An integral approach to sludge handling in a WWTP operated for EBPR aiming phosphorus recovery: simulation of alternatives, LCA and LCC analyses.

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ABSTRACT

As phosphorus is a non-renewable resource mainly used to produce fertilizers and helps to provide food all over the world, the proper management of its reserves is a global concern since it is expected to become scarcer in the near future.

In this work we assessed two different sludge line configurations aiming for P extraction and recovery before anaerobic digestion and compared them with the classical configuration. This study has been performed by simulation with the model BNRM2 integrated in the software package DESASS 7.1. Configuration 1 was based on the production of a PO_4-enriched stream from sludge via elutriation in the primary thickeners, while Configuration 2 was based on the WASSTRIP® process and its PO_4-enriched stream was mechanically obtained with dynamic

thickeners. In both alternatives recovery was enhanced by promoting poly-phosphate (poly-P) extraction under anaerobic conditions, for which both configurations were fully evaluated in a full-scale WWTP. Both were also optimized to maximize phosphorus extraction. Their costs and life cycles were also analysed.

The novelty of this research lies in the lack of literature about the integral evaluation of pre-anaerobic digestion P recovery from wastewaters. This study included a holistic approach and an optimization study of both alternatives plus their economic and environmental aspects.

In Configuration 1, the PO_4-P load in the recovery stream reached 43.1% of the total influent P load and reduced uncontrolled P-precipitation in the sludge line up to 52.9%. In Configuration 2, extraction was 48.2% of the influent P load and it reduced precipitation by up to 60.0%. Despite Configuration 1's lower phosphorus recovery efficiency, it had a 23.0% lower life cycle cost and a 14.2% lower global warming impact per hm^3 of treated influent than Configuration 2. Configuration 1 also reduced the TAEC by 17.6% and global warming impact by 2.0% less than Configuration 0.

1. INTRODUCTION

In nature, phosphorus (P) is directly linked to life and biological activity as it provides energy to cells and is present in genetic material, bones and other structures (Smit et al., 2009, Van Dijk et al., 2016). P causes pollution in aquatic ecosystems after excessive use of P-fertilizers and the discharge of untreated wastewaters, inducing algal blooms, reducing the dissolved oxygen in the water column and triggering the formation of P-deposits in sediments (Kroiss et al., 2011). These issues are avoided by proper crop management and removing P in wastewater treatment plants (WWTPs). P is currently removed in WWTPs biologically (enhanced biological phosphorus removal, EBPR), chemically or a combination of both methods (Cornel and Schaum, 2009). The combination of EBPR in the water line and anaerobic digestion in the sludge line is a triggering factor for the uncontrolled precipitation of P-compounds inside and downstream digesters, causing pipe blockages and deposits on the walls of pipelines, reactors and other equipment (Martí et al., 2017).

The main use of P is in the food supply, as more than 80% of the mined product is used in the food industry. An increase in its demand is expected in the near future as living standards and the global population increase, but also the depletion of its reserves (Ciéslik and Konieczka, 2017; Robles et al., 2020). P and phosphate rock are thus included in the list of Critical Raw Materials (EC, 2017). Around 90% of the highest quality reserves occur in Western Sahara and Morocco, China, South Africa, Jordan and the USA, which adds extra complexity to its future supply of food, as the resource has geopolitical interests and is subject to market fluctuations. (Guedes et al., 2014).

Several initiatives are now under way in an attempt to alleviate its scarcity, including reuse and recovery within its cycle. Egle et al. (2016) considered that implementing recovery techniques in WWTPs could reduce Central Europe's dependence on external P-supplies by up to a 40%. These techniques contribute to the evolution from the WWTP to the wastewater resource recovery facility concept, in which wastewater is not considered as a mixture of pollutants whose discharge into surface waters has to be avoided, but a rich source of valuable products.

P forms organic or inorganic compounds in WWTPs, the organic fraction being up to 60% of total P in the sludge (Le Corre et al., 2009). Its mineralization requires long retention times and takes place quantitatively in anaerobic sludge digestion. Inorganic P appears as i) orthophosphate PO_4, which is easily separated and afterwards assimilated by plants, and ii) polyphosphate (poly-P), the solid compound that some organisms (poly-P accumulating organisms, PAO) store after PO_4 uptake.

The main P-recovery product is struvite ($NH_4MgPO_4 \cdot 6H_2O$, MAP), which is considered a slow-release fertilizer with low solubility and high P content (13% by mass). MAP crystallizes at a pH of 7.5 when the molar ratio $Mg:NH_4:PO_4$ is around 1.3:4:1 (Shih et al., 2017; Peng et al., 2018). Other recovered products as calcium phosphates have lower fertilizing potential (Robles et al., 2020).

Direct application of P-enriched sludge after stabilization involves a simple recovery technique, however it is forbidden in many countries due to the likely presence of heavy metals, persistent organic pollutants and pathogens (Harrison *et al.*, 2006). This, together with the emergence of P-recovery awareness, has triggered the research, development and full-scale implementation of a wide variety of P-recovery techniques from different streams after anaerobic digestion: from sludge (such as Airprex®, Nuresys and Elophos®) or centrifuge centrate (such as Naskeo®, Ostara Pearl®, PHOSPAQ® or STRUVIA™). Under acidic conditions more P can be recovered, as a part of the organic matter is mineralized and the precipitated compounds re-dissolve after anaerobic digestion (Seaborne and Stuttgart processes). Nevertheless, the potential of these processes is limited as they cannot avoid the uncontrolled precipitation of the previously indicated P-compounds (Egle *et al.*, 2016). Other configurations, such as that described in Pastor et al. (2008), Martí et al. (2017) or the WASSTRIP® process (Cullen et al., 2013) aim to overcome this limitation by recovering P before anaerobic digestion.

Life cycle assessment (LCA) can quantify and compare the environmental impacts related to the process per functional unit (ISO, 2006) and makes possible its comparison. Its use has been promoted by the United Nations Environmental Program, however, it is rarely employed by decision-makers, despite its high potential (Guérin-Schneider et al., 2018). Post-digestion processes have already been compared by different approaches and methods in a number of studies (Rodriguez-Garcia et al., 2014; Bradford-Hartke et al., 2015; Remy and Jossa, 2015; Sena and Hicks, 2018), however, none have compared pre-digestion P-recovery technologies. Also, few studies have been carried out on the life cycle costing (LCC) of P-recovery and these only deal with costs for post-digestion technologies. The existing literature is based on assessing the costs related to infrastructure, energy, chemicals, personnel, maintenance, products, by-products and waste (Egle et al., 2016; Nättorp et al., 2017).

The novelty of this paper lies in its holistic assessment of technical, economic and environmental aspects of three different P recovery strategies from sludge: one after anaerobic digestion and two before anaerobic digestion. For this purpose, it is necessary to calibrate the model of the WWTP in order to obtain reliable results of the different P recovery strategies required for the LCA and LCC analysis. The post-anaerobic digestion strategy is based on classical P recovery from the centrates produced during the dewatering process, while the pre-anaerobic digestion strategies are based on enhancing P extraction and separation by means of a novel elutriation process and a process based on the WASSTRIP®.

2. MATERIALS AND METHODS

Both sludge line configurations assessed had the same goal: to increase P recovery and reduce uncontrolled P-precipitation during anaerobic digestion. This can be achieved by enhancing poly-P release from sludge and separating it before anaerobic digestion. The first configuration (C1) evaluated the potential to recover P by mixing both thickened sludges under anaerobic conditions in a mixing chamber and pumping the mixed sludge to the primary thickener to obtain a liquid PO_4-enriched stream to subsequently crystallize MAP (Figure 1, left). The second configuration (C2) was based on the WASSTRIP process: the fermented primary sludge was separated in the primary thickeners to obtain a VFA-enriched liquid stream which was added to the waste-activated sludge (WAS) in a release tank. A PO_4-enriched stream can then be mechanically separated from sludge and finally recovered by crystallizing MAP (Figure 1, right). In both configurations, VFA uptake and PO_4 release by PAO took place under the anaerobic conditions in the mixing chamber (C1) and release tank (C2).

