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Full Research Article

The circular economy and agriculture: new opportunities for re-using Phosphorus as fertilizer

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Abstract. The increasing demand of phosphorus (P) worldwide is posing important challenges on the market stability of fertilizers. Extracting more P would not guarantee high P quality and low prices. Globally, only the European Commission, in a recent document about the Circular Economy strategy, has begun to address the challenge of the dependence on phosphate rock. Based on a simple circular economy theoretical framework, this paper proposes an impact analysis of the use of recycled P as a substitute of chemical P fertilizers. Two new technologies applied to retrofit existing wastewater treatment plants (WWTP) are considered: Moving-Bed Bio-Reactors and Struvite Crystallization Modules. The analyses indicate that the introduction of these technologies prove to be economically sustainable for specific levels of inhabitant equivalent (IE) and that the profitability of struvite, as a substitute of chemical P, increases with increasing levels of P fertilizer prices and for increasing sizes of WWTPs.

Keywords. Circular economy, agriculture, phosphorus, fertilizer

JEL codes. O13, O33, Q15, Q52, Q56, R11

1. Introduction

Agriculture is the main user of phosphorus (P) as fertilizer (P_2O_5), with a share of about 90% of total phosphate rock (PR) consumption (Cordell *et al.*, 2009; Brunner, 2010; Van Vuuren *et al.*, 2010; EC, 2013). This datum would not provoke interest if P reserves were infinite and P extraction costs were marginally decreasing. The actual consumption dynamics of P, however, indicate that P reserves are not sufficient to meet the growing P demand worldwide, even at lower prices (Scholz *et al.*, 2013), and that improving P extraction or investing in searching new reserves might yield higher costs and lower quality of P fertilizers (Van Vuuren *et al.*, 2010). The consequences of such dynamics will negatively affect food security worldwide, especially in developing countries where the consumption of P fertilizers would likely prove to be economically unsustainable. On the other hand, the

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increased use of low-quality P fertilizers would have reduced effectiveness on crop productivity and would be responsible for increased water pollution, higher risks of eutrophication and wider contamination of heavy (radioactive) metals (von Horn *et al.*, 2009).

Although the increasing awareness about the likely impacts of P dynamics has led decision-makers to devise measures aimed at optimizing P-fertilizers use and reducing water pollution (Neset *et al.*, 2012), there is a perception that it may be impossible to significantly reduce dependence on PR as a source of P-fertilizers, as demonstrated by the absence of an international agenda addressing the issues of food security and environmental change (Cordell *et al.*, 2009). Beyond the theoretical/empirical interest of the scientific community and some actions at country level (Cordell *et al.*, 2009), only the European Commission (EC) seems to show a concrete interest in facing the issue from a wider political perspective by adopting a circular economy strategy (EC, 2015), in which the willingness to reduce the dependency on PR is demonstrated by the adoption of an action plan addressing, *inter alia*, the need to improve the recovery rate of secondary raw materials from wastes.

Based on the recent policy evolutions of the EU, and with the intention of contributing to the development of a new institutional setting inspired by a circular economy approach, the paper proposes first a general theoretical explanation of the circular economy concept as applied to phosphorus resources followed by a preliminary analysis to estimate the cost-effectiveness of two innovative technologies for the recovery of P from wastewater treatment plants (WWTP), based on Moving-Bed Biofilm-Reactor (MBBR) and Struvite Crystallization Modules (SCM), respectively. As a follow up step, the paper provides a territorial economic impact analysis of the use of struvite as a substitute of chemical P fertilizers by assuming that phosphorus recovery is an economically feasible process as long as environmental benefits[1] are considered (Hernandez-Sancho *et al.*, 2010). The chosen territory is the entire administrative area of the Emilia-Romagna region, for which data on fertilizer use and on urban WWTPs are readily available.

The paper continues in the next section with a brief presentation of the policy context and a literature review on economic analyses of P recovery. A theoretical framework introducing the concept of circular economy applied to the case of P-fertilizers is presented in the third section. Methodology and data are presented in the fourth section, followed by the presentation of results, a discussion and conclusions.

2. Background

2.1 Policy context

The awareness of the essential nature of phosphorus for life has increased significantly in recent years because the uncertainty in future availability, quality and especially price stability of P (and P fertilizers) is matched with population growth, the overhanging food security and especially the impacts of chemical P fertilizers on water resources. Interest

[1] Environmental benefits are what, in economic terms, are known as positive externalities, i.e., benefits that occur when the actions of firms and individuals have an effect on people other than themselves without economic compensation.

has, therefore, grown in facing the issue from a policy perspective, at least in the European Union (EU), with the adoption of a circular economy package based on the recovery and reuse of waste materials with the twofold objective of reducing the production of primary materials and the impact on the environment. However, given the scarce interest at the international level in the evolution of P, the action of the EU toward promoting the circularity of the economic systems represents an enormous step forward for setting the future policy agenda according to a sounder economic, social and environmental sustainability perspective. The new European policy course, launched in 2014 with the Europe 2020 strategy, has set the ground for driving the EU toward a circular economy framework. Major policy weight, indeed, has been attributed to the capacity of Member States (MS) to reduce the pressures and impacts of economic and social activities on natural resources, especially soil and water resources. This choice represents the 'natural' continuum – and in certain aspects the completion – of a fragmented policy path, initiated decades ago for facing single issues, but which evolved with the emanation of several directives concerning environmental protection (or the limitation of environmental pollution), such as the Urban Waste Water Treatment Directive 91/271/EEC (UWWTD), the Nitrates Directive 91/676/EEC, the Water Framework Directive 2000/60/CE (WFD), the Fertilizers Regulation 2003/2003/EC, the Waste Framework Directive 2008/98/EC. As stated by the EC (2015), a new paradigm for enhancing the competitiveness of the EU economy without impairing the environmental and natural resources is only possible by "closing the loop" and starting a transition process towards *a more circular economy*. Indeed, this statement introduces the circular economy plan adopted in 2015 by the EC with the specific objectives of reducing EU dependence upon the production and import of raw materials, highly subject to both scarcity of resources and price variability. The intention to drive an economic system towards the circularity of materials (especially secondary raw materials) largely rely upon the innovation capacity of the EU, based on the vast existing technological know-how, the enhancement of research potential (through the Horizon 2020 programme) and the financial support for knowledge-based investments through dedicated structural funds.

