

University of Nevada Reno

**Secure a sustainable, regional phosphorus supply:
Development of a strategic phosphorus reserve through recovery from wastewater**

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By

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ABSTRACT

Phosphorus (P) is vital to all living things. As the global population increases, so does the P demand. However, as a limited resource that can only be extracted by mining phosphate rock (PR), P has no substitute. This prompted the need to examine the recovery of P from water resource recovery facilities (WRRFs) as the majority of P consumed by humans ends up in WRRFs. Although the importance of P recovery has been established in the wastewater industry, the focus has been on recovery as struvite fertilizer for immediate use. The high implementation costs of struvite recovery coupled with low PR and chemical fertilizer prices make for an inadequate justification of the current approach of P recovery at WRRFs. There are three major streams in WRRFs where P can be recovered: sludge liquor, biosolids, and sludge ash. Using a simple payback, a 20-year present worth, and a Triple Bottom Line (TBL) analysis, this research systematically demonstrated that a promising way to encourage and justify WRRFs of all sizes to recover P is to establish a centralized facility where dewatered, undigested sludge from WRRFs of all sizes, within a region, would be transported to the facility for mono-incineration. Phosphorus would be extracted from the sludge ashes and precipitated to form calcium phosphate (Ca-P). The Ca-P, which has essentially the same composition as mined PR, would be deposited in a monofill as P reserve to be used in the future for diverse applications. One can argue that as of now, there is plenty of PR to be mined and the mining costs are inexpensive. Nevertheless, with more than 70% of the Earth's known reserves of PR located in Morocco and West Sahara, it is risky to jeopardize the nation's future P supply due to the possibility of geopolitical risks and supply chain

interruptions in/from these regions. By creating regional localized P reserves, the dependency on PR imports from foreign countries can be minimized.

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LIST OF ABBREVIATIONS

⁰ C	degree Celsius
Al-P	Aluminum phosphate (compound)
AlPO ₄	Aluminum phosphate
BEAM	Biosolids Emissions Assessment Model
Ca-P	Calcium phosphate (compound)
CapEx	Capital expenditure
CCME	Canadian Council of Ministers of the Environment
CH ₄	Methane
CHP	Combined heat and power
CO ₂	Carbon dioxide
CO _{2e}	Carbon dioxide equivalence
CSH	Calcium silicate hydrate
CSO	Combined sewer overflow
DAF	Design average flow
DCP	Dicalcium phosphate
DCPD	Dicalcium phosphate dehydrate
dmtpd	Dry metric ton per day
DT	Dry ton
EBPR	Enhanced biological phosphorus removal
ECHO	Enforcement and Compliance History Online
FBI	Fluidized bed incinerator
FeP	Iron phosphate (compound)

FeCl ₃	Ferric chloride
GHG	Greenhouse Gas
H ₃ PO ₄	Phosphoric acid
HAP	Hydroxyapatite
m ³ /day	cubic meter per day
MACT	Maximum achievable control technology
MAP	Magnesium ammonium phosphate
MGD	Million gallon per day
ML/d	Million liter per day
MSA	Municipal sludge ash
mt	metric ton
N	Nitrogen
N ₂ O	Nitrous oxide
NEBRA	North East Biosolids Residual Association
NPDES	National Pollutant Discharge Elimination System
OpEx	Operation expenditure
O&M	Operation and maintenance
P	Phosphorus
P ₂ O ₅	Phosphorus pentoxide
PAOs	Polyphosphate accumulating organisms
PF	Present worth factor
PFAS	Per- and polyfluoroalkyl substances
PPCPs	Pharmaceuticals and personal care products

PR	Phosphate rock
PW	Present worth
SPR	Strategic petroleum reserve
TBL	Triple bottom line
TP	Total phosphorus
TS	Total solids
TSP	Triple superphosphate
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
VS	Volatile solids
WEP	Water extractable phosphorus
WRRF	Water resource recovery facility
WT	Wet ton
wmtpd	Wet metric ton per day

CHAPTER 1

INTRODUCTION

1.1 Background

Phosphorus (P) is the building block for all life forms. It plays a critical role in all key biochemical functions in every living being. P is found in nucleic acids, cell membranes, and bones and teeth. It plays a role in energy metabolism, acting as the currency for energy transactions in adenosine triphosphate molecules (Elser et al., 2013). P is also a major building block for commercial fertilizers. More than 80% of phosphate rock (PR) mined is used as a fertilizer for food production (Scholz and Wellmer, 2018). As the global population increases, so does the demand for P in the food production industry. To illustrate the rapid increase in PR mining rate, the world population increased from 1.79 billion to 7.27 billion from 1913 to 2014 (~4 times) while PR extraction increased from 3.4 million metric tons (mt) in 1913 to 245 million mt in 2014 (~72 times) (Mew et al., 2018). With a population projection of 9.7 billion by 2050 (United Nations, 2015), the projected annual PR mining rate would be at 425 million mt by 2050 if the same mining rate per capita were to apply for the future. According to USGS (2019), there are approximately 272 billion mt of PR resources and 70 billion mt of PR reserves in the world. “Resources” is defined as “a concentration of naturally occurring solid, liquid, or gaseous material in or on the Earth’s crust in such form and amount that economic extraction of a commodity from the concentration is currently or potentially feasible” whereas “reserves” is defined as “the part of an identified resource that could be economically extracted or produced at the time of determination”. With a world production rate of 270 million mt per year, there are no imminent shortages of PR

(Scholz and Wellmer, 2018; USGS 2019). In the United States, it was reported that there are approximately 1 billion mt of PR reserves distributed among four states: Florida, North Carolina, Idaho, and Utah (USGS 2019). Since 1998, an annual PR production rate has been ranging from 25.8 million mt in 2010 to 44.2 million mt in 1998. It has been speculated that the U.S. production rate will remain steady at 29.5 million mt for the next five years (USGS 2019). If a static supply and demand relationship were to apply to PR reserves, it would take about 31 years to deplete the existing PR reserves in the U.S. However, the supply and demand relationship of a mineral commodity is never a static relationship. Factors such as exploitation of new mines, technologies advancement, and commodity recycling continually impact the dynamic relationship (Scholz and Wellmer, 2018). With new mines constantly being exploited, it is unlikely that PR reserves will be depleted in the U.S. anytime soon.

