoring
EF	Emission Factor
EPA	Environmental Protection Agency
EU	European Union
FOG	Fat-Oil-Grease
FTIR	Fourier Transformed Infrared Spectroscopy
FU	Functional Unit
FW	Food Waste
GC	Gas Chromatography
GEA	German Environmental Agency
GHG	Greenhouse Gas
GW	Grey Water
GWP	Global Warming Potential
HRT	Hydraulic Retention Time
IEA	International Energy Agency
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
LCA	Life Cycle Assessment
ML	Machine Learning
MLR	Mixed Liquor Recirculation
N	Nitrogen
N_2	Dinitrogen
N_2O	Nitrous Oxide
N_2OR	Nitrous Oxide Reductase
NH_2OH	Hydroxylamine
NH_4^+	Ammonium
NO	Nitric Oxide
NO_2^-	Nitrite
OFMSW	Organic Fraction of Municipal Solid Waste
P	Phosphorus
PANDA	Partial Nitrification, Denitratation and Anaerobic Ammonia Oxidation
PD	Partial Denitrification
PD/A	Partial Denitrification-Anammox
PE	Population Equivalent
PN/A	Partial Nitritation-Anammox
SBR	Sequencing Batch Reactor
SCENA	Shortcut Enhanced Nutrient Abatement
SCND	Shortcut Nitrification-Denitrification
SHARON	Single Reactor System for High Activity Ammonium Removal over Nitrite
SNAD	Simultaneous Partial Nitritation-Anammox and Denitrification
SRT	Solid Retention Time
TCD	Thermal Conductivity Detector
TKN	Total Kjeldahl Nitrogen
TN	Total Nitrogen
TSS	Total Suspended Solids
UK	United Kingdom
US	United States
VS	Volatile Solid
WEF	Water Environmental Federation
WFD	Water Framework Directive
WRRF	Water Resource Recovery Facility
WWTP	Wastewater Treatment Plant

The indirect GHG emissions, referred to as scope 2 emissions, are due to energy and electricity consumption by WWTPs. The remaining indirect emissions, referred to as scope 3 emissions, are associated with transportation and production of chemicals outside of the plant [2]. The total sum of GHG emissions from all the scopes in known as the carbon footprint (CF) and reported based on the carbon dioxide equivalent (CO_{2e}) unit [8].

In comparison with the transportation and energy sectors, the wastewater sector has fewer studies on net-zero carbon pathways [1]. However, many countries are beginning to move towards low-carbon operation and undertaking actions to decarbonize municipal water management [9]. Net-zero targets are included in near-term policies and plans, and the discussion on the net-zero condition has been evolving recently in line with the targets of the Paris Agreement [10].

Metcalf et al. [11] outlined criteria for a net-zero transition in wastewater treatment, including the overall energy balance, process-related GHG emissions, chemical consumption and sludge disposal. Therefore, the net-zero pathways for wastewater sector is more diverse and complex compared to other sectors. A multi-criteria approach that considers both wastewater reclamation and other sustainability goals need to be defined specifically for the wastewater sector [1]. The net-zero carbon condition refers to a prospective WWTP which aims to offset certain CF that is sum of scope 1,2 and 3 emissions in wastewater treatment and thus attain net-zero GHG emission. The ne-zero carbon WWTP employs novel strategies to mitigate N2O and CH4 emissions via process modification. Furthermore, this plant needs to mitigate CF via energy optimization as well as chemicals and transportation optimization. The number of scientific papers on the topic has increased rapidly in the last decade (Fig. 1), confirming growing attention from the scientific community.

This review aims to provide comprehensive insights into effectively reducing CF and achieving a net-zero carbon condition in municipal WWTPs. The paper covers crucial aspects, including relevant regulations, GHG sources, and quantification methods and tools for CF assessment. Process optimization, energy usage, and reduction and offsets of indirect emissions are discussed to minimize CF. Practices for net-zero carbon wastewater treatment and a multi-criteria approach to decarbonize WWTPs are thoroughly reviewed with an on environmental protection, resource recovery, and reducing impact. GHG emissions in WWTPs, particularly N_2O and CH_4, and their sources are explored.

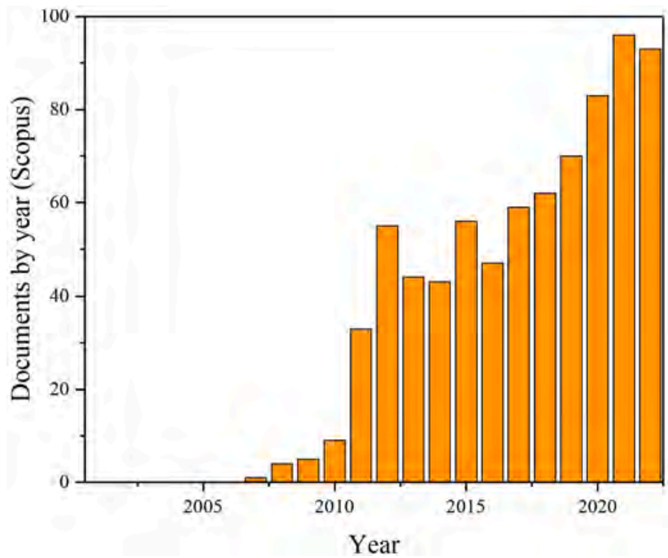

Fig. 1. Number of publications covering CF of WWTPs documented in the Scopus database since 2007 (analysis based on the keywords: CF and wastewater).

Methods for estimating the emissions and effective mitigation strategies are also discussed. Energy optimization in WWTPs is addressed, emphasizing reduced consumption and increased production through co-digestion and biogas utilization. The review explores the optimization of chemical dosages and transportation to minimize emissions. Additionally, the study investigates the potential of recovering N_2O from wastewater as an energy source. The findings of study carry significant implications, supporting climate change mitigation, environmental protection, and sustainable wastewater treatment with a reduced CF. The review underscores the need for comprehensive strategies to achieve net-zero carbon conditions in municipal WWTPs.

2. Evolving wastewater regulations and recommendations towards a net-zero carbon condition

Fig. 2 presents a summary of historical regulations and recommendations in the wastewater sector and outlines the recent trends that would affect the sustainability of wastewater treatment. For a long time, the regulations focused on protecting the aquatic environment, while only recently the efficient use/recovery of resources and reducing the impact on the entire environment have been postulated.

Fig. 2. Summary of significant wastewater management regulations/ recommendations.

2.1. Protection of the aquatic environment

Based on the Public Health Act established in United Kingdom (UK) in 1875, the authorities were obliged to provide clean water, dispose and reuse all sewage. A century later (1972), the Environmental Protection Agency of United States (US EPA) published its definition of the minimum standards for secondary treatment, including major parameters, such as total suspended solids (TSS) and biochemical oxygen demand (BOD). A few of US EPA regulations brought changes in planning and designing WWTPs, including the Clean Water Act (CWA) (1972) that defined the minimum standards for discharging wastewater. Further bio-solids regulations were set in 1993 for pathogen and heavy metal content as well as safe handling and use of bio-solids [11].

Meanwhile, across the European Union (EU), the first directive concerning municipal wastewater treatment (91/271/EEC) was adopted in 1991 to protect the aquatic environment from unfavorable effects of municipal wastewater discharge. In 1998, another directive (98/15/EC) clarified the requirements in relation to the discharges from municipal WWTPs to areas which are potentially subject to eutrophication [12].

In the beginning of 21^{st} century, the EU Water Framework Directive (WFD) was adopted, establishing a framework for water policy (2000/60/EC). The WFD established water quality standards and discharge controls to mitigate the impact of human activities on surface water quality [13]. In 2015, the United Nations set a global target recommendation through its sustainable development goals to reduce untreated wastewater by half by the year 2030 [14].

2.2. Efficient reuse and recovery of resources

In 2011, Water Environmental Federation (WEF) stated "*WWTPs are not waste disposal facilities, but rather water resource recovery facilities that produce clean water, recover nutrients (such as phosphorus (P) and nitrogen (N)), and have the potential to reduce the nation's dependence upon fossil fuel through the production and use of renewable energy*" [15]. *In 2012, WEF officially started using the novel term of water resource recovery facility (WRRF), instead of WWTP* [16].

In 2017, according to the new German Environmental Agency regulation (GEA) [17], sewage sludge must be recycled to recover P in plants with population equivalent (PE) over 50,000. The regulation is aimed at closing the P cycle and replacing over 50% of the imported P.

2.3. Reducing impact on the entire environment

The landmark Paris Agreement [18] aimed at limiting the global warming by prohibiting the temperature rise <2 °C above the pre-industrial levels, while IPCC [7] encouraged to follow even more ambitious goal leveled at 1.5 °C. In order to achieve this milestone, countries are aiming to achieve net-zero carbon condition by 2050. In 2019, the European Green Deal was introduced, which re-evaluated the EU's commitment to the net-zero concept with the introduction of the 'Fit for 55' package. The package outlines the path towards carbon-neutral transition, including short-term targets for 2030, such as a minimum reduction of 55% in GHG emissions compared to the 1990 level [19].

According to the International Energy Agency (IEA), developing and deploying clean energy technologies is crucial for achieving net-zero carbon conditions [20]. In practice, achieving net-zero globally will require a two-pronged approach, i.e. removing carbon from the atmosphere and restricting human-produced emissions. Based on the IPCC [7] definition, the net-zero carbon entails balancing anthropogenic GHG emissions with their intentional removal from the atmosphere. Although the proposed net-zero pathways can be found in various sectors, no framework for the wastewater sector has been complied yet.

Municipal WWTPs can be a significant element of circular economy (CE) due to incorporation of resource recovery and energy production, while minimizing the environmental impact on the aquatic environment. Salminen et al. [21] presented a concept of a water-smart CE which would decrease energy and water loss and recover valuable resources and reuse wastewater. The authors suggested taxation of water abstraction and tax relief on the recycled materials and instruments targeting pollutants in wastewater.

To proactively respond to climate change and reduce GHG emissions, China committed in 2020 to peak carbon emissions by 2030 and achieve carbon neutrality by 2060, known as the “dual carbon” goal [22]. According to the European Environment Agency, the EU’s GHG emissions fell by 23.2% since 1990. The EU’s global contribution declined from 15% to 8% between 1990 and 2018. Despite this, the EU aims to cut carbon emissions by 55% by 2030 (compared to 1990 level) and achieve carbon neutrality by 2050 as per their new policy announcement [23]. Under the Paris Agreement, Australia also set its target to reduce GHG emissions in 2030 by 26–28% from 2005 level and achieve net-zero emission by 2050 [24]. Germany and the UK led 50% of the EU’s GHG emission reduction from 1990 to 2019. Romania, France, Italy, Poland, and the Czech Republic contributed 33% of the EU’s total reduction during that period. Over 29 years, the UK and Germany were responsible for nearly half of the overall net reduction in the EU-KP. In 2019, Germany, Spain, and Poland played a significant role in the EU-KP’s net GHG reduction, accounting for over half of the total absolute reduction [25]. In the same year, the Netherlands’ GHG emissions dropped by 3.2% compared to 2018, with total emissions approximately 18% lower than the 1990 levels [26].