For comparison purposes, both configurations were assessed technically, economically and environmentally, in the Murcia-Este WWTP (Spain). The first step was the calibration of the Biological Nutrients Removal Model No.2 (Barat et al., 2013), BNRM2, with information from three complete analytical campaigns to characterize the whole WWTP performance. The different sludge line management strategies were then simulated on DESSAS 7.1 software

(Ferrer et al., 2008) to stablish the optimal operating conditions and finally were environmentally and economically evaluated.

2.1. Current Murcia Este WWTP configuration

The Murcia-Este WWTP mainstream consists of primary settling, an A2O biological reactor with a total volume of 41,405 m^3 (3.9% of the volume is anaerobic, 23.3% anoxic and 72.8% aerobic) and secondary settling. The sludge line configuration is made up of two primary gravity thickeners and two dissolved air flotation thickeners (DAF) for waste-activated sludge (WAS). However, only one DAF thickener is currently being used. The thickened WAS is electromagnetically hydrolysed and mixed with the primary sludge in a 79.2 m^3 chamber and then digested and dewatered to a final concentration of 20-25% of TSS. Figure 2, shows the layout of the Murcia-Este WWTP with the different streams characterized for model calibration.

2.2. Layout of the sludge line configurations evaluated

The determining factor for the implementation of both configurations was extracting as much poly-P as possible with the minimum investment cost by using as many elements of the current WWTP layout as possible.

C1 was based on the elutriation of mixed sludge in the primary thickeners and increasing the mixing chamber volume from 79 m^3 to 609 m^3 (Figure 3) with one of the secondary thickeners. In this alternative the following effects were tested: 1) different elutriation flows (Q_{elut}) (from 0 to 1,000 m^3 d^{-1}); 2) reduced primary sludge flows (from the current Q_{ps} of 5,685 m^3 d^{-1} to 1,500 m^3 d^{-1}); 3) reduced sludge digestion flow (Q_{dig}) (from the current operation flow, 787 $m^3 d^{-1}$ to 500 m^3 d^{-1}).

C2 was based on the WASSTRIP configuration (Figure 4). In C2 the primary sludge was fermented in the primary thickeners and the VFA concentration in the overflow stream was increased by elutriation of the thickened sludge in the primary thickeners. The VFA-enriched primary thickener overflow was mixed with the WAS in the secondary thickeners, which were converted into anaerobic P-release tanks (V_{tot}= 1,060 m^3). The next step was the separation of an enriched PO_4-P stream with a dynamic thickener. The effect of increasing Q_{elut} from 0 to 1,000 m^3d^{-1} while reducing Q_{ps}, from 5,685 to 1,500 m^3d^{-1} were tested in this configuration.

2.3. Analytical method

Total and volatile suspended solids (TSS, VSS), total and soluble chemical oxygen demand (COD_{tot}, COD_{sol}), total and soluble biological demand (BOD_{tot}, BOD_{sol}), total and soluble phosphorus (P_{tot}, P_{sol}), total and soluble nitrogen (N_{tot}, N_{sol}), orthophosphate (PO_4), ammonium (NH_4), nitrate (NO_3), and soluble metals (Mg_{sol}, $Ca_{sol,}$ K_{sol}) analyses were carried out following Standard Methods (APHA, 2005). Alkalinity and VFA were analysed as described in Moosbrugger et al. (1992).

2.4. Assessing precipitation

Sludge line precipitation was estimated following the method described in Martí et al. (2008) by applying mass balances for volatile solids, orthophosphate, magnesium and potassium in the mixing chamber, anaerobic digestion, secondary digester and centrifuge for the current configuration, plus primary thickener for C1 and the P-release tanks and dynamic thickeners for C2.

2.5. Calibration and simulations

The model calibration and all simulations were carried out on DESASS 7.1 software (Ferrer et al., 2008), which runs the BNRM2 (Barat et al., 2013), which includes all the equations, parameters and components to accurately reproduce the biological and chemical processes in a WWTP.

For calibration purposes, three intensive analytical campaigns where carried out at different points of the water and sludge lines under normal operating conditions. These included an exhaustive characterization of different parameters (see Section 2.3) in the streams indicated in Figure 2. A total of 15 samples were analysed in each campaign (with between 16 and 18 parameters in each sample). All the samples were composite for 24 h (continuous sampling in the water line and manual composite samples in the sludge line).

The model was calibrated by tuning the different parameters to minimize the differences between the analytical results of three analytical campaigns and their respective simulations, after which it was used to simulate the configurations above-mentioned.

2.6. Life cycle cost

The life cycle cost was assessed for each configuration calculating the annual cost, considering the sludge line differential elements. It was considered the OPEX cost associated to: chemicals, sludge disposal and energy consumption for operation and maintenance of thickening, digestion (including the maintenance cost associated to the uncontrolled precipitation), dewatering and combined heating and power (CHP). The CAPEX considered for C1 and C2 were: valves, pumps, variable-frequency drives, stirrers, flow indicators and sludge blanket height controller indicators and transducers, pipelines, PO_4 analysers and dynamic thickeners (see Tables 1 and 2). CAPEX were calculated at a 10% annual discount rate (r) and a project lifetime of 20 years (t) according with Pretel et al. (2016). The cost analysis is given in Tables 3 and 4. Maintenance included replacement of pumps, valves, stirrers, blowers, anti-scaling investment and other differential elements. All the sludge was assumed to be used for agricultural applications.

Energy production was estimated at a methane calorific power of 38,000 kJ m^{-3} and 35% efficiency (Seco et al., 2018). As the same personnel were assumed to run the infrastructure after implementing both configurations, this item was excluded from the LCC calculation as it was not a differential element. Maintenance costs were related to the replacement of equipment after its lifetime, based on information retrieved from WWTP operators, equipment providers and the cost of associated operations to reduce the effects of uncontrolled P precipitation. The total annual equivalent cost (TAEC) was calculated as the sum of the annual capital expenses (CAPEX) and operational requirements (OPEX), according to Eq.(1) (Ferrer et al. 2015):

$$TAEC = \frac{r(1+r)^t}{(1+r)^t - 1} \cdot CAPEX + OPEX \quad (1)$$

Return on investment (ROI) was calculated assuming a 95% MAP crystallization efficiency and a market value of between €55 and €300 $TmMAP^{-1}$ (de Vries et al., 2017).

2.7. Life cycle assessment

The life cycle assessment (LCA) was performed according to UNE-EN ISO 14040 (ISO, 2006). The inventory of materials and processes, which lists the volumes of inflows from the natural environment and the outflows released into it, was compiled using the Ecoinvent Database v3 via SimaPro 8.5 (PRé Consultants; The Netherlands). The Hierarchist ReCiPe(H) v 1.02 midpoint method was employed to carry out the assessment of global warming (GW, quantified as kgCO_2 eq.). The three scenarios were evaluated with functional units of 1 t of PO_4-P in the recovery stream and a treated wastewater inflow of 1 hm^3. Only the differential elements were considered.