Within the broad framework of the action plan for the circular economy, a section is dedicated to recovering and recycling materials, aiming at generating a new lifecycle for wastes and new trading opportunities for secondary raw materials. Phosphorus is included in this section as a secondary raw material recovered from sewage and food wastes and recycled through organic and other typologies of fertilizers (e.g. struvite). However, the route for full implementation of recycled P must pass through a further and parallel process for setting the production and trade standards of secondary raw materials. In fact, in 2015, the EC received a working/position document from the Fertilizer Working Group (2014), in which the opportunities for fostering market access for recycled phosphorus is proposed as a valid alternative to the current and future perspective of the dependence of the EU on chemical P fertilizers. This document, *de facto*, proposes the modification of Regulation 2003/2003 on fertilizers in order to allow for the commercialization of recycled P. The opportunity to recover and recycle phosphorus has emerged as a complementary action of the previous EU policies on the reduction of pollution loads in water resources, specifically the UWWTD and WFD, which open to the possibility of reusing treated wastewater.

In particular, the UWWTD states that all generated wastewater agglomerations of size between 2,000 and 10,000 inhabitant equivalent (IE) must set up collection and treatment systems by December 2005. Therefore, one of the main challenges for European authorities in the achievement of the good ecological status of water bodies, as stated in the WFD, is to implement the appropriate treatment of wastewater, even in small agglomerations. This objective is of crucial importance, especially in some territories where urbanisation is characterised by extensive and less populated areas and where investments in sludge networks and advanced treatment plants are relatively costly and financially unsustainable.

This is the case, for example, in Emilia-Romagna, where there are about 1,900 treatment plants (89% of total across the region) serving less than 2,000 each. Out of 1,900, about 73% (1,400) involves only the first treatment stage (Imhoff tanks) able to retain, at the exit, only the 10% of the introduced P charges. Indeed, such conditions, which are common across Europe, ought to convince the EC about the need to identify additional sensitive areas, and their related catchments, characterised by punctual and critical pollution loads. This, in turn, would imply the need to upgrade the treatment facilities dedicated to a significant quantity of discharges and the development of new facilities in the near future. In this context, it is crucial to identify the most feasible technologies, from an integrative point of view, to be tackled through new wastewater management projects, designed according to each specific wastewater scenario[2].

2.2 Literature review

The fact that about 90% of global P production (extraction from PR) is used in agriculture as fertilizer has generated noteworthy interest with respect to the dependence of life on PR such to question the long-run sustainability of the resource. Indeed, although PR are generated through natural cycles, in terms of social and environmental sustainability, P is considered a finite resource due to the disparity between the rates of natural generation and human use.

Since most of the P absorbed by plants and animals could be returned to agriculture, for example through crop residues, manure and WWTP effluents, P can be considered as a renewable resource, provided that it can be recovered and returned to agriculture as a convenient and non-polluting input. However, in order to meet the growing global demand for food, agricultural production has become more specialised and – for many commodities – globalised, implying the unlikely return of the P extracted to the soil of production.

Another important cornerstone of the sustainability of P production is the fact that it is essential for life. Indeed, the growing global demand for food has given rise to increased demand for P fertilizers, globally. The main – and relatively less costly– source of P fertilizers is the Phosphate Rocks (PR), the known reserves of which are located mostly in Morocco, the United States and China. Extracting P fertilizers (mostly P_2O_5) from PR (and distributing it worldwide) is cheaper than returning P to soil and waiting for it to be available to plants. Therefore, from an economic perspective, P needs to be treated as a non-renewable resource. This perspective is strengthened by the fact that P has no substitutes in nature and there is no way to obtain it by synthesis.

[2] By wastewater scenario we intend the combination of pollution sensitiveness of the areas (which depends upon the catchments location) and the typology and charge of pollutants.

The main implication deducible from these considerations, together with the diffuse distribution of relatively small treatment plants unable to retain phosphorus, is that the most practical alternative to non-renewable P would be to invest in technologies capable of recovering 'used' P and to make it available for use. In such a perspective, P recovery is an economically feasible process as long as environmental benefits are taken into consideration (Hernandez-Sancho *et al.*, 2011).

Taking as an example the data on the balance of P charges (ratio between P in entry and exit) in Emilia-Romagna (ARPA, 2012), the level of effectiveness for P retention in WWTPs increases with the level of treatment[3]. In particular, for the first stage the retention level is 10%, whilst for second and third is about 57%and 85%, respectively. However, the total IE served by each typology represents 2%, 11% and 87%, respectively, indicating that most P is still lost in low- and medium-sized treatment plants. From these data, it can be deduced that innovations for the recovery of P and improving water treatments are mostly needed to retrofit the low- and medium-sized plants, in order to raise the effectiveness of P recovery and to lower the investment and operational costs.

In fact, one of the promising technologies[4] for the recovery of P from WWTP, as well as from other types of effluents, is represented by a reactor to add as a module into existing treatment plants, and capable of precipitating P into struvite[5], a mineral composed of P, N and Mg (Booker *et al.*, 1999; Laridi *et al.*, 2005).