Nonetheless, as Schulz et al. (2017) indicated, even though most of the mineral commodities are in sufficient amounts in the earth for many years of supply, their availability can be impacted by politics, social constraints, environmental regulations, land-use restrictions, and economics. PR reserves are geographically concentrated in a handful of countries and regions such as China, Morocco, Western Sahara, and the United States (U.S.). The rest of the world depends on P imports from these countries (Lwin et al., 2017). This uneven and limited distribution creates complexities from a geopolitical standpoint. Social and political challenges, along with supply chain risk and high prices, will begin well before the PR reserves run out (Childers et al., 2011).

To fully comprehend the concern over the exploitation of P mining and supply, we should also pay attention to the fact that the majority of the mined P is lost to the

environment and causes eutrophication due to a broken P cycle. Out of the 80% mined PR, only 5% to 10% is absorbed by humans, with the rest lost along the supply chain from mined PR to food for consumption (Jacobs et al., 2017). Much of the P loss in the open or linear “human” P cycle ends up in natural water bodies. This P causes eutrophication; accumulates in soils due to animal manure and excessive application of fertilizers; or ends up in landfills (Alvarez et al., 2018). Eutrophication causes serious environmental damage and decreased biodiversity in the impacted water bodies (Mayer et al., 2016). Once the P is “lost” to the oceans, it is almost impossible to economically recover due to the low concentration in water (Mew et al., 2018). As a result, P is continuously replaced with newly mined PR. This further exacerbates environmental pollution through the mining processes and the excess P loss to the environment. The only sustainable path to improve water quality and to allow P to flow efficiently is a closed-loop P cycle that would reduce input (via recovery), reduce accumulation in soils, and reduce runoff into natural waters.

As the majority of P consumed by humans ends up in water resource recovery facilities (WRRFs) in the form of wastewater, WRRFs become ideal locations for P recovery. In WRRFs, unintentional phosphate precipitation causes scaling in pipelines, pumps, and other highly turbulent areas of sludge processing systems, leading to operational problems and increased operation and maintenance (O&M) costs. The precipitation is worse in enhanced biological P removal (EBPR) facilities where the biomass produced by EBPR systems is high in P content, and the same is released into the liquid phase after digestion and dewatering. The concentrated streams of sludge liquor are the primary source of unintentional precipitation and scaling in post-digestion

solids handling systems. In the U.S., as wastewater effluent limits become more stringent, the number of states requiring nutrient (N and P) removal is increasing.

Phosphorus is not truly removed from WRRFs unless it is recovered, and the P discharge to the environment from WRRFs is eliminated. Hence, P recovery in WRRFs should be viewed as a necessity rather than a luxury treatment process.

In essence, P recovery reduces PR mining demand, reduces pollution from PR mining, reduces O&M costs in WRRFs, and offers environmental benefits such as water quality improvements by reducing P discharge into streams and lakes as well as reducing excessive P released into soils through land application of biosolids (Steen, 2004). Given all the benefits, little P recovery is currently practiced at WRRFs. This is due to economic constraints as the cost for P recovery is several times higher than the market price of PR (Molinos-Senante et al., 2011). In particular, P recovery for small to medium size WRRFs (defined here as between 3,800 cubic meters per day (m^3/d) to 38,000 m^3/d) in suburban areas may not be economically viable due to the low recovery percentages and increased costs (such as chemical and energy costs) of P recovery. In order to encourage WRRFs of all sizes to adopt P recovery, the operational costs, chemical usage costs, energy costs, and transportation costs of the recovered products must be sufficiently low. To date, economic feasibility evaluations for P recovery have been performed on a facility-to-facility basis. Small WRRFs are not able to justify P recovery when compared to larger WRRFs that benefit from the economy of scale. There has not been any study conducted on the economic feasibility of P recovery from a regional perspective. In a circular economy where the practice of reducing or minimizing the use of raw materials, reusing the materials that are not wasted, and recycling or reprocessing

waste materials is promoted (Ghisellini, 2016), the development of sustainable P recovery requires P to be recovered for more diverse applications (Vasenko and Qu, 2017). A strategy to recover P in forms that can be economically stored as a reserve for future use is yet to be explored.

1.2 Approach to P recovery at WRRFs

There have been comprehensive reviews and studies on methods to recover P from WRRFs (Egle et al., 2016; Peng et al., 2018); however, P recovery methods have been focused on the immediate use of recovered P as fertilizer. In general, phosphorus can be recovered from wastewater by processing sludge liquor, biosolids (treated sludge), and/or sludge ash. These streams/sources within a WRRF dictate the approach through which P can be recovered. A total phosphorus (TP) flow diagram that can be employed in a typical WRRF is depicted in **Figure 1** (Law and Pagilla, 2018). Unintentional P recovery through land application of biosolids for disposal or for beneficial use has been the most practiced approach by WRRFs in the U.S. Approximately 55% to 60% of the total biosolids produced in the U.S. are land applied (Lu et al., 2012). Furthermore, a report published by the North East Biosolids and Residuals Association (NEBRA) in 2007 indicated that 55% of biosolids produced in the U.S. by WRRFs are land applied, 28% are disposed of in landfills, 15% are processed in incinerators, and the remaining 2% is disposed of in biosolids-only surface applications. The P recovery approaches for each stream in the WRRF are briefly reviewed below.