Different authors consider that the infrastructure's contribution to the LCA is negligible in comparison to the operational impacts (Bradford-Hartke et al., 2015; Remy and Jossa, 2015; Kjerstadius et al., 2017). In the present study, minimum modifications of the current sludge line configuration (C0) were required to set up the C1 and C2 configurations. Neither configuration required new reactors (existing elements were used, such as secondary thickeners not used in the current sludge management configuration), only pumping, pipes and new dynamic thickeners for C2 being necessary. For these reasons and due to the lack of reliable information, the environmental impact related to the infrastructure were omitted, focusing on the operation of the three P-recovery configurations.

3. RESULTS

3.1. Characterization

Table 5 shows the average values of the different parameters analysed during the three analytical campaigns together with standard deviations. It is important to note the significant fermentation in the primary thickener, which raised the VFA concentration from 95 mg COD/l to 794 mgCOD/l. This fermentation is a key aspect of the study since the mix of VFA and secondary sludge before the anaerobic digestion (in the mixing chamber in C1 and in the release tank in C2, see Figure 1) triggered poly-P release. Also important is the increased ammonium concentration during anaerobic digestion due to hydrolysis and the reduced phosphate concentration, which confirmed the high level of phosphorus precipitation (see Section 3.4). Different mass balances were applied for phosphate, magnesium and potassium in the mixing chamber, anaerobic digestion, secondary digester and centrifuge, following Martí et al. (2008) to quantify uncontrolled P precipitation. The analytical values determined the content of P, Mg and K in organic matter, with values of 1.7 mgP·100 $mgVSS^{-1}$, 0.5 mgMg·100 $mgVSS^{-1}$, 1.1 mgK·100 $mgVSS^{-1}$ (considering 32.7% of primary and 67.3% of secondary sludge). Uncontrolled P precipitation in the sludge line was 577.9 kgP d^{-1}, anaerobic digestion being the hot spot, as 95.7% of the precipitation took place there. Under the current WWTP configuration, only 7.4% of the influent P load was present as PO_4-P in the primary and secondary thickening overflows and centrates from centrifuges (3.6% available only from centrates). This low P-availability confirmed that the classical recovery of phosphorus from centrates drastically reduces the potential recoverable P due to its precipitation.

3.2. Calibration

The BNRM2 model was calibrated to fit the simulated results with the three analytical characterizations of the Murcia-Este WWTP under steady state conditions. As can be seen in Figures S1-S4, the model was able to reproduce the biological and physical processes that took place in the entire WWTP, which was fitted with the default values of the model parameters proposed in BNRM2, with the exception of some parameters related to biological phosphorus removal, hydrolysis and fermentation. The modified parameters are listed in Table 6.

The model simulations revealed significant differences in the behaviour of the bacteria responsible for biological P removal (PAO) and organic matter hydrolysis and fermentation (acidogenic bacteria) before and after the mixing chamber, probably due to electromagnetic hydrolysis of the biological sludge before the mixing chamber (see Figure 2, between streams 7 and 7'), which probably reduced the bacteria's metabolism and performance. It is remarkable the small volume of the anaerobic zone in the biological reactor (3.9% of the biological reactor), which contrasts with the ability of the PAO bacteria developed in the reactor to reach high P removal efficiencies (around 97%). For this reason, different parameters involved in the PAO metabolism were modified, such as PHA accumulation rate (q_{PHA}) and volatile fatty acids half saturation constants (K_A and K_{PRO}). Acidogenic activity (μ_{Acid}) was modified in order to reproduce the high degree of fermentation observed at the bottom of the primary thickeners. Hydrolytic activity (k_H) was changed in order to fit the experimentally measured soluble COD balance in the gravity thickeners. This parameter was increased in the mixing chamber to simulate the effect of the electromagnetic hydrolyser after secondary thickening.

3.3. Phosphorus extraction evaluation

3.3.1. Current configuration

The three analytical campaigns established the baseline of the current WWTP. Under the current sludge line configuration (C0) the PO_4-P concentration in the overflows of the primary and the secondary thickener were 5.2 and 3.0 mg L^{-1}, which represents 3.8% of the influent P load. The NH_4-N concentration in the primary and secondary thickener overflows were 53.4 and 5.8 mg L^{-1} respectively. 29.5% of the stored poly-P was released in the mixing chamber and PO_4-P concentration in the centrifuge centrate was 49.2 mg L^{-1}, 3.6% of the influent P, with a NH_4-N concentration of 921.1 mg L^{-1}. The sludge blanket height in the primary thickener was 0.80 m and 8301 Nm^3 d^{-1} of biogas was produced under the current configuration.

P recovered as MAP from the centrifuge centrate would give a production of 91.5 tMAP $year^{-1}$, assuming 95% crystallizer efficiency. It is important to note that the PO_4-P concentration in this recovery stream was 3.6% of the influent P load.

3.3.2. Alternative configurations C1 and C2

The proposed configurations (see Figures 3 and 4) were optimised by modifying different operating conditions. Figures 5 and 6 show the simulation results in terms of potential P recovery from the P-enriched streams and phosphate concentration under different operational conditions. Figures S5 and S6 show the P mass balances in the optimized scenario for each configuration.

As can be seen in Figure 5, reducing Q_{ps} from 5,685 to 1,500 $m^3 d^{-1}$ increased PO_4 concentration in the recovery stream from 82.6 to 250.2 mgP L^{-1}. However, as shown Figure 6, the influent extraction efficiency was reduced from 50.7% to 36.7%, while the sludge blanket height rose from 0.65 to 1.15 m (data not shown).

According to the specifications of the crystallization process, it was necessary to achieve as high a phosphate concentration as possible in the recovery stream and to keep it above 80 mgP L^{-1}. As the operational limitations of the primary settlers in the Murcia-Este WWTP did not allow reducing Q_{ps} below 2,500 $m^3 d^{-1}$ it was decided to fix Q_{ps} around 2,500 $m^3 d^{-1}$. In this condition, the phosphate concentration was 169.1 mgP L^{-1} with an extraction efficiency of 43.1%.

Raising Q_{elut} to 1,000 $m^3 d^{-1}$ increased PO_4-P concentration in the recovery stream by about 3%. The negligible effect of Q_{elut} on P extraction was due to the mixing chamber volume, and its high hydraulic retention time (around 24 hours) promoted fermentation and extraction of the PAO internal Poly-P, so that it was not necessary to mix primary and secondary sludge in the mixing chamber. These results indicate that the optimal C1 operational conditions were Q_{ps} = 2,500 $m^3 d^{-1}$ with no elutriation flow. Considering a 95% efficiency of P recovery as MAP, the annual production after crystallizing MAP would be 1,053.3 tMAP $year^{-1}$. The reduction of Q_{dig} from the current value of 787 $m^3 d^{-1}$ to 500 $m^3 d^{-1}$ increased the extraction efficiency from 43.1% to 48.3%. However, this reduction of the sludge digestion flow was not considered, due to a drastic rise in sludge blanket height up to 1.57 m, which compromised overall C1 performance due to the risk of the overflow of the primary thickeners.

Changing Q_{ps} and Q_{elut} in the C2 simulations showed that reducing Q_{ps} from 5,685 to 1,500 $m^3 d^{-1}$ without elutriation did not significantly increase the PO_4-P concentration in the recovery stream (from 46.2 to 56.9 mgP L^{-1}). Raising Q_{elut} up to 1,000 $m^3 d^{-1}$ increased the concentration in the recovery stream, reaching up to 104.2 mgP L^{-1} for Q_{ps} = 2,500 $m^3 d^{-1}$ and 135.2 mgP L^{-1} for 1,500 $m^3 d^{-1}$. The recovery efficiencies were 44.2%, 48.2 and 47.4% for Q_{ps} of 1,500 $m^3 d^{-1}$, 2,500 $m^3 d^{-1}$ and 5,685 $m^3 d^{-1}$, respectively. Q_{ps} = 2,500 $m^3 d^{-1}$ with Q_{elut} = 1,000 $m^3 d^{-1}$ was considered the optimal operating conditions for this configuration (Figure 5 and Figure 6). The annual MAP production increased by up to 1,176.8 t year^{-1} with the same crystallization efficiency as in C1.

