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NUTRIENT RECOVERY FROM A WASTEWATER TREATMENT PLANT FOR AGRICULTURAL APPLICATIONS IN THE CONTEXT OF THE CIRCULAR ECONOMY: A SUSTAINABLE ANALYSIS

Thesis Report and Annexes

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Resum

El fòsfor és considerat un recurs essencial degut a la seva importància en el desenvolupament de plantes, animals i humans. Es tracta d'un recurs no renovable i com a conseqüència al ràpid creixement de la població mundial, la demanda de fòsfor per a aliments i fertilitzants per a cultius es veurà inevitablement augmentada, situant les seves reserves en risc d'esgotament, la extinció de les quals es preveu durant el pròxim segle. Per altra banda, el contingut de nitrogen al sòl és limitat, el que es pretèn resoldre mitjançant la seva fixació en les seves formes reactives com els aminoàcids, nitrats i amoni; no obstant, la fixació natural del nitrogen no és suficient per a cobrir tota la demanda energètica i alimentària de la creixent població mundial, pel que la seva producció antropogènica està sorgint actualment mitjançant processos industrials (procés Haber–Bosch) amb un alt cost energètic i altes emissions de GEH.

En aquest treball, es va dur a terme una avaluació ambiental i econòmica de la recuperació de fòsfor i nitrogen d'una EDAR i la seva posterior aplicació com a fertilitzants en un camp de cultiu. Posteriorment, es va procedir a la comparació d'aquesta configuració amb l'aplicació directa de fertilitzants minerals comercials al camp de cultiu. Per a la comparació de les dues alternatives es van dur a terme tant un anàlisi de cicle de vida (ACV) com un anàlisi de cost de cicle de vida (CCV). L'ACV es va dur a terme mitjançant el software OpenLCA v1.10.2 i la base de dades ecoinvent v3.6.

La configuració de la línia de fangs a avaluar es va implementar a gran escala a l'EDAR Murcia-Este i es va basar en la producció d'una corrent enriquida amb PO_4^- a partir del procés d'elutriació de fangs en els espessidors primaris. La corrent rica en fòsfor es va tractar en una unitat de cristal·lització d'estruvita, precipitant com a fosfats, amoni i magnesi en forma de perles d'estruvita. La recuperació de nitrogen en forma de sals d'amoni (p.e., nitrat d'amoni) es va realitzar mitjançant columnes d' intercanvi iònic de zeolita com a procés de preconcentració i una etapa de contactors de membrana de fibra buida (HFMC).

En la majoria de categories ambientals, l'escenari de recuperació de nutrients va resultar en majors impactes ambientals respecte a l'escenari base, atribuïts principalment als alts consums d'àcid nítric i sulfat de potassi. Els resultats de l'ACV van suggerir la necessitat d'estratègies de millora de l'acompliment ambiental de l'escenari de recuperació de nutrients centrant-se en la reducció dels consums d'aquests fertilitzants. A més, com s'esperava, l'anàlisi CCV va resultar en costos més elevats per a l'escenari de recuperació de nutrients. Finalment, després d'analitzar els resultats, es podria plantejar el disseny d'un pla de negoci entorn a el model de recuperació de nutrients avaluat i els nutrients recuperats per al seu ús com a fertilitzants per tal de promoure la transició d'un model d'economia lineal a un model d'economia circular tenint en compte els aspectes mediambientals, econòmics, aspectes tècnics i socials que pogués comportar.

Resumen

El fósforo es considerado un recurso esencial debido a su relevancia en el desarrollo de las plantas, animales y humanos. Se trata de un recurso no renovable y como consecuencia al rápido crecimiento de la población mundial, la demanda de fósforo para alimentos y fertilizantes para cultivos se verá aumentada inevitablemente, poniendo sus reservas en riesgo de agotamiento. Por otro lado, el contenido de nitrógeno en el suelo es limitado, lo que se pretende solventar mediante la fijación de este en sus formas reactivas como aminoácidos, nitratos y amonio; no obstante, la fijación natural del nitrógeno no es suficiente para cubrir toda la demanda energética y alimenticia de la creciente población mundial, por lo que su producción antropogénica está surgiendo actualmente mediante procesos industriales (proceso Haber–Bosch) con un alto coste energético y altas emisiones de GEI.

En el presente estudio, se realizó una evaluación ambiental y económica de la recuperación de fósforo y nitrógeno de una EDAR y su posterior aplicación como fertilizantes en un campo de cultivo. Posteriormente, se llevó a cabo la comparación con la aplicación de fertilizantes inorgánicos comerciales directamente en el campo de cultivo. Para comparar las dos alternativas, se llevaron a cabo tanto un análisis de ciclo de vida (ACV) como un análisis de coste de ciclo de vida (CCV). El ACV se realizó a través del software OpenLCA v1.10.2 y la base de datos ecoinvent v3.6.

La configuración de la línea de lodos a evaluar se implementó a gran escala en la EDAR Murcia-Este y se basó en la producción de una corriente enriquecida con PO_4^- a partir de elutriación de lodos en los espesadores primarios. La corriente rica en fósforo se trató en una unidad de cristalización de estruvita, precipitando como fosfatos, amonio y magnesio en forma de perlas de estruvita. La recuperación de nitrógeno como sales de amonio (p.e., nitrato de amonio) se llevó a cabo mediante columnas de intercambio iónico de zeolita como proceso de preconcentración y una etapa de contactores de membrana de fibra hueca (HFMC).

En la mayoría de categorías ambientales, el escenario de recuperación de nutrientes resultó en mayores impactos ambientales respecto al escenario base, atribuidos principalmente a los altos consumos de ácido nítrico y sulfato de potasio. Los resultados del ACV sugirieron la necesidad de estrategias de mejora del desempeño ambiental de este escenario centrándose en la reducción de los consumos de estos fertilizantes. Además, como se esperaba, el análisis CCV resultó en costes más elevados para el escenario de recuperación de nutrientes. Finalmente, tras analizar los resultados, se podría plantear el diseño de un plan de negocio entorno al modelo de recuperación de nutrientes evaluado y los nutrientes recuperados para su uso como fertilizantes con el fin de promover la transición de un modelo de economía lineal a un modelo de economía circular teniendo en cuenta los aspectos medioambientales, económicos, aspectos técnicos y sociales que pudiera conllevar.

Abstract

Phosphorous is considered as an essential resource since it plays a key role in the development of plants, animals and humans. It is a non-renewable resource and as a consequence to the fast growth of the world population, phosphorous demand for food and fertilizers for crops will increase inevitably putting its reserves in risk of depletion, which are estimated to be exhausted in the next century. On the other hand, nitrogen content in soils is limited, which is proposed to be sold by fixing it in reactive forms such as amino-acids, nitrate and ammonia; however, natural fixing is insufficient for covering all the food and energy demand of the rising world population, so the anthropogenic production of nitrogen is currently emerging by industrial processes (i.e., Haber–Bosch process) that are high energy intensive and produce significant GHG emissions.

In this work, an environmental and economic assessment of phosphorus and nitrogen recovery from a wastewater treatment plant and their further application as fertilizers in a crop field was carried out. Later on, it was compared with the crop field application of commercial mineral fertilizers. To compare both alternatives, a lifecycle assessment (LCA) and a lifecycle costing (LCC) were carried out. The LCA was performed by means of the software OpenLCA v10.1.2 and the ecoinvent v3.6 database.

The sludge line configuration to be assessed was full-scale implemented in Murcia-Este wastewater treatment plant (WWTP) and it was based on the production of a PO_4-enriched stream from sludge via elutriation in the primary thickeners. Phosphorus rich stream was treated in a struvite crystallizer unit, precipitating as phosphates, ammonium and magnesium as struvite beads, while nitrogen recovery as ammonium salts (e.g., ammonium nitrate) was recovered by means of zeolite ion exchangers columns as a pre-concentration process and a hollow fibber membrane contactors (HFMC) stage to produce liquid fertilizer.

In most of environmental categories, nutrient recovery scenario resulted in higher environmental impacts than baseline scenario, which were mainly attributed to high requirements of nitric acid and potassium sulphate, directly applied on the crop field and in the case of nitric acid also in the ammonium nitrate recovery unit. LCA results suggested that some strategies would be needed to improve the environmental performance of the nutrient recovery scenario focusing on the reduction of these fertilizers requirements. Moreover, as expected, LCC analysis resulted in higher costs for nutrient recovery scenario. Finally, after analysing the results, a business plan around the assessed nutrient recovery model and the recovered nutrients for their use as fertilizers could be designed in order to promote the transition from a linear economy model to a circular economy model taking into consideration the environmental, economic, technical and social aspects that could entail.

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I would like to thank professors César Valderrama and José Luis Cortina, director and co-director of this masters' thesis, for their dedication and availability for solving any doubt that may arise along the development of the project; their contributions have been an essential point, especially at the very beginning when the project methodology and perspective was needed to be defined. Their guidance over the master's thesis allowed me to expand my knowledge and to familiarize with the life cycle assessment and life cycle costing methodology applied to a real project. On the other hand, I would like to thank Kishore Ganesan for his work on the OpenLCA software and the quantification of the environmental impacts.

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1. Introduction

In the current transition to a Circular Economy model, closing the nutrients cycles has become a critical challenge that has the aim of migrating to a more effective and sustainable resource management, from an economic and an environmental perspective. Mineral fertilizer demand increases continuously and the dependence of the agricultural sector on these fertilizers, which are based on fossil resources, can be considered as an important thread for the future of livestock and human food. In the group of key actions for waste and recycling there are also different opportunities for optimizing the nutrient management. Nitrogen (N) and phosphorous (P) recycling from waste streams is needed to be carried out, as well as from the municipal wastewater networks, manure or industrial effluents. Technologies for carrying out these processes currently exist, although they are not applied by the same way around the world. Moreover, among these technologies for wastewater treatment and nutrient recovery, only a limited number is implemented at industrial scale, being the quantity of nutrients recovered a minimum percentage of this streams potential (Encinas et al., 2020). Nutrients found in waste streams are mostly compounds of carbon (C), N and P. All of them are important for sustenance of various life forms (Sengupta et al., 2015).

Phosphorous can be considered as an essential resource since it plays a key role in the development of plants, animals and humans. It is the main component of conventional mineral fertilizers; thus, it is an element which is vitally required for the plants to grow and therefore, it is critically important for food production. It is estimated that 148 million of tonnes of phosphate rock are used every year, being a 90% of the total demand employed in the food production sector. Moreover, P is a non-renewable resource and there are no elements that can replace it as a raw material for essential nutrients in the production of fertilizers (Cordell et al., 2009; Encinas et al., 2020; Pradel & Aissani, 2019). As a consequence to the fast growth of the world population and of rising living standards in emerging and developing countries, P demand for food and fertilizers for crops will increase inevitably putting P reserves in risk of depletion, which are estimated to be exhausted between the present and the next century (Cordell et al., 2009; Egle et al., 2016; Roldán et al., 2020). In its mineral form, P is found highly concentrated on earth locked in igneous and sedimentary deposits, being the mining of these rocks the most viable method of extraction (Sengupta et al., 2015). About 86% of P reserves are found in Western Sahara and Morocco (71,4%), China (4,7%), Algeria (3,1%), Syria (2,6%), Brazil (2,4%), South Africa (2,1%), Jordan and the USA, which means that the resource availability of supply is subject to geopolitical risks and market fluctuations (Pradel & Aissani, 2019; Roldán et al., 2020). Moreover, as these current reserves of P get exhausted the quality becomes more variable, which leads to lower P concentrations and higher impurities; as a consequence, the cost of extraction, processing and shipping is being drastically increased (Pradel & Aissani, 2019; Sengupta et al., 2015). Europe imports more than a 90% of its P needs, with only one active mine in Finland; this is why in recent years, European

Commission included P and phosphate rock in the list of Critical Raw Materials, which remarks the high economic importance that is associated to its supply, and emphasized the change towards a more circular use of nutrients by promoting the recovery of P from local sources (Rosemarin et al., 2020; Rufí-Salís et al., 2020). Research, governments and industry have recognised the importance of their respective roles for exploiting municipal wastewater as a P source, which has the potential to substitute a significant part of phosphorus rock demand; by this way, Circular Economy practice is increased and at the same time, the overall environmental impacts from current P use practices, such as air emissions and eutrophication of water bodies, are reduced considerably (Amann et al., 2018).

Recovery of P can be performed from both livestock waste and wastewater, being recovered either in the liquid phase of the waste, from the sludge due to its concentration or from the sludge ash (Sarvajayakesavalu et al., 2018). The simplest methods for P recovery are the direct use of the active sludge or from the digestate as fertilizer or its composting. At the same time, waste can contain significant quantities of organic contaminants potentially dangerous (e.g., aromatic hydrocarbons) and Potential Toxic Elements (PTEs). Legal regulations for the use of this wastes as fertilizers are increasingly strict, especially the ones that define the maximum allowed PTEs concentrations in the waste that are spilled on the soil, and that is the reason why the technologies for waste treatment for the indirect recovery of P are taking more relevance; chemical precipitation, biological P removal and crystallization are the main examples of technologies used for P recovery from liquid waste. From solid waste, the most common recovery processes are the digestion, precipitation and acidification. Finally, for P recovery from the ashes, thermochemical and thermomechanical processes can be used (Encinas et al., 2020).

The development of a sustainable management of P in wastewater treatment plants (WWTPs) is required since the presence of pollution problems such as eutrophication due to P release in wastewater effluents, which refers to the pollution in aquatic ecosystems after an excessive use of P-based fertilizers and the discharge of untreated wastewaters, inducing algal blooms, reducing the dissolved oxygen in water column and forming P-deposits in sediments. Although it is usual that WWTPs have the aim to remove phosphate rather than to recover it, which avoid environmental problems such as eutrophication, several studies confirm the possibility of P recovery from wastewater due to its potential given the large quantity of municipal wastewater that is generated; however, it is important to consider its low phosphates content. That is the reason why WWTPs are crucial to enhance P recovery. Thus, several initiatives are now under study with the aim to P reuse and recovery, by means of which is considered to reduce Central Europe’s dependence on external P-supplies by up to a 40%, turning WWTPs to a wastewater resource recovery facility concept which does not consider wastewater as a mixture of pollutants but a rich source of valuable products (Bouzas et al., 2019; Roldán et al., 2020).