3. Identification of sources of GHG emissions in wwtps

The largest contribution to the total CF in WWTPs comes from scope 1 CF, which is due to the direct fugitive emissions from wastewater treatment and anaerobic digestion (AD) processes. A few studies which are presented in Table 1 [4,27,28] reported that the direct emissions contributed to over 60% of the total CF.

The indirect energy-related emissions, which are caused by the electricity purchased from the grid, contributed on average to the total CF with a lower share of approximately 20%. Those emissions play a minor role especially in the countries, such as Finland and Austria, that have a high share of renewable energy sources [27,28]. In contrast, Hu et al. [29] analyzed the CF of WWTPs in China and found a lower share of 29% for the direct emissions, while 45% of the total CF was related to sludge handling. This high share resulted from a large number of studied WWTPs performing sludge incineration and landfilling. In the case of plants without on-site electricity production (via e.g., biogas production through AD and energy co-generation), a dominant percentage share can also be attributed to the indirect emissions [30]. The indirect emissions related to chemicals consumed at the plant and transportation showed a marginal effect on the CF of WWTPs [4].

Kadam et al. [31] presented a discussion on achieving carbon neutrality in municipal WWTPs based on feasible technical aspects. Their proposed plant incorporates various advanced primary treatment techniques, leading to efficiently organic recovery and reducing the oxygen demand for oxidation of organic compounds. Consequently, biogas production during biological processes significantly increases. The optimized approach also involves converting concentrated primary sludge to biomethane through anaerobic digestion. Additionally, H_2 gas derived from N upgrades plays a crucial role in enhancing biomethane quality by reducing CO_2 content in the biogas. These findings demonstrate the potential of municipal WWTPs to achieve high process efficiency and energy utilization. To achieve a net-zero carbon condition, a hybrid system integrating both GHG emission reduction solutions and energy recovery is suggested [32]. Specific solutions for the three scopes of emissions are discussed in Sections 5 to 7.

Wu et al. [33] performed a comprehensive CF analysis of different wastewater treatment configurations. The authors reported that scope 1 emissions accounted for 23–83% of the total CF in different configuration scenarios. Scope 1 emission could be the primary contributor to CF for configurations with anaerobic processes. This is due to the reduction in scope 2 and scope 3 emissions resulting from energy recovery and reduced sludge production. Recent studies by Yao et al. [34] reported N_2O emissions alone to exceed 50% of the total CF based on site-specific monitoring of BNR WWTPs employing such configurations as A/O, anaerobic-anoxic-oxic (A_2O), and SBR. The contribution of scope 2 emissions is also highly variable and can range from 14 to 68% of the total CF of WWTP (Table 1). Scope 2 emissions could be minimized by reducing the energy consumption, but most wastewater treatment configurations with relatively low GHG emissions involve energy recovery, which offsets the scope 2 emissions. Scope 3 emissions play a minor role in the overall GHG emissions in WWTPs, contributing only 1–13% to the overall CF.

3.1. WWTP emission hotspots

Fig. 3 shows the potential sources of different GHGs that are emitted directly and indirectly. The preliminary stage of wastewater treatment is known as a potential source of CH_4 emissions. Anaerobic conditions often prevail in sewer systems, and intense CH_4 fluxes have been

Table 1
CF distribution in WWTPs - overview.

Reference -*number of studied plants*	Contribution to the total CF (%)										
	Scope 1					Scope 2	Scope 3				
	Bioreactor	AD	Recipient	Sludge handling	Total		Grit and screening handing	Chemicals	Sludge handing (3rd parties)	Transport	Total
Parravicini et al. [27] – *2 WWTPs*	42.9	26.3	9.1	–	78.4	15.5	–	4.2	–	1.7	5.8
Hu et al. [29] – *344 WWTPs*	29		–	–	29	26	–	–	45	–	–
Maktabifard et al. [30] – *6 WWTPs*	58	3	1	10	72	26	1	1	–	–	2
Awaitey [28] – *4 WWTPs*	52	1.5	2	11	66.5	18	0.5	8	–	7	15.5
Min	29	3	0	0	29	15.5	0	0	0	0	0
Max	58	26.3	9.1	11	78.4	26	1	8	45	7	15.5
Average	45.5	10.3	4	10.5	61.5	21.4	0.8	4.4	15	2.9	17.1
Deviation	10.9	11.3	3.6	0.5	19.2	4.7	0.3	2.8	21.2	2.9	16.8

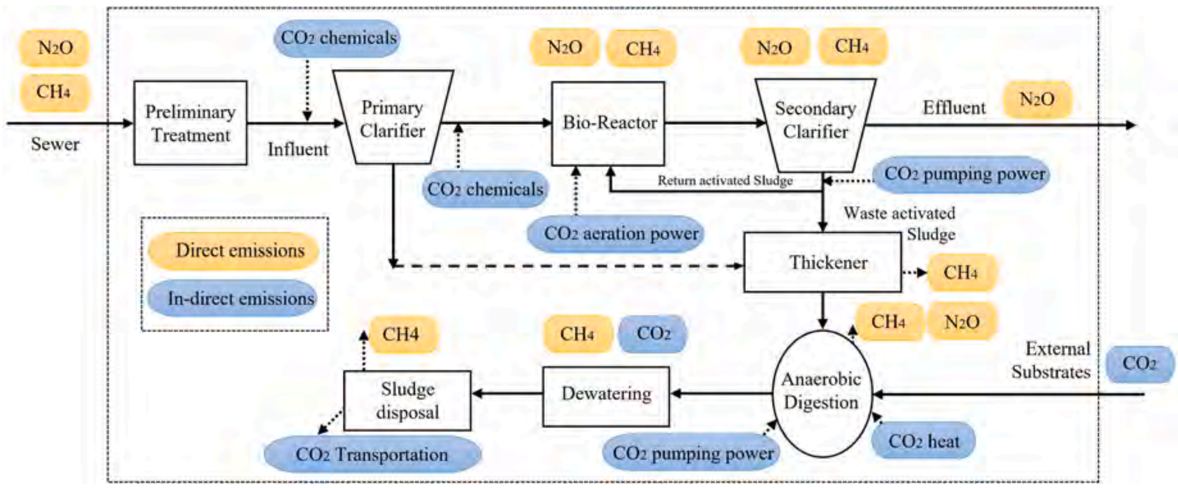

Fig. 3. Sources of the direct and indirect GHG emissions in WWTPs.

reported at the headworks of the WWTP, especially in aerated grit chambers. Other potential CH_4-forming spots are primary clarifiers, and then anaerobic compartments in the biological reactors. CH_4 can subsequently be stripped in the aerobic compartments of the reactors [35]. Due to the leakage of gases from internal facilities in the plant, biogas production installations may become significant CH_4 sources. Aerobically stabilized sludge, when stored for several months, can also contribute to CH_4 emissions due to the fact that anaerobic conditions start to prevail in the long-term perspective [36]. The effluent BOD load can contribute to further direct CH_4 emissions from the natural processes in recipients [27].

The GHG emissions from the preliminary treatment and sludge treatment stages do not attract as much attention as the emissions from the biological stage employing biological nutrient removal (BNR). Bioreactors have been recognized as the main source of GHG emissions in WWTPs. Campos et al. [37] reported that, the dominant N_2O production occurs in the bioreactors (90%) while for CH_4 emissions the sludge line with AD was responsible for over 70% and the remaining portion originates from bioreactors. N_2O produced under favorable conditions (either anoxic or aerobic) is emitted via either saturation-induced liquid-gas transfer or stripped to the air via aeration [38]. The effluent N load also becomes the source of direct N_2O emissions in the recipients.

3.2. N_2O emissions

WWTPs are considered significant anthropogenic sources of atmospheric N_2O, contributing 3–10% of the total emissions [5]. The latest IPCC report [7] shows a progressive increase in the atmospheric N_2O concentration, which has been levelled at the rate of 0.95 ppb/year for the last decade (2010–2019). The report emphasizes that wastewater treatment is the fourth largest sector responsible for N_2O emissions. The N_2O emissions from this sector increased from 0.2 Tg N/year to 0.35 Tg

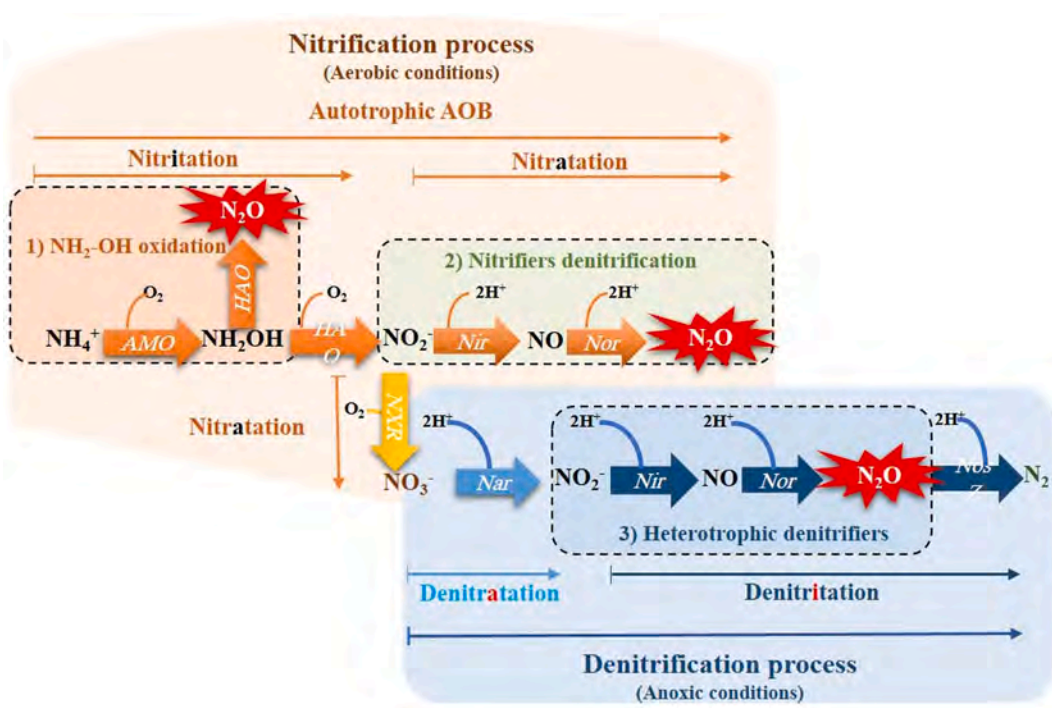

Fig. 4. Biological pathways of N_2O production and sink in the bioreactors (responsible genes: *AMO*: ammonia monooxygenase; *HAO*: hydroxylamine oxidoreductase; *NXR*: nitrite oxidoreductase; *Nar*: nitrite oxidoreductase; *Nir*: nitrite reductase; *Nor*: nitrite reductase; *NosZ*: Nitrous oxide reductase).