Struvite can be used as a slow release fertilizer at high application rates without the danger of damaging plant roots (Bridger *et al.,* 1962; Lunt *et al.,* 1964). Likewise, because struvite is insoluble in natural water, eutrophication problems and infiltration in groundwater are prevented, hence representing another advantage in its use as fertilizer. Therefore, phosphorus recovery as struvite can be seen as a basic process for achieving sustainable development.

The other valid alternative for recovering P from WWTP is represented by an innovative technology based on the concept of microbiological accumulation and digestion of waste P, capable of being accumulated as an organic compost. Such technology is implemented by the means of a moving-bed biofilm reactor[6], inoculated with specific micro-bacterial material, on which the effluent is left to stream (McQuarrie *et al.*, 2011; Barwal *et al.*, 2014).

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[3] The available (and traditional) technology organizes the wastewater process in three subsequent levels: primary, allowing the sedimentation of large and solid waste (Imhoff tanks); secondary, providing biological oxidation and a low level of microbial disinfection; tertiary or advanced, with filtration and a high level disinfection (mostly using advanced technologies such as UV ray). The higher the level the less is the risk for human health in using treated wastewater. According to the standard set by Environment Protection Agency (EPA) in the USA, wastewater from primary depuration must not be used; the second level allows for superficial uses such as furrow irrigation for orchards and industrial crops not intended for human and animal consumption, for recharging non-potable water-tables and the preservation of humid habitats and minimum vital flows of rivers; the tertiary level allows for irrigation of crops intended for human consumption, the recharge of water bodies for bathing (or more generally for recreational uses) and the recharge of potable water-tables (UNEP, 2011).

[4] Promising in terms of potential large diffusion.

[5] The presence of struvite as a mineral precipitate into the WWTP has been discovered because it usually accumulates into the pipes creating obstacles to the flow of the treated water. In order to keep the plants working in an efficient way, the pipes need to be continuously cleaned from struvite, implying additional maintenance and operating costs.

[6] MBBR is composed by a bio-plastic film, inoculated with specific microorganisms capable of catching and retaining selected materials. The films become carriers once inoculated. Such carriers work inside a tank and they are set in movement when the wastewater flows inside the tank. In the last a stage the whole system becomes a reactor (a system in which a chemical-physical reaction occurs).

The costs of recovering phosphate during wastewater treatment can be calculated at 2€ per kg P as a minimum and may total more than 8€ per kg P under specific conditions (Dockhorn, 2007; Dockhorn, 2009 Schaumm, 2007). PR[7] in the United States is sold at between $35 and $50 per tonne (US Geological Survey) depending on purity. These values show that there are no economic incentives for implementing P recovery technologies in the wastewater sector since it is cheaper for the fertilizer industry to continue using PR as feedstock.

However, it is important to remember that the recovery of P from wastewater involves noteworthy environmental benefits because it prevents eutrophication in the receiving environment, and increases the availability of a non-renewable resource. In this light, when the economic feasibility of projects with environmental effects is analysed, beyond the internal impacts, environmental benefits should also be considered (Hernandez *et al.* 2006). It follows that the economic performance of new technologies potentially improving environmental conditions should be evaluated in terms of their pollutant abatement capacity and total benefits, including the related monetized environmental benefits.

3. Theoretical insights surrounding the circular economy concept applied to phosphorus resources

As previously stated, P is an essential building block of life. Indeed, it is an irreplaceable element of modern agriculture, as there is no substitute for its use either as animal feed or as fertiliser (EC, 2013). The reserves of PR in the European Union (EU) are very limited and not sufficient to meet the internal demand of P, especially fertilizers. Hence, given that the EU is a net importer of P, the prospective growth in EU domestic fertilizer demand would determine, beyond import dependency, a resulting rise in the extraction costs of P from PR.

Moreover, the current use of P, especially as chemical fertilizers, is partly inefficient as the share not taken up by plants leaks out causing water pollution and eutrophication. Such externalities constitute the increasing costs to be borne by society. Therefore, the prospective increase in the demand and externalities of chemical P-fertilizer use would imply the likely increase in private and social costs. This scenario provides for incentives for the exploitation of alternative sources of P, such as the one presented in this paper, and the related theoretical implications concerning the circular economy framework.

A possible interpretation of the depicted scenario is offered in Figure 1 where the demand for P is satisfied by two alternative sources of supply, characterised by two different technologies: P extracted from PR, namely *technology A*, and P extracted from secondary raw materials (recycled from treated wastewater), namely *technology B*. It is assumed that the marginal costs required to supply P through *technology A* increases with increasing demand, while the costs to bear with *technology B* are independent on the quantity demanded.

Panel 1 of Figure 1 describes a scenario in which the current demand for P is satisfied exclusively by *technology A*, as the supply costs for *technology B* are relatively higher. On the other hand, panel 2 of Figure 1 describes a scenario where the prospective higher

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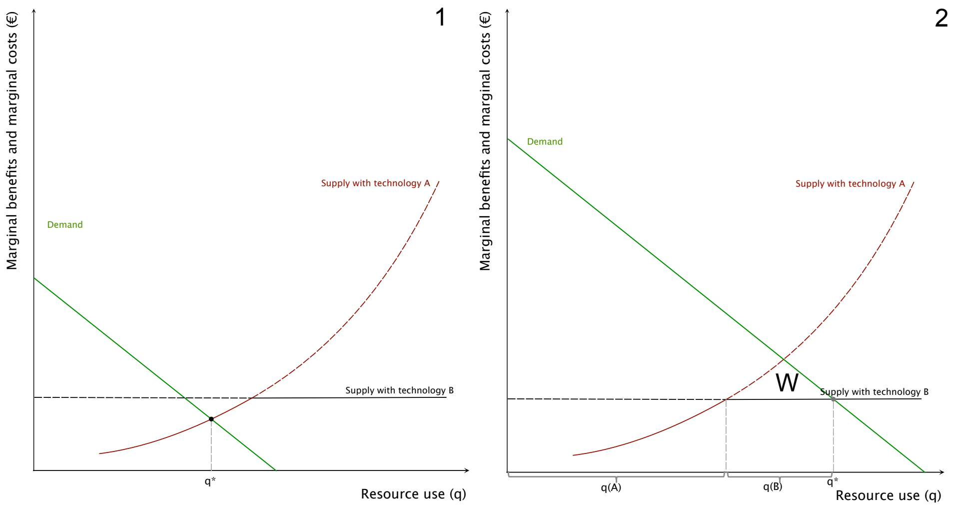
[7] High quality PR have a title of about 30% in P_2O_5.