Nowadays, in WWTPs, P is removed biologically, by means of what is called Enhanced Biological Phosphorous Removal (EBPR), chemically or by a combination of both methods. Combining EPBR in the water line and anaerobic digestion (AD) in the sludge line is a triggering factor for the uncontrolled precipitation of P-compounds inside and downstream digesters, which causes pipe blockages and deposits on the walls of pipelines, reactors and other equipment (Roldán et al., 2020). In WWTPs where chemical or biological P removal is implemented, between 75 and 90% of the total P entering the WWTP is transferred into the sewage sludge (Bouzas et al., 2019). Inorganic P compounds formed in WWTPs are orthophosphate (PO_4) and polyphosphate (Poly-P). In the EBPR process, phosphates and other ions (i.e., Mg^{2+}, K^{2+}) are removed from wastewater and accumulated by means of Polyphosphate Accumulating Organisms (PAO) as Poly-P granules after PO_4 uptake (Roldán et al., 2020). Later on, during the AD, Poly-P is released to the liquid phase, making the liquors rejected from digested sludge dewatering show high concentrations of P, ammonia and magnesium, that make these streams very appropriate for recovering P as struvite through a crystallization process (Martí et al., 2010). Nevertheless, most of the P that is dissolved in the sludge is lost due to precipitation during AD and post-AD steps, causing relevant operational and maintenance problems in the WWTP and therefore, increasing the related costs. Thus, in order to guarantee phosphorus rich streams and obtain high struvite production in the crystallization process, uncontrolled P precipitation should be reduced (Bouzas et al., 2019).

Among the implemented P recovery techniques, its recovery by struvite ($NH_4MgPO_4{\cdot}6H_2O$) crystallization is one of the most promising and efficient technologies for treating sludge rejected streams and one of the most widely recommended, especially in WWTPs with EBPR, taking place before and after AD (Martí et al., 2010; Pradel & Aissani, 2019). Struvite is the main N and P recovery product and a valuable slow-release fertilizer for agriculture with low solubility, high P content and with the possibility of being commercialized. Nowadays, struvite product criteria are under study for the possibility of being used as direct fertilizer or as a raw material for fertilizer manufacturing, with the aim to integrate it into the EU Fertilizer Regulation (Bouzas et al., 2019; Roldán et al., 2020). Beyond the implementation of struvite in the agricultural sector replacing demand for traditional fertilizers and its commercialization, P recovery by means of its crystallization is directly related with the prevention of uncontrolled spontaneous P precipitation in the sludge treatment line, which is formed naturally in WWTPs due to high concentrations of PO_4^{3-} and NH_4^+ along with the presence of Mg^{2+} and a basic pH environment (8.5 – 9.5) in a molar ratio of 1:1:1, and can cause blockages of pipes and get rise to large and costly maintenance problems (Rufí-Salís et al., 2020; Sena et al., 2021; Sena & Hicks, 2018). Struvite crystallization is normally applied in the dewatering stream after AD, thus not preventing problems such as the aforementioned and therefore, worsening P recovery due to the losses that are given during the process by this uncontrolled crystallization (Bouzas et al., 2019).

On the other hand, although the high abundance of N in the atmosphere (78%v.) in a highly stable and non-reactive form N_2 gas, its content is limited in soils. This limitation is proposed to be solved by fixing that nitrogen in reactive forms such as amino-acids, nitrate and ammonia, that are essential for plant growth (van der Hoek et al., 2018). Nevertheless, natural fixing is insufficient for covering all the food and energy demand of the rising world population, so the anthropogenic production of nitrogen is currently emerging (Sengupta et al., 2015). At the same time, its abundancy in the atmosphere makes its supply unlimited, which puts the recovery of N in a lower level of priority than P recovery. The Haber–Bosch was invented in 1909 and provides an industrial fixation of N_2 gas into ammonia; since it was invented, the production of N-based fertilizers supported the largest historical increase in food production capacity and the productivity of agricultural crops was more than quadrupled. The introduction of the Haber–Bosch process affected the nitrogen cycle since the food produced by using N-based fertilizers produced through it is excreted mainly as urea and ammonium by human metabolism and then, discharged to the sewer. In order to avoid the eutrophication of water, on the conventional activated sludge process in the current wastewater treatment technology, this reactive nitrogen is biologically converted to its non-reactive form, N_2 gas, by means of nitrification/denitrification or deammonification processes and then released into the atmosphere (van der Hoek et al., 2018). Despite the nitrogen cycle is anthropogenically closed through the combination of the aforementioned processes, the Haber–Bosch process for fertilizer production is highly energy intensive and produces substantial GHG emissions. That is why the recovery of N from wastewater in WWTPs for its reuse is one of the challenges to sustainably operate in these facilities (Chen & Valderrama, 2018).

As aforementioned, N can be recovered from wastewater through ammonia precipitation as struvite, but this process is more focused on P recovery. Ion exchange/adsorption-based processes by means of zeolites, for example, which is the most popular ion exchanger/adsorbent for nitrogen recovery, are considered a good alternative to obtain concentrated streams of reactive nitrogen. Other options such as bioelectrochemical systems can recover nitrogen as ammonia gas and ammonium sulphate ($(NH_4)_2SO_4$). Air stripping of ammonia from the anaerobic digestate has been reported to recover over the 90% of nitrogen and finally, hydrophobic membrane separation is able to recover almost 100% of nitrogen without secondary pollutants in the permeate (Sengupta et al., 2015; van der Hoek et al., 2018).

The LIFE program is the EU's funding instrument for the environment and climate action. Established in 1992, its general objective is to contribute to the implementation, updating and development of EU environmental and climate policy and legislation by co-financing projects with European added value (*LIFE Programme & Clim'Foot | Clim'Foot*, n.d.). LIFE ENRICH Project *(Enhanced Nitrogen and Phosphorous Recovery from wastewater and Integration in the value Chain)*, initiated in 2017 and with a duration of three years and a half, is a LIFE project coordinated by *Cetaqua Water Technology Centre* in Murcia-Este WWTP whose goal is to contribute to Circular Economy through the recovery of nutrients from WWTP and its valorization in agriculture, either direct use on crops or through the fertilizer industry. The products obtained are proposed to be mixed in order to find optimal mixtures and the agronomic properties of these products will be validated at full-scale through field tests in order to ensure the viability of them. Finally, it is proposed to define the whole value chain, involving several partners from different sectors, in order to ensure the replicability in other case studies or other EU regions. By means of this project, it is proposed to define a business model for the entire nutrient recycling value chain for Spain and assess the replicability of it to other European countries such as UK, France, Germany and Poland, considering technical, economic and environmental issues as well as the legal framework in each particular case. It is also proposed to validate a treatment train integrating different technologies for the recovery of both N and P from wastewater in existing WWTPs, as well as to increase the efficiency of P recovery by implementing new elutriation schemes for enhanced production of different forms of struvite, to develop membrane contactors technology for the production of ammonium salts and contribute to their regulation in future fertilizer directives. Finally, LIFE ENRICH wants to promote the agronomic value of digested sludge as a source of nutrients (N and P) and organic carbon and to define the optimal fertilizer mixtures of struvite, ammonium salts and digested sludge for crops of interest, demonstrating the agronomic properties of the recovered products. Thus, the most relevant environmental problems that LIFE ENRICH faces are the depletion of phosphate rock reserves, the high carbon footprint of production and use of chemical fertilizers and of the N-removal processes in WWTPs, the eutrophication and the large quantities of sewage sludge that are generated in WWTPs.

Life Cycle Assessment (LCA) is a comprehensive approach to evaluate the environmental aspects and the potential environmental impacts associated to a product, which may refer to goods, technologies or services, during its entire life cycle, that involves the pathway from the raw material extraction, materials processing, manufacture, distribution, use and the waste management stage, including the end-of-life disposal, recycling or reuse of this product (Muralikrishna & Manickam, 2017). By this way, this method has been developed and widely applied within research and industry, and more specifically in the field of nutrient recovery from wastewater treatment (Lam et al., 2020; Pradel et al., 2016; Remy & Jossa, 2015; Sena & Hicks, 2018). Thus, LCA provides a method to evaluate and compare different wastewater-based nutrient recovery technologies from environmental sustainability perspectives, quantifying environmental impact potentials of the processes under study and providing insights of potential trade-offs between different environmental impacts (Lam et al., 2020). This methodology was standardized in the International Organization for Standardization (ISO) series UNE-EN ISO 14040/14044, defining the steps to follow to develop a LCA; this steps correspond to the goal and scope definition, where system boundaries, functional unit (FU), analysed scenarios and data quality are presented, followed by the life cycle inventory (LCI), that gives a description of material and energy flows within the product system and especially its interaction with the environment, consumed raw materials and emissions to the environment, then the life cycle impact assessment (LCIA), where the results of LCI are translated into the potential contributions to the selected environmental impact categories for the assessment (e.g., global warming, acidification) (Finnveden & Potting, 2014). Finally, the interpretation of the results would be the last step of the methodology; the results obtained in the previous sections are evaluated taking into consideration the goal and the scope firstly defined, with the aim to reach conclusions and recommendations, for which it is necessary a major contribution analysis, and the corresponding sensitivity and uncertainty analyses.

Life Cycle Costing (LCC) can be considered as the equivalent of LCA, but in economic terms. Thus, this method refers to an analysis method which encompasses all costs associated with a product, technology or process over its whole life cycle (UNEP SETAC Life Cycle, 2001). It can start to address the economic dimension of sustainability by estimating capital, operational and maintenance costs, as well as costs related to upstream and downstream processes (Guest et al., 2009). Although LCC and LCA are different context-specific assessment techniques, combining them may be a good decision in order to evaluate different alternatives (Lam et al., 2020).

This work aimed to assess both the nutrient (N and P in form of ammonium salts and struvite) recovery technologies implemented in Murcia–Este WWTP by LIFE ENRICH project and the application of these nutrients as fertilizers on agricultural crops in environmental and economic terms, which is an innovative perspective since there are no studies of these characteristics available in the literature. The nutrient recovery strategies assessed involve pre-anaerobic and post-anaerobic digestion processes. Later on, the results obtained were proposed to be compared by means of the LCA and LCC methodology, to the whole process of manufacturing and agricultural application of a conventional fertilizer. All the necessary information and data gathered to carry out this study was provided by *Cetaqua Water Technology Centre*.

2. State-of-the-art

Some recent and different reviews about the topic have studied the LCA methodology applied on nutrient recovery from wastewater from several perspectives (Lam et al., 2020; Pradel et al., 2016; Sena & Hicks, 2018). In the case of Sena & Hicks, (2018), the review was focused explicitly on struvite precipitation from wastewater treatment, while Lam et al., (2020) focused on LCA studies that considered nutrient recycling from wastewater for agricultural land application.

The environmental assessment of P recovery technologies takes an important role in the improvement on the environmental performance with increased resource circularity. Thus, considering the limited research works present on the literature, further research is needed to establish a comprehensive knowledge of the environmental performance of struvite recovery processes (Rufí-Salís et al., 2020).

Various studies have been reported assessing post-anaerobic digestion processes in recent years. Linderholm et al., (2012) assessed the environmental impact of four ways to supply agricultural land with a phosphorous fertilizer, comparing mineral fertilizer, certified sewage sludge, struvite precipitated from wastewater and phosphorous from sludge incineration; based on a unit of mass of phosphorus (P_2O_5) applied to agricultural land as FU, sewage sludge direct application resulted the most efficient alternative in terms of energy and emissions of GHG. Struvite and digested sludge presented more favourable results, in terms of GHG emissions, than phosphorous from sludge incineration. Another study, Rodriguez-Garcia et al., (2014) applied LCA to compare the potential environmental implications of some technologies, among which was struvite crystallization process and other two related to N recovery. Using the mass of P removed from the sludge as FU, these technologies were first assessed individually in a pilot plant and then implemented in a full-scale WWTP, showing a much lower environmental impact for N recovery technologies than struvite crystallization in the pilot plant and not relevant differences when applied in the full-scale model, which suggested to assess N and P recovery technologies not individually but as a part of a WWTP. Moreover, Bradford-Hartke et al., (2015) compared two different struvite precipitation alternatives, one was the recovery from the dewatering stream resulting from the AD in a biological P removal WWTP and the other, the recovery on the reverse osmosis brine stream of a plant with biological nutrient removal. Struvite crystallization from the dewatered digestate reported net environmental benefits in most impact categories and offered lower potential values of toxicity than other recovered phosphorus products, while P recovered as chemical solids and struvite from brine gave a net environmental burden; however, the resources consumed were not offset by the avoided fertilizer. The study concluded that P recovery as struvite from brine may require additional resources such as supplemental ammonia and power use, which did not make the benefits of avoided fertilizer outweigh them as a general rule, being relatively small compared to the impacts related to the necessary operational changes at WWTP to P recovery technologies implementation. Despite P is a valuable resource, its recovery it is not always beneficial to the environment, thus this study highlights the importance of including multiple impact indicators such as global warming, ozone depletion, fossil fuel depletion, toxicity and salinity in the LCA in order to select the most appropriate P recovery method to achieve an environmental benefit. The study of Remy & Jossa, (2015) was based on the analysis of the potential environmental impacts of a number of selected processes for P recovery from sludge, liquor, or ash following the methodology of LCA; results confirmed that the amount of recovered P depends exclusively on each of the different recovery processes, as well as the energy demand and the associated environmental impacts.

In the specific context of this work, a limited number of studies reported an assessment based on pre-anaerobic digestion processes. Roldán et al., (2020) studied for the first time two different sludge line configurations aiming for P removal and recovery in an enhanced biological P recovery (EBPR) WWTP before AD and compared them with the classical configuration of the WWTP (i.e., without these technologies implemented), simulating them and analysing their costs and life cycles. Both configurations were based on the production of a PO_4-enriched stream; the first alternative from the sludge through elutriation and the second one by dynamic thickeners. The sludge line configuration based on the elutriation showed lower global warming impact values, as well as a reduction in uncontrolled P-precipitation. More recently, Sena et al., (2021) compared a WWTP with biological treatment with three other scenarios; the baseline case considered the classical operation of a WWTP, which was compared with a full-scale WWTP scenario after the implementation of a struvite recovery system and with the struvite recovery system case isolated from the rest of the processes in the plant in the full-scale WWTP scenario. Results showed an important difference in total impacts, being reduced considerably in when the struvite recovery technology is implemented, thus obtaining a better environmental performance of the whole plant. The isolated struvite recovery scenario results showed net environmental benefits in six of the ten considered impact categories; balancing the positive and negative impacts, independent struvite recovery system was concluded to be neutral.

On the other hand, there are no studies that have environmentally assessed both the struvite recovery and its application on agricultural crops as fertilizer but there are some studies that assessed its performance as a fertilizer, concluding that it is comparable to that of conventional fertilizers and reducing the heavy metal application to soil in a significant way (Rufí-Salís et al., 2020). Liu et al., (2011) evaluated struvite from swine wastewater as a slow-release fertilizer for maize production in a successful way. No significant differences were found when comparing the maize produced from recovered struvite to the obtained from a commercial fertilizer. Antonini et al., (2012) assessed a selection of six urine-derived struvite fertilizers as phosphorous source for plants (i.e., ryegrass, maize) in terms of composition and effectiveness, concluding that urine-derived struvite can be considered as a valuable compound for this purpose and when combined with other soil conditioners, it is an efficient fertilizer which covers the magnesium and more than a half phosphorus demand of crops. In addition, both struvite recovery and recycling studies can be found in the literature (Gell et al., 2011; Uysal et al., 2014), although as abovementioned, they did not include the environmental assessment but focused only on the struvite crop application performance.