N/year between the 1980s and 2010s.

The N_2O emissions occur dynamically and normally remain beyond the control of the plant operators. Even small changes, such as dissolved oxygen (DO) variations, can significantly affect the liquid N_2O production by microorganisms [3].

3.2.1. Identification of the dominant pathways

N_2O production pathways involve three microbiological reactions, which require either aerobic or anoxic conditions (Fig. 4). In two pathways mediated by ammonia oxidizing bacteria (AOB), N_2O can be an intermediate product of hydroxylamine (NH_2OH) oxidation and the final product of autotrophic denitrification. Moreover, N_2O is an intermediate product of denitrification by heterotrophs. If that process is not disturbed, N_2O is further reduced to dinitrogen (N_2) in the final step of denitrification and this pathway can thus be a sink of N_2O [39]. Based on a literature review of N_2O emissions from WWTPs, Vasilaki et al. [39] formulated four main operational factors responsible for liquid N_2O production in the conventional nitrification-denitrification systems, including (i) low DO conditions, nitrite (NO_2^-) accumulation and changes in the ammonium (NH_4^+) concentration in aerobic compartments, (ii) low chemical oxygen demand (COD) to N ratio and NO_2^- accumulation in anoxic bioreactors, (iii) alternation of the anoxic to aerobic environments in intermittent compartments, (iv) sudden fluctuations (shocks) in the processes conditions.

Diverse contributing factors and dynamic reactions can occur simultaneously in the bioreactors, which are beyond operational control in full-scale WWTPs. Furthermore, heterotrophic denitrification can consume liquid N_2O produced via the different pathways. Regardless of the various production and consumption pathways of liquid N_2O, the stripping process should be considered as the process leading to N_2O emissions.

3.2.2. Additional sources of N_2O emissions in WWTPs

Although the direct N_2O process emissions released from bioreactors has the highest contribution to the total CF of WWTPs, a minor fraction of N_2O is also emitted from other units of the plant. Hwang et al. [40] performed N_2O measurements in different units of a WWTP, including the primary clarifier and secondary clarifier, as well as the sludge thickener and anaerobic digester. Small quantities of N_2O (0.012 g N_2O/kg TN) were produced in the digester, accounting for less than 1% of the total N_2O emissions. Regarding the primary and secondary clarifier, N_2O emission factors (EFs) of 0.22–0.26 g N_2O/kg TN were reported based on the full-scale measurements of Hwang et al. (2016). For comparison, Solís et al. [41] applied an EF of 0.01 kg N_2O–N/kg TN to predict the direct N_2O emissions from sludge storage, assuming uncovered storage throughout the year. Another study by Caniani et al. [42] found interesting results in the disinfection unit, observing high N_2O EF of 0.008 kg CO_{2e}/kg COD, due to the interaction between disinfecting agent and NH_2OH.

3.3. CH_4 emissions

The latest IPCC [7] report shows a progressive increase in the atmospheric concentration of CH_4, which has been levelled at the high rate of 9.3 ppb/year in the last decade. On a global scale, the wastewater sector accounts for approximately 5–7% of anthropogenic CH_4 emissions, ranking fifth after livestock (32%), oil and gas (25%), landfills (13%), and coal mining (11%) [43]. In the United States, CH_4 emissions from wastewater treatment increased from 10% to 14% between 1990 and 2019 [44]. A recent study by Song et al. [45] utilized the updated data from municipal WWTPs in US and estimated 10.9 ± 7.0 MMT CO_{2e}/year of annual CH_4 emissions which is about the double amount of emissions estimated by IPCC [36] (4.3–6.1 MMT CO_{2e}/year).

CH_4 is mainly produced during the anaerobic degradation of organic matter in bioreactors and digesters. The effectively utilized CH_4 can assist the plant to move towards energy neutrality and indirectly offset CO_2 emissions. On the other hand, CH_4 would increase the plant CF when emitted through incomplete combustion or leakage [46].

Studies at a regional level are usually based on CH_4 EFs [47]. Case studies of CH_4 emissions at the plant level are less frequent in the literature [48]. Zhao et al. [49] found that the CH_4 emissions from municipal WWTPs in China were over three times higher compared to the US plants. Although an empirical approach is highly uncertain, it is still a suitable way to estimate CH_4 emissions from WWTPs. Zhang et al. [6] concluded that it would be crucial to perform extensive measurements under various treatment processes, scales, and locations to obtain more accurate CH_4 emission predictions.

A full-scale study of Ribera-Guardia et al. [50] reported the long-term CH_4 emissions from aerobic compartments of a plug-flow bioreactor. The peak CH_4 emissions were found in the first aerobic zone, while decreasing towards the end of the bioreactor. The authors assumed that CH_4 was produced under anaerobic conditions in the initial stages of the plant and then stripped in the aerobic compartment of bioreactors. The highest liquid CH_4 concentrations were detected in the plant influent (0.55 mg CH_4/l) and reject water from the anaerobic digesters (0.52 mg CH_4/l).

Hwang et al. [30] determined the CH_4 EFs in different units of a municipal WWTP in South Korea. The sludge thickening process produced the largest CH_4 emissions (2.09 g CH_4/kg BOD) considerably more than aeration basin (0.72 g CH_4/kg BOD), primary clarifier (0.26 g CH_4/kg BOD) and secondary clarifier (0.67 g CH_4/kg BOD). CH_4 was primarily released from the digesters with a high CH_4 EF of 227 g CH_4/kg BOD. Wei et al. [51] analyzed four scenarios of sludge treatment and disposal. The highest contribution to CH_4 emissions was incineration (45.1%), followed by sanitary landfills (23%), land utilization (17.7%), and building materials (14.2%).

In a plant-wide model, implemented by Solís et al. [52], CH_4 emissions were calculated by a modified Anaerobic Digestion Model No. 1 (ADM1) [53]. Fugitive emissions were included as 2.7% of the produced biogas which was un-combusted in combined heat and power (CHP) units or slipped from anaerobic digester. The remaining amount of biogas was assumed to be completely combusted and CH_4 was fully converted to CO_2 while generating heat and electricity. It was assumed that CH_4 from the digester effluent was fully stripped to the atmosphere. Furthermore, the direct emissions due to sludge storage were estimated at 8.7 kg CH_4 per ton of volatile solids (VS). Overall, for the studied WWTP with the flow rate of 21,000 m^3/d, the CF was 19,000 kg CO_{2e}/d and CH_4 emissions contributed 5.8% to the total CF.

4. Estimation methods of GHG emissions

4.1. GHG emissions inventories

European inventory: The annual EU GHG inventory 1990–2019 report [54] contains the CH_4 and N_2O emissions from the wastewater sector. Both N_2O and CH_4 emissions account for 0.6% of the total EU GHG emissions in 2019 with 35.5 Mt CO_{2e} (domestic wastewater) and 9.3 Mt CO_{2e} (industrial wastewater). Between 1990 and 2019, the total emissions from wastewater treatment in EU decreased by 43.7%. Due to the application of new wastewater treatment processes, CH_4 and N_2O emissions from domestic wastewater decreased by approximately 50% and 17%, respectively [54].

US inventory: According to the US GHG Emissions and Sinks 1990–2020 Inventory Report [55], the wastewater treatment sector significantly contributed to the total N_2O country-wide emissions. It was the second largest anthropogenic source of N_2O emissions in 2020, levelled at 23.5 Mt CO_{2e}.

The wastewater sector contributed to 5.5% of N_2O fluxes, which resulted in 0.4% of the total GHG emission in the US. Comparing to 1990, a 41.8% increase was reported due to the country's growing population and a peaking protein consumption. The wastewater sector was ranked seventh of CH_4 emission sources with its 18.3 Mt CO_{2e}

fluxes. During the period of 1990–2002, CH_4 emissions remained stable, but a decreasing trend was observed later due to a reduction in the use of on-site septic systems [55].

Australian inventory: In the quarterly updated national inventory GHG (Australia's Greenhouse Accounts [56]), the levels of ozone layer depleting gases are given for different sectors, including wastewater handling. In 2010, wastewater treatment contributed to the total Australia's emissions in 0.51%, reflecting 0.8 Mt CO_{2e}. After a decade, the emissions increased to 0.9 Mt CO_{2e} constituting 0.72% of the total GHG emissions in the continent.

Chinese inventory: The most recent Chinese national GHG report from 2014 revealed that N_2O emissions from WWTPs were 1.1 Mt of N_2O, accounting for 5.6% of the national N_2O emissions. CH_4 emissions from wastewater treatment were 2.7 Mt of CH_4, accounting for 4.9% of China's total CH_4 emissions [6].

4.2. Direct measurements

There are various methods on *in-situ* measurements of GHG emissions that can be applied in WWTPs. In hermetic facilities, the direct GHG emissions can be analyzed continuously in the ventilation air by combining the concentration results with the airflow parameters [35]. In the case of open-to-air plants, collection of flux gas requires a dedicated sampling technique. In order to aggregate samples in open-surface tanks, different gas hoods and flux chambers are installed in multiple measurement points to address spatial and temporal variability along the wastewater train. In both cases, liquid N_2O concentrations should be simultaneously examined [57] and they need to be complemented by the analysis of conventional qualitative and quantitative operating parameters. The analytical techniques for measuring GHG concentration in collected flux-gas samples include Fourier Transformed Infrared Spectroscopy (FTIR) or Gas Chromatography with a Thermal Conductivity Detector (GC TCD).

Duan et al. [58] provided a critical review on using isotope technology for direct N_2O measurements, including both natural abundance and labelled isotope approaches. However, the authors emphasized that the accuracy and reliability of these techniques requires further improvement.

In order to reliably validate N_2O emission models, data from full-scale measurement campaigns are essential. However, the number of plants which measure N_2O emissions is currently very limited [59]. The long-term (yearly) N_2O monitoring data are available in the literature for Viikinmäki WWTP in Helsinki [60] and Kralingseveer WWTP [35].

4.3. CF calculation tools

General requirements that are dedicated to CF calculation can be found in International Organization for Standardization (ISO) 14,064 standard [61]. The IPCC (2006) with 2019 refinement for the national GHG inventories [36] is a frequently applied guidance note for the estimation of GHG emissions from WWTPs.