Figure 1. Welfare impacts caused by the implementation of a circular economy strategy for phosphorus resources.



● – Social optimum
q(A) – Resource consumption with technology A
q(B) – Resource consumption with technology B
W – Welfare gain obtained by combining supply technologies

demand for P is satisfied by both *technology A* and *B*. In the context of the latter scenario, P would be supplied by *technology A* up to a given amount of resource use, say q(A), while the residual amount of the resource, required to satisfy the entire demand, say q(B), would be supplied through *technology B*.

It follows that level q(A) could be interpreted as the threshold value below which the resource is supplied solely through traditional technologies and above which the resource is supplied also through new technologies, while q* remains the point identifying the optimal level of overall phosphorous use.

Scenario 2 shows two interesting features. First, by decreasing the cost of technology B, we reach a solution in which the social optimum is given by a combination of the two technologies rather than the substitution of one with the other. We can expect that the combination will be more in favour of technology B, the more its cost is reduced. Second, by reducing the cost of B, the overall amount used increases. This could be interpreted as a sort of rebound effect and should be considered carefully if an environmental effect is included in the model (which is not the aim of this paper).

The uptake of alternative technologies, therefore, seeks to modify the equilibrium points of resource usage, q*, and to induce related changes in private and social welfare. Indeed, welfare changes are determined by the cost loss brought about by the transition from the traditional to the alternative technology for the supply of phosphorus. It can be deduced that the supply of recycled P is feasible when it contributes to an increase in the welfare of society.

However, if a scenario of invariance of P-fertilizer demand is considered, a gradual increase in recycled P-fertilizer use might occur as well because of a reduction in the costs required to supply P with alternative technologies or by an increase in the supply costs of traditional technologies.

All of these conditions might be plausible to explain the coexistence of alternative technologies, but is not yet sufficient to guarantee the transition to a full circular economy strategy (sole use of recycled resources) because of lock-in effects (Zeppini *et al.*, 2014). Namely, a technology that is dominant in a particular application domain, could be resistant to competing alternatives even if the latter can be considered socially desirable. Scholars attribute this phenomenon mainly to the fact that the introduction of new technologies might not reach the critical mass of adopters causing the transition (Bikhchandani *et al.*, 1992; Arthur, 1989) or a critical price (Solomon, 2000).

4. Methodology

Cost effectiveness analysis (CEA) and cost benefit analysis (CBA) are the two main methods adopted for carrying out economic assessments. CEA usually compares monetary costs and physical benefits (i.e., nutrient recovery). CBA, for its part, compares monetarily valued costs and benefits (i.e. nutrient recovery and the shadow prices of water quality improvement). CEA avoids the controversial monetization of intangible assets, such as the environment, and is usually designed for the comparative assessment of alternative measures, rather than for a clear-cut judgment on the feasibility of a project or a policy.

CBA is designed to assess the viability of the intervention, as it requires an estimation of costs and of both tangible and intangible benefits. This approach could also be adopted to deal with the theoretical background previously described as it allows for an assessment of the impacts brought about through the transition from traditional supply technologies to recycling. However, in the present study we assume that recycling is preferable to non-recycling, so the problem we face is to compare the two most recent wastewater treatment technologies. Although the two innovative technologies are related to distinct phases of the wastewater treatment process and they can be combined together to upgrade current WWTPs, it is assumed that due to budget constraints the regulator seeks to identify the most cost effective option. In the following section, a cost effectiveness (CE) analysis has been implemented to compare the two alternative systems as regards the P abatement capacity, which, for the considered technologies, is equivalent to the recovery capacity[8]:

$$z = \max_{i,j}\left\{\frac{NC(x)_i}{A_i};\frac{NC(x)_j}{A_j}\right\} \tag{1}$$

[8] Recovery capacity is not the same as recycle capacity, however, both the considered technologies enable recycling. The step of recovery is limited to the capacity of the technology of retaining the P that otherwise would have been left in the effluent, while the recycling is the further step in which the recovered P is made available for being traded. As for the technology producing struvite, both steps coincide, while for MBBR the sludge obtained from the digestion might be subject to specific processes before being traded.

where *z is the recovery capacity, x* is the size of the treatment plant in terms of IE, NC(x) is the net cost obtained by the difference between costs C(x) and benefits B(x) and A is the level of P abatement for the two alternative options, i and j.

5. Data

The methodology has been implemented by simulating the uptake of the described technologies in the Emilia-Romagna region. The territory of Emilia-Romagna can be considered as a unique case study given the availability of reliable data, provided by ARPA (the Regional Agency for Environmental Protection) on fertilizer use as well as on the wastewater sector. Fertilizer prices have been extracted by the Chamber of Commerce of Modena (a province of Emilia-Romagna) where imported fertilizers are traded throughout the Emilia-Romagna region.

Table 1. Evolution of fertilizer consumption in the Emilia-Romagna region.