Less studies have been developed on the LCC of P recovery context, and none of them included the assessment of its crop field application. In addition, these only deal with costs for post-digestion processes and technologies and are based on the costs related to infrastructure, energy, chemicals, personnel, maintenance, products, by-products and waste (Roldán et al., 2020). Egle et al., (2016) compared 19 different P recovery technologies from wastewater in economic, environmental and technical terms; depending on the specific circumstances, these techniques can be beneficial from an economic and technical perspectives. However, they cannot be considered much effective due to the recovery rates are relatively too low. The approaches that recover P from sewage sludge use to apply more complex technologies that have a good efficiency in front of heavy metals removal, although costs associated are high in comparison to other alternatives. Sewage sludge ash is presented as the most promising P source, making the ash-based recovery pathways a good option for P recovery; the recovery rate is relatively high and despite the costs are highly related to the final product purity requirements, they are relatively low. Nättorp et al., (2017) analysed, in terms of LCC, a selection of P recovery processes based on municipal sewage sludge, sludge liquor or sludge incineration ash as input material. CAPEX, material and revenue, energy and revenue and personnel costs contributed to the global cost of the different assessed processes. Considering all the different processes studied, the cost of P recovery per population equivalent represented at most 3% of the wastewater disposal cost.

3. Materials and methods

3.1. Initial configuration of Murcia-Este WWTP

Murcia-Este WWTP initial configuration consists of primary settling, then biological removal of P is carried out by means of microorganisms called Phosphorous Accumulating Organisms (PAO), which are able to accumulate phosphates (PO_4^{3-}) in the biological reactor with an anaerobic–anoxic–oxic configuration (A_2O) with a total volume of 41,405 m^3 (3.9% anaerobic, 23.3% anoxic and 72.8% aerobic) and then, a secondary settling stage. The biological sludge, which is rich in P, is introduced in the sludge line to be thickened and it is mixed with the primary sludge to be introduced into the anaerobic digester, where the unspecific precipitation of struvite uses to take place; as commented before, this precipitation causes considerable maintenance problems on the digestor and on the dehydration centrifuges. Moreover, digested sludge dehydrability is increased, increasing its management price as waste. The sludge line counts with two primary gravity thickeners and two dissolver air flotation thickeners (DAF) for waste-activate sludge (WAS), being only one DAF currently used. Thickened WAS is electromagnetically hydrolysed and mixed with the primary sludge in a 79.2 m^3 mixing chamber and then digested and dewatered to a final concentration of 20–25% of TSS (Roldán et al., 2020).

N is partially removed in the biologic reactor by nitrification and denitrification processes, being emitted to the atmosphere as atmospheric nitrogen (N_2). A fraction of the non-removed N is accumulated in the biologic sludge in form of organic nitrogen, following the same route as P and giving as well the precipitation of struvite when being released from the anaerobic digester as ammonium (NH_4). The Process Flow Diagram (PFD) of the current configuration of Murcia Este WWTP is shown in *Figure 3.1*.

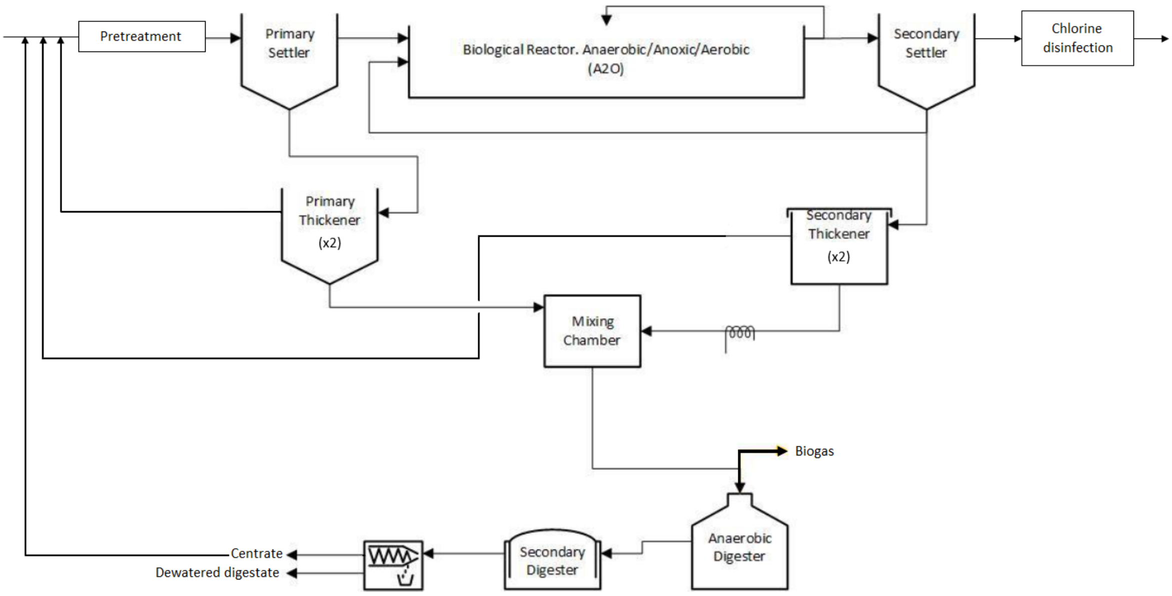

Figure 3. 1. PFD of the initial configuration of Murcia Este WWTP.

3.2. LIFE ENRICH nutrient recovery model configuration

The nutrient recovery model (*Figure 3.2*), proposed by LIFE ENRICH Project, is based on the implementation of a sludge line configuration which allows the elutriation process of P. This P is accumulated by PAO, giving Poly-P. During the elutriation process, a liquid stream is concentrated in the form of phosphates; apart from obtaining a phosphate rich stream, the phosphorous content in the sludge is reduced, avoiding the struvite unspecific precipitation in the biologic reactor. In a pilot-scale project, this phosphate concentrated stream is treated in a struvite crystallizer, where phosphates, ammonium and magnesium are precipitated in form of struvite crystals. Ammonium is contained in that stream, however magnesium could be limiting, so it might be needed to dose synthetic magnesium. The struvite crystals obtained could be applied either directly on the crop field or as a precursor to produce fertilizers.

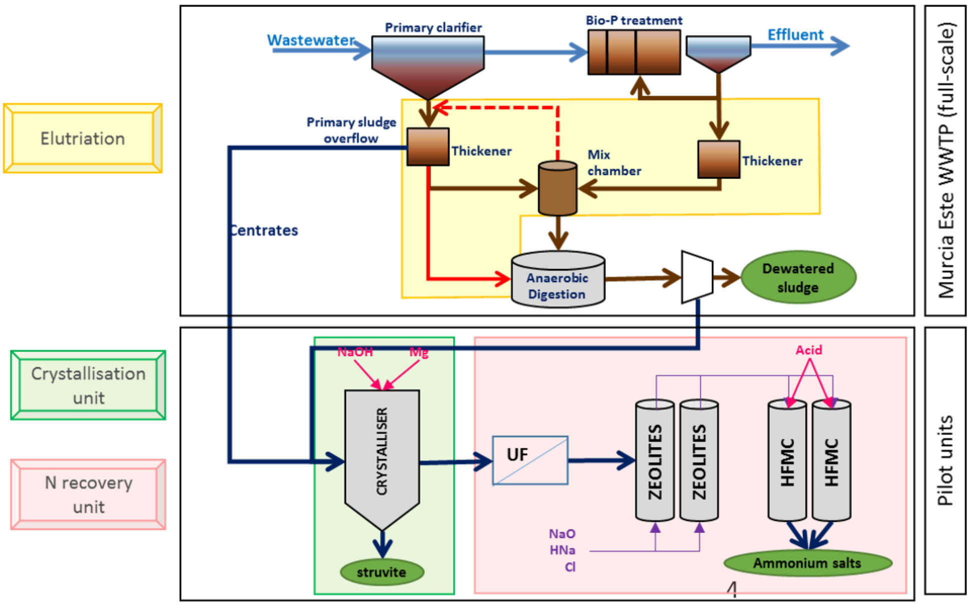

Figure 3. 2. Simplified PFD of the nutrient recovery configuration model proposed by LIFE ENRICH Project (Source: *Cetaqua Water Technology Center)*.

The ammonium recovery is carried out over the crystallizer effluent or over the dehydrated sludge centrates by means of zeolite columns through an ionic exchange stage, which concentrate the ammonium in a highly basic stream to be introduced into HFMC that allow the generation of an ammonium salt (i.e., ammonium nitrate NH_4NO_3) by means of counter flowing an ammonia gas stream with an acid stream, which can be also used either directly on the crop field or as a precursor to produce fertilizers. A more detailed PFD of the configuration model proposed by ENRICH for the nutrient recovery is shown in *Figure 3.3*.

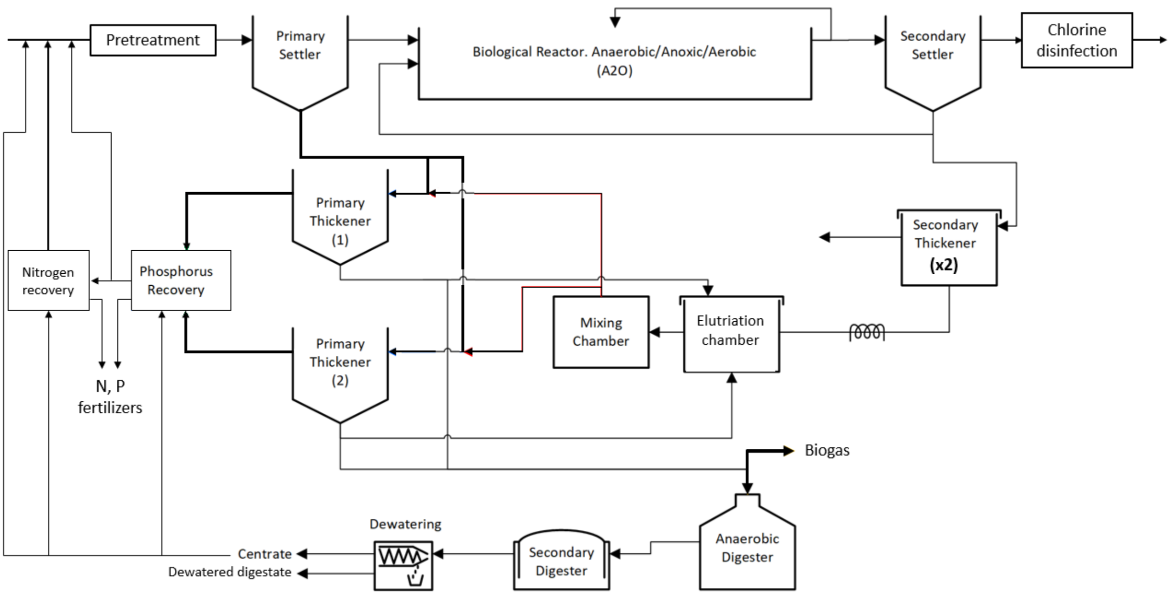

Figure 3. 3. Detailed PFD of the nutrient recovery configuration proposed by LIFE ENRICH Project.

3.3. Life Cycle Assessment

LCA methodology was applied in order to evaluate the potential emissions and energy demand required for the different nutrient recovery technologies from wastewater implemented in the initial configuration of Murcia-Este WWTP by LIFE ENRICH Project and the application of the nutrients recovered, which were N in form of ammonium nitrate (NH_4NO_3) and struvite (N and P), as fertilizers to the agricultural sector; this scenario was compared to the defined base scenario, which consisted of the agricultural application of a commercial mineral (inorganic) fertilizer, therefore obtained from phosphate rocks and Haber–Bosch processes. LCA was performed according to UNE-EN ISO 14040:2006 (ISO, 2006a, 2006b) and later on, to complete the analysis, an economic comparison between the two defined scenarios was carried out by means of a LCC assessment. The following sections describe the goal and the scope of this LCA, going into detail with the description of the different compared scenarios, the system boundaries and the FU. The life cycle inventory (LCI) was analysed describing the material and energy flows (inputs and outputs) considered within the system boundaries. Then, the impact assessment was developed and lastly, the results interpretation.

3.3.1. Goal and scope definition

The main objective of the LCA was to assess both the production and the agricultural application of N and P fertilizer as recovered products from wastewater in front of the ones of conventional inorganic fertilizers. The following subsections describe the system boundaries, the FU and the two scenarios defined for the assessment.

3.3.1.1. System boundaries

The defined system boundaries include the processes that are related to the sludge treatment (i.e., elutriation), anaerobic digestion, nutrient recovery and the agricultural crop application of them as fertilizers. Therefore, all the processes related to the water treatment line of the WWTP have been excluded of the scope of this LCA and primary and secondary sludge has been considered as the raw materials for the nutrient production.

The production of sludge-based N and P fertilizers, as Pradel & Aissani, (2019) reported, involves the environmental burdens of the wastewater treatment line allocated to sludge, where it is produced, and the environmental burdens of the sludge treatment and the following steps until the desired nutrients are recovered. The global sludge treatment process depends on the scenario considered, and is a chain of multiple steps; in this case, elutriation and thickening (*Figure 3.4*). The steps that follow the sludge treatment are the anaerobic digestion and dewatering by centrifugation, P recovery and N recovery. In the case of the mineral fertilizers, the system boundaries refer to the whole production process of these, and also includes the phosphate rock extraction in the case of P-based fertilizers and the Haber–Bosch processes for the production of N-based fertilizers.

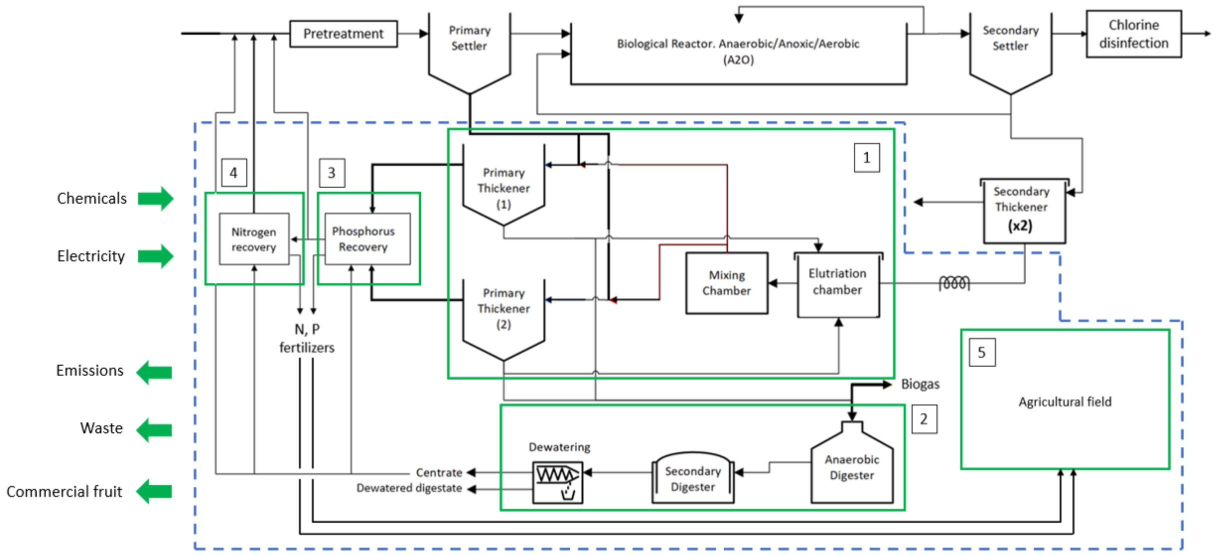

Figure 3. 4. System boundaries of the LCA. 1: Sludge treatment line; 2: Anaerobic digestion + centrifugation; 3: Phosphorus recovery by struvite crystallization; 4: Nitrogen recovery as ammonium salts; 5: Agricultural crop application.