Life cycle assessment (LCA) is a broad technique to systematically evaluate multiple environmental impacts associated with municipal water cycle infrastructure, including wastewater collection and treatment [62]. The CF is part of LCA that quantifies a negative impact on the climate change. The LCA can be an accurate tool for calculating the indirect GHG emissions including high level of details in inventory data [63]. The direct GHG releases to air from the wastewater and sludge lines of WWTPs are incorporated via additive calculation procedures, often supported by secondary data estimations sourced from, either empirical or mechanistic models as well as in-situ measurement campaigns [63].

Several tools are available specifically for the CF assessment of WWTPs, usually as open-source files or applications. These are designed to deliver CF results based on the user's data input comprising operational parameters of the analyzed WWTP or its part. Table 2 shows an overview of CF calculation tools specifically dedicated to WWTPs. The Carbon Footprint Calculation Tool (CFCT) [64] has been utilized frequently in the literature [4,30] which is an adjustable spreadsheet developed originally for Swedish case studies [65] and has been updated since then.

Table 2
Overview of the CF assessment tools for WWTPs.

	Tool name (if available)	Country of origin	Year	Methodology	Open source
1	Quantifying the GHG emissions of WWTPs (as part of a thesis project)	Netherlands	2009	ISO	YES - source code for MatLab
2	Biosolids Emissions Assessment Model (BEAM)	Canada	2010–2011	IPCC, WRI	YES - Excel
3	Worldbank Organization - Sustainable Urban Energy and Emissions Planning Toolkits - Energy Balance and GHG Inventory Spreadsheet	USA	2011–2012	IPCC	YES - Excel
4	Waste Sector GHG Protocol Calculation Tool	France	2013	GHG Protocol	YES - Excel
5	Energy Performance and Carbon Emissions Assessment and Monitoring (ECAM)	Germany	2015	IPCC	YES - online
6	GESTABoues for wastewater treatment sludge management	France	2016	Own - based on literature	YES - after free registration
7	C-FOOT-CTRL	UK/Greece	2019	Own - based on literature	NO
8	Queensland Water GHG Calculator	Australia	2010–2019	National Greenhouse and Energy Reporting Standard	YES - excel
9	Calculation of the CF from Swedish WWTPs (SVU 12–120)	Sweden	2012–2021	Own - based on literature	YES - excel

When planning the CF calculation for a WWTP, two main approaches for setting boundary conditions of the CF analysis can be applied. The first method covers a full life cycle of a WWTP within the construction, operation and demolishment stages [66]. Another method would be an in-depth analysis of the operation stage with detailed considerations of process emissions [4].

Overlooking the construction and demolition phases of WWTPs and supporting buildings is a common simplification in LCA. The impacts of these phases are usually considered insignificant compared to the emissions from the operational stage [67]. The inclusion of new aspects in WWTPs, such as energy and resource recovery, would broaden the scope of carbon accounting. Therefore, Li et al. [46] proposed a system expansion approach, while assuming that the products recovered from wastewater, such as struvite, vivianite, biodiesel, bioplastics, biochar and protein, should substitute similar products on the market. This would result in offsetting carbon emissions from the system.

4.4. Empirical EFs

Simple empirical models are the most common approach to estimate N_2O emissions from wastewater and N_2O EFs are advantageous for providing a better insight into the WWTPs CF. It is important to note that the specific functional units (FUs) used to report N_2O EF might vary in the literature. The influent total Kjeldahl nitrogen (TKN) load was used most frequently, but the total removed N load or influent NH_4 load have occasionally been reported in the literature as well (Table 3).

The latest IPCC guidelines [7] recommended 1.6% kg N_2O/kg influent N-load as the EF based on the results of various monitoring campaigns. The reported N_2O EF for other processes, such as nitrite "shunt" or deammonification, are typically higher than full nitrification/denitrification. For example, Gustavsson and La Cour Jansen [68] estimated 6% kg N_2O/kg $NH_{4\ removed}$, EF for the nitritation process, while the EF for partial nitrification anammox process was 3% kg N_2O/kg $N_{removed}$ [69].

Continuous, long-term monitoring campaigns can substantially improve the N_2O EF accuracy [70]. A high inconsistency in the annual EFs (0.1–8% of the N load) was found after monitoring fourteen full-scale WWTPs in Switzerland, but the EFs were strongly correlated with the effluent NO_2^- concentrations. The authors proposed a national EF calculated from the weighted (weights based on the fraction of N loading in the country-scale) EFs of carbon removal (EF: 0.1–8% depending on the expected variability in plant performance), nitrification only (EF: 1.8%), and full N removal (EF: 0.9%). This approach allowed to estimate country-specific N_2O emissions from WWTPs. In Switzerland, the average EFs and total annual N_2O emissions ranged between 0.9 and 3.6% and 410–1690 ton N_2O (corresponding to 0.3–1.4% of the total GHGs), respectively [70].

The fixed EF approach neglects the impact of operational conditions and wastewater characteristics on N_2O emissions. Valkova et al. [3] suggested that the N_2O EF by IPCC overestimates N_2O emissions for the plants with the high efficiency of total nitrogen (TN) removal. The authors proposed that the annual direct N_2O emissions can be predicted based on the annual average TN removal efficiency. In ten Austrian WWTPs with high N removal efficiency (83–92%), the N_2O EF of 0.12% ± 0.1% kg N_2O/kg TKN was found. The N_2O EFs do not capture emission dynamics, potentially leading to overestimation or underestimation of emissions [71]. To improve the accuracy of N_2O emissions and CF assessment in process optimization and mitigation strategies for individual WWTPs, local measurements and mathematical models are preferred over fixed EFs.Table 3 shows significant variations in N_2O EFs depending on the estimation methodology, plant location, and study conditions. The scale of estimation, from global inventories through individual facilities, also impacts the results. Nayeb et al. [72] found high uncertainty in the CF estimation using national-scale inventories. Maktabifard et al. [30] demonstrated over 50% uncertainty in total CF estimation when considering a wide range of N_2O EFs from literature. High uncertainties were especially observed in the facilities dominated by the direct emissions.

Table 3
Measured and estimated N_2O EFs from WWTPs in different studies.

Reference	N_2O EF benchmark			Type of study
	kg N_2O/kg TKN	kg N_2O/kg $N_{removed}$	kg N_2O/kg NH_4 removed	
Solís et al. [41]	0.55%			Practical modelling
Li et al. [73]				Practical modelling
Maktabifard et al. [59]	0.94%			Practical modelling
Zaborowska et al. [74]	1.6%			Practical modelling
Domingo-Félez and Smets [75]			1.2–4.6%	Theoretical modelling
Massara et al. [76]	1–11%			Theoretical modelling
Pocquet et al. [77]			1%	Theoretical modelling
Ni et al. [78]	0.69–3.5%			Practical modelling
Foley et al. [79]		1.57%		LCA
Gruber et al. [80]	1–2.4%			Long term monitoring campaign
Sun et al. [81]	0.2–1.6%			Sampling measurements
Kosonen et al. [60]	1.9%			Long term monitoring campaign
Wang et al. [82]	0.095–3.44%			Online measurements
Hwang et al. [40]	1.606%			Sampling measurements
Duan et al. [83]	0.58%			Long term monitoring campaign
IPCC [36]	1.6%			Inventory report
EEA [84]	0.035%			Inventory report
NGER [85]	0.7–0.8%	1%		Inventory report
WERF [86]		0.01–2.5%		Inventory report
IPCC [87]		0.32–0.5%		Inventory report

4.5. Mathematical models

Mechanistic models are considered powerful tools for determining liquid N_2O production. These models can help identify the dominant N_2O production pathways and mitigation strategies. In comparison with empirical models, the use of mechanistic models may significantly reduce the uncertainty of the total CF results, as the CF results are highly sensitive to N_2O emissions [59].

Single-pathway (AOB denitrification/NH_2OH oxidation) and two-pathway models have been suggested for N_2O production by AOBs, while heterotrophic denitrification could be modelled as three or four-step process [88]. These models are added as extensions to existing activated sludge models (ASMs). For example, Kim et al. [89] combined the ASM for Nitrogen (ASMN) [90] and ADM1 [53] in the ASMN for GHGs (ASMN_G). Blomberg et al. [91] extended the ASM3 [92] with a dynamic N2O prediction and implemented the model in a full-scale WWTP in Finland. Zaborowska et al. [93] extended the ASM2d (ASM2d-N_2O) and included both N and P removal processes to model a full-scale WWTP in Poland. Research attention has primarily focused on modelling N2O emissions, but efforts are being made to develop models considering CO_2 and CH_4 emissions from wastewater (ASMs) and sludge treatment (ADMs). Mannina et al. [94] proposed a simple mass balance-based model with N_2O from nitrification and denitrification. A modified Benchmark Simulation Model No. 2 (BSM2) incorporated GHG emissions were considered (N_2O and CO_2 from AS, CO_2 and CH_4 from AD) along with the indirect emissions from the electricity and chemicals [95,96]. These models investigated the influence of different operational scenarios on the effluent quality, operational cost and GHG emissions. The DO control in the aerobic zone was a commonly studied strategy for GHG mitigation strategy. Other model-based studies considered the influent COD/N ratio [94], TSS control in the primary clarifier and CEPT [95,97], MLR ratio [96] and sidestream deammonification [74]. The advantage of the modelling approach was the possibility to take into account the processes interactions and identification of trade-offs between conflicting objectives [98]. In the approach proposed by Arnell et al. [97], the plant-wide model was combined with LCA to account for the global environmental impact due to external resource use.

The models of N_2O production/consumption should be integrated with a stripping model to predict N_2O emission. Baresel et al. [99] initially applied the saturation-induced liquid-gas transfer and the stripping N_2O emission model in the anoxic and aerobic compartments, respectively. These models were adopted in several subsequent studies [59,91,93].

The inclusion of all the known N_2O production pathways may result in over parameterized complex models. These models require more calibration efforts due to a higher number of coefficients. Moreover, calibration of the models by using short-term measurement data would reduce the accuracy and increase the uncertainty of the model [59].

Although mechanistic models have explicit advantages in determining N_2O mitigation strategies for WWTPs, their practical use in GHG inventories is still limited. Another emerging mathematical model is Machine Learning (ML), which can transform WWTP data into knowledge. By training ML models with outputs from ASMs, the prediction capabilities can be improved. These trained ML models are highly effective for online monitoring and providing decision support and are widely used in industry. Limitations of ML approaches may arise from data scarcity for learning and testing. To address this, Mehrani et al. [100] proposed a hybrid model combining a mechanistic model with ML to simulate N_2O production in a sequencing batch reactor. Szelag et al. [101] examined various ML models for N_2O emission prediction, using global sensitivity analysis to select a ML model as an alternative to the mechanistic model.