Regarding anaerobic digestion performance, biogas production increased by around 1.5% in C1 and 7.4% in C2 due to the higher organic load in the anaerobic digesters due to the increase of the COD load in the digesters (see Table 7).

3.4. Reduction of uncontrolled P precipitation

The implementation of both sludge line configurations significantly reduced uncontrolled precipitation in relation to the current configuration (C0) of around 52.9% in C1 and 60.0% in C2. The digesters remained as the main P sink in the sludge line with around 90% of total precipitation. The available PO_4 in the overflows and centrates increased from 65.8 kgP d^{-1} (7.4% of the influent P load) in the original configuration (C0) to 421.4 kgP d^{-1} for C1 and 476.1 kgP d^{-1} for C2 (Figure 7), considering all the overflows and centrates, which represent 47.4% and 53.6% of the influent P load respectively.

3.5. Life cycle costs

TAEC per ton of PO_4-P in the recovery stream and per hm^3 of treated wastewater are shown in Figures 8a and 8b respectively for each configuration. The cost was divided into OPEX and CAPEX, considering 10% depreciation for 20 years considering the differential elements in the sludge line and its operation between the three studied configurations. Regarding the results normalized per ton of PO_4-P (Figure 8a), TAEC for C0 was 22,565 € and was reduced by 92.8% in C1 and 91.7% in C2.These reductions were clearly due to the large amount of phosphorus recovered in the alternatives focused on phosphorus extraction and recovery before anaerobic digestion (C1 and C2) compared to the current configuration, which recovers phosphorus after anaerobic digestion (C0).

However, normalizing the cost with a conventional functional unit, i.e. the influent flowrate of treated wastewater (in hm^3), the TAEC changed drastically, obtaining values of 7,696 € hm^{-3}, 6,341 € hm^{-3} and 8,236 € hm^{-3} for C0, C1 and C2 respectively (Figure 8b). In this case C1 again obtained the lowest TAEC (17.6% lower than C0) due to the significant reduction of maintenance costs to mitigate uncontrolled P precipitation in the sludge line and higher biogas production, despite the slight increase in the energy requirements (around 3.2% increase from C0 to C1 mainly due to the thickening process and mixing chamber agitation). Despite the greater reduction in maintenance costs and higher biogas production, C2 obtained the highest TEAC due to higher energy and chemical consumption and the acquisition cost of new dynamic thickeners.

CAPEX was 6.9% of TAEC for C1 and 13.2% for C2. This difference was mainly due to the cost associated with the dynamic thickeners to obtain the PO_4-enriched stream (C2 required the acquisition of 4 dynamic thickeners to treat 4,530 $m^3\,d^{-1}$ of sludge). C1 did not need any specific equipment to obtain this PO_4-enriched stream. This configuration employed the existing primary thickeners to separate the soluble PO_4-P in a specific stream.

Assuming 95% efficiency in the subsequent crystallization phase and a MAP market price between 55 and 300 € $tMAP^{-1}$, the investment in the sludge line would be recovered in between 5 to 27 months for C1 and from 11 and 61 months for C2.

The results obtained in the LCC analysis confirmed that the sludge management configuration based on the elutriation in the primary thickeners (C1) presented the lowest cost per ton of PO_4-P in the recovery stream and per hm^3 of treated wastewater.

3.6. Life cycle assessment

SimaPro 8.5 was used to compare the differential elements in operating the different configurations. Figure 9a shows the global warming (GW) impact of the three approaches per t of PO_4-P in the recovery stream. For C0, GW was the highest, 46,007 kg $CO_2eq\cdot tPO_4\text{-}P^{-1}$. GW for C1 and C2 was very similar, 3,917 kg $CO_2eq\cdot tPO_4\text{-}P^{-1}$ and 4,088 kg $CO_2eq\cdot tPO_4\text{-}P^{-1}$ respectively. Despite the higher P recovery in C2, the GW impact of this configuration is similar to the one obtained in C1 due to its higher emissions associated to the increase in the energy and chemicals consumption. However, using the influent flowrate of treated wastewater (hm^3) as the functional unit to normalize the impacts (Figure 9b), C2 emissions increased up to 14.2% over C0, and its GW impact was 15,690 kg $CO_2eq \cdot hm^{-3}$, whilst for C1 it was reduced by up to 2.0% (Figure 9b). Despite the fact that C1 produced less biogas, it required substantially less energy for sludge pumping and thickening. The lower impact of C1 was mainly related to higher biogas production. A similar behaviour between alternatives was observed in other impact categories, such as, terrestrial acidification, freshwater eutrophication and mineral resource scarcity. Further information on other environmental indicators can be found in Figures S7-S20 and Table S1.

Same as in the cost analysis, the results obtained in the LCA confirmed that the sludge management configuration based on the elutriation in the primary thickeners (C1) presented the lowest impacts per ton of PO_4-P in the recovery stream and per hm^3 of treated wastewater.

4. DISCUSSION

Currently, most studies on recovering phosphorus as struvite from wastewater focus on crystallization of the centrates after the digestion process (Egle et al., 2016; Nättorp et al., 2017; Sena and Hicks, 2018) and disregard the enormous amount of phosphorus that is lost due to uncontrolled precipitation during and after anaerobic digestion. Only a few studies have explored new sludge management strategies to minimize this phosphorus loss, for example those based on phosphorus extraction and separation in a specific stream before the anaerobic digesters (Lizarralde et al., 2019, Martí et al., 2017). These strategies have been evaluated in different studies (Bouzas et al., 2019; Cullen et al., 2013) but have never been compared to a real case study under technical, economic and environmental criteria. The present work tested both configurations jointly with the classical system using the centrates from anaerobic digesters.

Lizarralde et al. (2019) simulated different phosphorus management strategies in the Sur-WWTP (Madrid, Spain), including, among others, a WASSTRIP-based process and achieved 23.8% of the influent phosphorus present as phosphate in the extraction stream. This is lower than the one obtained in the present work (48.5%), mainly due to the lower efficiency of the biological phosphorus removal in the water line (73% in the Sur-WWTP and 96% in the Murcia-Este WWTP), which considerably reduced potential phosphorus recovery. However, the results obtained in the present study are in agreement with those obtained by Gysin et al. (2018), who found that the potential recovery of the WASSTRIP process was between 45% and 60% of the influent P-load.

Bouzas et al. (2019) evaluated the overall performance of the Calahorra WWTP (Spain) after the implementation of the new sludge management based on the phosphorus extraction by elutriation in the primary thickeners (equivalent to the C1 in the present work) and found that this sludge management system reduced uncontrolled precipitation by 43%, in agreement with the present study (52.9%). However, there are significant discrepancies in the amount of phosphate present in the extraction stream. Bouzas et al. (2019) obtained 19% of influent phosphorus in the extraction stream as phosphate, while this was 43.1% in the present study. These discrepancies could be attributed to the low capacity of the primary thickener of the Calahorra-WWTP to admit more solids in the elutriation stream and the lower phosphorus removal efficiency than in Murcia-Este WWTP (between 80-92% in the former and 96% in the latter). However, further research is needed in a full-scale experiment to confirm the simulated results of the present work.