As the study is based on a WWTP located in Murcia, Spain, this is the geographical scope of this assessment. As temporal reference, this LCA is related to the year 2019. Each process was characterized by *Cetaqua Water Technology Centre* by computing the corresponding mass balances, nutrient requirements and required fertilized area, as well as all the external system inputs, that refer to energy consumption and chemicals and the outputs, which correspond to the emissions to air, water and soil.

3.3.1.2. Functional unit

Due to the great importance that it has, the results of the LCA are largely influenced by the selection and definition of the FU (Sena et al., 2021). (Lam et al., 2020) reviewed 65 LCA studies that considered nutrient recycling from wastewater for agricultural land application. The amount of wastewater treated (influent/effluent), the amount of sludge disposed and the amount of nutrient recycled or removed were the three major types of FU among all of these nutrient recovery related LCA studies. As a novelty, this work is mainly focused on the recycled nutrients performance as fertilizers once they are recovered from wastewater. Thus, FU must be directly related to the commercial fruit production on the crop field. Since the baseline scenario is proposed to be a business-as-usual option to the nutrient recovery scenario, the commercial production is the same per m^2 of soil, so the production is proposed to be the same in both scenarios. In this study, the FU that has been chosen is the area of soil fertilized by P, therefore “1 m^2 of P-fertilized soil”.

3.3.1.3. Description of scenarios

3.3.1.3.1 Baseline scenario

The baseline scenario refers to the reference scenario that has been defined to be compared to the other defined scenario, in which nutrient recovery technologies are implemented in an existing WWTP for recovering N and P to be used as fertilizers for the agricultural sector, specifically for the production of tomatoes.

The reference scenario considers the factors that are directly and indirectly related to the application of a commercial solution (i.e., conventional fertilizers) the aforementioned purpose, treating the same area as defined in the nutrient recovery scenario. The agricultural crop application was performed using monopotassium phosphate (MKP), potassium nitrate (KNO_3), potassium sulphate (K_2S), nitric acid (HNO_3) and calcium nitrate ($Ca(NO_3)_2$) as fertilizers. In the case of P-fertilizers, as it is common in LCA studies, phosphate rock was considered as a natural resource and then, environmental impacts of its extraction and the processes related to the specific production of each of them were considered. The industrial processes for the production of N-based fertilizers, thus referred to the Haber–Bosch process, are also considered in this scenario. As the idea was to cover the same fertilized area (i.e., same commercial fruit production) as in the case of the nutrient recovery scenario but employing commercial solutions, the water supply from the water supply network for crop irrigation was considered as equal and therefore, not considered in either of the two assessed scenarios since the results are not affected by this factor. The same reason leads the LCA to not considering the crop wastes. Other consumptions such as fuels and electricity were not considered for the crop field application and thus, not considered in this scenario. Thus, the block diagram for the baseline scenario is shown in *Figure 3.5*.

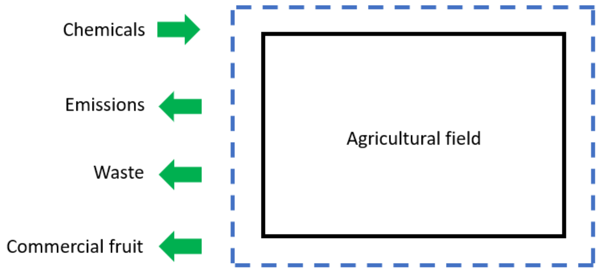

Figure 3. 5. Baseline scenario block diagram.

3.3.1.3.2 Nutrient recovery model configuration (ENRICH Model)

In this scenario, N and P are recovered from Murcia-Este WWTP, whose initial configuration is described in section 3.1 and which is located in Murcia, Spain. The influent of the WWTP, therefore the starting point for the sludge-based nutrient recovery, is 99,124 m^3 per day. As the sludge was considered as raw material for the nutrient recovery, impacts and fuel and energy consumptions related to the water treatment line, composed by the corresponding pre-treatment and the primary treatment, were considered out of the scope of the study.

The process for N and P recovery involves five units (*Figure 3.4*), whose environmental impacts, as well as energy and chemicals consumptions, were considered in the LCA. The sludge is treated in an elutriation process, which is composed by a primary and a secondary thickening stage before entering to the elutriation chamber, where a liquid stream is concentrated in the form of phosphates and sent to the struvite recovery stage, which consists of a struvite crystallizer, without passing through the AD process. The sludge stream is transferred to the AD and centrifugation stage, where it is dehydrated. The resulting dewatered digestate is considered as a waste, while the dehydrated sludge centrates is sent to the nitrogen recovery unit, where it passes through an ionic exchange stage by means of zeolite columns and introduced to a HFMC to obtain the final N-based recovered fertilizer in form of ammonium salts, being ammonium nitrate in particular. The recovered struvite and ammonium nitrate are applied as fertilizers in the crop field. The different units involved in the nutrient recovery process and crop field application are shown in the block diagram exposed in *Figure 3.6*. They have been considered as black boxes with material, chemicals and energy inputs and outputs.

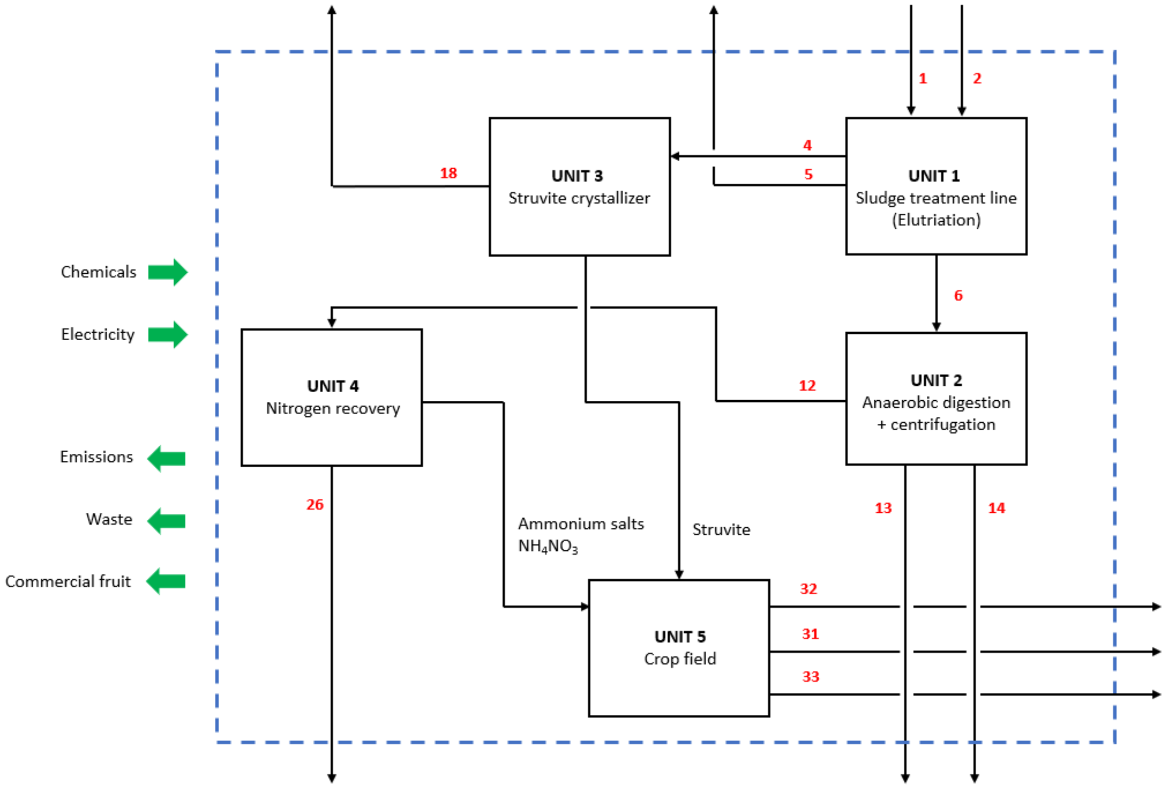

Figure 3. 6. Nutrient recovery scenario block diagram.

Due to the flocculant properties that polyelectrolytes present, its dosage in sludge treatment is widely used for enhancing the dewaterability of the sewage sludge prior to dewatering (Thapa et al., 2009). Referring to anaerobic digestion, since it is a process that produces odorous gases, ferric chloride is commonly added to the anaerobic digestion influent to mitigate these odours; moreover, it is proven that this additive also improves the anaerobic digestion performance (Park & Novak, 2013). As magnesium could be a limiting reactant in the precipitation of struvite, this process is performed by using magnesium dichloride as synthetic Mg, as well as sodium hydroxide with the aim of increasing the pH. During ammonium nitrate recovery, the dehydrated sludge centrates passes through an ultrafiltration (UF) membrane stage before the ion exchange step by using zeolites, where sodium hydroxide and sodium chloride pellets are applied. The resulting ammonia gas is counter flowed with nitric acid in a HFMC, obtaining the final ammonium salts. In this scenario, the crop field application differs from the baseline scenario mainly on the use of the recovered fertilizers from wastewater. Nitric acid, potassium sulphate and calcium nitrate are also applied by the same way and for the same purpose than in the baseline scenario but in lower quantities due to the substitution of struvite and ammonium nitrate by the commercial MKP and potassium sulphate, so as in the baseline case, the industrial processes for the production of these fertilizers is considered in the study. The crop field area to be treated per day was computed in this scenario and defined in the baseline case with the aim to be covered by means of commercial solutions. Considering the P amount required to fertilize 1 m^2 of soil for tomato production and with the total amount of P present in the recovered struvite, crop field area to be treated by the fertilizers was determined.

Energy consumptions related to the different processes were considered. In the case of the anaerobic digestion and centrifugation, the electricity requirements for the consumptions of sludge thickening, digestor pumping, primary decanter and anaerobic digestor were considered. Biogas generated during AD was considered in the study, while the electrical consumptions of struvite and nitrogen recovery units were also taken into consideration within the scope of the LCA.

3.3.2. Life Cycle Impact assessment

Once all the necessary information for the LCI was gathered, the two aforementioned scenarios were modelled using the LCA software OpenLCA v1.10.2 and the ecoinvent v3.6 database, which offers the possibility of balancing all the different mass and energy balances. As seen in *Figure 3.6*, the nutrient recovery scenario global process was divided into 5 different units, that were modelled and linked by means of the aforementioned LCA software. Each scenario was environmentally assessed through the quantification of its potential environmental impacts. The selected categories for the LCIA and the units to represent them are shown in *Table 3.1*.

Table 3. 1. Selected environmental impact categories for the LCA.

Indicator	Unit
Acidification	molc H^{+}_{eq}
Climate change	kg CO_{2eq}
Freshwater ecotoxicity	CTU_e
Freshwater eutrophication	kg P_{eq}
Human toxicity, cancer effects	CTU_h
Human toxicity, non-cancer effects	CTU_h
Land use	Kg C deficit
Marine eutrophication	Kg N_{eq}
Mineral, fossil & renewable resource depletion	kg Sb_{eq}
Ozone depletion	kg CFC -11_{eq}
Particulate matter	kg $NMVOC_{eq}$
Terrestrial eutrophication	mol N_{eq}
Water resource depletion	m^3 $water_{eq}$

3.4. Life Cycle Costing

In this work, LCC was carried out for each of the scenarios compared. The considered OPEX cost was associated to the chemicals consumed for the processes involved in both scenarios, and the working staff cost and energy consumption for operation and maintenance of the primary thickeners, anaerobic digester including the maintenance cost associated to struvite uncontrolled precipitation and struvite and N (in form of ammonium nitrate) recovery units was considered for the ENRICH scenario, which included the nutrient recovery technologies. The biogas generated in the anaerobic digestion was assumed to be used for substituting the equivalent energy consumption in the plant, so the equivalent cost was considered as a saving, and the whole amount of nutrients recovered was assumed to be applied in the agricultural application, having a neutral impact in the OPEX. CAPEX considered was only associated to this last configuration, considering the capital costs related to the elutriation process, which included the costs associated to the adaptation for prototypes installation, civil works, tanks and deposits, the corresponding equipment and flowmeters, the costs associated to the crystallizer for the struvite recovery and finally, the CAPEX associated to the ammonium recovery stage, which included the zeolites and HFMC. This CAPEX was calculated at a 2% annual depreciation rate and a project lifetime of 30 years. The annual equivalent CAPEX (AEC) was calculated according to *Eq. 3.1*.

$$AEC = CAPEX \cdot \frac{r(1+r)^t}{(1+r)^t - 1} \qquad \textbf{(Eq. 3.1)}$$

Where r is the annual depreciation rate and t the project lifetime.

The total annual equivalent cost (TAEC) was calculated by the sum of the annual CAPEX and the operational expenses (OPEX), according to *Eq. 3.2*.

$$TAEC = AEC + OPEX \qquad \textbf{(Eq. 3.2)}$$

The estimated specific costs for the OPEX and CAPEX calculations are shown in *Tables 3.2 and 3.3*, respectively.

Table 3. 2. Specific OPEX costs.

OPEX	Unit	
WWTP		
Electricity	€/kWh	0,09
Polyelectrolite	€/kg	2,15
$FeCl_3$ 40%	€/kg	0,18
Antifouling	€/kg	2,80
$MgCl_2$	€/kg	0,10
NaOH 50%	€/kg	0,52
NaCl (pellets)	€/kg	0,42
HNO_3 58%	€/kg	0,39
Sludge disposal	€/tn	20,82
Zeolites	€/kg	0,10
Staff	€/year	372.000,00
Maintenance and renewals	€/kg P in centrates	0,52
Crop field		
HNO_3 58%	€/kg	0,42
K_2S	€/kg	0,42
$Ca(NO_3)_2$	€/kg	0,50
KH_2PO_4	€/kg	1,28
KNO_3	€/kg	0,98

Table 3. 3. Specific CAPEX costs.