Fertilizers (000 t)	Period	Nitrate	Phosphatic	Potassic	Compost	Organic
	<2007	227	48	9	123	38
Average	2007-2011	178	39	8	104	52
	>2011	219	25	7	91,5	64
Δ% total	2004-2011	-15.11%	-61.40%	-37.50%	-33.85%	50.00%
Δ% excluding crises	pre07-post11	-3.67%	-47.55%	-19.23%	-25.81%	68.42%
Δ% per ha	2004-2011	3.19%	-55.32%	-52.00%		

Source: own elaboration on ARPA data (elaborated from ISTAT).

Table 1 shows the evolution of fertilizer consumption in Emilia-Romagna during the period 2004-2013. Given the occurrence of the global economic crises, which has resulted in significant price volatility and a price peak between 2007 and 2011 (Table 2 and Figure 1), the series has been truncated in two sub-periods in order to assess the evolution by isolating the crises period.

Chemical fertilizers have recorded a gradual reduction in consumption during the considered decade. In particular, by excluding the crises period in which the reduction was largely induced by the price peak, it can be noted that Nitrate fertilizers, for which substitutability makes it an elastic product, the reduction reached about 4%, while it has been more accentuated for the phosphate with a fall of about 48%, indicating a very low elasticity.

On the contrary, organic fertilizers, having a function more related to an amendment than a fertilizer, have experienced a consumption increase of about 70%. Such data indicate an actual change in the regional agricultural sector toward a major use (attitude) of substitutes for replacing chemical products.

Table 2 and Figure 2 clearly show that, excluding the crises period of 2008-2011, the price for chemical P in Emilia-Romagna region has steadily increased, from 230 €/t in January 2007 to about 500 €/t in October 2015, an increase of about 100%. Nitrates fertilizers, for their part, recorded a positive increase in prices up until November 2011 and maintained a steady pace thereafter at an average level of 350 €/t.

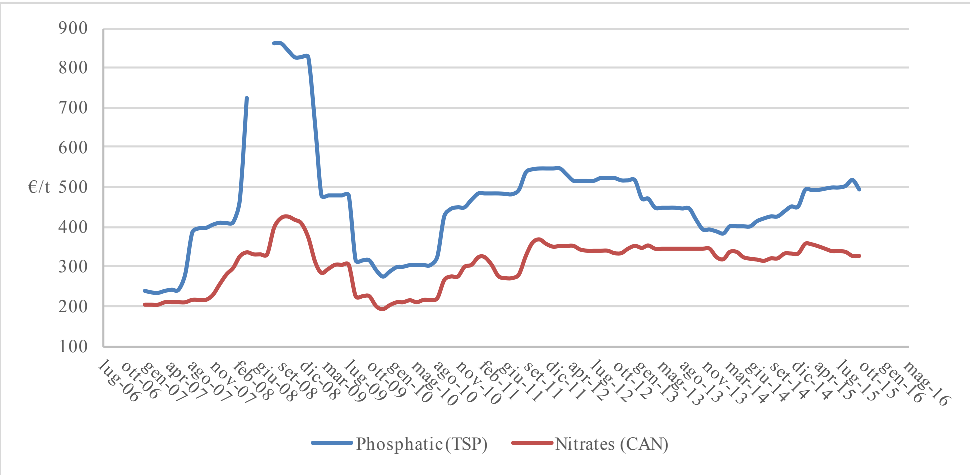
The high price volatility recorded during the period 2007-2011, with a peak in 2008 at 870 €/t, has yielded an average reduction in chemical P fertilizer consumption of about 19%. The estimated elasticity of demand of about -0.19 indicates that chemical P fertilizer is inelastic with respect to price variations.

Table 2. Evolution of fertilizer prices in the Emilia-Romagna region.

		Nitrates (CAN)	Phosphatic (TSP)
	<Jun 2007	209	237
Average	>Jul2007-2010	280	463
	>2010	334	474
Δ%	2007-post 2011	59.74%	100.56%

Source: own elaboration on ARPA data (elaborated from ISTAT)

Figure 2. Evolution of N and P chemical fertilizer prices in the Emilia-Romagna region.



Source: own elaboration on CCIAA Modena.

Such evidence is coherent with the typology of a product unique to chemical P fertilizers obtained from a non-renewable (fossil) resource and exchanged in markets characterised by both monopolistic competition and natural monopoly forms (few States selling similar goods).

As mentioned in the previous paragraphs, the potential substitution of chemical P fertilizer could come from the recycling of P present in sludge and WWTP effluents. In order to evaluate the recovery potential, an overview of the WWTP system in Emilia-

Romagna region is presented. The Emilia-Romagna region, in the last decade, has considerably invested in the improvement of the efficiency and effectiveness of WWTPs and related networks. Data collected biannually by ARPA since 2005 indicate the realisation of economies of scale due to specific investments in infrastructure, for modernization, new urban sludge networks and connections to treatment plants, which have augmented the treatment capacity and increased the effectiveness of the plants with the resulting improvement to the efficiency of the treatment system. Further steps forward have been realised by improving the nutrient recovery capacity and the return of the muds into the economic system, especially to agriculture. Such basic condition results to be highly favourable for more important and longer-run engagements in investments for developing new technologies, to be applied, in particular, to nutrient recovery.

Table 3 presents the situation highlighted by ARPA in 2012 with regard to WWTPs in the Emilia-Romagna region by typology (expressing the current level of technology) and the classes of urban conglomerations served. During the period 2005-2012, as concerns systems greater than 2,000 IE, reductions in the number of agglomerates, in the number of plants (from 245 in 2005 to 222 in 2012) and industrial IE have been recorded, with improvements in the efficiency and reduction in environmental pressures.

Table 3. WWTPs in the Emilia-Romagna region per typology and conglomeration classes in 2012.