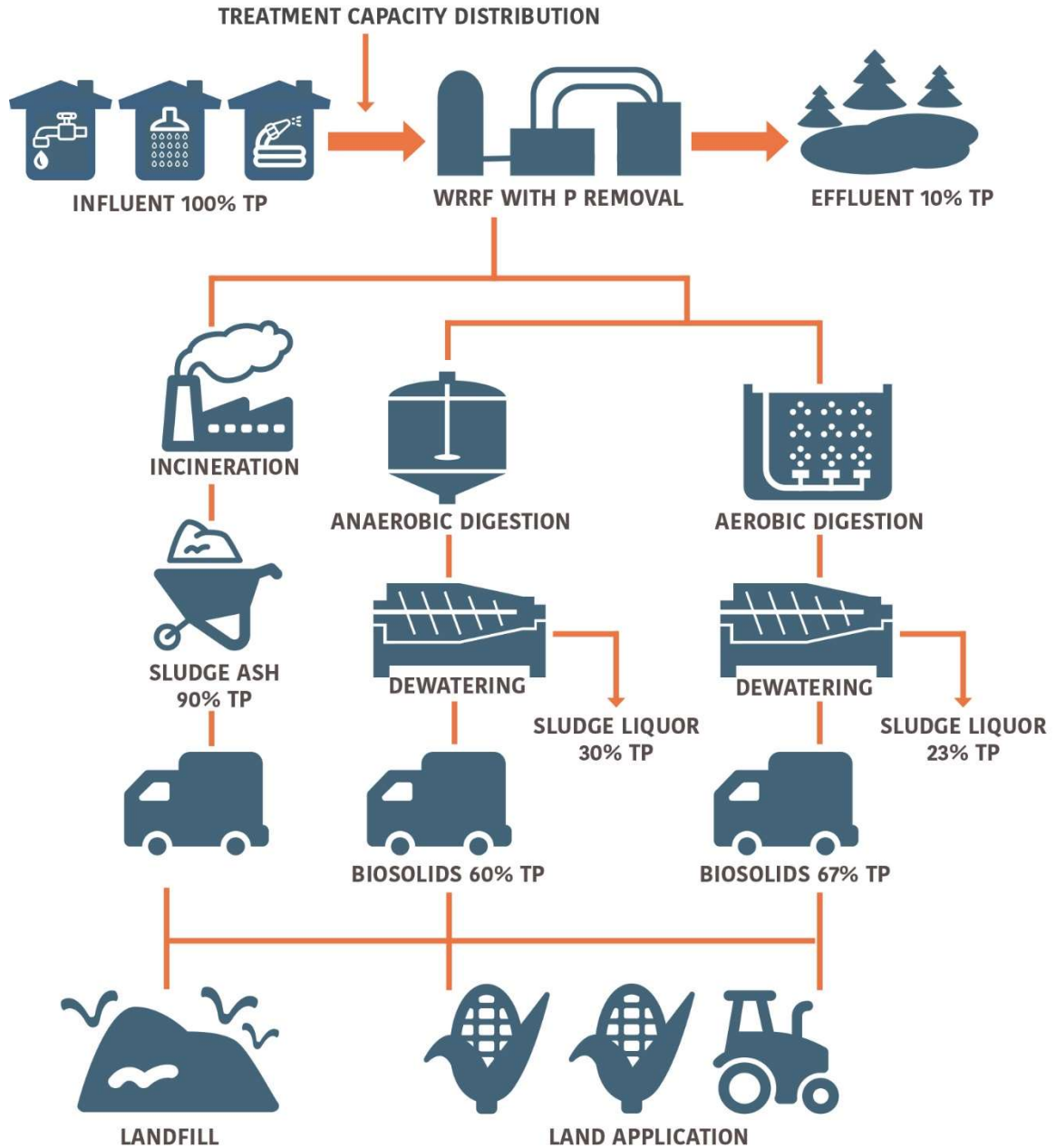


Figure 1. Total P mass flow diagram as a percentage of influent TP in a typical WRRF (Law and Pagilla, 2018).

1.2.1 Recovered P via a land application

The most commonly practiced P recovery approach for beneficial reuse in the U.S. is direct land application of biosolids produced at WRRFs. Most of the WRRFs

prefer direct land application since it is a relatively inexpensive method to dispose of biosolids and it benefits the soil conditions. Advantages of land application of biosolids include soil property improvements such as moisture and nutrient retention ability, macronutrient supply for plant growth (mainly N and P), and micro-nutrient (such as iron, copper, zinc, etc.) supplementation for plant growth.

Some of the challenges with the land application include the need for biosolids storage during winter season when land application is prohibited on frozen ground and the need for transportation to haul away the biosolids from WRRFs. There are restrictions and regulations on allowable pollutant content in the biosolids, pollutant loading to the land, and rate of N and P application based on agronomic conditions. Furthermore, management strategies to minimize N and P runoff into adjacent streams and rivers may be needed. Hence, land application of biosolids options incurs operational costs for WRRFs, and possibly, for landowners. Public perception is another hurdle for land application due to odor concerns for residential areas close to the land application sites (USEPA, 2000). Public concern over heavy metals is also on the rise. When land applied, biosolids may contain heavy metals that would increase the content in the soil and may leach to groundwater supplies (Wuana and Okieimen, 2011). While there may not be acute risk, long-term risk monitoring of metal concentrations is necessary (Zhou et al., 2017). In Europe, land application is restricted for the same reason due to contaminants in biosolids (Scholz and Wellmer, 2018). Accumulation of trace organic compounds in biosolids is another increasing concern. The increasing use of prescription drugs and personal care products coupled with incomplete removal of these pharmaceutical and personal care products (PPCPs) by conventional wastewater

treatment processes has caused the concentrations of PPCPs in treated effluent and biosolids to rise (Mohapatra et al., 2016). A study conducted by Kinney et al. (2006) identified 87 organic wastewater contaminants from nine (9) biosolids samples produced by different WRRF treatment processes across seven (7) states. Twenty-five (25) of these contaminants (including triclosan and antihistamine) were detected in all biosolids samples, 55 were detected in at least one (1) biosolids sample, and a minimum of 30 and a maximum of 45 were detected in any one (1) biosolids sample. Using biosolids that contain PPCPs as soil supplements may potentially increase plant uptake of the PPCPs, contaminate food crops, and increase human and animal exposure through dietary intake of vegetables grown in biosolids-supplemented soil (Wu et al. 2015). Therefore, it is anticipated that restrictions on land application in the U.S. and elsewhere will become more stringent which may limit the amount of biosolids that can be land applied. From the fertilization standpoint, phosphorus availability of biosolids is low. Brandt et al. (2004) indicated that while triple superphosphate, a commercialized mineral fertilizer, has a water extractable phosphorus (WEP) content of 85.2%, typical biosolids have a WEP content of 2.5%. Plant P availability in most reclaimed P fertilizers is also higher than that of commercial fertilizer (Moller et al., 2017). Nonetheless, this status quo or “do nothing” approach of biosolids use as P source represents a more traditional way to recover and reuse P. As can be seen from **Figure 1**, the extent of P recovered from typical WRRFs can be as high as 90% of the influent TP if all the biosolids are land applied.