5. CONCLUSIONS

This study confirms that the implementation of any of the proposed configurations to modify sludge management before anaerobic digestion is economically viable and highly efficient in comparison with the currently widespread post-digestion recovery technologies. Optimal C1 operation increased PO_4 concentration in the recovery stream by up to 169.1 mgP L^{-1}, corresponding to 43.1% of the influent P load, and reduced uncontrolled P-precipitation by 53% in comparison to C0. On the other hand, the optimal operation of C2 was able to recover 48.2% of the influent P load with a concentration of 104.2 mgP L^{-1}, reducing uncontrolled precipitation by up to 60% in comparison to C0.

Despite its slightly lower recovery potential, C1 shows a significantly lower CAPEX and OPEX, mainly because this configuration does not require the acquisition of dynamic thickeners to obtain the PO_4-enriched stream from sludge and has lower sludge pumping flows and a smaller environmental impact. The analysis of the implementation of both alternatives in the Murcia-Este WWTP showed C1 has a lower TAEC, ROI and environmental footprint and is therefore considered the most favourable sludge line configuration for P recovery.

These results show how performing complete holistic studies for each situation are essential to differentiate between apparently similar options, can make substantial progress towards a new paradigm and contribute to overcoming the limitations of the current technologies.

6. ACKNOWLEDGEMENTS

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Table 1
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Table 1. Acquisition costs for C1.

Element	nº	Total cost (€)	Source
Pneumatic guillotine valve DN 200	1	634.85	BEDEC, 2019 / Murcia-Este WWTP operators
Pneumatic guillotine valve DN 150	17	9,062.02	BEDEC, 2019 / Murcia-Este WWTP operators
Manual valves	2	674.30	BEDEC, 2019
Motorised guillotine valve DN 150	3	5,803.74	BEDEC, 2019 / Murcia-Este WWTP
VFD 7.5 kW	12	11,395.74	Murcia-Este WWTP operators
Agitators	2	17,261.02	Murcia-Este WWTP operators
Level indicator	1	1,146.66	Murcia-Este WWTP operators
Sludge blanket flow indicator	2	8,500.00	Murcia-Este WWTP operators
Flow indicator	7	11,030.11	Murcia-Este WWTP operators
Pumping 1	3	6,830.25	Bombas Ideal S.A., 2018 / Murcia-Este WWTP operators
Pumping 2	3	6,830.25	Bombas Ideal S.A., 2018 / Murcia-Este WWTP operators
P analyzer	1	23,285.88	Endress+Hauser, 2018 / Murcia-Este WWTP operators
PLC adjustment	1	13,789.87	Murcia-Este WWTP operators
SCADA programming	1	7,123.20	Murcia-Este WWTP operators
SCADA+PLC programming	1	20,913.07	Murcia-Este WWTP operators
Ditch digging	24	1,578.24	Murcia-Este WWTP operators
Filling 1	8	148.48	Murcia-Este WWTP operators
Filling 2	16	347.20	Murcia-Este WWTP operators
Polyethylene pipes	287 (m)	6,483.33	Frans Bonhomme, 2019 / Murcia-Este WWTP operators
Total		131.925,14	

Table 2
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Table 2. Acquisition costs for C2

Element	nº	Total cost (€)	Source
Pneumatic guillotine valve DN 200	1	634.85	BEDEC, 2019 / Murcia-Este WWTP operators
Pneumatic guillotine valve DN 150	22	11,727.32	BEDEC, 2019 / Murcia-Este WWTP operators
Manual valves	2	674.30	BEDEC, 2019
Motorised guillotine valve DN 150	3	5,803.74	BEDEC, 2019 / Murcia-Este WWTP operators
VFD 7.5 kW	9	8,592.30	Murcia-Este WWTP operators
VFD removal	3	0.00	Murcia-Este WWTP operators
Agitators	4	34,522.04	Murcia-Este WWTP operators
Level indicator	2	2,293.32	Murcia-Este WWTP operators
Sludge blanket flow indicator	2	8,500.00	Murcia-Este WWTP operators
Flowindicator	3	4,727.19	Murcia-Este WWTP operators
Pumping 1	3	12,201.33	Bombas Ideal S.A., 2018 / Murcia-Este WWTP operators
Pumping 2	3	6,830.25	Bombas Ideal S.A., 2018 / Murcia-Este WWTP operators
Dynamic thickeners	4	180,400.00	Alfa Laval Iberia, 2018
P analyzer	1	23,285.88	Endress+Hauser, 2018 / Murcia-Este WWTP operators
SCADA+PLC programming	1	20,913.07	Murcia-Este WWTP operators
Ditch digging	24	1,578.24	Murcia-Este WWTP operators
Filling 1	8	148.48	Murcia-Este WWTP operators
Filling 2	16	347.20	Murcia-Este WWTP operators
Polyethylene pipes	287 (m)	6,483.33	Murcia-Este WWTP operators
Total		329,662.84	

Table 3
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Table 3. Energy and polymer consumption for C0, C1 and C2.

	Energy consumption (kWh year^{-1})		
	C0	**C1**	**C2**
Primary sludge pumping to primary thickeners	157,680	100,740	100,740
Thickened primary sludge pumping to mixing chamber	19,272	0	0
Thickened primary sludge pumping to digestion	0	47,304	0
Thickened primary sludge towards primary thickeners (elutriation) and mixing chamber	0	0	56,064
Thickened biological sludge pumping to mixing chamber	43,800	0	0
Thickened biological sludge pumping to primary thickeners	0	43,800	0
Biological sludge pumping to dynamic thickening	0	0	271,560
Secondary thickener recirculation pumping	297,840	297,840	0
Secondary thickener blower	19,272	19,272	0
Dynamic thickener	0	0	198,414
Mixing chamber stirring	6,938	6,938	6,938
P-release tanks stirring	0	46,428	92,856
Anaerobic digestion + sludge dewatering + CHP	1,523,047	1,523,047	1,523,047
	Polyelectrolyte consumption (t year^{-1})		
	C0	**C1**	**C2**
Secondary thickeners	21.8	22.5	0.0
Dynamic thickeners	0.0	0.0	38.1

Table 4
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Table 4. OPEX costs.

	Cost		Reference
External energy cost	0.138	€ kWh^{-1}	Pretel et al. (2016)
Co-generated energy cost	0.008	€ kWh^{-1}	Wang et al. (2019)
Polyelectrolyte (polyacrylamide)	2.350	€ kg^{-1}	Pretel et al. (2016)
Polyelectrolyte consumption (secondary thickeners)	3.0	kg $tTSS^{-1}$	Murcia Este WWTP operators
Polyelectrolyte consumption (dynamic thickeners)	5.0	kg $tTSS^{-1}$	Alfa Laval Iberia, 2018
Agricultural application of the sludge	4.8	€ $tTSS^{-1}$	Pretel et al. (2016)
Uncontrolled P-precipitation mitigation	0.489	€·t P-precipitated	Neetheling and Benisch (2004)

Table 5
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Table 5. Average characterization in the different sampling points (see Figure 2).