5. Mitigation of GHG emissions via process modification

From the CF perspective of WWTPs, the direct emissions from the treatment processes usually play the most significant role. This chapter presents N_2O mitigation strategies in order to reach a net-zero condition and explores the implementation of innovative removal processes, source separation, and nutrient recovery for decarbonizing WWTPs.

5.1. Approaches to N_2O mitigation strategy

N_2O mitigation strategies are mainly focused on [83]: (i) Optimizing aeration mode and DO set-point, (ii) Preventing DO gradients via mixing optimization, (iii) Avoiding NH_4^+ peaks, (iv) Preventing NO_2^- accumulation, (v) Ensuring complete denitrification by supplying sufficient carbon source or/and increasing hydrolysis in the primary clarifiers.

Duan et al. [83] evaluated N_2O production by adopting a multi-pathway mechanistic model. The employed multi-criteria strategy in an Australian plant, which was focused on reducing DO levels, resulted in 35% mitigation of N_2O emissions with the EF reduced from 0.89 to 0.58% of the influent TN load. This could mainly be associated with the declines in N_2O generated via the NH_2OH oxidation pathway. Based on lower DO levels simultaneous nitrification and denitrification was encouraged and thus nitrite accumulation was minimized, leading to less N_2O production via AOB denitrification pathway. Another study by Mampaey et al. [102] reported a decline of over 50% in N_2O emissions when one-stage granular SHARON (partial nitrification) reactor cycles were reduced by 1 h, which also falls in the mitigation category of optimizing aeration.

One of the most effective ways to recognize N_2O mitigation strategies is testing different operational modes [39]. Rodriguez-Caballero et al. [103] suggested that the optimum control strategy in sequencing batch reactors (SBRs) would be shortening aerobic-anoxic cycles (20 min of aeration with a short anoxic phase).

5.2. Novel processes for N removal

Nitrogen removal systems, which are based on anaerobic ammonia oxidation (anammox), can be categorized into partial nitritation-anammox (PN/A), simultaneous PN/A and denitrification (SNAD), partial denitrification-anammox (PD/A), and denitrifying anaerobic methane oxidation (DAMO)-anammox (DAMO/A) systems (Fig. 5). PN/A and SNAD can effectively treat ammonia-rich wastewater, while reducing energy consumption and sludge production compared to the traditional nitrification-denitrification processes. In contrast, PD/A and DAMO/A are used to treat NO_3^--rich wastewater, with a focus on reducing energy consumption (no oxygen demand and reducing C/N ratio to < 3) and achieving sustainable removal efficiencies [104]. Therefore, anammox-based systems are becoming a promising technology in WWTPs for energy neutrality [105].

There are three main pathways of N_2O production in anammox-based systems, including NH_2OH oxidation (($NH_4^+ \rightarrow NH_2OH \rightarrow N_2O$)) and autotrophic denitrification both mediated by AOB ($NO_2^- \rightarrow NO \rightarrow N_2O$ or $NH_2OH \rightarrow N_2O$ or $NH_2OH + NO \rightarrow N_2O$), as well as heterotrophic denitrification ($NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O$) mediated by diverse groups of denitrifiers. The N_2O consumption ($N_2O \rightarrow N_2$) is a unique N_2O degradation (nosZ-dominated) catalyzed by denitrifiers [106]. Other microorganisms present in anammox-based systems, such as anaerobic ammonia oxidizing bacteria (AnAOB), dissimilatory nitrate reduction to ammonia (DNRA) heterotrophs and DAMO autotrophs do not contribute to N_2O emission in anammox systems [107–109].

PN/A. The reported overall N_2O emissions in PN/A systems are highly variable (dependent on the scale) and range from 0.08% to 6.6% of N-load [5,110]. The mitigation of N_2O emission in PN/A systems can be achieved by adjusting various operation parameters, such as NO_2^-/NH_4^+ and C/N ratios, aeration strategies, DO set points, temperatures, and pH [111,112].

N_2O emission is dominated by NH_2OH oxidation pathway at low DO levels (<0.5 mg O_2/L), low inorganic carbon, high pH or low NO_2^- concentration, while AOB denitrification pathway dominates at higher inorganic carbon, lower pH or higher NO_2^- concentration [5]. Properly selecting intermittent aeration strategies can lead to frequent shifts in the predominating pathways of N_2O production and help mitigate N_2O emissions [113]. Beier et al. [114] aimed to predict the production of N_2O in a side-stream system using a novel modelling approach. During the periods when the system was aerated, the average EF was achieved 0.05 mg N_2O/mgNH_4^+. During the anoxic phases, the production factor was determined to be 33%. The overall EF of oxidized NH_4^+ was found to be 27%, with denitrification being the primary source of N_2O production.

A high NO_2^-/NH_4^+ ratio can result in an increased N_2O production [115], but a low NO_2^- and high NH_4^+ loading is expected to minimize N_2O production rates [116].

SNAD. N_2O is produced through the oxidation of NH_2OH by AOB and reduction of NO_2^- to NO, then to N_2O under aerobic conditions [117]. N_2O production is also linked to heterotrophic denitrification under anoxic conditions [118]. The N_2O production in SNAD systems was found to be slightly lower (by 4%) compared to PN/A systems due to lower NO_2^- accumulation [82]. This reduction may be due to the consumption of NO_2^- and/or NO during anammox metabolism [119]. Moreover, complete heterotrophic denitrification by the NosZ enzyme was identified as a promising strategy for mitigating N_2O emissions [120]. For comparison, N_2O emission was mitigated by 80% and 70% in a SNAD biofilter compared to traditional nitrification and nitrification-denitrification biofilters [117].

PD/A. In PD/A systems, diverse microbial communities compete for NO_2^-. The presence of AnAOB and heterotrophic denitrifying/Dissimilatory Nitrate Reduction to Ammonium (DNRA) in this system is crucial for N_2O mitigation. N_2O is not produced during anammox metabolism and should not be produced by heterotrophs catalyzing partial denitrification (PD) ($NO_3^- \rightarrow NO_2^-$) (Fig. 5) [121]. Low N_2O emissions were

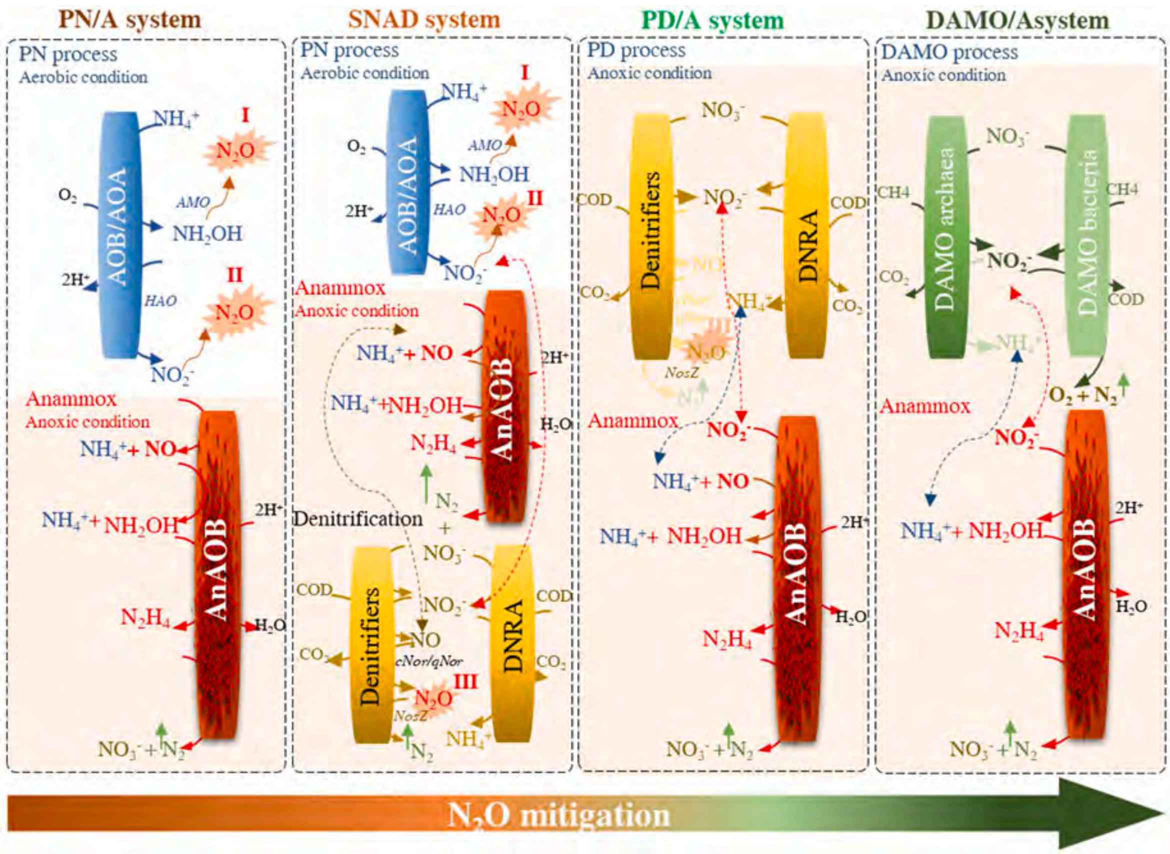

Fig. 5. Anammox based N removal systems.

observed in PD/A systems either with or without COD addition. Mitigation of N_2O was successful (EF = 0.002%) with low levels of COD addition and remained low during long-term PD/A performance [122]. However, adding COD resulted in approximately 50% reduction in N_2O emissions compared to the period without COD addition, indicating further reduction through the activity of NosZ enzyme in denitrifiers [122]. The N_2O conversion to N_2 gas was comparable to N_2O production from NO_2^- by heterotrophs under COD-limited conditions [123]. Thus, PD/A and DAMO/A have gained a significant attention for applications in WWTP as an alternative process to nitrification-denitrification and pursuing efficient N removal in a sustainable way. The most important challenge for a stable PD/A performance is a balance in NO_2^- by the activities of coexisting denitrifiers and AnAOB [123]. Moreover, low process temperatures and low influent C/N ratios could limit the startup of PD/A systems [121]. In contrast, heterotrophs can produce N_2O when PD is not successfully performed to only produce NO_2^- during unbalanced COD addition [124]. For example, 30% of total N_2O was observed to be produced in the AnAOB-dominated anoxic zone, potentially by heterotrophic denitrification or other unknown pathways [125]. Proper control of denitrifiers through adjusting COD addition (proper C/N ratio and COD type) appears to be crucial for N_2O mitigation in PD/A systems [126]. Comprehensive research should be taken for a deep understanding PD/A microbial metabolism and microbial growth rates, which are needed to improve the efficiencies of N and COD removal rates in WWTPs in the future.