1.2.2 Recovered P as fertilizers for immediate use

A more pro-active approach to P recovery at WRRFs is intentional recovery for immediate use as fertilizer. Phosphorus recovered as a slow-release fertilizer, struvite, from wastewater has been on the rise (Le Corre et al., 2009). In particular, P recovery from sludge liquor is the most common approach as it involves simple steps and is likely the most economical approach when compared with P recovery from biosolids or sludge ash (Sartorius et al., 2011). Phosphorus recovery from dewatered biosolids and sludge ash is relatively more complicated. However, up to 90% of influent P can be recovered from these processes compared to a maximum of 30% of influent P that can be recovered from the sludge liquor alone, as shown in **Figure 1**. To recover P from biosolids or sludge ash, wet chemical or thermal treatment is necessary to dissolve the bound P in the biosolids or sludge ash (Petzet and Cornel, 2013b). There are commercially available P recovery processes that use biosolids or sludge ash as sources. These processes, including Aqua Reci and LEACHPHOS, use biosolids and sludge ash, respectively, to produce fertilizers (Law and Pagilla, 2018). P recovery from sludge liquor is less complicated than P recovery from biosolids or sludge ash. Sludge liquor from dewatering processes contains large amounts of P and ammonium that are released after the anaerobic digestion process. By adjusting the pH and adding magnesium chloride or magnesium hydroxide, struvite can be intentionally recovered using a fluidized bed upflow reactor. Alternatively, P can be recovered from sludge liquor as brushite, a form of Ca-P, or as vivianite, a form of iron phosphate, that can be used as fertilizer (Law and Pagilla, 2018). Commercially available technologies include Ostara (as struvite) and CalPrex (as brushite).

Advantages for P recovery as fertilizer include the creation of a potential revenue stream for WRRFs. Operational benefits include improvements on biosolids dewaterability, which may reduce biosolids disposal costs. However, a challenge to recover P as fertilizer for immediate use is that the price for phosphate rock and phosphate-based fertilizers is considerably lower than P recovered in WRRFs (Jacobs et al., 2017). Also, P recovery for small WRRFs (less than 3,800 m³/d) may not be economically viable due to the low efficiency of P recovery and increased capital costs (Sarvajayakesavalu et al., 2018). In the U.S., 78% of all WRRFs are small facilities treating less than 3,800 m³/d flow (USEPA, 2012). Phosphorus recovery as fertilizer is not economically viable to the majority of the WRRFs in the U.S.

1.2.3 Recovered P as a commodity strategic reserve for future use

Phosphorus recovery at WRRFs includes recovery for future use. As discussed in the last section, the current price for phosphate rock and phosphate-based fertilizers is lower than P recovered as struvite from WRRFs (Cornel and Schaum, 2009). The lack of an economic driver makes P recovery from WRRFs less attractive in comparison. Therefore, the recovery of P into a suitable and economical form coupled with this product's storage as a reserve for future use becomes a more favorable approach to P recovery. For example, the same streams used for struvite fertilizer recovery can be used for Ca-P recovery which can then be used as feedstock for fertilizer production when the fertilizer costs justify the processing costs. An advantage of Ca-P recovery is that the chemicals used in Ca-P recovery (mostly calcium hydroxide or lime) are more readily available than those used in struvite recovery (mostly magnesium-based chemicals such as magnesium hydroxide or magnesium chloride and sodium hydroxide for pH

adjustment), and therefore the operating costs can be reduced. Another advantage is that since Ca-P in the form of hydroxyapatite (HAP) is comparable and can be substituted as rock phosphate, the recovered product offers more diverse applications (Mayer et al., 2016). Therefore, there is an expansion of potential end-users as Ca-P serves as a feedstock to manufacturing phosphoric acid and fertilizers as well as other industrial applications. A final advantage is that the recovered Ca-P can be stored at a regional site thus minimizing the dependence of rock phosphate imports due to potential geopolitical conflicts and/or supply chain disruptions in the future.

The concept of commodity strategic reserves can be applied for recovered P from WRRFs. Commodity strategic reserves have been adopted in the world since World War II. Commodities such as petroleum, uranium, rare-earth metals, and seeds are being reserved in various countries to cope with price spikes and supply shortages, which may arise from natural disasters, wars, or other dire situations (Resiman, 2012). For example, the U.S. strategic petroleum reserves consider only crude oil, rather than refined products, as a strategic reserve. That is because crude oil is less expensive to acquire and store, easy to transport, and has more diverse end uses (United States Department of Energy). Understanding that there is no substitution for P and that P is essential to all life forms, a strategic P reserve in the form of widely usable Ca-P is necessary to secure the future supply. Surprisingly enough, P is not part of the commodity strategic reserves. Recovered P from municipal and agricultural wastes could be the strategic P reserve near urban areas, which can be refined into fertilizers or other needs in the future as needed.

1.3 Research goal

The goal of this research is to demonstrate a strategic, novel, and economically viable approach for wastewater P recovery and storage that can involve all WRRFs of all sizes in a geographical region. An innovative concept is to recover P as calcium phosphate (Ca-P) and store the product in a regional, dedicated landfill (monofill) for future use. Since the phosphate minerals in PR are essentially Ca-P in various forms, recovery of P as Ca-P allows for a diverse application of P, thus broadening its applicability beyond fertilizer. With strategic locations of the monofills, the regional approach lowers transportation costs and allows for local P supply under local control without the dependency of importing P from other states or countries. This is particularly important in the future when potential social and geopolitical conflicts may impact the global supply chain of P, resulting in skyrocketing of PR and fertilizer costs.

CHAPTER 2

LITERATURE REVIEW

2.1 Introduction

Much of the focus on P recovery from WRRFs has been on refined struvite or magnesium ammonium phosphate (MAP, $MgNH_4PO_4$) forms, which are readily usable as slow-release fertilizers for agricultural applications. The recovery of P in struvite form requires addition of magnesium as magnesium is the limiting constituent present in the sludge liquors of WRRFs (Fattah et al., 2008; Liu et al., 2013) for struvite precipitation.