CAPEX	Unit		Supplier
Elutriation			
WWTP adaptation for prototypes installation	€	8.953,00	Instalaciones Hergasa 2006, S.L.
Civil works	€	2.720,30	Instalaciones Hergasa 2006, S.L.
Tanks and deposits	€	56.160,00	Toro Equipment, S.L.
Equipment	€	3.405,38	Logistium Servicios Logisticos
Flowmeters	€	3.334,11	Logistium Servicios Logisticos
Struvite recovery			
Crystallizer	€	1.395.000,00	-
Ammonium recovery			
Zeolites	€	256.000,00	-
Membrane contactors	€	500.000,00	-

4. Results and discussion

4.1. Life Cycle Inventory

In both scenarios compared, LCI input and output data were provided by *Cetaqua Water Technology Centre*, the company that carried out the LIFE ENRICH Project, and modelled using the ecoinvent v3.6 database. In the case of the nutrient recovery scenario, the data were extracted from the material and energy balances of LIFE ENRICH Project, which was developed in Murcia-Este WWTP. In the baseline scenario, the data were determined by fixing the same value of P-fertilized area and then fertilizing it employing commercial solutions. The main characteristics of the WWTP, nutrient recovery technologies and the crop field application are listed in *Table 4.1*.

Table 4. 1. Main characteristics of the WWTP, nutrient recovery technologies and crop field application.

	Unit	Value
Amount of wastewater treated	m^3/d	9.9124,33
Sludge production from wastewater treatment (dry basis matter)		
Primary sludge	m^3/d	4.449,00
Biological sludge	m^3/d	2.218,00
Centrates production from sludge dewatering	m^3/d	620,67
Biogas production from sludge anaerobic digestion[1]	Nm^3/d	8.161,77
Final products obtained by N and P recovery technologies		Struvite, Ammonium nitrate
Amount of struvite	kg/d	2.450,06
N_{TOT}	kg/d	310,01
P_{TOT}	kg/d	140,00
Amount of ammonium nitrate	L/d	3.423,00
N_{TOT}	kg/d	838,64
Area to be fertilized by P	m^2/d	16.316,16
Tomato commercial production	kg/m^2	10,00
	kg/d	16.3161,60

Primary sludge characteristics		
DQO	mg/L	3.592,00
N_{TOT}	mg/L	298,70
$N\text{-}NH_4$	mg/L	50,48
P_{TOT}	mg/L	37,76
$P\text{-}PO_4$	mg/L	5,72
TSS	mg/L	2.759,13
Biological sludge characteristics		
DQO	mg/L	8.504,00
N_{TOT}	mg/L	529,00
$N\text{-}NH_4$	mg/L	2,90
P_{TOT}	mg/L	201,00
$P\text{-}PO_4$	mg/L	13,20
TSS	mg/L	7.141,00
Centrates characteristics		
DQO	mg/L	758,40
N_{TOT}[2]	mg/L	1.105,32
$N\text{-}NH_4$	mg/L	921,10
P_{TOT}	mg/L	62,15
$P\text{-}PO_4$	mg/L	32,04
TSS	mg/L	2.612,00

[1]Biogas production is calculated based on the ratio Nm^3 of biogas:m^3 of centrates, which is set at 13,15.

[2]N_{TOT} content in centrates is calculated based on the ratio N_{TOT}:$N\text{-}NH_4$ content, which is set at 1,2.

Four different categories of streams were defined, corresponding to material, chemicals added, energy and waste streams. For the nutrient recovery scenario, it was needed to model material, chemicals, energy and waste streams; material flows were the ones resulting from the WWTP performance, chemical flows consisted of the chemicals needed to be added to the different units to carry out the different processes, energy flows were related to the electricity consumption and to the biogas generated during the anaerobic digestion and finally, only one waste stream was considered, corresponding to the dewatered digestate resulting from the centrifugation after the anaerobic digestion. On the other hand, as baseline scenario consists of the direct application of commercial chemicals (i.e., fertilizers) to the crop field, it was not necessary to model material and energy flows, but only chemical flows. Since the area to be fertilized is the same in the baseline scenario as in the nutrient recovery scenario, waste streams resulting from the crop field application are considered equal in both scenarios compared, as well as irrigation water and therefore, not included in the scope of the study.

Table 4.2, *4.3*, *4.4*, *4.5* and *4.6* summarize LCI material inputs and outputs, chemical inputs, LCI energy flows and LCI waste flows, respectively.

Table 4. 2. LCI material inputs for the nutrient recovery scenario. Stream numbers according to Figure 3.6. Values presented per FU: 1 m^2 of area P-fertilized.

Material Inputs	Unit	Sludge treatment line (Elutriation)		Anaerobic digestion + centrifugation	Struvite crystallization	Nitrogen recovery
Stream	-	1	2	6	4	12
Definition	-	Primary sludge	Biological sludge	Anaerobic digestor inlet	SN thickener 1	Centrates
Total flowrate	m^3	2,73E-01	1,36E-01	4,29E-02	2,60E-01	3,80E-02
COD	mg	9,79E+05	2,32E+06	1,84E+06	2,14E+05	2,88E+04
N_{TOT}	mg	8,14E+04	1,44E+05	1,27E+05	2,40E+04	1,20E+00
$N-NH_4$	mg	1,38E+04	7,91E+02	5,19E+04	1,78E+04	4,20E+04
P_{TOT}	mg	1,03E+04	5,48E+04	1,23E+04	2,47E+04	3,50E+04
$P-PO_4$	mg	1,56E+03	3,60E+03	8,58E+03	2,38E+04	2,36E+03
TSS	mg	7,52E+05	1,95E+06	1,57E+06	1,38E+05	1,22E+03
Mg	mg	-	-	-	2,30E+04	9,94E+04

Table 4. 3. LCI material outputs for the nutrient recovery scenario. Stream numbers according to Figure 3.6. Values presented per FU: 1 m^2 of area P-fertilized.

Material outputs	Unit	Sludge treatment line (Elutriation)	Struvite crystallization	Nitrogen recovery
Stream	-	5	18	26
Definition	-	SN thickener 2	Crystallizer effluent	Zeolite's effluent
Total flowrate	m^3	1,05E-01	2,60E-01	3,80E-02
COD	mg	1,32E+04	2,14E+05	1,98E+04
N_{TOT}	mg	2,11E+03	1,54E+04	7,99E+03
$N-NH_4$	mg	5,48E+02	9,23E+03	5,71E+03
P_{TOT}	mg	5,58E+02	5,74E+03	2,36E+03
$P-PO_4$	mg	4,00E+02	4,75E+03	1,22E+03
TSS	mg	1,37E+04	1,38E+05	2,74E+03
Mg	mg	-	4,82E+03	-

Table 4. 4. LCI chemical inputs. Values presented per FU: 1 m^2 of area P-fertilized.

Chemical Inputs	Unit	Nutrient (N, P) recovery	Baseline
Sludge treatment line (Elutriation)			
Polyelectrolyte	kg	1,38E-02	-
Anaerobic digester + centrifuges			
$FeCl_3$ 40%	kg	4,19E-02	-
Antifouling	kg	3,49E-04	-
Struvite crystallization			
$MgCl_2$	kg	4,36E-03	-
NaOH 50%	m^3	2,18E-05	-
Nitrogen recovery			
NaOH 50%	m^3	1,23E-05	-
NaCl (pellets)	kg	3,96E-02	-
HNO_3 58%	L	1,38E-01	-
Zeolites	kg	7,61E-02	-
Crop field			
Struvite	kg	1,50E-01	-
N_{TOT}	kg	1,90E-02	-
P_{TOT}	kg	8,58E-03	-
NH_4NO_3	kg N	5,14E-02	-
HNO_3 58%	L	1,45E-01	1,92E-01
K_2S	kg	3,29E-01	6,40E-02
$Ca(NO_3)_2$	kg	7,70E-02	7,40E-02
KH_2PO_4	kg	-	8,30E-02
KNO_3	kg	-	2,28E-01

Table 4. 5. LCI energy flows. Values presented per FU: 1 m^2 of area P-fertilized.

Energy flows	Unit	Nutrient (N, P) recovery
Anaerobic digester + centrifuges		
Electricity for thickening and digestor pumping	kWh	8,56E-02
Electricity for primary decanter and digester	kWh	2,19E-01
Biogas production from sludge anaerobic digestion	Nm^3	5,00E-01

Struvite crystallization		
Electricity for crystallizer	kWh	8,02E-02
Nitrogen recovery		
Electricity for nutrient recovery unit	kWh	6,71E-02

Table 4. 6. LCI waste flows. Values presented per FU: 1 m^2 of area P-fertilized.

Waste flows	Unit	Nutrient (N, P) recovery	Baseline
Anaerobic digester + centrifuges			
Dewatered digestate			
Dry weight	kg	1,00E+00	-
N_{TOT}[1]	kg	4,95E-02	-
P_{TOT}[2]	kg	3,02E-03	-
Crop field			
Crop field waste	tn	-	-
Leachates	m^3	-	-

[1] N_{TOT} content in dewatered sludge is calculated based on the ratio kgN in digestate:kgN in centrates, which is set at 1,18.

[2] P_{TOT} content in dewatered sludge is calculated based on the ratio kgP in digestate:kgP in centrates, which is set at 1,28.

4.2. Life Cycle Assessment

The software OpenLCA v1.10.2 and the ecoinvent v3.6 database were used to quantify the environmental impacts and therefore, to compare both scenarios assessed in this study. All the values were normalized per 1 m^2 of P-fertilized soil area, and are shown in *Table 4.7*.

Table 4. 7. LCIA results for each scenario. All values are normalized per 1 m^2 of P-fertilized soil area.

Indicator	Unit	Nutrient (N, P) recovery	Baseline
Acidification	molc H^+_{eq}	1,93E-02	1,04E-02
Climate change	kg CO_{2eq}	2,43E+00	1,45E+00
Freshwater ecotoxicity	CTU_e	5,74E+01	2,68E+01
Freshwater eutrophication	kg P_{eq}	5,58E-02	3,80E-04
Human toxicity, cancer effects	CTU_h	2,03E-07	1,25E-07

Human toxicity, non-cancer effects	CTU_h	1,60E-06	4,36E-07
Land use	kg C deficit	4,06E+00	1,26E+02
Marine eutrophication	kg N_{eq}	1,94E-01	1,44E-03
Mineral, fossil & renewable resource depletion	kg Sb_{eq}	7,61E-05	6,16E-05
Ozone depletion	kg CFC-11_{eq}	2,25E-07	7,52E-08
Particulate matter	kg PM 2.5_{eq}	1,55E-03	7,90E-04
Photochemical ozone formation	kg $NMVOC_{eq}$	6,87E-03	3,91E-03
Terrestrial eutrophication	molc N_{eq}	4,33E-02	2,55E-02
Water resource depletion	m^3 $water_{eq}$	-3,34E-01	-3,40E-01

Table 4.7 clearly shows that nutrient recovery scenario environmental impacts were higher in almost all the impact categories. The only indicator for which this scenario presented lower impacts is to the land use category. Regarding the different stages of the nutrient recovery process, the primary and the biologic sludge obtained from the primary settler and secondary thickener, resulted the major contributors on the total impacts of this scenario on freshwater eutrophication (83% and 15%) and made the elutriation stage to contribute with a 97% of the freshwater eutrophication impacts, as represented in *Figure 4.1*. In this indicator, the other stages were almost negligible, only the nitrogen recovery unit amounted 1% and it was due to the centrates feed stream coming from the centrifugation process. Similar to freshwater eutrophication, marine eutrophication (*Figure 4.2*) impacts were affected by the primary and biologic sludge by a 64% and a 31%, respectively, and despite the 3% of the impacts were attributed to the wastewaster effluent of the struvite crystallizer, no more significant contributions resulted.

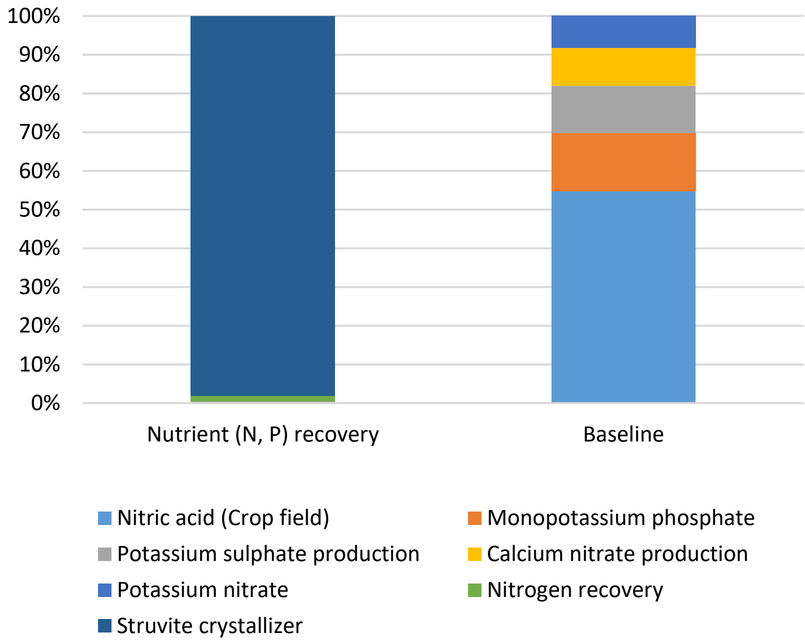

Figure 4. 1. Freshwater eutrophication main contributors of each scenario.

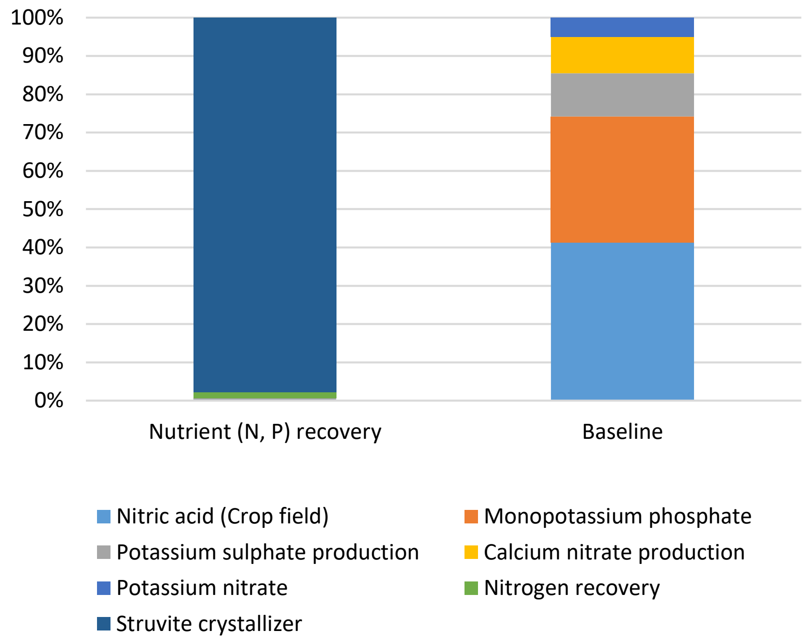

Figure 4. 2. Marine eutrophication potential main contributors of each scenario.