DAMO/A. DAMO microorganisms can be classified as DAMO archaea and DAMO bacteria. DAMO-archaea can reduce NO_3^- to NO_2^- (substrate for AnAOB) by CH_4 oxidation under anoxic conditions, leading to N_2O and CH_4 mitigation. In contrast, DAMO-bacteria perform full denitrification ($NO_3^- \rightarrow N_2$) by utilizing CH_4 as an electron donor [127]. Combining DAMO microorganisms and AnAOB have many advantages: i) CH_4 could be decreased to 15% in the effluent wastewater [128]; ii) minimal amounts of N_2O were emitted [129]; iii) excellent nitrogen removal efficiency (99%) was achieved and cost savings were 49% [130]. The DAMO/A systems exhibit negligible N_2O emissions due to the absence of N_2O-generating enzymes in DAMO archaea and AnAOB, while DAMO bacteria effectively remove excess NO_2^- [104]. Thus, DAMO/A has significantly attracted attention for applying it in WWTP as an alternative process to nitrification-denitrification and realizing optimal N removal efficiency and environmental sustainability. However, according to Samedo et al. [131], Thauera linaloolentis strain 47 Lol^T contains NosZ, which converts N_2O to N_2. However, Mania et al. [132] found that when NO_3^- levels are high, the regulation of NosZ activity and N_2O reduction may not be possible. Consequently, elevated NO_3^- levels can significantly contribute to environmental impact and result in higher net N_2O emissions [131]. More research is still needed to understand DAMO/A microbial metabolism.

5.3. Source separation systems

In source separation systems, black water (BW), grey water (GW) and food waste (FW) are separated from municipal wastewater and solid waste as shown in Fig. 6. Separating the different kinds of wastewater at source can potentially increase the treatment efficiency, nutrient recovery and biogas yield. Kjerstadius et al. [133] compared the CF of conventional systems and a source separation system in Swedish WWTPs. The LCA showed that the source separation system could decrease the overall CF by 34 kg CO_{2e}/PE·year, primarily due to an increasing the biogas production.

Other studies, e.g. Remy [134], were in-line with the findings of Kjerstadius et al. [133] regarding the reduction of the net CF after source separation. In contrast, Thibodeau [135] reported a CF of 65 kg CO_{2e}/PE·year for a source separation scenario, which was higher by 12 kg CO_{2e}/PE·year in comparison with the conventional system. The wet

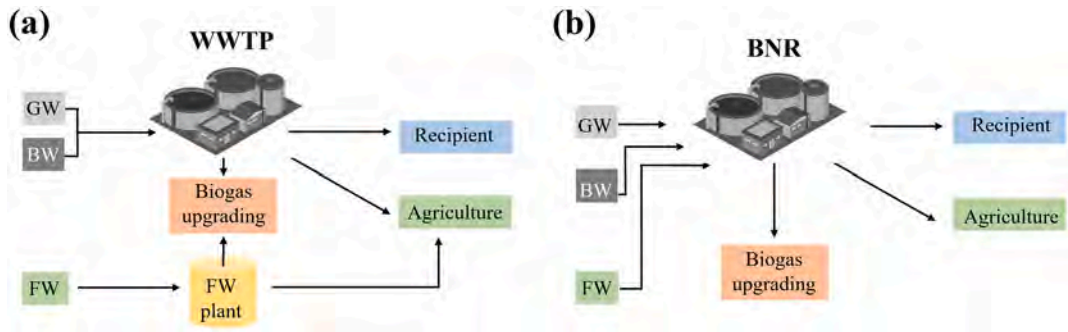

Fig. 6. (A) Conventional systems vs (B) source separation system of wastewater treatment (adopted from Kjerstadius et al. [133]).

fraction of BW was entirely returned to agriculture, leading to an increase in the CF resulting from transportation. However, the increased CF could be balanced out by selling bio-products as fertilizer.

Another benefit of source separation could be an increased recycle of nutrients to farmlands. The range of excess return of nutrients (N and P) was 1.7–4.3 kg N/PE·year and 0.06–0.54 kg P/PE·year, respectively [133–135]. The increased recovery of nutrients and utilizing them on farmlands could partially offset CF [4] and further decrease the indirect CF of WWTPs. Besson et al. [136] performed an LCA study and compared the environmental impacts of a central WWTP with source separation systems. Their results showed that for maximizing nutrient recovery and reducing GHG emissions, urine and BW separation are more favorable scenarios than a centralized WWTP. The separation allowed for mitigating N_2O emissions by 60% and avoiding energy-intensive N fertilizer production (Habor-Bosch process), which emits a large amount of GHGs (8.6 kg CO_{2e}/kg N) [136].

In another study, Tian et al. [137] utilized urine wastewater derived from a source separation system to pretreat WAS for AD. The authors observed a significant improvement in cumulative methane production, showing a 23% increase after pretreatment with urine wastewater at a volumetric proportion of 1:8. The presence of urea and hydrolyzed free ammonia in urine wastewater had a synergistic effect on sludge disintegration, leading to a reduction in sludge volume and increased availability of substrates for biological treatment processes. These findings highlight the potential of urine wastewater for pretreatment to enhance the efficiency of AD processes and improve biogas production from WAS. Badeti et al. [138] conducted a theoretical investigation of the impact of urine diversion on the biological treatment processes at a decentralized WWTP. Their simulations showed that 33% of aeration energy could be saved with 90% urine diversion. Furthermore, direct N_2O and CO_2 emissions in the treatment processes could be reduced by 98% and 25%, respectively, while indirect GHG emissions could be reduced by 20%. Source separating sanitation was considered in a new city district of Hiedanranta, Finland, Lehtoranta [139] considered source separating sanitation and applied LCA to compare it with a conventional sanitation system. The study revealed that with a separating system, 3 to 10 times more N could be recovered compared to the conventional system. Based on their environmental performance, source separating systems with improved nutrient recovery showed reduced climate and eutrophication impacts.

5.4. Systems with nutrient recovery

5.4.1. P recovery from wastewater

The P load in urban wastewater could theoretically replace up to 50% of the mineral P fertilizer used in agriculture. Moreover, P mining from the phosphate rock is associated with environmental concerns, such as air pollution and water contamination [140]. Amann et al. [141] used LCA to evaluate 18 P recovery technologies in terms of the energy demand and GWP. The analysis showed that a wide-range of GHG emissions can be estimated through the implementation of different P recovery strategies. The recovery from the liquid phase reduces the net GHG emissions. For the recovery from sludge, high GHG emissions were attributed to the processes which were already practiced in the full-scale. The recovery from sewage sludge ash showed the highest potential for an efficient P recycling. This method could annually decrease the cumulative energy demand and GWP by 0.096% and 0.1% per PE, respectively [141].

Another LCA study by Duan et al. [142] analyzed and compared the environmental impacts of different P removal and recovery methodologies. For chemical P removal and recovery pathways, the net specific CF was 44.5 kg CO_{2e}/kg $P_{removed}$ and −3.76 kg CO_{2e}/kg $P_{recovered}$, respectively. The negative value indicated the potential savings of CF due to P recovery. In another review, Zhao et al. [143] explored potential pathways for P recovery in WWTPs with a special focus on the A-2B-centered process, which involves an anaerobic fixed bed reactor to capture COD for improving energy efficiency. Simultaneously, this process allows for P recovery through further treatment with P crystallization. The A-2B-centered process exhibited the lowest specific energy consumption and GHG emissions when considering the amount of P recovered, indicating. Its promising approach in terms of both energy efficiency and environmental impact. The management of P resources in WWTPs encompasses multifaceted concerns related to environmental protection, energy efficiency, and GHG emissions. By exploring and implementing efficient recovery paths like the A-2B-centered process, WWTPs can take significant steps towards sustainable P management.

5.4.2. N recovery from wastewater

Recovery of N as a nutrient would decrease both direct environmental impacts and energy consumption. Nitrification-denitrification could be energy intensive and without any additional benefits aside from meeting the effluent standards. N recovery allows for the simultaneous treatment of wastewater, while consuming less energy and recovering resources. Beckinghausen et al. [144] performed a comprehensive review of approximately 50 N recovery techniques. The authors focused on techniques that could produce fertilizers. The highlighted technique was permeable membrane, which consumed smaller amount of energy (1–1.2 kWh/kg N). Moreover, it showed the potential to recover NH_4^+ from wastewater and the market value of the recovered product (ammonium sulphate). Other techniques, such as vacuum membrane distillation, can recover large amounts of energy in the form of heat (27 kWh/kg N), but the process is highly energy demanding (60 kWh/kg N). These techniques require further investigation to evaluate the costs of energy, as the amount of recovered heat highly effects the overall energy balance [145]. Innovative technologies, such as membranes, sorbents, electrodialysis, bioelectrochemical process, and even microalgae, are used to enhance nutrient recovery from wastewater, moving beyonnd conventional crystallization methods. Moreover, at the full-scale scale, some physicochemical methods, such as air stripping and struvite formation, are employed to recover P and N from sidestreams [146].

6. Mitigation of GHG emissions via energy usage optimization

Wastewater treatment has a high potential to decarbonize and move towards net-zero carbon condition, since water reclamation from wastewater is considered to be the most energy-intensive water infrastructure [1].

6.1. CF due to energy consumption

Energy efficiency has received growing attention as the number of WWTPs is growing globally and the effluent quality criteria become more strict [147]. The water industry is estimated to account for 2.1% in the US and 0.8% in the EU of the total electricity consumption [148]. Around 23 MMT CO_{2e} was generated during electricity consumption, which accounts for 37.7% of the entire CO_{2e} release in the water industry [149]. Aeration, pumping, and heating are the main energy consumers in WWTPs [149]. In a typical WWTP, aeration can account for up to 60% of all electricity usage, followed by sludge treatment and return activated sludge (RAS) recirculation from secondary clarifiers [150].

Decarbonization of the energy sector can lead to a reduction in the indirect emissions in WWTPs. Coal-fired electricity production is being replaced by natural gas and more countries are substituting fossil fuels with renewable resources, such as wind, solar, hydropower, and nuclear power [149]. Hydropower and nuclear power can be seen as the cleanest energy source with a maximum carbon intensity of 40 g CO_{2e}/kWh and 110 g CO_{2e}/kWh respectively, while coal is the most carbon-intensive energy source (Table 4) with the maximum carbon intensity of 1689 g CO_{2e}/kWh [151]. Currently, about 88% of the grid energy is satisfied by fossil fuels. Over the next 20 years, it is planned to increase the usage of renewable energy by 25%, which would decrease the CF related to the energy mix by up to 42%.