Sampling point	Flow rate (m^3 d^{-1})	TSS/TS* (mg L^{-1})	VSS/VS** (%)	COD_{tot} (mg L^{-1})	COD_{sol} (mg L^{-1})	VFA (mg L^{-1})	P_{tot} (mg L^{-1})	N_{tot} (mg L^{-1})	PO_4-P (mg L^{-1})	NH_4-N (mg L^{-1})	Ca_{sol} (mg L^{-1})	Mg_{sol} (mg L^{-1})	K_{sol} (mg L^{-1})	pH
0	97,701 ± 18,839	404 ± 65	79.7 ± 4.9	654 ± 102	194 ± 96	105 ± 20	9.1 ± 2.2	76.8 ± 7.0	4.9 ± 0.5	50.6 ± 1.8	132.5 ± 12.8	66.1 ± 4.2	37.5 ± 5.5	7.5 ± 0.1
1	92,983 ± 18,203	289 ± 66	80.9 ± 2.8	572 ± 114	196 ± 95	105 ± 49	7.8 ± 2.3	66.8 ± 9.5	4.5 ± 0.7	50 ± 2.7	130.3 ± 11.2	63.4 ± 5.8	34.6 ± 9.2	7.5 ± 0.1
2	157,301 ± 23,381	2,869 ± 601	79.0 ± 0.7	3,863 ± 644	n.a.	n.a.	86.8 ± 24.6	210 ± 43.8	0.1 ± 0.0	4.3 ± 3.5	142.1 ± 10.4	59.4	34.9 ± 0.9	7.2 ± 0.1
3	97,100 ± 18,723	10.3 ± 10.2	83.7 ± 2.7	38 ± 11	34 ± 57	n.a.	0.3 ± 0.0	16.3 ± 0.2	0.1 ± 0.0	3.3 ± 1.7	134.2 ± 13.2	61.2 ± 5.5	32.8 ± 2.6	7.3 ± 0.1
4	4,718 ± 1,690	2,868 ± 902	73.9 ± 7.5	3,592 ± 971	223 ± 6	97 ± 15	37.5 ± 13.7	326.7 ± 40.1	5.4 ± 0.5	55.7 ± 4.6	142.4 ± 18.0	66.1 ± 8.1	32.9 ± 4.1	7.3 ± 0.1
5	2,232 ± 217	7,147 ± 1,151	78.7 ± 0.9	8,504 ± 1,129	33 ± 10	n.a.	201.0 ± 50.7	529.2 ± 142.7	2.7 ± 0.2	2.9 ± 2.2	136.0 ± 9.9	63.6 ± 3.6	34.9 ± 3.6	6.9 ± 0.0
6	220 ± 56	46,830 ± 14,090	75.2 ± 3.2	51,307 ± 10,715	1,049 ± 326	794 ± 194	312.1 ± 105.5	n.a.	26.6 ± 9.1	87.5 ± 4.2	182.6 ± 15.4	81.8 ± 11.8	52.1 ± 4.8	6.4 ± 0.1
7	567 ± 89	36,040 ± 2,995	79.1 ± 2.0	32,789 ± 2,822	119 ± 62	n.a.	736.1 ± 150.9	n.a.	91.8 ± 18.6	15.4 ± 4.1	140.7 ± 8.0	72.6 ± 3.0	96.9 ± 22.8	n.a.
7'	567 ± 89	33,906 ± 2,545	78.2 ± 1.5	32,906 ± 2,743	126 ± 51	n.a.	669.1 ± 59.9	n.a.	110.0 ± 26.9	19.3 ± 4.6	134.2 ± 17.1	78.1 ± 4.2	96.8 ± 8.7	n.a.
8	4,498 ± 1,742	529 ± 72	83.3 ± 3.7	823 ± 335	102 ± 39	n.a.	9.3 ± 1.4	n.a.	5.3 ± 0.8	40.7 ± 7.5	141.2 ± 15.1	69.8 ± 5.6	41.5 ± 11.9	7.5 ± 0.1
9	1,665 ± 139	130 ± 22	72.2 ± 3.0	125	29 ± 8	n.a.	5.3 ± 0.4	n.a.	3.8 ± 0.2	5.2 ± 3.2	142.7 ± 18.7	61.6 ± 7.5	26.1 ± 1.0	7.4 ± 0.1
10	787 ± 70	40,547 ± 3,614	77.0 ± 2.3	42,895 ± 9,546	674 ± 388	474 ± 324	536.9 ± 55.9	n.a.	136.0 ± 47.9	72.4 ± 32.1	127.7 ± 2.8	84.7 ± 8.3	94.8 ± 34.4	6.5 ± 0.1
11	787 ± 70	28,211 ± 1,352	63.7 ± 2.0	24,323 ± 1,578	788 ±116	76 ± 12	505.1 ± 40.1	n.a.	73.1 ± 12.1	879.4 ± 56.7	89.2 ± 31.4	11.3 ± 16.2	372.0 ± 19.3	7.4 ± 0.0
12	787 ± 70	28,718 ± 1,122	64.0 ± 4.6	23,708 ± 2,117	1,700 ± 1,483	143 ± 8	544.6 ± 167.7	n.a.	55.8 ± 13.7	958.5 ± 191.8	71.4 ± 16.6	8.2 ± 10.5	389.5 ± 47.2	7.6 ± 0.6
13	668 ± 58	2,612 ± 1,580	74.2 ± 2.7	n.a.	n.a.	n.a.	138.7 ± 15.6	n.a.	49.2 ± 8.4	921.1 ± 122.7	60.8 ± 23.1	2.9 ± 5.0	345.8 ± 16.2	8.1 ± 0.2

* TSS were determined in streams 0, 1, 2, 3, 4, 5, 6, 8, 9, 13 and TS were determined in streams 6, 7, 7', 10, 11, 12

** VSS were determined in streams 0, 1, 2, 3, 4, 5, 6, 8, 9, 13 and VS were determined in streams 6, 7, 7', 10, 11, 12

Table 6
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Table 6. Fitted parameters for the biological model of the Murcia-Este WWTP.

	Bibliography	Water & Sludge Lines	Mixing Chamber
Phosphate-accumulating organisms			
q_{pha} (d^{-1})	3.00 (ASM2d)	9.00	1.55
K_A (mg COD L^{-1})	4.00 (ASM2d)	3.30	3.30
K_{PRO} (mg COD L^{-1})	4.00 (ASM2d)	3.30	3.30
Acidogenic bacteria			
μ_{Acid} (d^{-1})	1.28 (BNRM2)	8.00	8.00
k_H (d^{-1})	43.00 (BNRM2)	13.30	75.00

Table 7

Click here to download Table: Table 7.docx

Table 7. Fitted parameters for the biological model of the Murcia-Este WWTP.

Configuration	Organic load anaerobic digestion (kP d^{-1})	Biogas production (Nm^3 d^{-1})
C0	30,577	8301
C1	31,498	8426
C2	32,292	8913

Figure 1

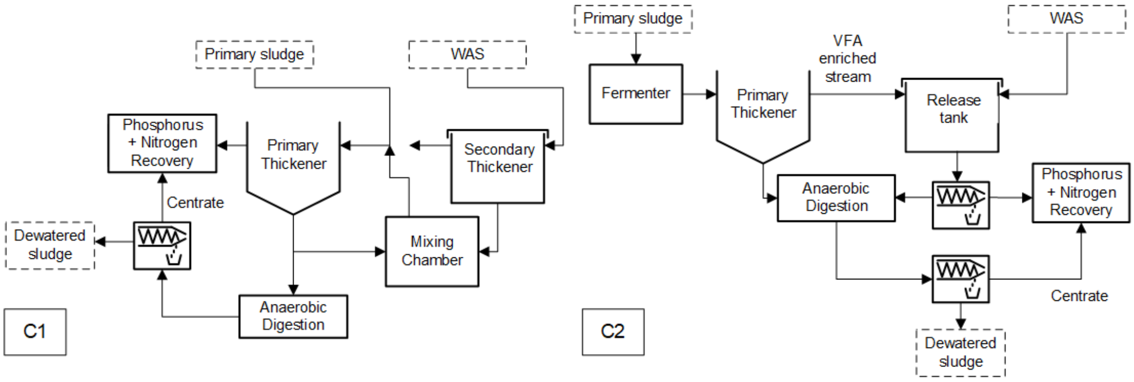

Figure 1. Principles for both assessed configurations. Configuration 1 (left) and Configuration 2 (right).