The anaerobic digestion and centrifugation stage contribution resulted almost negligible in all of the indicators evaluated; its higher contribution was a 2% of the total impacts on water resource depletion due to the iron chloride needed. Nitrogen recovery stage was the major contributor within the nutrient recovery process, followed by the struvite crystallization unit. In general, the relevance of nitrogen recovery unit among the other WWTP units on the total environmental impacts relayed in the nitric acid needed for this stage; acidification (19%), climate change (25%), photochemical ozone formation (21%) and terrestrial eutrophication (28%) were the impact categories most affected, as plotted in *Figures 4.3, 4.4, 4.5, 4.6*, respectively. Despite the significant contribution of this unit, the highest impacts were attributed to the crop field stage, which consisted of the application of the commercial inorganic fertilizers, being nitric acid and potassium sulphate the most significant ones due to the large amounts of nutrients directly applied on the agricultural soil and the emissions released from their industrial production processes.

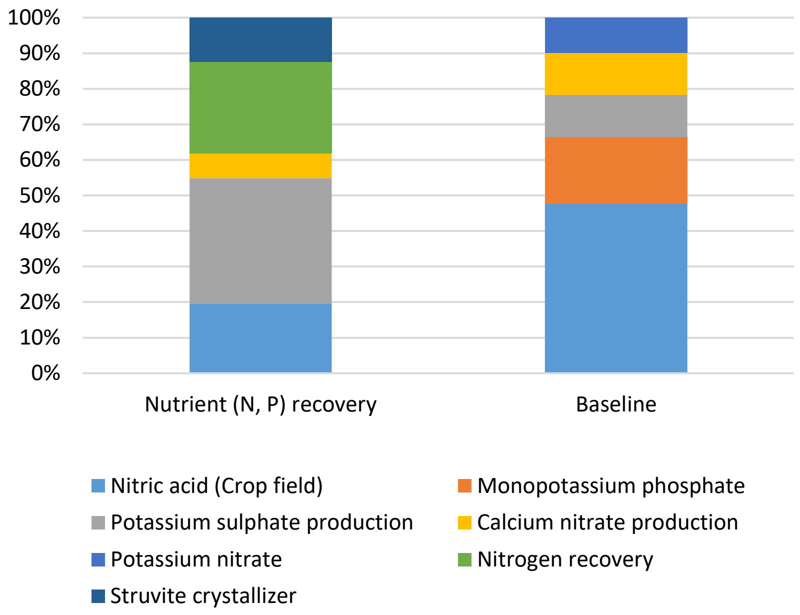

Figure 4. 3. Acidification potential main contributors of each scenario.

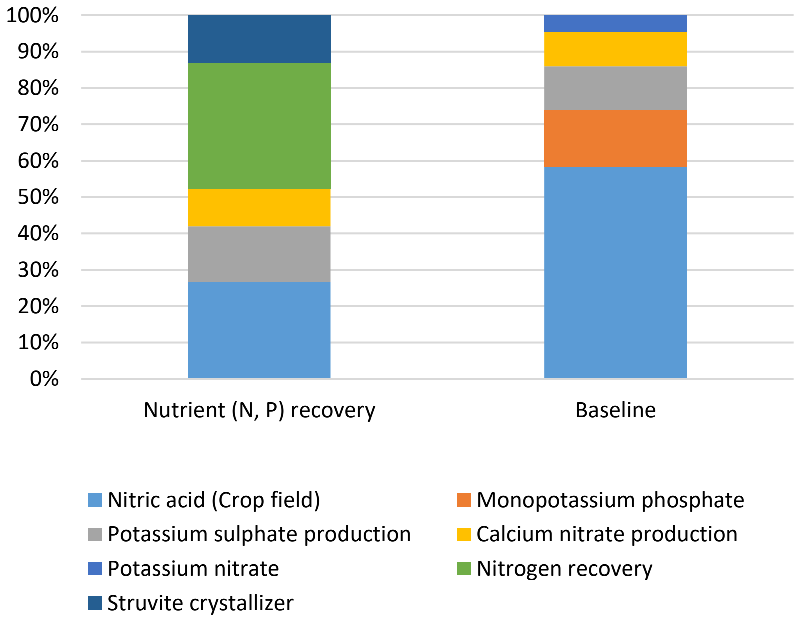

Figure 4. 4. Climate change potential main contributors of each scenario.

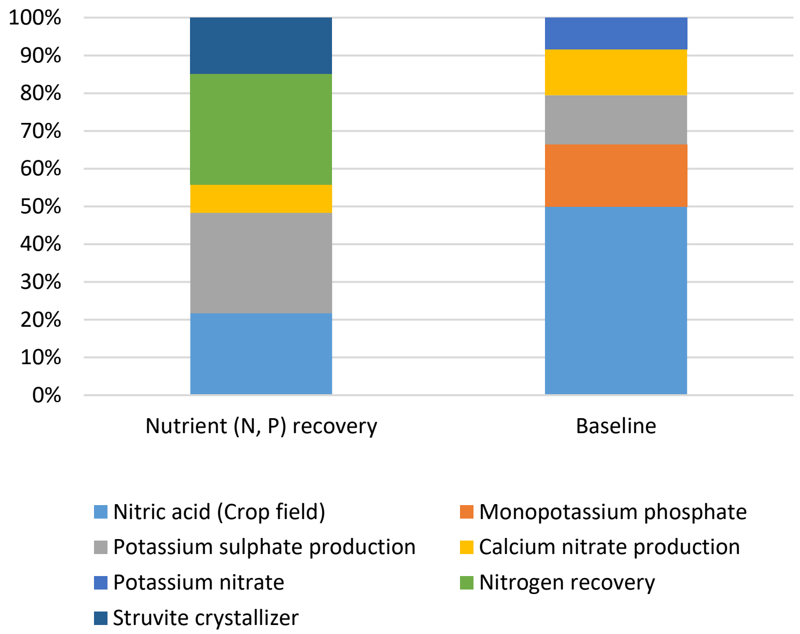

Figure 4. 5. Photochemical ozone formation potential main contributors of each scenario.

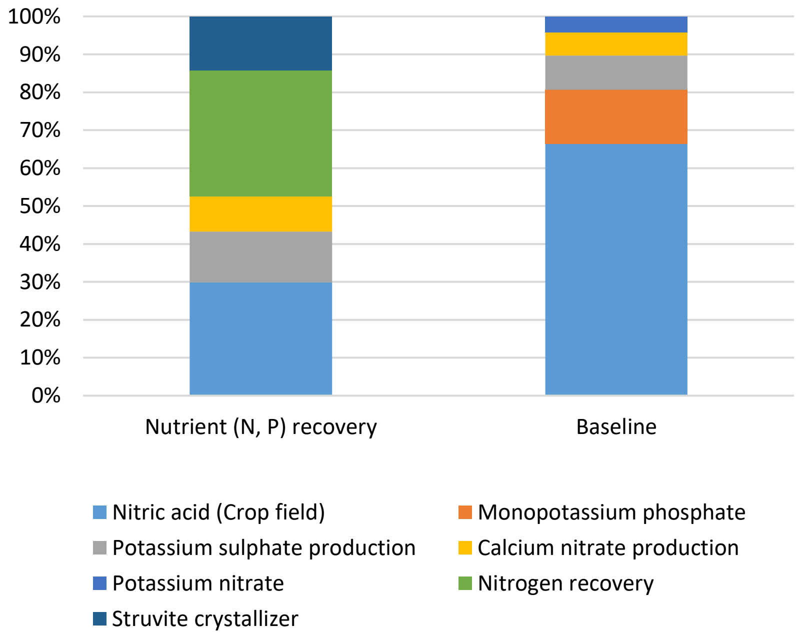

Figure 4. 6. Terrestrial eutrophication potential main contributors of each scenario.

Zeolite ion exchangers used for this stage had a relevant role in carcinogenic human toxicity impacts (42%), as shown in *Figure 4.7*, while in other impact categories in which nitrogen recovery was the major contributor within the nutrient recovery process, such as freshwater ecotoxicity (*Figure 4.8*), mineral depletion (*Figure 4.9*) and ozone depletion (*Figure 4.10*), the contributions of nitric acid and zeolites were similar (12 – 11%, 11 – 8%, 21 – 9%, respectively). As remarkable, as plotted in *Figure*

4.10, ozone depletion category was affected by the NaCl pellets used in the ammonium recovery unit with a 10% of the total impacts, being the highest contribution of these.

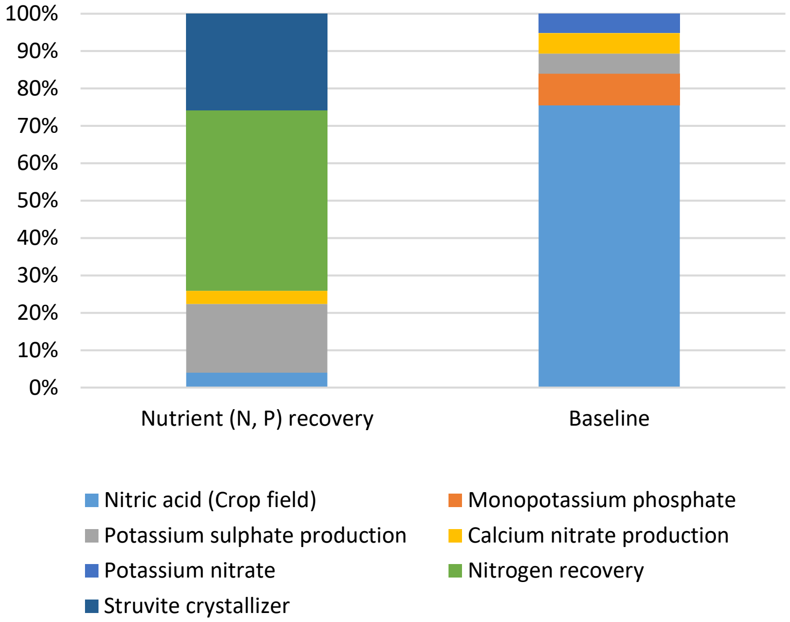

Figure 4. 7. Human toxicity (cancer effects) main contributors of each scenario.

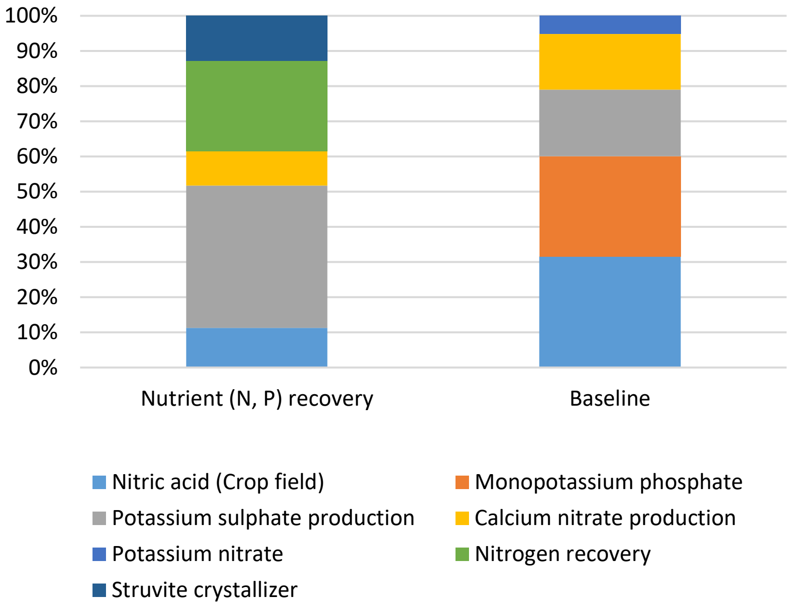

Figure 4. 8. Freshwater ecotoxicity potential main contributors of each scenario.

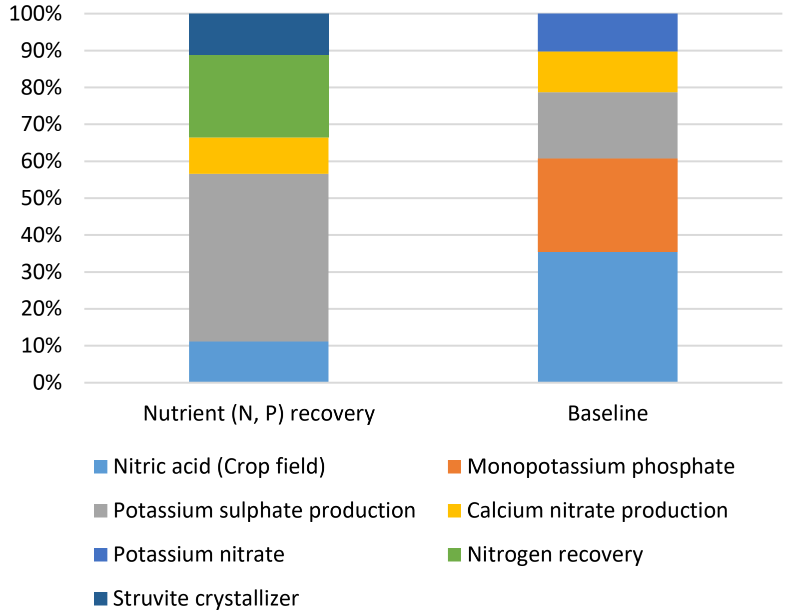

Figure 4. 9. Mineral depletion potential main contributors of each scenario

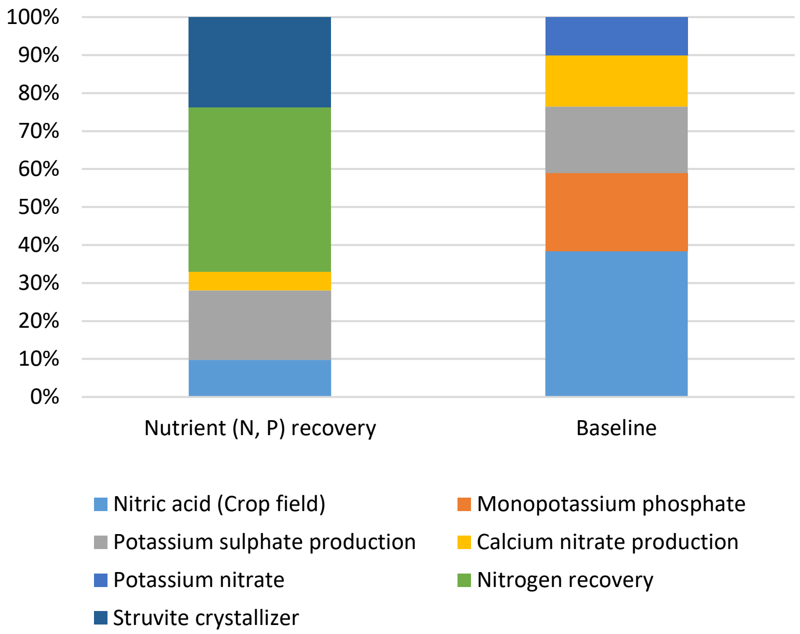

Figure 4. 10. Ozone depletion potential main contributors of each scenario.