Furthermore, alkaline chemicals that enhance the precipitation of struvite by controlling the pH are also required in most cases (Le Corre et al., 2009; Liu et al., 2013). In addition to struvite recovery, there have been developments in phosphate recovery technologies that produce different types of fertilizers. It is apparent that fertilizer form is the most widely studied and currently accepted end product of P recovery from WRRFs.

Phosphorus in wastewater can be removed chemically, biologically, or a combination of both. Chemical P removal is effective. However, P removal using chemicals in the mainstream treatment of the WRRF generates a large amount of solids due to the added chemicals that will need to be disposed of or managed. EBPR produces fewer solids when compared with chemical P removal. A disadvantage with EBPR is that the stored P in the biomass is re-released into solution during solids handling processes such as in anaerobic digestion. The re-release of phosphate during and after anaerobic digestion causes the formation of scale under different conditions. The most common precipitate form or scale is MAP or struvite. Other precipitation forms include

ferrous phosphate (vivianite) and Ca-P (hydroxylapatite, HAP, as the most stable form) (Cornel and Schaum, 2009). Phosphate precipitation poses operational challenges at WRRFs. Struvite can be found inside anaerobic digesters, digested sludge pumps, pipelines, dewatering equipment, and dewatered supernatant drain lines. The scale clogs pipelines and increases the wear and tear of equipment, which results in increased O&M costs at the facilities. With a better understanding of when, where, and why precipitates may form, unintended precipitates may be eliminated, and a controlled environment can be promoted so that precipitates can be formed intentionally for beneficial recovery of P. **Table 1** presents a summary of common precipitates, their chemistries, and conditions that promote precipitation in a WRRF.

Table 1. Summary of the common P precipitates, their chemistries, and conditions that promote precipitation at a WRRF.

Precipitates	Molar ratio	Conditions			Ref	WRRF processes
		pH	pKs	T, °C		
Magnesium ammonium phosphate (Struvite) $MgNH_4PO_4 \cdot 6H_2O$	1:1:1	7 > pH > 6.3	13.36	21	c	<ul style="list-style-type: none"> Anaerobic digestion Digested sludge piping/pump/mixer with high turbulence conditions Dewatering equipment, in particular centrifuge dewatering with high turbulence Filtrate/centrate piping
	1.2:3:1	7.5 – 9.0	NR	25	e	
	1.04:1.11:1	7.0	13.26	25	b	
	1.93:1:1.26	10	NR	35	a	
	2:1:1	10	NR	25	d	
	Ca:PO₄	pH	pKs	T, °C	Ref	
Calcium phosphate (Hydroxyapatite, HAP) $Ca_5H(PO_4)_3OH$	1.67:1	9	58.5	21.5	g	<ul style="list-style-type: none"> Anaerobic digestion
	6.67:1	7.5				
	1.67:1	11.5	40.5	21	h	

Precipitates	Molar ratio	Conditions			Ref	WRRF processes
		pH	pKs	T, °C		
Calcium phosphate (Octacalcium phosphate, OCP) $\text{Ca}_4\text{H}(\text{PO}_4)_3$	5:1	7.5 - 8.0	50.6	21.5	g	• Anaerobic digestion
	1.33:1	8 - 10	46.9	21	h	
Calcium phosphate (Brushite, DCPD) $\text{CaHPO}_4 \cdot 2\text{H}_2\text{O}$	1:1	< 5.0	NR	25	f	• Acid phase anaerobic digestion
	1:1	7.4	19	21	h	
	Fe:PO ₄	pH	pKs	T, deg C	Ref	
Ferrous phosphate (Vivianite) $\text{Fe}_3(\text{PO}_4)_2 \cdot 8\text{H}_2\text{O}$	1.5:1	6.3 – 7.0	NR	30	j	• Anaerobic digestion
Ferric phosphate $\text{FePO}_4 \cdot 2\text{H}_2\text{O}$	2:1	5.5 – 6.0	26.4	25	i	<ul style="list-style-type: none"> • Acid phase anaerobic digestion process • Anaerobic digester if FeCl_2 is added for struvite control • Where excess iron concentration. with a lower pH – e.g., sludge conveyance • In digested sludge pipeline (higher temp) • Sludge heat exchanger and associated piping

NR: Not Reported

Ref: a. Chen et al., 2013; b. Ohlinger et al. 1998; c. Muster et al., 2013; d. Xavier et al., 2014; e. Hao et al., 2008; f. Abbona et al., 1986; g. Song et al., 2001; h. Hermassi et al., 2015; i. Zhang et al., 2010; j. Cheng et al., 2015.

As shown in **Table 1**, the conditions which favor precipitation of struvite, Ca-P, and vivianite can be found in areas where unintentional phosphate precipitation occurs within a WRRF. These locations represent high pH (or low pH for brushite), high constituent concentrations, and at mostly above 20°C temperature.

Table 1 also shows that phosphate precipitation can occur under different molar ratios of the constituents as well as under different pH since the wastewater characteristics and ambient conditions are different in different WRRFs. When the conditional solubility of a precipitate is exceeded, unintentional precipitation occurs. Channeling the unintentional precipitation into a useful product that can be harvested, repurposed, and reused is one step closer in closing the loop of a circular economy for P management.

2.2 WRRF P recovery sources and technologies review

Phosphorus entering a WRRF can exit via two routes: as part of the effluent discharged, or as part of the waste solids. Phosphorus can be recovered from treated effluent, sludge liquor, municipal sludge, and municipal sludge ashes. **Figure 2** presents P recovery potential in various stages in a WRRF with EBPR process employed for P removal from wastewater.

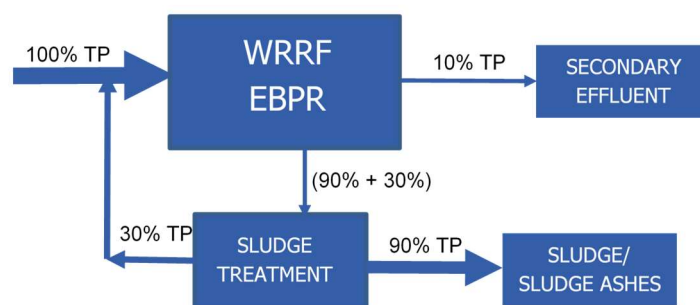


Figure 2. Phosphorus recovery potential in various stages of wastewater treatment.