Delre et al. [132] evaluated the effect of electricity usage on the CF of seven different WWTPs located in Denmark and Sweden. In the Danish WWTPs, where the energy mix mostly consists of fossil fuels, energy consumption accounted for up to 29% of the CF. In the Swedish WWTPs, where nuclear and hydropower account for most of the energy mix, the CF-related energy consumption was as low as 3%. In the US, the fuel mix varies substantially by the state, resulting in energy carbon intensity ranging from 0.12 kg CO_{2e}/kWh in Idaho to 1.15 kg CO_{2e}/kWh in Washington, DC [82]. As shown in Fig. 7, the Midwest and Southwest regions contribute heavily to CF. This is due to the heavy reliance on fossil fuels in these regions. In the Midwest, 66.5% of the energy mix is satisfied by coal, while only 6% of the energy mix is from renewable energy sources. In contrast, in the Southwest regions, natural gas and coal make up 45% and 22% of the energy mix. On the other hand, approximately 50% of the energy mix in some regions is fueled by renewable energy, and thus the specific GHG emissions were found as low as 0.38 kg $CO_{2e}/m^3_{wastewater}$ [148].

Although the energy mix has a noticeable effect on the indirect CF of WWTPs, other factors also play an important role. For example, the Owls Head WWTP (New York, US) has a substantially lower CF related to energy compared to the Gloversville-Johnstown WWTP (New York, US), even though the energy mix is similar in those plants. This is because Owls Head WWTP uses chlorine as a disinfectant, while Gloversville-Johnstown WWTP uses ultraviolet (UV) for effluent disinfection, which requires electricity consumption [82].

Table 4
EFs for different sources of electric energy [151].

Energy Source	min (g CO_{2e}/kWh)	max (g CO_{2e}/kWh)
Coal	675	1689
Oil	510	1170
Natural gas	290	930
Biogas	50	700
Hydropower	3	40
Nuclear power	4	110

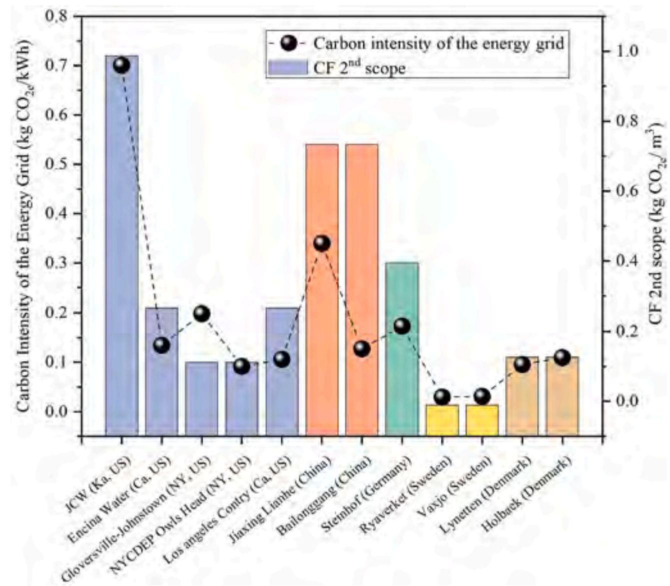

Fig. 7. Comparison between carbon intensity of the energy in different countries and scope 2 emissions of selected WWTPs [82,152].

6.2. Reducing energy consumption

Minimizing energy consumption can play a pivotal role in the CF reduction. There are two major approaches to decrease the power use in WWTPs: (i) reduction through operational changes and (ii) application of innovative, energy-efficient wastewater treatment processes [153]. Table 5 shows the energy mix and CF reduction under different scenarios in several case studies.

The DO, solids retention time (SRT), and mixed liquor recirculation (MLR) are crucial operational parameters in the bioreactor, which highly determine the operation of the aeration system and electricity consumption. Energy consumption could be decreased by 7–9% through the adjustment of SRT, DO, and MLR [154]. Mamais et al. [150] also obtained a 4500 MWh/year energy reduction by optimizing the SRT and DO concentration in a WWTP. Borzooei et al. [159] concluded that theoretically a reduction of up to 5000 MWh/year in energy consumption would be possible through operational changes in the largest Italian WWTP (2.1 million PE).

The advanced equipment also brings benefits to the CF. Li et al. [157] installed a TN online monitoring equipment at both influent and effluent, with a carbon source dosing system. As a consequence, the TN treatment efficiency increased by 1%, while the energy consumption and carbon emission decreased by 5.6% and 12.7%, respectively. Li et al. [157] updated the traditional blower to magnetic suspension centrifugal blowers, resulting in a 15–24% decrease in the aeration energy and a 4.6–7.7% decrease in the CF. Additionally, Ke and Lu [160] noted that the energy efficiency of the magnetic suspension blower could be improved by 20%.

ML is one of the novel techniques for reducing the energy demand of WWTPs. Cao and Yang [161] developed a model (OS-LEM) to control DO supply of aerobic bioreactors, which can reduce the energy consumption by 40% in comparison with the conventional on/off DO control. Similarly, Ramli et al. [158] used ML to build a prediction model to reduce the energy consumption of WWTPs by forecasting it one month ahead of time, which estimated a 2.2% reduction in energy consumption.

Table 5
Reported energy reductions in WWTPs under different scenarios.

Reference	Scenario	GHG emission related to energy ($kgCO_{2e}/m^3$)	Reduction in Energy Demand		Total GHG emission decrease (%)	Remarks
			(MWh/year)	%		
Mamais et al. [150]	Reduction in SRT and DO	–	4500	–	–	Psyttalia (Greece) WWTP in Greece was analyzed.
Daskiran et al. [154]	Reduction in SRT and MLR	0.1	3510	–	0%	Emission rates matched the active scenario in the WWTP
	Reduction in SRT and DO	0.06	3530	–	−15%	The emission rates were about 0.17 kgCO2/m3 more than the active scenario in the WWTP
	Reduction in MLR and DO	0.05	4661	–	−9%	The emission rates were about 0.1 kgCO2/m3 more than the active scenario in the WWTP
Borzooei et al. [155]	Reduction in SRT	–	5000	–	–	In Castiglione Torinese WWTP (Italy) different scenarios where SRT was optimized were tested
Longo et al. [147]	Minimizing energy consumption of pumps, motors, and other electro-mechanic devices	–	–	10–20%	–	WWTP in Missouri Southeast, USA
Torregrossa et al. [156]	optimizing pumping, blowers, and AD	–	–	50–80%	–	Various WWTPs in Europe
Li et al. [157]	Upgrading to a magnetic suspension blower	–	–	15–24%	4.6–7.7%	14 WWTPs in North-Central China were examined
	Accurate aeration system modification	–	–	6.3%	10.37%	The implementations were studied on two WWTP in China
	Intelligent precise aeration system for sewage treatment	–	–	5.3%	10.93%	The implementations were studied on a WWTP in China
Ramli et al. [158]	Optimizing pumping, blowers, and AD	–	–	3%	–	WWTP in Peninsulay (Malaysia)

6.3. Increasing energy production inside WWTPs

6.3.1. Co-digestion

Co-digestion is proposed as a straightforward approach for improving macronutrient balance, correcting moisture, and/or diluting inhibitory or toxic compounds [162], which possessed a positive effect on energy production and CF reduction. Most energy neutral plants located in developed countries adopt biogas recovery through co-digestion [163]. Koch et al. [164] reported more than double energy recovery when adding a ratio of 10% FW as a co-substrate in a WWTP. The biogas output increased by 17% by co-digesting 7% of fat with the mixed sewage sludge [30]. Sarpong and Gude [165] reported that the utilization of fat-oil-grease (FOG) for co-digestion with low-strength wastewater can increase energy production by 0.08 kWh/m^3, while the total required energy of the studied plant was reported as 0.32 kWh/m^3. Wickham et al. [166] demonstrated a biogas production increase of up to 191% using carbonated soft drinks as a co-substrate. An increase of 81% in biogas production was achieved by adding 3% of glycerol as a co-substrate [167]. Similarly, biogas production was increased by up to 209% by using grease trap water as a co-substrate [168]. Many plants, employing co-digestion, have reached 100% energy neutrality and even become net energy positive, including Zurich Wedholzli WWTP-Switzerland (42 GWh/year), Point Loma WWTP-US (193 GWh/year), Grevesmuhlen WWTP-Germany (193 GWh/year), and Sheboygan Regional WWTP-US (32 GWh/year) [169]. Table 6 provides the potential increase in the biogas production rate for various co-substrates.

6.3.2. Biogas utilization and upgrading

The produced biogas can be utilized internally for valorization in combustion engines to supply electricity and heat generation. Most WWTPs that produce biogas through AD in developing countries do not have thermal energy recovery or electricity production, thus the biogas is flared and wasted [163]. A decade ago, the USEPA estimated that if 544 US WWTPs with AD utilize CHP, the reduction in energy consumption would be equivalent to the emissions from 430,000 cars [173]. The Psyttalia WWTP deployed CHP and revealed the lowest annual energy consumption among 10 studied WWTPs in Greece, at 15 kWh/PE

Table 6
Biogas production rate increase after using different co-substrates.

Co-substrate	Condition	Energy production without co-substrate (kWh/d)	Energy Production with co-substrate (kWh/d)	Biogas Production without co-substrate (L/gVS/day)	Biogas Production with co-substrate (L/gVS/day)	Reference
Microalgae	–	774	1189	–	–	[170]
Cheese whey	–	774	1531	–	–	
The organic fraction of municipal solid waste (OFMSW)	Volumetric ratio 75:25; HRT 20 d	7688	8418	–	–	[171]
Dry waste (DW)	Volumetric ratio 80:20; Hydraulic Retention Time (HRT) 16 d	677	10,639	–	–	
OFMSW + DW	Volumetric ratio 68:23:9; HRT 18 d	6037	8972	–	–	
Grease (G)	TS: G; 50:50	–	–	0.48	0.78	[172]
Septage (SP)	TS:SP; 90:10	–	–	0.48	0.46	
Whey (W)	TS: W; 70:30	–	–	0.48	0.75	

[150].

The produced biogas can be upgraded to biomethane, which has the same CH_4 quality as natural gas and could be utilized as a fuel for vehicles or pumped into the grid [174]. Biogas upgrading, involving CO_2 removal, employs gas sorption and separation techniques. Physical and chemical scrubbings effective but energy-intensive, while pressure swing adsorption is less energy-intensive, but more complex. Membrane separation technologies are environmentally friendly with a relatively low energy consumption, whereas cryogenic separation demands high energy for cooling [175]. Following the technological advances in biogas upgrading, a transition away from CHP and towards upgrading biogas to biomethane has been trending [176]. Biogas is already being converted to biomethane in 15 EU countries and 10 of those countries are injecting biomethane into the natural gas grid. The countries with the highest number of biomethane plants include Germany (185 plants), UK (80 plants), and Sweden (61 plants), while the other countries are lagging behind [174].