Struvite recovery unit was the major contributor on the categories of non-carcinogenic human toxicity (51%), land use (32%) and marine eutrophication (98%), as *Figure 4.11, 4.12* and *4.2* show, mainly due to the wastewater effluent generated in the crystallizer with specific contributions of 49%, 29% and 95%, respectively. This late mentioned contributed, in a minor proportion, to other impact categories such as carcinogenic human toxicity (22%) (*Figure 4.7*) and ozone depletion potential (17%) (*Figure*

4.10). The presence of sodium hydroxide in the struvite crystallizer made this stage to contribute to water resource depletion category with a significant proportion (62%), as shown in *Figure 4.13*.

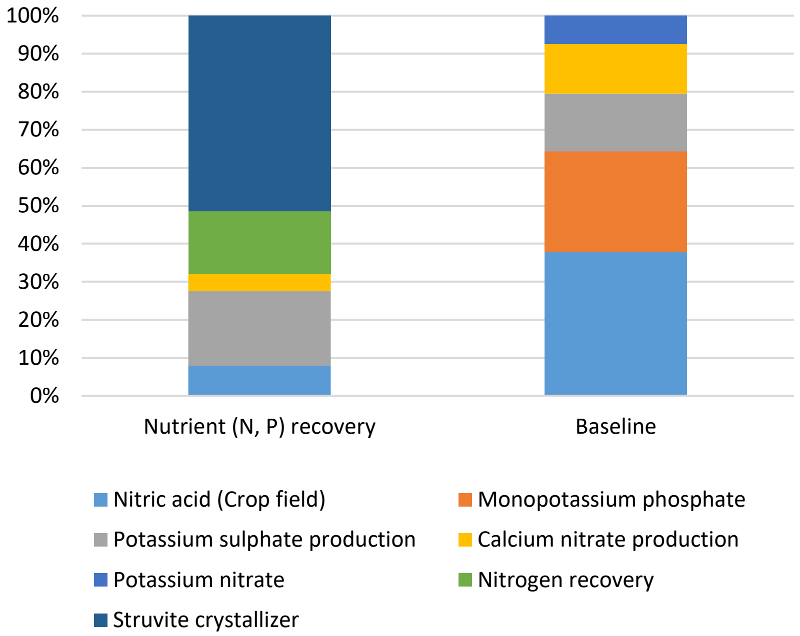

Figure 4. 11. Human toxicity (non-cancer effects) main contributors of each scenario.

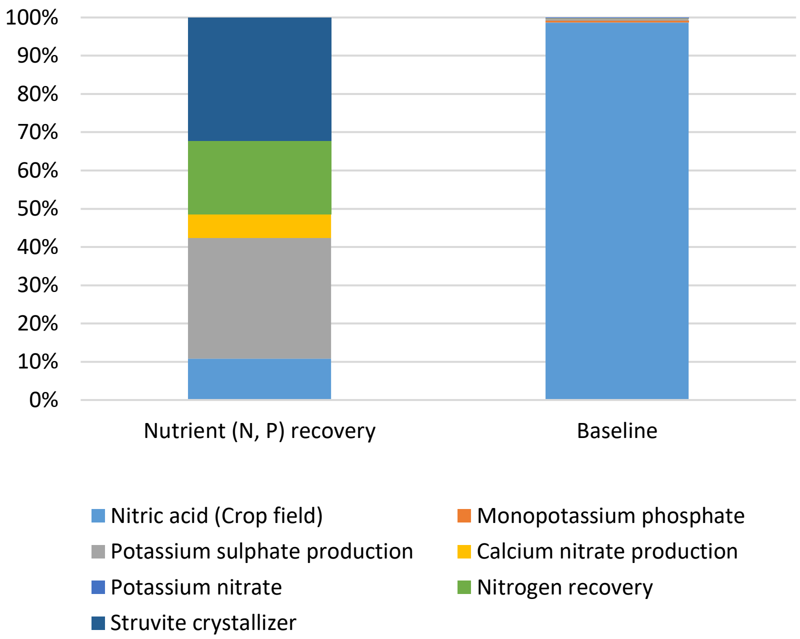

Figure 4. 12. Land use main contributors of each scenario.

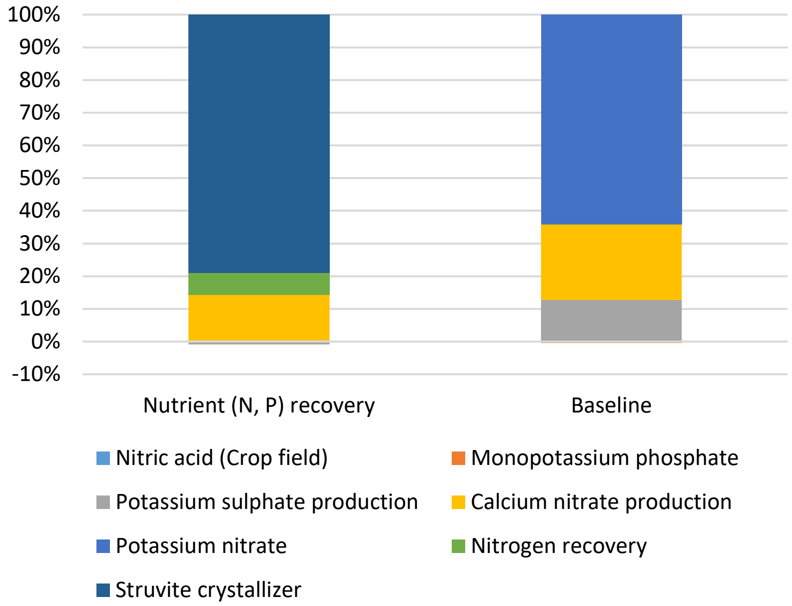

Figure 4. 13. Water resource depletion potential main contributors of each scenario.

The baseline scenario impacts were mainly affected by nitric acid, being the major contributor in the most of the impact categories selected. As plotted in Figure *4.12*, the major contribution of nitric acid in the baseline scenario resulted in the land use category (99%), followed, as most remarkable proportions, by the carcinogenic human toxicity category (75%), terrestrial eutrophication (66%) and climate change (58%), as shown in *Figure 4.7*, *4.6* and *4.4*, respectively. Environmental impacts related to monopotassium phosphate were also considerably relevant, mostly in marine eutrophication (33%) (*Figure 4.2*), freshwater ecotoxicity (29%) (*Figure 4.8*) and particulate matter emissions (26%) (*Figure 4.14*).

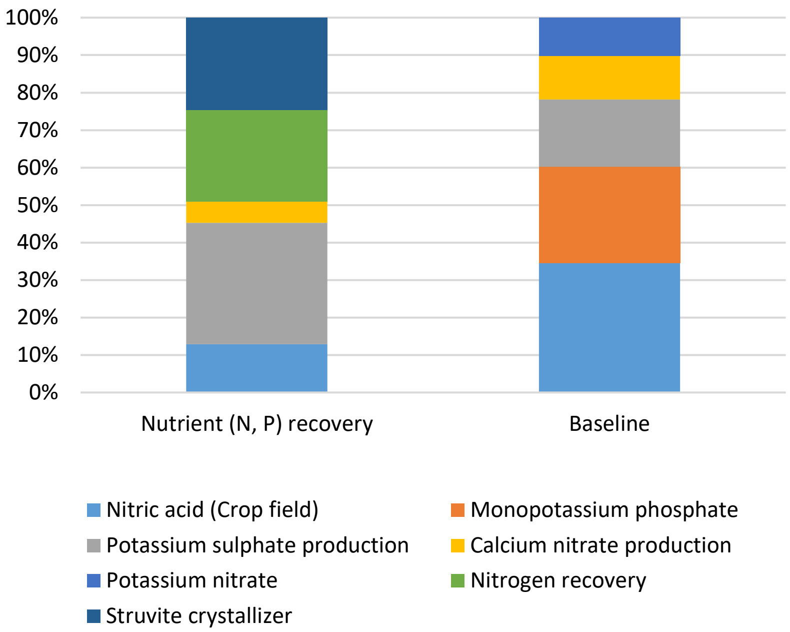

Figure 4. 14. Particulate matter emissions main contributors of each scenario.

Following monopotassium phosphate, potassium sulphate contributed its maximum on freshwater ecotoxicity (19%), mineral depletion (18%) and particulate matter emissions (18%) (*Figure 4.8, 4.9* and *4.14*, respectively). Calcium nitrate and potassium nitrate environmental impacts resulted in the lower contributions, being similar although the ones from calcium nitrate slightly higher than the attributed to potassium nitrate. Calcium nitrate major contribution was given in freshwater ecotoxicity indicator (16%), followed by ozone depletion (14%) and non-carcinogenic human toxicity (13%) (*Figure 4.8, 4.10* and *4.11*). As lowest impacts on this scenario, potassium nitrate major contributions were attributed to mineral depletion (10%), particulate matter emissions (10%) and acidification (10%) (*Figure 4.9, 4.14, 4.2*).

In nutrient recovery scenario, some indicators such as acidification (62%), freshwater ecotoxicity (61%) or mineral depletion (66%) were especially affected by the mineral fertilizers used in the crop field stage due to, as commented before, emissions of gases (e.g., NO_x) during the industrial production processes caused by the combustion of fuels (*Figure 4.3*, *4.8*, *4.9*). The amount of nutrients contained in these mineral fertilizers that was directly deposited on the agricultural soil had a great contribution on these categories. In other cases, as climate change (52%) (*Figure 4.4*), land use (49%) (*Figure 4.12*), particulate matter emissions (51%) (*Figure 4.14*), photochemical ozone formation (56%) (*Figure 4.5*) and terrestrial eutrophication (53%) (*Figure 4.6*), the contributions of the crop field stage were similar as the ones from the WWTP. The mineral fertilizers used in the crop field stage of this scenario were nitric acid, potassium sulphate and calcium nitrate, for which the amounts needed on this scenario, as seen in LCI results section (*4.1*), were lower than in the baseline scenario in the case of nitric acid and higher in potassium sulphate and calcium nitrate. The amount of potassium sulphate that was needed in this scenario was considerably higher than in the baseline scenario and from the other mineral fertilizers, which is reflected on the results as potassium sulphate was the major contribution among the total scenario impacts in acidification potential (35%) (*Figure 4.3*), freshwater ecotoxicity (41%) (*Figure 4.8*), land use (32%) (*Figure 4.7*), mineral depletion (45%) (*Figure 4.9*) and particulate matter emissions (32%) (*Figure 4.14*). By the other hand, the nitric acid used in the crop field stage contributed significantly to acidification (20%) (*Figure 4.4*), climate change (27%) (*Figure 4.4*), photochemical ozone formation (22%) (*Figure 4.5*) and terrestrial eutrophication (30%) (*Figure 4.6*). Calcium nitrate contributions were the lowest obtained, affecting the most to climate change (10%) (*Figure 4.3*), freshwater ecotoxicity (10%) (*Figure 4.8*) and mineral depletion (10%) (*Figure 4.9*).

Once analysed the results, since the impacts attributed to the nutrient recovery scenario were higher than it was expected, it would be needed to make some proposals in order to improve the nutrient recovery scenario. First, due to nitric acid contributed significantly in most of the evaluated environmental categories, requirements should be reduced either for the nitrogen recovery unit as for the crop field application. In the case of the crop field application, some alternative fertilizers may be used instead of nitric acid, as well as fertilizer mixtures, which would be a good option to reduce the nitric acid consumption requirements and therefore, nitrogenous gases emissions, such as NO_x, generated during its production process. Given the large amount of potassium sulphate needed in the crop field stage of this scenario and specifically the non-renewable resource depletion that it entails, the aforementioned would be applicable to the potassium sulphate requirements. Referred to the ammonium salts recovery unit, a reduction in the requirements of nitric acid appears as the only option since it is needed to produce ammonium sulphate when reacting with ammonia, varying the configuration of the nitrogen recovery unit can be considered as a real option to reduce and optimize the high consumptions of this product with the amount of ammonium nitrate recovered. Other important factor was the wastewater generated in the struvite recovery unit, for which a better management would be good to achieve; for this, changes in the configuration of the nutrient recovery treatment should be assessed. For example, the wastewater effluent generated could be sent directly to the ammonium nitrate recovery unit, an option that it is probable that would need to a higher consumption of nitric acid in that stage due to a larger influent to be treated, but it is an alternative that could be worth to prove and by this way, evaluate the changes in the LCA results.

4.3. Life Cycle Costing

Table 4.8 and *4.9* show OPEX and CAPEX results for each scenario. As aforementioned, total cost was divided into CAPEX and OPEX, considering 2% depreciation for 30 years and all the elements listed in the tables. Since the baseline scenario just consists in the direct agricultural application of commercial fertilizers, CAPEX was not considered in the cost calculation and therefore, it only affected to the nutrient recovery scenario.

Table 4. 8. OPEX results.

	Baseline		Nutrient (N, P) recovery	
	€/d	€/m^2	€/d	€/m^2
WWTP				
Electricity	0,00	0,00E+00	655,77	4,02E-02
Chemicals	0,00	0,00E+00	1.927,00	1,18E-01
Sludge disposal	0,00	0,00E+00	1.605,48	9,84E-02
Zeolites	0,00	0,00E+00	124,20	7,61E-03
Energy substituted by biogas	0,00	0,00E+00	-725,58	-4,45E-02
Struvite	0,00	0,00E+00	-857,52	-5,26E-02
Ammonium Nitrate	0,00	0,00E+00	-1.257,95	-7,71E-02
Subtotal	0,00	0,00E+00	1.471,40	9,02E-02
Staff	0,00	0,00E+00	1.019,18	6,25E-02
Maintenance and renewals	0,00	0,00E+00	20,10	1,23E-03
Total ENRICH Project	0,00	0,00E+00	2.510,69	1,54E-01
Crop Field				
Struvite	0,00	0,00E+00	857,52	5,26E-02
Ammonium Nitrate	0,00	0,00E+00	1.257,95	7,71E-02
Nitric acid (HNO_3) 58%	1.322,00	8,10E-02	998,39	6,12E-02
Potassium sulphate	439,83	2,70E-02	2.261,01	1,39E-01
Monopotassium phosphate	1.733,43	1,06E-01	0,00	0,00E+00
Potassium nitrate	3.642,71	2,23E-01	0,00	0,00E+00
Calcium nitrate	608,53	3,73E-02	633,20	3,88E-02
Total Crop Field	7.746,50	4,75E-01	6.008,06	3,68E-01
Total OPEX	7.746,50	4,75E-01	8.518,75	5,22E-01

Table 4. 9. CAPEX results.