As shown in the figure, the highest potential of P recovery is from municipal sludge, which can account for 90% of the influent P. Recycle sludge liquor can recover as high as 30% of the influent P loads with EBPR in the mainstream treatment, or about

10% if chemical P removal process is used. P recovery from treated effluent is not as cost-effective since only 10% of the influent P can be recovered (Cornel and Schaum, 2009; Hester and Harrison, 2013; Desmidt et al. 2015; Egle et al., 2016). The most widely used and accepted de facto P recovery method has been the direct land application of stabilized biosolids onto agricultural farmland as a nutrient supplement and/or for soil conditioning (Blocher et al., 2012; Hester and Harrison, 2013). In many European countries, direct land application has become stricter due to safety concerns regarding heavy metals in the biosolids. In Belgium and Switzerland, it is illegal to recycle biosolids as fertilizer on farmland for food safety reasons (Franz, 2008; Marchi et al., 2015). In the United States, land application is subject to restrictions of heavy metal content as well as the biosolids application rates. The rates cannot be more than nutrient requirements of the vegetation to minimize nutrient runoff and the potential leaching into groundwater. Historically, nitrogen has been the controlling factor that governs the amount of biosolids that can be land applied per acre per year. However, there has been a shift from nitrogen to P being the controlling nutrient due to concerns of P loss to surface water (Lu et al., 2012). Furthermore, any future considerations of emerging contaminants such as pharmaceuticals and personal care products (PPCPs) and per- and polyfluoroalkyl substances (PFAS) in biosolids could pose additional concerns for land application. This makes land application of biosolids a less attractive method of P recovery and reuse.

Currently, there are commercially available P recovery technologies using secondary effluent, sludge liquor, municipal biosolids, and municipal sludge ashes (MSA). The technical maturity of the technologies varies from full-scale implementation to laboratory testing stages. Many of these technologies also apply to biosolids, and

MSA generated from chemical P removal processes and are not limited to EBPR only. Technologies recovering chemical solids tend to be more involved, less cost-effective, and require more chemicals for P recovery. A summary of commercially viable P recovery technologies is presented in **Table 2**. As shown in **Table 2**, many of the technologies focus on recovering phosphorus as fertilizer in the forms of struvite and calcium phosphate. This refinement of the recovered P to be in readily usable fertilizer form adds cost to the overall production and hence makes it less competitive than commercial P fertilizers.

2.2.1 Phosphorus recovery from secondary effluent

Phosphorus from secondary effluent can be recovered from precipitation; however, a large quantity of chemicals will be required (Hester and Harrison, 2013). A method to recover P from secondary effluent is by a combination of ion exchange and precipitation. For example, the REM-NUT technology is based on using ion exchange and precipitation technology for struvite recovery (Egle et al., 2015; Sengupta et al., 2015). Phosphorus in the brine after the ion exchange can be recovered as struvite (with magnesium addition) through precipitation. Studies indicated that the recovery potential from the influent P ranges from 50% to 70% (Egle et al., 2015). However, there has been no pilot or full-scale implementation of this technology at this point (Egle et al., 2016). Overall, effluent P removal by chemical means leads to chemical solids, which is a form of P recovery. These solids could potentially be stored for future use as a source of P to be refined later in a strategic P reserve. The suitability of these solids for storage as P reserve can be improved by eliminating the entrainment of either influent solids or biosolids. However, as shown in **Figure 2** only about 10% of influent P can be recovered

from effluent P and hence may not be practical from a standalone P recovery point of view.

Table 2. Summary of commercially viable P recovery technologies.

	Status	Operational since	Final Products	Reference
Secondary Effluent				
REM-NUT®	Lab scale	2003	MAP	b, c, f
Sludge Liquor				
DHV Crystalactor®	Full scale	1988 (closed) 1993	MAP/CaP	c, d, g, l
NuReSys™	Full scale	2006	MAP	b
Ostara®	Full scale	2007	MAP	c, g, l
P-Roc®	Pilot	2005	CaP	c, m
CalPrex™	Pilot	2011	CaP (DCPD)	o
AirPrex™	Full scale	2009	MAP	c, l
Multiform™	Full scale	2012	MAP	g
Quick Wash®	Pilot	2007	CaP	n
Struvia™	Pilot	2013	MAP	f
Municipal Sludge				
Seaborne (Gifhorn)	Full scale	2007	MAP/CaP	a, b, c, d
Stuttgart	Pilot	2015	MAP	a, b, c
LOPROX/ PHOXNAN	Pilot	2012	H ₃ PO ₄	b, e
Aqua Recii	Pilot	2004	CaP	c, f, g
MEPHREC	Pilot	2016	P-slag	a, f, g
ExtraPhos® (Budenheim)	Pilot	2017	CaP (DCP)	a, c
Cambi/KREPRO	Pilot	2002	MAP/CaP/FeP	c
Municipal Sludge Ashes				
AshDec	Pilot	2008	AlP/FeP	b, c, g, i, j
PASCH	Pilot	2011	MAP/CaP	c, f
LEACHPHOS	Pilot	2012	CaP	f
EcoPhos®	Full scale	2016	H ₃ PO ₄ /CaP	a, c, f
RecoPhos®	Full scale	2012-2015 (currently closed)	P 38 fertilizer	c, f

Municipal Sludge Ashes	Status	Operational since	Final Products	Reference
Fertilizer Industry	Test	--	Commercial fertilizers	a, c
Thermphos	Full scale	2008-2011 (currently closed)	Pure P ₄	b, c
SEPHOS	Lab scale	2007	AlPO ₄ /CaP	c, d, k
BioCon	Pilot	2001	H ₃ PO ₄	b, c, g
Eberhard	Lab-scale	2008	CaP	c, h
EPHOS	Lab scale	2016	CaP	c
LOTUS Gifu City	Full scale	2010	CaP	a, c
P-Bac (Inocre) [®]	Pilot	2013	MAP/CaP	c
SESAL-Phos	Lab-scale	2012	CaP (HAP)	c, k
TetraPhos	Pilot Full scale (under construction)	2015 2019	H ₃ PO ₄	a, f

Ref.: a. Kabbe, 2017; b. Hester et al., 2013; c. Egle et al., 2015; d. Cornel and Schaum, 2009; e. Blocher et al., 2012; f. Egle et al., 2016; g. Barnard et al., 2012; h. Franz, 2008; i. Mattenberger et al., 2008; j. Adam et al., 2009; k. Petzet et al., 2012; l. Desmidt et al., 2015; m. Berg et al., 2006; n.