Figure 2

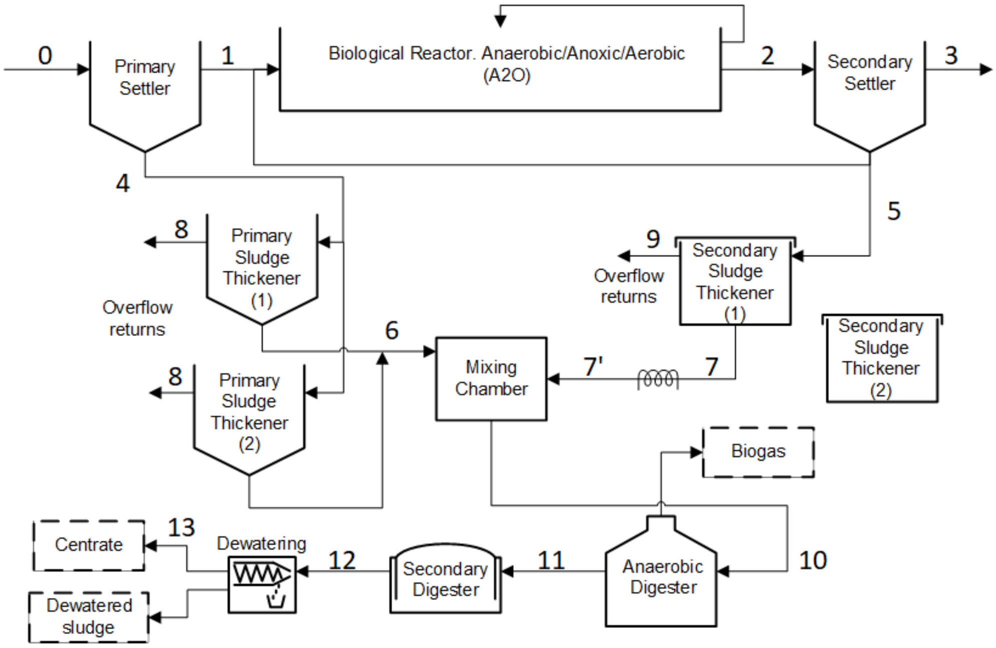

Figure 2. Current WWTP layout and main flows during the study of the different alternatives. Sampling points numbered.

Figure 3

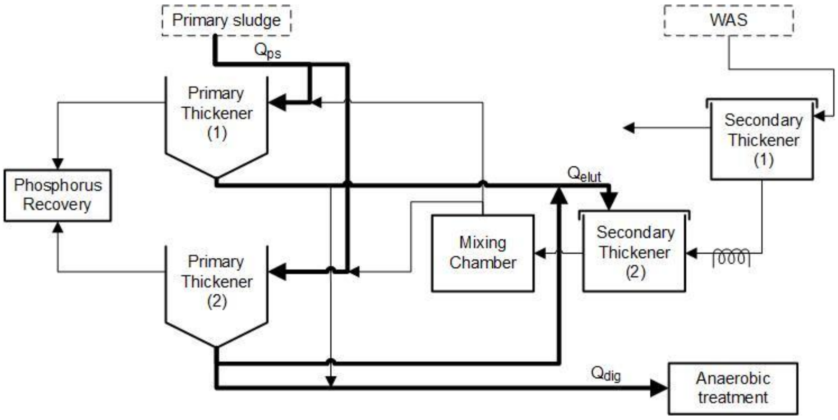

Figure 3. C1 scheme configuration in the Murcia-Este WWTP.

Figure 4

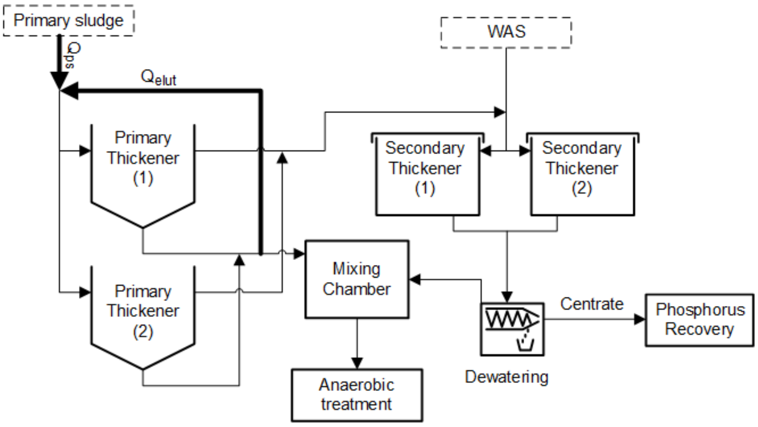

Figure 4. C2 scheme configuration in the Murcia-Este.

Figure 5

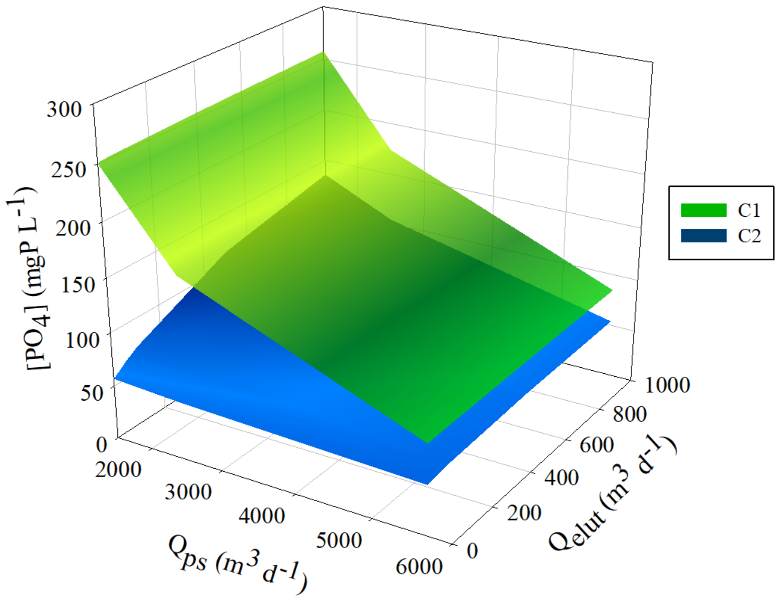

Figure 5. PO_4 concentration in the recovery stream for C1 (green) and C2 (blue) depending on Q_{ps} and Q_{elut}.

Figure 6

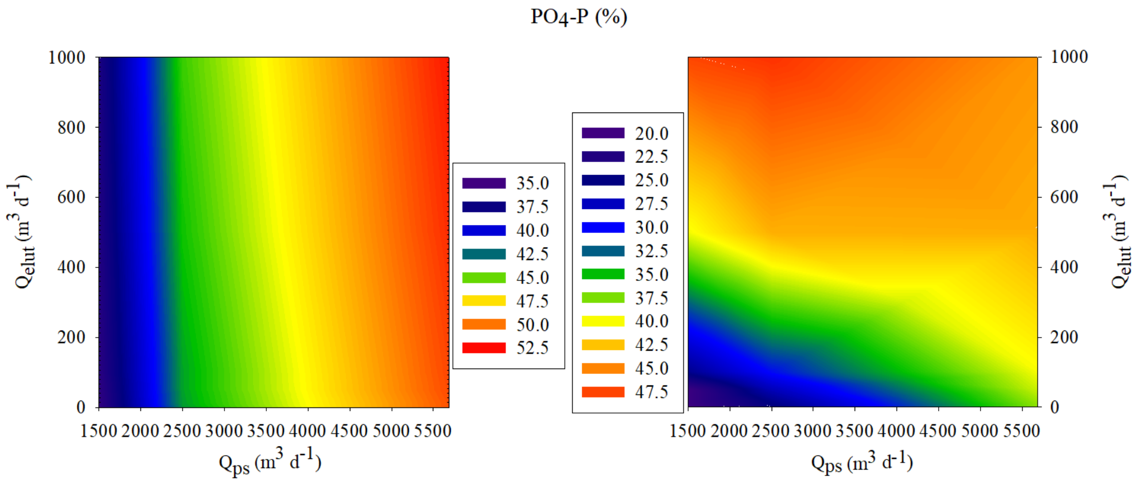

Figure 6. C1 (left) and C2 (right): percentage of the PO_4 load from the influent P in the recovery stream depending on Q_{ps} and Q_{elut}.