Urban conglomeration classes	Number of plants (per typology)				Project potential (per typology)			
(IE)	I (n°)	II (n°)	III (n°)	tot (n°)	I (IE)	II (IE)	III (IE)	tot (IE)
0 – 1,999	1,377	469	31	1,877	174,515	311,940	47,045	533,500
2,000 - 10,000	0	65	75	140	0	397,515	531,620	929,135
10,001 - 15,000	0	2	21	23	0	18,500	457,700	476,200
15,001 – 100,000	0	2	37	39	0	96,000	1,684,400	1,780,400
>100,000	0	0	20	20	0	0	4,681,333	4,681,333
Total	1,377	538	184	2,099	174,515	823,955	7,402,098	8,400,568

Source: ARPA Emilia-Romagna.
Note: Typology I stays for primary, II for secondary and III for tertiary (or advanced).

Indeed, since 2005, there has been a noteworthy reduction in the pollution loads at the exit stage of the WWTP. In particular, recent data shown in Table 4 point to an improvement in abatement capacity[9] from 2009 to 2012 of about 4.2% and 2.6% for secondary and advanced WWTPs, respectively, a stagnation for primary plants and a reduction for plants equipped with *DeN* and *DeP* technologies.

Improvements in the reuse of muds resulting from the treatment systems have been recorded since 2007. As shown in Table 5, the reuse of muds in agriculture has increased by 40%, contributing to the substitution of costly chemical fertilizers and a reduction in dumping.

[9] Abatement capacity is the relative difference between pollutants loads entry and exit.

Table 4. 2009-2012 differences in P loads per WWTP typology in the Emilia-Romagna Region.

	Plants (n°)	Flow $(m^3/y)/10^3$	Projected IE (IE)	Treated IE (IE)	Nitrates loads		Phosphatic loads		Abatement capacity
					entry (t/y)	exit (t/y)	entry (t/y)	exit (t/y)	
Primary (I)	1,377	7,782	174,515	85,286	342.4	291.1	49.8	44.8	10.0%
Secondary (II)	538	44,694	823,955	535,748	2,018.7	637.7	247.6	105.3	57.5%
Advanced (III)	184	386,585	7,402,098	4,402,509	17,686.8	4,060.5	2,385.2	356.4	85.1%
Δ% 2009-2012									
I	-6.39%	-17.95%	-16.35%	-17.52%	-17.53%	-17.51%	-17.55%	-17.50%	-0.6%
II	4.67%	-3.02%	6.30%	8.29%	15.84%	-13.00%	7.79%	2.13%	4.3%
III	3.95%	-10.47%	1.37%	1.09%	7.73%	-11.00%	8.18%	-5.29%	2.6%
Denitrific (DeN)	*80*	*29,775*	*463,540*	*324,399*	*1,179.8*	*302.6*	*153.9*	*58.6*	*61.9%*
Dephosphat (DeP)	*9*	*6,480*	*137,800*	*109,278*	*256.9*	*89.8*	*27.7*	*8.5*	*69.3%*
DeN + DeP	95	*350,330*	*6,800,758*	*3,968,832*	*16,250.0*	3,668.1	2,203.6	289.3	86.9%
Δ% 2009-2012									
DeN	9.59%	-5.43%	4.76%	10.92%	-2.41%	-6.78%	-10.31%	-4.72%	-3.5%
DeP	-10.00%	-88.33%	277.02%	-84.01%	-88.31%	-90.99%	-90.07%	-84.93%	-13.1%
DeN+DeP	*1.06%*	*1.61%*	*16.56%*	*17.45%*	*24.88%*	*13.18%*	*25.61%*	*11.91%*	*1.9%*
Total	**2,099**	**439,061**	**8,400,568**	**5,023,543**	**20,047.9**	**4,989.3**	**2,682.6**	**506.5**	

Source: ARPA Emilia-Romagna.
Note: DeN and DeP refer to treatment plants equipped with specific technologies adopted for the improvement of the N and P recovery.

Table 5. Quantity of reused and disposed muds per typology in the Emilia-Romagna region.

	Reuse		Disposal			Total
	Agriculture	Other reuse	Dumping	Incineration	Other	
2007	8,309	9,514	33,550	8,043	8,379	67,794
2009	8,766	14,874	25,817	7,840	0	57,297
2012	11,860	19,555	14,751	10,071	0	56,237
Δ %	40.51%	67.51%	-72.82%	25.87%	-100.00%	-20.17%

Source: ARPA Emilia-Romagna.
Note: values in tonne/yr of dry substance.

The most important aspect emerging from the presentation of the data in Tables 4 and 5 is the potential for achieving the abatement of P in primary plants and the large room for improvement in the abatement of secondary and advanced plants by employing the proposed new technologies. In support for such statement, a simulation of the possible market prices for muds is realized on the basis of the relative (to the P title[10]) prices of chemical P fertilizer, conditional on a simple analysis of market opportunities for the development of fertilizer products based on muds, such as soil improver and compost.

Table 6. Simulation of market price for WWTP muds relative to the title in P and N and market opportunity

Muds from WWTPs (2007-2012)		Fertilizers regional market – since 2007	
Abatement	N from 65 to 75%	Chem fertilizers prices:	N raised by 60%
	P from 68 to 81%		P raised by 100%
		Fertilizers consumption:	N (chem) stable
Reuse (agriculture and compost) from 30 to 56%			P (chem) reduced by 50%
			Organic raised by 70%
Value of P	0,5 – 0,6 €/Kg	Value of P (chem – P_2O_5)	1,1 €/Kg
Value of N	0,6 – 0,7 €/Kg	Value of N (chem)	0,7 €/Kg

Source: own elaboration on ARPA Emilia-Romagna and CCIAA Modena data.
Note: *Chem* stays for chemical.