<http://phosphorusplatform.eu/images/download/ScopeNewsletter78.pdf>; o. <http://www.nrutech.com>.

2.2.2 Phosphorus recovery from sludge liquor

Sludge liquor, or concentrated sidestreams, is likely the most studied and practiced source for P recovery. Using sludge liquor as a source, approximately 30% of the influent P can be recovered, as shown in **Figure 2**. To increase the overall P recovery, waste activated sludge fermentation has been used to release additional phosphate into the feed stream (Petzet and Cornel, 2013).

Struvite recovery from sludge liquor requires magnesium addition (in the form of magnesium chloride or magnesium hydroxide) since magnesium is the limiting constituent for intentional struvite precipitation from sludge liquors (Fattah et al., 2008; Liu et al., 2013). Furthermore, the pH of the system will need to be adjusted, either by

sodium hydroxide addition or CO₂ stripping via aeration, in order to increase the pH to an optimum level for efficient P recovery (Fattah, 2012). The process typically involves a fluidized bed reactor with concentrated sludge liquors entering at the bottom of the reactor. In the United States, successful commercialized technologies for struvite recovery include Ostara®, Airprex™, and Multiform™. Full-scale installations are available in the U.S. In Europe, DHV Crystalactor® and NuReSys™ are more common. Similar to the struvite recovery process, a fluidized bed reactor is used to recover Ca-P using sludge liquor. Cornel and Schaum (2009) suggested that since Ca-P is comparable with the composition of phosphate rock, Ca-P precipitation is mostly driven by kinetics instead of thermodynamic considerations. Spontaneous precipitation of Ca-P may only occur under high supersaturation conditions. Ca-P formation goes through different phases depending on the pH of the solution, from the most unstable form of brushite (DCPD) to the most stable form of hydroxylapatite (Hosni et al., 2007). The various phases of Ca-P precipitation lower the ionic activities of the precipitating constituents and change the supersaturation continuously, resulting in low repeatability and accuracy of precipitation experiments (Valsami-Jones, 2004). It has been suggested that these concerns related to spontaneous precipitation of Ca-P can be overcome by using seeding materials (Valsami-Jones, 2004). Seeding materials can also help to increase the rate of precipitation in the presence of competitive ions such as magnesium, barium, lead, and copper ion, making it more practical to recover Ca-P (Rosa et al., 2010).

Seeding Materials: Studies have shown that the use of calcium silicate hydrate (CSH) in the form of tobermorite as seeding materials to promote calcium phosphate precipitation served several benefits (Jiang et al., 2010; Petzet and Cornel, 2013). First,

calcium is released by the seed material, and therefore calcium addition is not necessary. Second, calcium hydroxide is formed, and pH can be raised without additional chemical addition for pH adjustment. The increase in pH promotes Ca-P precipitation. Third, using CSH can eliminate the need to remove carbonate in the system before calcium precipitation. Also, when CSH no longer releases calcium, tobermorte "disappears" and HAP is formed. Without CSH seeding, the carbonate in the system will need to be removed first by lowering the pH with sulfuric acid, followed by CO₂ stripping to avoid carbonate formation. After that, pH is then raised by sodium hydroxide for calcium phosphate precipitation.

Furthermore, Petzet and Cornel (2013) reported that CSH could react with digested sludge when it is added to anaerobic digesters directly to react with the P released in the digester. This is the basis of the FIX-Phos technology. In the FIX-Phos technology, CSH releases calcium ions in the digesters, coupled with the rising pH, and triggers Ca-P formation (Petzet and Cornel, 2013). The Ca-P formed, with a P content of 8% to 10%, can remain in sludge for land application, or can be separated after the digestion process and used as fertilizer (Petzet and Cornel, 2013). Another technology, P-Roc, also uses CSH as seeding material for Ca-P recovery. However, instead of a fluidized bed reactor, a stirred reactor is used (Berg et al., 2005).

Other seeding materials include amorphous CSH (A-CSH), synthetic CSH, xonotlite, calcite, magnetite, and Juraperle (a form of calcium carbonate) were studied in the past years. A-CSH, with a high Ca: Si ratio of greater than 2, offered better settleability, filterability, and dewaterability than conventional CSH (with a lower Ca:Si ratio) or calcium hydroxide (Okano et al., 2013 and 2015). Synthetic CSH using

polyethyleneglycol (PEG) as the pore-generation agent showed enhanced solubility of calcium ions and hydroxide ions, which further increased the supersaturation (Guan et al., 2013). Using xonotlite, crystallization occurred with a pH higher than six and the P removal efficiency increased by increasing the bicarbonate concentration through the increase of pH and xonotlite concentrations (Chen et al., 2008). Studies on calcite as seeding materials showed that neither the removal of carbonate nor the addition of sodium hydroxide was necessary and therefore operating costs can be reduced (Song et al., 2006a). Also, used calcite can increase the crystallization rate. Use of magnetite mineral (Fe_3O_4) as seeding materials offers the potential to use magnetic separation to separate the calcium phosphate precipitated from the solution (Karapinar et al., 2004). The use of juraperle as seeding materials improved the phosphorus removal efficiencies after repeated use (Song et al., 2007). While many different types of seeding materials have been studied and some showed promising results, none of these seeding materials has been applied in commercialized Ca-P recovery technologies.