	Baseline			Nutrient (N, P) recovery		
	€	AEC (€/year)	€/m^2	€	AEC (€/year)	€/m^2
Elutriation	0,00	0,00	0,00E+00	74.572,79	3.329,67	5,59E-04
Struvite recovery	0,00	0,00	0,00E+00	1.395.000,00	62.286,64	1,05E-02
Ammonium recovery	0,00	0,00	0,00E+00	756.000,00	33.755,34	5,67E-03
Total CAPEX	0,00	0,00	0,00E+00	2.225.572,79	99.371,65	1,67E-02

Once the results were normalized per m^2 of P-fertilized soil, the OPEX obtained in the baseline scenario was 0,475 €·m^{-2} in front of the 0,522 €·m^{-2} obtained for the nutrient recovery scenario, which indicated that OPEX incremented by a 10% in the case of the scenario of recovering nutrients from wastewater by implementing the different technologies described in previous sections. This trend on the OPEX was to be expected not only due to the chemicals and energy needed for the recovery of nutrients at the WWTP but also for the chemicals needed to the crop field application. Due to the recovery of nutrients to substitute commercial fertilizers in the agricultural field, the results obtained on the OPEX showed that this cost was lower in the case of the nutrient recovery scenario than in the baseline scenario, although it was compensated when considering the CAPEX related to the nutrients recovery at the WWTP making the total costs related to nutrient recovery scenario higher than the ones for baseline scenario. CAPEX obtained in the nutrient recovery scenario was 0,0167 €·m^{-2}, corresponding to a very low portion (3% approximately) of the TAEC (*Figure 4.15*).

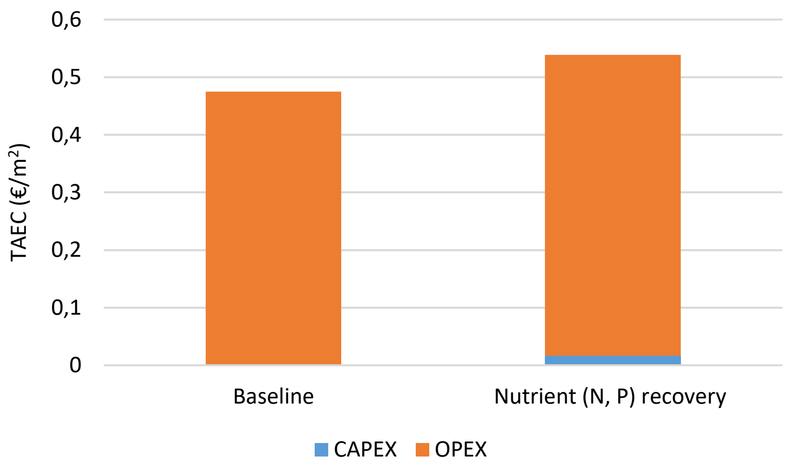

Figure 4. 15. Total annual cost estimations (TAEC) for the different scenarios compared per m^2 of P-fertilized soil.

The results obtained in the LCC analysis confirmed, as it was expected, that the nutrient recovery scenario composed by the stages of sludge management (elutriation), anaerobic digestion and centrifugation, struvite recovery, nitrogen recovery and crop field application presented the highest cost per m^2 of P-fertilized soil. However, the benefits related to the use of inorganic fertilizers represent a significant impact on the environment that will cause environmental externalities that should be included in the economic analysis. As shown in *Table 4.8*, baseline scenario's OPEX is mostly affected by potassium nitrate requirements, contributing to a 47% of the total OPEX, followed by monopotassium phosphate (22%) and nitric acid (17%). For this scenario, calcium nitrate and potassium sulphate were the minor contributors to the OPEX. Regarding the nutrient recovery scenario, 71% of the OPEX was attributed to the crop field stage, in which the consumption of potassium sulphate affected majorly due to the large amount required. The remaining 29% relayed on the nutrients recovery at the WWTP, where the consumption of chemicals and the sludge disposal after the anaerobic digestion and centrifugation stage contributed the most; among the chemicals used, nitric acid for the nitrogen recovery stage at the WWTP resulted in the highest economic impacts.

The CAPEX of the nutrient recovery scenario, as shown in *Figure 4.16*, was most affected by the cost of the struvite crystallization unit (63%), followed by the costs of the ammonium salts recovery unit (34%) and finally, the elutriation unit (3%).

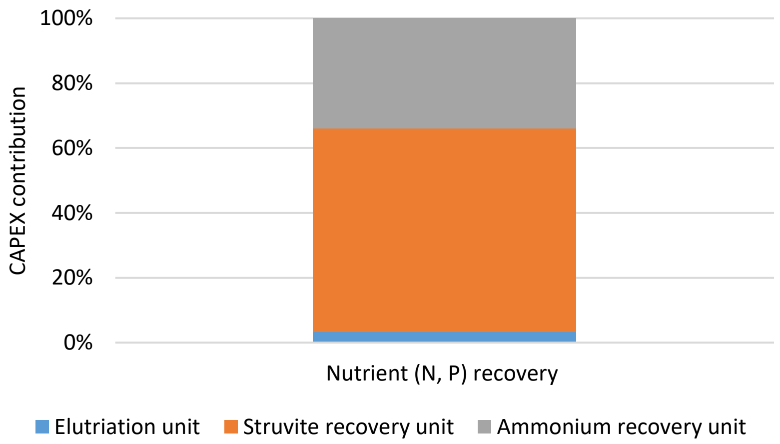

Figure 4. 16. CAPEX contributions of nutrient recovery scenario.

From the perspective of nutrient recovery at the WWTP, results indicated that the total cost taking into consideration both the OPEX, excluding the expenses of the crop field application, and CAPEX was 0,17 €·m^{-2}, which is less than the 32% of the total OPEX and CAPEX costs of nutrient recovery scenario assessed in this LCA (0,54 €·m^{-2}) (*Figure 4.17*).

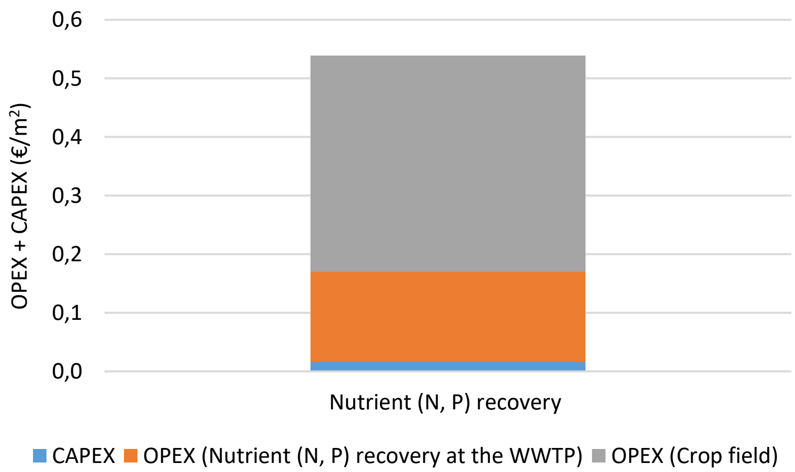

Figure 4. 17. OPEX and CAPEX of nutrient recovery scenario. Costs per m^2 of P-fertilized soil.

A seen in LCI results (*4.1*), the amounts of recovered fertilizers needed for the crop field application were 0,15 kg·m^{-2} for the struvite and 0,0513 kg·m^{-2} for the N contained in the ammonium nitrate, therefore 0,2013 kg·m^{-2} as total amount of fertilizers (*Figure 4.18*).

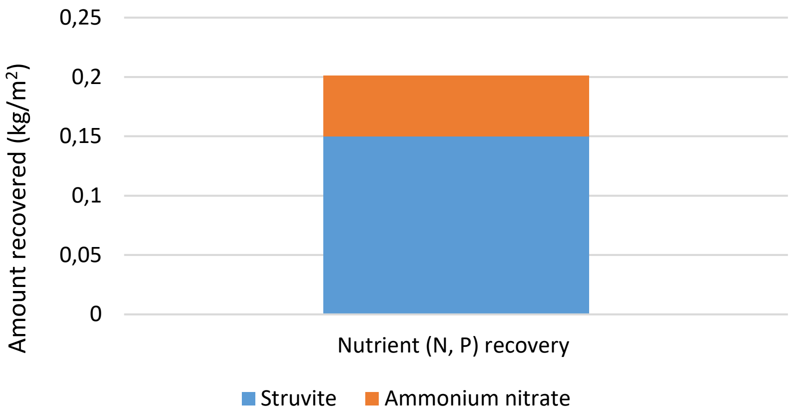

Figure 4. 18. Amount of nutrients recovered in the WWTP.

Establishing the total OPEX and CAPEX cost aforementioned as the cost to balance with the revenues from the fertilizers, it would be needed a total cost of 0,85€·kg^{-1} of fertilizers in order to balance the total nutrient recovery expenses. Considering that 75% of the fertilizer requirements are attributed to struvite and the remaining 25% is attributed to the N contained in ammonium nitrate, the minimum cost that would balance the total expenses derived from OPEX and CAPEX would be 0,63 €·kg^{-1} for the struvite and 0,22 €·kg^{-1} for the ammonium nitrate.

4.4. WWTP perspective

Despite the results showed that it is possible to implement the LIFE ENRICH nutrient recovery model in Murcia-Este WWTP, an economic and technical viability analysis would be needed in order to assess the feasibility of this project and optimize the process, if necessary, to make it more feasible; nutrient recovery yields and operational costs (OPEX) could be adjusted in order to find the optimal operation conditions. As seen previously, implementing the nutrient recovery technologies offers some benefits to the WWTP such as a reduction on the sludge disposal, energy savings due to biogas generation and a reduction or avoidance of the uncontrolled precipitation of struvite, that lead to a considerable reduction on the maintenance costs of the WWTP. Moreover, the investment costs (CAPEX) of these technologies must be taken into consideration. For the project to be worthwhile, it is important that the total expenses, thus OPEX and CAPEX, related to the nutrient recovery processes are, at least, balanced by the revenues obtained, which would be the ones generated from the recovered fertilizers. If the ENRICH project results are positive, a scale-up assessment could be a good option in order to analyse the possibility of a full implementation in the WWTP, or to expand the nutrient recovery technologies to other WWTPs.

On the other hand, the definition of a business plan would involve price, revenues, sales, logistics and potential distributors definition, taking into consideration the nutrient recovery costs and the feasibility study carried out previously, with the possibility of replicability of the model in other case studies. The stakeholders involved in the value chain involve nutrient producers, fertilizer companies, end-users and administrations, and raising awareness to all of them would be necessary in order to introduce them all the environmental and economic advantages or disadvantages that this proposed nutrient recovery model presents.

4.5. Social perspective

The object of the present study leads to a discussion or debate about the social acceptance of the final recovered product or derivatives, which are the fertilizers recovered from residues, in this case wastewater, that can be either the pure struvite and ammonium salts obtained or some derivate fertilizers manufactured from these late mentioned. Thus, the use of these alternative fertilizers or the consumption of the final product obtained in the crop field is a possible object of social rejection. This possible rejection can be given mainly by the facts that these nutrients are recovered from municipal waste, which is not a conventional source, and by the possibility of high content of pathogens or micropollutants such as heavy metals.

Considering the environmental aspects of the possibility of using and consuming the products obtained by the nutrient recovery model developed in the ENRICH project, it is clearly observable that this is a much more sustainable alternative for producing nutrients (N and P) in front of the possibility of phosphate rock reserves depletion or the use of the Haber-Bosch process, which is nowadays the main industrial process for the production of ammonia, with the sustainability and environmental drawbacks that these options entail. In economic terms and regarding the results obtained, as the cost of the nutrients as raw materials for fertilizer industries or directly for the farmers, which are the main potential end users, is expected to be higher than the one for conventional products, the resulting cost of the commercial agricultural products obtained through them would be higher. From the perspective of these final end consumers, both the environmental and economic aspects must be considered and put on a balance in order to decide if it is worth to acquire these recycled products. Despite the cost of conventional fertilizers is expected to be lower than the one of the fertilizers that are recovered from wastewater, the environmental impacts of its production process are much higher, that is why in order to raise awareness between the society about all the advantages and the added value that nutrient recycling presents and promote the transition to a circular economy model, communication activities would be useful to be carried out not only in an academic context, but to reach a more general and diverse audience.

5. Conclusions

This project aimed to evaluate the nutrient recovery process proposed by LIFE ENRICH Project and its application on the crop field using the LCA and LCC methodology and to compare the use of inorganic fertilizer for the same agricultural application as a baseline scenario.

Results showed higher impacts for the nutrient recovery scenario than for the baseline in most of the environmental categories. Due to the nitrogenous (NO_x) emissions from anthropogenic activities such as the combustion of fuels for the production of mineral fertilizers such as nitric acid, its contribution on the total impacts was significant either in the nitrogen recovery unit as in the crop field stage, while in the baseline scenario it was the major contributor for all the categories, followed by monopotassium phosphate, potassium sulphate, calcium nitrate and potassium nitrate. Struvite recovery stage followed the nitrogen recovery unit as the second major contributor within the nutrient recovery process being its influence relayed on the wastewater effluent generated in the crystallizer, that was the major contributor of this in most of the categories, especially in marine eutrophication and human toxicity indicators. Due to the large amount of nutrients contained in the sludge feed streams of the nutrient recovery system, proceeding from the water treatment line, elutriation unit presented the highest impacts on freshwater eutrophication. Not significant contributions resulted from anaerobic digestion and centrifugation. Despite the significant contribution of nitric acid and due to the large amount of potassium sulphate needed for the crop field stage of the nutrient recovery scenario, potassium sulphate showed the highest impact contributions in some categories such as acidification, freshwater ecotoxicity, mineral depletion or particulate matter emissions. Moreover, apart from climate change and terrestrial eutrophication impacts, in which nitric acid contributed the most among the mineral fertilizers, potassium sulphate was the most affecting inorganic fertilizer. LCA results suggested that some strategies would be needed to improve the environmental performance of the nutrient recovery scenario; these should be focused mainly on reducing nitric acid and potassium phosphate requirements at the crop field stage, as well as the requirements of nitric acid at the nitrogen recovery unit.

LCC results, as expected, showed higher costs for nutrient recovery scenario than for baseline scenario; it was important to take into consideration that the nutrient recovery scenario counted, additionally to the CAPEX, not only on the chemicals and energy consumptions of the nutrient recovery at the WWTP, but also por the chemicals needed for the crop field application.

Finally, the results obtained in this project offer the possibility of designing a business plan around the nutrient recovery model proposed by the LIFE ENRICH project and around the nutrients recovered for their use as fertilizers and by this way, promote the transition from a lineal economy model to a circular economy model. Different aspects such as the environmental, economic and technical must be considered for the business plan design. However, in order to move forward to the transition to a circular economy model, the social aspects that affect to the acquisition of these recovered products must be considered due to they are presented as object of debate about the social acceptance mainly due to its origin.

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