Indeed, as illustrated in Table 6, while chemical P consumption declined during the period 2007-2012, organic fertilizers and the reuse of muds in agriculture (as compost or soil improver) increased in response to the price rise of N and P. In particular, it is important to highlight that the increase in reuse of muds occurred in concomitance to the improvement in the P abatement capacity of muds, which implies a higher level of P returned to the land. From all such information it would be reasonable to deduce that there is evidence of a substitution of muds and organic fertilizers for chemical P ferti-

[10] The title of P is the physical content of phosphorus in the chemical compound, expressed in percent terms.

lizers. On the basis of such a deduction, a simulation of the value of P contained in the muds would provide for a reference for the comparative assessment of alternative P sources obtained by innovative technologies.

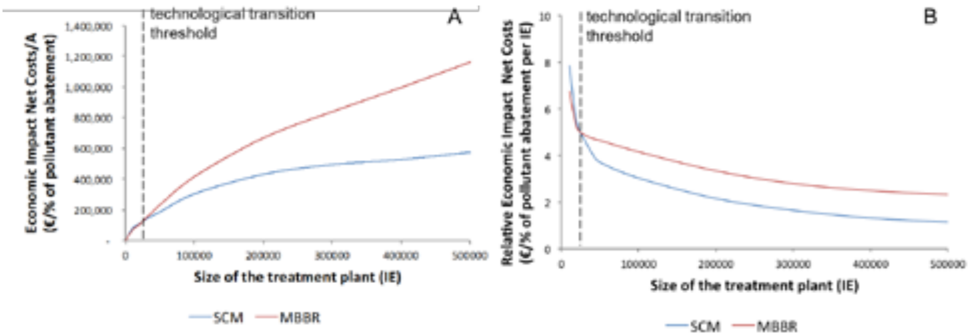
Supposing that, in terms of nutrient availability and chemical title, mud is a full substitute for P and N chemical fertilizers, the simulation results, shown in Table 6, indicate that P components of muds present a half relative value with respect to chemical fertilizers, while the value of N does not vary.

This simulation, therefore, provides preliminary information regarding the relative advantage in using P from muds as a possible substitute of chemical P. A second consideration is related to the absence of heavy metals in muds, such as uranium and cadmium, which are instead present in chemical P extracted from PR and which, inevitably, remain in soil and leach into water.

6. Results

The results of the investment and costs analysis are depicted in Figure 3. The adoption of MBBR technology (red line) does not significantly alter the maintenance and operating (M&O) costs up to the size of 10,000 IE (for technical reasons[11]), while beyond such a level the costs follow an exponential increase. On the other hand, the results show that the costs for the SCM (blue line) marginally decrease as the size of WWPTs increase (in terms of IE).

Figure 3. Economic impact trends of two alternative treatment processes with increasing size of the treatment plant: Absolute (A) and Average (B) values.



Source: own elaboration.

More specifically, Figure 3 shows the trend of the cost-effectiveness indicators per increasing size of the treatment plants for both innovative technologies. Comparing both innovative technologies, MBBR and SCM, Panel A and B show the estimated evolution of the costs-pollutant abatement ratio for increasing size of the WWTP. In particular, the

[11] The technical efficiency of MBBR depends on the flow of wastewater. In particular, MBBR modules can efficiently treat wastewater up to a predetermined flow. This implies that higher flows need to be split and treated by more than on module of MBBR. MBBR technologies do not show increasing returns to scale.

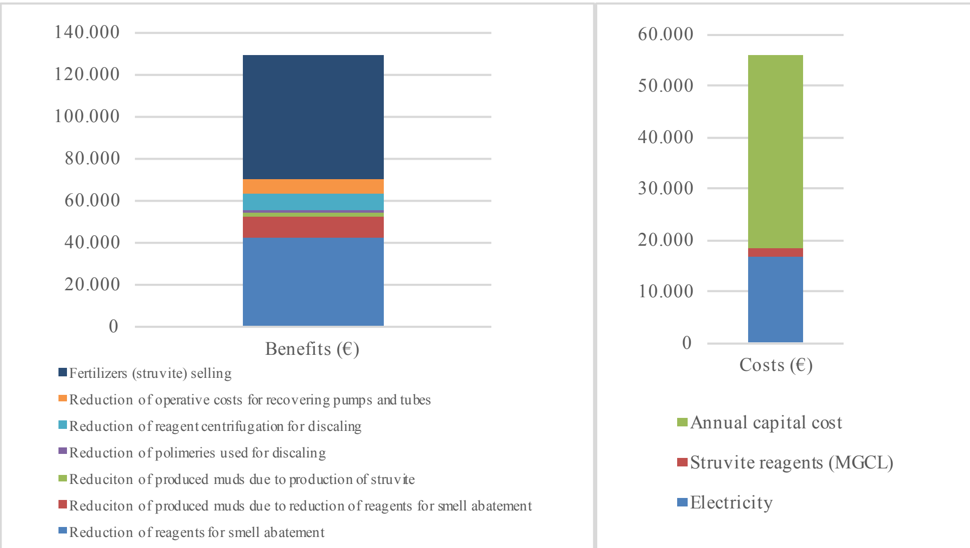
abatement in Panel A is measured in per cent levels, regardless of the WWTP size, while in Panel B it is measured in per cent referred to IE. The graphs reveal the existence of a threshold for a size of about 10,000 IE, below which MBBR proves to be more cost-effective and above which the SCM is more cost-effective.

In the Emilia-Romagna region, as exposed in Table 3, about 70% of the treatment plants are smaller than the estimated threshold, but do not reach the 5% of the wastewater treated in the region. This consideration, on one hand, implies the wider opportunity offered by the new technology to retrofit primary stage (small) plants at relatively low costs, improving P abatement and reducing pollution loads on water resources. On the other hand, however, the scarcity of technical information does not allow us to elaborate a more comprehensive cost-benefit analysis on the use of MBBR.