Figure 7

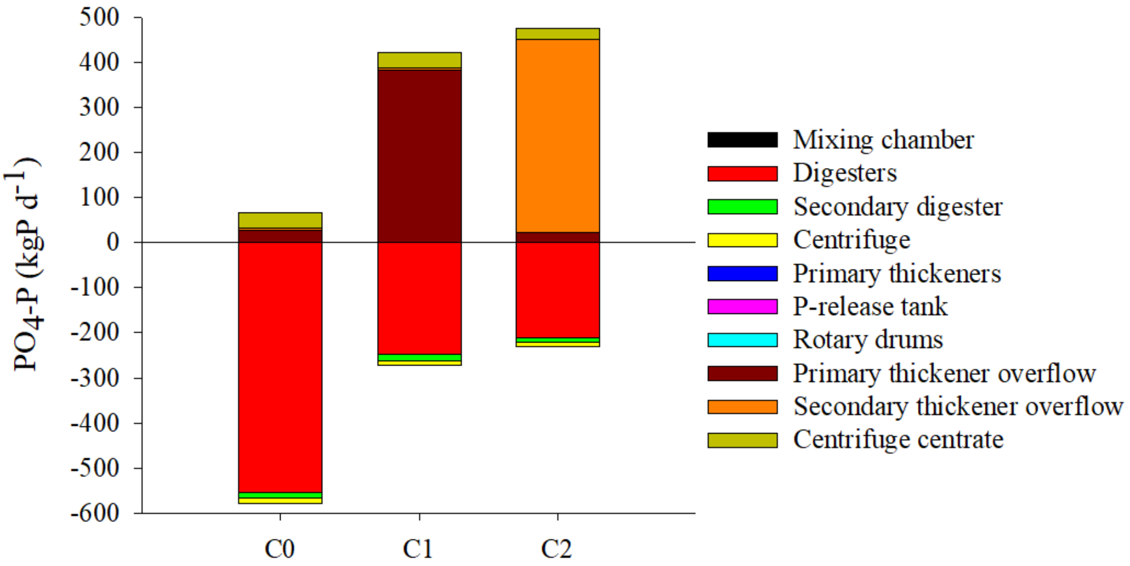

Figure 7. Precipitation of PO_4 in the sludge line (negative values) and available (positive values) in the liquid streams generated under the different configurations.

Figure 8

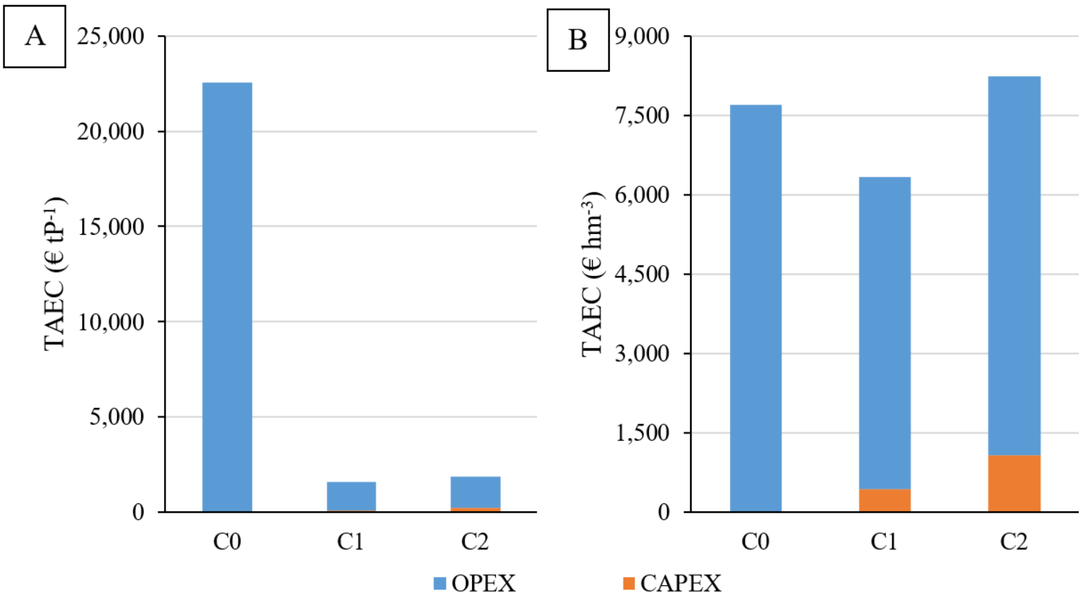

Figure 8. Total annual cost estimations for the different configurations (a) per ton of PO_4-P in the recovery stream and (b) per hm^3 of treated wastewater.

Figure 9

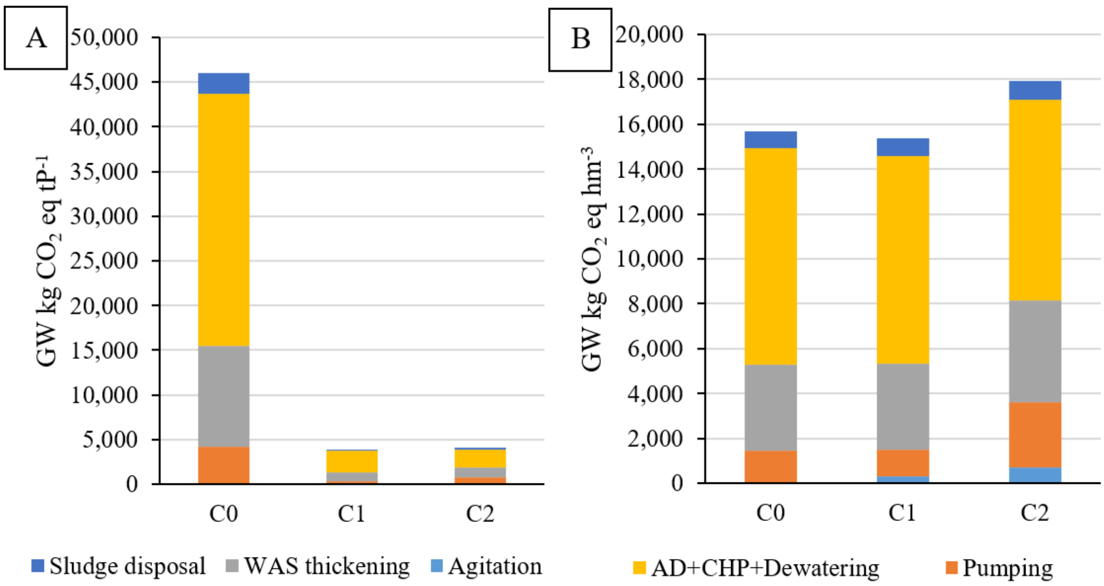

Figure 9. Global warming impact of the three configurations (a) per ton of PO_4-P in the recovery stream and (b) per hm^3 of treated wastewater.