To address GHG release from biogas upgrading, emerging technologies focus on CO_2 utilization through methanation, combining CO_2 with H_2 to produce CH_4 via chemical and biological processes. Implementing this technology requires investment and operational costs, especially for H_2 production through electrolysis. The biomethanation process efficiency is a limitation, but benefits include carbon neutrality and continuous fuel delivery for grid balance [175].

Nguyen et al. [169] compared the CHP and biomethane for biogas utilization from WWTPs. It was concluded that the CHP was effective with a negative CF, while the biogas upgrading to biomethane resulted in a higher CF due to a high energy input during the upgrading process. Ravina and Genon [177] also concluded that CHP of biogas with the thermal energy utilization produces the lowest CO_{2e} ($-0.277 tCO_2/t$ biogas). The biomethane upgrade (-0.13 tCO_2/t biogas) to the Italy's national grid enabled to store biomethane at a lower cost for further utilization. Thus, upgrading biogas to biomethane could be a more sustainable scenario than on-site biogas consumption in CHP units, especially if the gas energy content is not fully cogenerated and the produced thermal energy is not capitalized. Regarding the CH_4 loss during upgrading, a sustainable system should reduce CH_4 slip below 4%, with a target value of 0.05% [177].

7. Mitigation of GHG emissions via chemicals and transportation optimization

To achieve full decarbonization and the net-zero carbon condition in WWTPs, both upstream and downstream, indirect emissions are ought to be reduced. Ambiguous GHG emissions from WWTPs are primarily related to chemical agent usage and transportation of by-products (grit, screenings, and sludge) [28].

7.1. Reducing chemical consumption

Chemicals are used at different stages of wastewater treatment and sludge management. To minimize their contribution to the CF, not only consumption cuts can be implemented, but also some sustainability criteria introduced to the purchase procedure may result in reducing the indirect GHG emissions. Factors linked to the usage of chemical agents in the context of upstream indirect emissions are leveled at 1964 kg CO_{2e}/kg for inorganic agents and 1909 kg CO_{2e}/kg for organic agents (Ecoinvent database 3.6).

The inorganic agents are commonly used to support enhanced biological P removal. Metal salts, such as aluminum (III) sulphate, iron (III) chloride, and lime in the form of $Ca(OH)_2$ or CaO_2, are added to trigger precipitation. The use of lime compounds results in a pH increase, which requires the addition of acid chemicals to adjust the pH level of the WWTP's effluent. Moreover, both metal salts and calcium agents result in an increase in sludge volume, leading to higher GHG emissions associated with sludge disposal. It is preferable to use recycled inorganic chemical agents instead of new resources in order to avoid emissions related to the extraction and manufacturing processes of raw materials. Organic agents and polymers are regularly applied for flocculation, coagulation, and dewatering processes. Recycling could effectively reduce the indirect GHG fluxes. Instead of synthetic, polymers of bio-based origin can be applied as a more environmentally friendly choice [178,179].

The application of chemical agents influences the entire wastewater treatment process and its stages [180]. The enhancement of primary sedimentation via chemically forced flocculation contributes to a higher chemical consumption in the overall GHG balance. Ultimately, it becomes an advantage due to the biogas and energy production bust. In contrast, the chemical agents used to support dewatering processes may become a target in lowering CO_{2e} indirect emissions via overall chemical agent consumption cuts at WWTPs.

7.2. Reducing transport-related CF

Conveying activities and services, such as upstream transportation of chemical agents and downstream transport of wastewater treatment by-products (grit, screenings, and sludge), contribute to the indirect emissions of WWTPs. It is often challenging to implement a transport-related emission reduction plan based on switching to more environmentally-friendly means of transportation, such as rail transport, or shortening the transportation distance. Therefore, more advanced reduction scenarios are sought.

According to the Department for Environment, Food and Rural affairs (DEFRA) [181] EF database, the amount of GHG emitted to the atmosphere depends on (1) engine standard, which translates into the type of fuel combusted, (2) size of the vehicle – rigid, and (3) the laden. Emission reduction can be achieved via a detailed transportation plan, including optimization of laden for each material transported, adaptation of the vehicle rigid type (elimination of oversizing), and optimization of fuel type engines.

Transport-related concerns resulted in the promotion of various biofuels: especially bioethanol from sugar and starch crops and biodiesel from oilseed crops [182]. Triggered by the Renewable Energy Directive I and II [183,184], the process of replacing fossil-origin gasoline and diesel with biofuels presents at least 65% GHG savings in comparison with the conventional fuels.

8. Recovering N_2o from wastewater as an energy source

Recovering N_2O as an energy source would be a win-win strategy. The process would be a step-forward towards net-zero operation by offering both the possibility of energy generation and mitigation of N_2O emissions. There are limited studies available in the literature that investigate N_2O recovery, which has also been being used as a powerful oxidant for energy generation. Wu et al. [185] suggested that applying Coupled Aerobic-Anoxic Nitrous Decomposition Operation (CANDO) might be one of the bioprocesses with a high potential to achieve energy neutral wastewater treatment paradigm. The process involves three steps: i) partial nitrification of NH_4^+ to NO_2^-; ii) partial anoxic reduction of NO_2^- to N_2O; and 3) N_2O conversion to N_2 with energy recovery by either catalytic decomposition or co-combustion of N_2O with CH_4. Yu et al. [186] documented a novel process that recovers N_2O by inhibiting the activity of the nitrous-oxide reductase (N_2OR) in denitrifying bacteria. N_2O recovery was demonstrated in batch experiments in which NO was the only electron-acceptor. N_2O recovery level in their study reached approximately 70% of the total gas production.

The most investigated N_2O recovery process is CANDO, which has been tested with various configurations and real wastewater [187]. However, achieving stable long-term operation requires further experiments. Promising alternatives include autotrophic photoelectrotrophic denitrification, sulfur-driven denitrification, and hydrogenotrophic denitrification [188]. These autotrophic processes are currently immature technologies and need intensive investigations. N_2O recovery from high-strength wastewater is more favorable than low-strength (mainstream) wastewater due to a higher energy and economic potential. Recently, studies such as Deng et al. [189] and Huo et al. [190] have developed reliable models to simulate N_2O recovery from Fe(II) EDTA-NO. The simulation results of Huo et al. [190] suggested that with a S/N mass ratio of around 1, high-purity N_2O could be more rapidly and efficiently recovered from Fe(II)EDTA-NO (N_2O recovery efficiency reached up to 85%). Deng et al. [189] highlighted that sufficiently increasing the headspace volume of the reactor was considered an ideal strategy to obtain ideal N_2O production of 86.6%. Duan et al. [188] critically reviewed the progress of research on N_2O recovery and counted (i) unstable nitritation, (ii) the low energy potential and (iii) environmental risks associated with intentional N_2O emissions in mainstream treatment, as technical and economic challenges.

9. Challenges and limitations while implementing net-zero carbon practices

Both direct (fugitive, process-related) and indirect (due to energy and chemicals consumption, transportation) GHG emissions have some potential for reduction. However, implementing mitigation strategies for the net-zero carbon concept in WWTPs comes with recognized limitations and trade-offs. Effluent quality, operational cost and GHG emissions are potentially conflicting objectives [95,97,191]. For example, energy savings in the AS bioreactor, achieved by decreasing aeration intensity, may result in increased N_2O emissions [192]. Co-digestion of external substrates reduces energy-related emissions and increases process-related GHG emissions [30]. Those studies proposed either separate evaluation criteria or integrated plant performance indicators. LCA complements decision-making to emphasize environmental impact [97]. In a full-scale WWTP modelling study [74], three sustainability criteria were considered: effluent TN concentration, overall energy balance and GHG footprint. Investigating the DO set-point in the aerobic zone and MLR rate showed improvements in GHG emissions and effluent quality. Technological upgrades, such as CEPT and deammonification in reject water treatment, enhanced energy neutrality and reduced GHG footprint, but did not improve effluent quality simultaneously.

10. Conclusions

Several steps may be considered when moving WWTPs towards net-zero carbon conditions. The direct emissions from treatment processes can contribute to more than 60% of the CF in WWTPs, while energy consumption with over 30% contribution to the CF has also a high potential for decarbonization of WWTPs. The remaining emissions are of lesser importance and can be offset by optimizing chemical usage and transportation. The first major step towards net-zero carbon condition is identification of emission hotspots and quantification of those emissions via either measurements, reliable models or EFs. The second step is GHG mitigation via process modification and energy usage optimization. The implementation of novel N removal processes such as PD/A and DAMO/A, could reduce GHG emissions and energy consumption while ensuring reliable N removal efficiencies. Other techniques such as source separation systems could potentially allow mitigation of N_2O emissions by 60% while avoiding energy-intensive N fertilizer production. Nutrient recovery methods are another approach which offered negative value for the net CF. Permeable membrane N recovery offered 1 kWh/kg N of energy savings and P recovery led to -3.76 kg CO_{2e}/kg $P_{recovered}$ CF savings. Upgrading biogas to biomethane could be a more sustainable scenario than on-site biogas consumption in CHP units, especially if the thermal energy is not capitalized. Recovering and utilizing N_2O for energy production is a promising method which leads to both direct and indirect CF reductions.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper

Data availability

No data was used for the research described in the article.

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Paulina Szulc-Kłosińska Poznań,

PhD Candidate's Name and Surname

STATEMENT

about the percentage contribution to the publication titled

Net-zero carbon condition in wastewater treatment plants: A systematic review of mitigation strategies and challenges

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List of the Authors	Percentage Contribution	Author's signature
Mojtaba Maktabifard	20%	
Hussein Al-Hazmi	15%	
Paulina Szulc-Kłosińska	**15%**	
Mohammad Mousavizadegan	10%	
Xianbao Xu	10%	
Ewa Zaborowska	10%	Podpisuję z Cencert Podpisany elektronicznie przez Ewa Katarzyna Zaborowska 10.10.2025 16:58:11 +02'00'
Xiang Li	10%	
Jacek Mąkinia	10%	

PhD Candidate's contribution:

Conceptualization of Chapter 4, text editorial and preparation

Paulina Szulc-Kłosińska
PhD Candidate's Name and Surname

Poznań, 29.09.2025

STATEMENT

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Hussein Al-Hazmi	15%	
Paulina Szulc-Kłosińska	**15%**	Szulc Kłosińska
Mohammad Mousavizadegan	10%	
Xianbao Xu	10%	
Ewa Zaborowska	10%	
Xiang Li	10%	
Jacek Mąkinia	10%	

PhD Candidate's contribution:

Conceptualization of Chapter 4, text editorial and preparation

Paulina Szulc-Kłosińska
PhD Candidate's Name and Surname

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