For the SCM, instead, a preliminary cost-benefit analysis has been elaborated on the basis of both the presented technical data at the regional level and the technical data of the SCM, for a plant size of 120,000 IE.

Figure 4 indicates the relative costs and benefits for the employment (retrofit) of a SCM based on a secondary depuration stage WWTP (about 55% abatement of P). The annual benefits are computed by accounting for the reduction of operative costs, mainly due to lower maintenance and operations, the relative reduced mud production and the revenue from the sale of struvite as fertilizer. The costs, instead, take into account the annualized capital costs, the specific factors needed to separate the struvite and the electricity. The balance returns positive, especially for technical reasons due to the fact that the struvite is a direct residue of the treatment procedure that obstructs the tubes and, hence, needs to be removed (a cost for the depuration procedure).

Figure 4. Costs and benefits of the use of a SCM in a 120,000 IE plant.



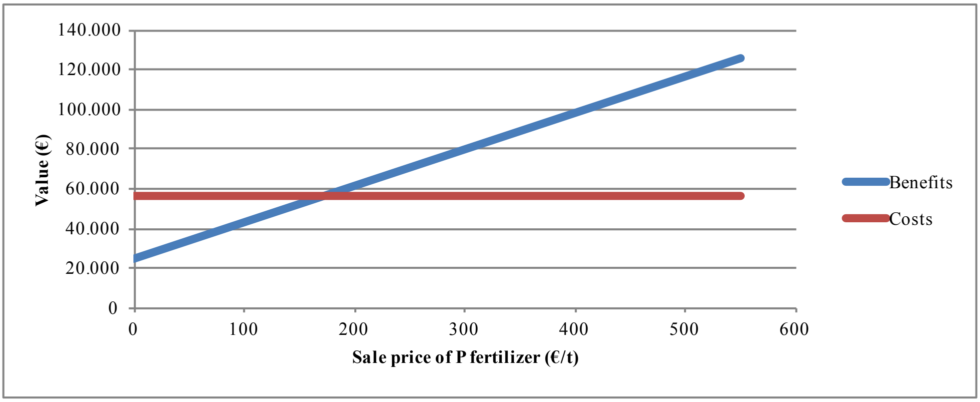
Source: own elaboration.

Based on the cost and benefit estimations for struvite production and by allowing for a range of possible prices for P fertilizer, a market-based cost-benefit analysis is provided in Figure 5.

As shown, a net convenience in investing in a SCM retrofit for large-sized WWTPs might be obtained for prices of P fertilizers greater than the 150-200 €/t range. Over such range, the profitability should linearly increase for higher values of P prices.

This simulation, together with the results of the CEA, implies non-linear profitability of installation of SCMs for increasing WWTP size.

Figure 5. Market-based cost-benefit analysis for the employment of a SCM in a 120,000 IE plant.



Source: own elaboration.

With regard to the adoption of SCM, a concise representation of the results could be expressed in a twofold perspective: the production costs (transformation value) of struvite diminish for increasing sizes of WWTPs (in terms of IE) and the relative profitability of installing (retrofitting) a SCM, given by the substitution value of the struvite, increases for increasing prices of chemical P-fertilizers.

Having estimated the economic value of the struvite on the basis of the value of current P fertilizer title, the results of the profitability analysis indicate a substitutability rate of 1.2 (in terms of P title), which would be sufficient to cover about 2% of the regional territory. The environmental impacts of MBBR and SCM adoption can be identified in the significant reduction in P-nutrient leaching in water bodies, leading to a consistent reduction of environmental damage.

These considerations, however, make it possible to infer that, from an environmental perspective, the implementation of the chemical P removal processes is better off for the SCM than for the MBBR processes.

7. Discussion and Conclusions

The analytical approach and related results have shown that the implementation of innovative technologies applied to WWTPs (in particular for the introduction of SCM) have the potential to improve pollution abatement and to reduce abatement costs. Furthermore, the results indicate that the production of an alternative and sustainable source of P fertilizer, namely struvite, show decreasing marginal costs for increasing size of the WWTP (in terms of IE). This outcome supports the outlined theoretical framework of circular economy according to which the alternative supply of recycled P-fertilizers would become profitable and, hence, represent an environmentally and economically sustainable substitute to chemical P-fertilizers, inducing a virtuous cycle of consumption and production of fertilizers.

Although the approach is limited to specific conditions (P recovery from WWTPs only and from one region), the paper is able to demonstrate that, for an outlook of increasing food demand and the need for enhancing food security while preserving/improving environmental conditions, the investments in (social, economic and environmental) sustainable technologies might stimulate confidence in devising measures/policies that respond to a circular economy paradigm.

Improvements for this line of research might be possible by widening the scope of the analysis by including animal manure as raw material for P recovery and by considering a larger geographical territory.

Another factor that limits the analytical reach of the paper is the lack of specific technical information concerning the functionality of the MBBR technology, which has prevented a specific cost-benefit analysis.

The main message this article intends to convey is that economic approaches, especially within the framework of the circular economy, that are employed to evaluate environmentally beneficial innovative alternatives, should play a greater role in contributing to satisfy environmental targets, in an economically sustainable manner, through the identification of the most promising innovations. This can be instrumental in determining a rational and efficient level of pollution/abatement in a watershed, which is considered to be a very important policy objective (Ancev *et al.*, 2006). According to Ancev *et al.* (2006), indeed, the efficient target should not be identified as a single number for any given watershed. Rather, its value will be dependent on the abatement options available, and on the policies targeted at incentivizing the adoption of those options.

As an expected consequence, it can be stated that instead of imposing exogenous environmental targets, policy makers ought to consider the costs of environmental damages and the abatement options available to the polluters. The targets can be set in such a way that the total costs - i.e., the sum of abatement and damage costs - are minimized (Ancev *et al.*, 2006).

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