Although P recovery as struvite has been widely adopted, the number of commercially available technologies for Ca-P recovery from sludge liquor is on the rise. Technologies such as DHV Crystalactor®, P-RoC, and FIX-Phos, originated from Europe, utilize seeding materials to reduce the chemicals for pH adjustments and to increase crystallization rates (Cornel and Schaum, 2009; Desmidt et al., 2015; Egle et al., 2015; Petzet et al., 2012; Petzet and Cornel, 2013). Most Ca-P precipitation occurs under high pH conditions ($\text{pH} > 9$) (Valsami-Jones, 2004). For both the DHV Crystalactor® and P-RoC processes, phosphate-rich pellets are formed that can be used as rock phosphate substitutes in the phosphate industry (Cornel and Schaum, 2009). End

products of the FIX-Phos process can also be used as fertilizers. DHV Crystalactor® and FIX-Phos have been implemented in full scale while the P-Roc process has been piloted. Commercially available processes originating in the United States include Quick Wash™ and CalPrex™. Quick Wash™ uses citric or hydrochloric acid to first solubilize organically bound phosphorus into soluble phosphorus. After dewatering, P is then recovered by the addition of lime to form Ca-P (Szogi et al. 2010). CalPrex™ takes advantage of an acid phase digestion process to recover phosphate as brushite that can be used as fertilizers (Tabanpour, 2016). Both processes are in the pilot stage.

Vivianite ($\text{Fe}_3(\text{PO}_4)_2 \cdot 8\text{H}_2\text{O}$) is another typical precipitate formed in WRRFs from sludge liquor. When ferric chloride (FeCl_3) is used at WRRFs for odor control in sewers or struvite control in anaerobic digesters, ferric is reduced to ferrous which may cause vivianite formation when phosphate is present. Vivianite can be collected and used as iron supplement in calcareous soils (Zhang, 2011). A study conducted by Wilfert et al. (2015) showed that transformation of vivianite by oxidation or by exposing it to sulfide can induce P release and that the reaction rate can be increased if the oxidation is induced by microbes. Therefore, vivianite has been used as a slow-release iron and P fertilizer. The study also showed that vivianite is stable in the absence of oxygen, under a pH condition between 6 and 9, high ferrous iron and ortho-P concentrations, and that the solubility of vivianite decreases as temperature increases. Although there have been studies on vivianite, there is limited research on recovery of P as vivianite for beneficial reuse such as fertilizers. Furthermore, there are no known, or commercially viable technologies exist for recovery of P as vivianite. Therefore, P recovery as vivianite is not focused here.

2.2.3 Phosphorus recovery from municipal biosolids

In municipal biosolids, P is bound either biologically or chemically with the solids. Biosolids consist of 90% of the influent P, as shown in **Figure 2**. To recover P from biosolids, the first step is to dissolve the bound P either through wet chemical or thermal treatment (Hester and Harrison, 2013). The Seaborne process uses sulfuric acid to dissolve the bound P in digested sludge, followed by separation of dissolved P from heavy metals (Cornel and Schaum, 2009). By adding sodium hydroxide and magnesium oxide, phosphate is precipitated as struvite to be used as fertilizer. The process can also recover nitrogen as ammonium sulfate (Cornel and Schaum, 2009; Hester and Harrison, 2013; Desmidt et al., 2015; Egle et al., 2015). A full-scale facility in Gifhorn, Germany has been in operation since 2007 (Cornel and Schaum, 2009). Similar to the Seaborne process, the Stuttgart process also uses sulfuric acid for dissolving the bound P and heavy metals. A full-scale pilot is currently in operation in Offenburg, Germany where 5% of the total flow is treated by the process. The dissolved heavy metals are treated by citric acid as complexing agent while the phosphate and ammonium are treated with sodium hydroxide and magnesium oxide for struvite precipitation (Antakyali et al., 2013).

Besides the wet chemical process, a wet oxidation process has been developed for P recovery from biosolids. The PHOXNAN/LOPROX process combines low-pressure wet oxidation for P and heavy metal dissolution, followed by a nanofiltration process to separate P from heavy metals. The separated P is then recovered into fertilizers (Blocher et al., 2012). However, this process is most applicable for facilities where EBPR or P

removal with aluminum salts is employed. If ferric salts are used to remove P, the oxidized iron will react immediately with the released P, thereby lowering the recovery potential (Blocher et al., 2012; Egle et al., 2015). The PHOXNAN/LOPROX process was initially developed for treating industrial wastewater. There are several full-scale installations of the process; however, they are all for industrial pretreatment rather than P recovery from biosolids.

Another method of P recovery from biosolids is a metallurgical smelt-gasification process (Egle et al., 2015; Hester and Harrison, 2013). The smelting gasification technology is mostly thermal treatment of the biosolids. Using this process, biosolids are smelted with metallurgical coke or limestone for slag formation in 2,000°C furnace. The P-rich slag is tapped at 1,450°C, cooled, granulated in a water bath, and the slag is collected which can be used as fertilizer (Barnard et al., 2012). All of the organic matter in the biosolids is degassed due to the high temperature. Heavy metals will accumulate below the P-rich slag and can be collected separately from the phosphates. A pilot plant using MEPHREC®, a technology utilizes the metallurgical smelt-gasification process, was reportedly under construction and the performance of the process is unknown (Egle et al., 2016) and no additional information was published since the first report of this method. Last but not least, the Aqua Reci process uses supercritical water oxidation at a temperature above 375°C and a pressure above 220 bar to decompose organic contaminants. The oxidation is then followed by chemical treatment to participate phosphate from the inorganic residual ash (Stendahl and Jäfverström, 2004).

2.2.4 Phosphorus recovery from municipal sludge ash (MSA)

Ashes from incinerated biosolids contain 4% to 11% P content (Franz, 2008; Biswas et al., 2009; Ottosen et al., 2013; Donatello et al., 2010). It also represents about 90% of the influent P load. At a temperature of 800°C to 900°C during incineration, all organic matter and pathogens are destroyed (Hester and Harrison, 2013; Egle et al., 2015). As a result, heavy metal content in MSA is higher than the feed sludge on a mass basis. Since P in MSA is not readily available for plant growth, MSA would have no value if it is to be applied directly to farmlands. Phosphorus dissolution, using either wet chemical or thermal treatment, is the first step to recover P from MSA, followed by heavy metal removal or recovery (Ottosen et al., 2013; Guedes et al., 2014). Since P dissolution is required, P recovery from MSA is not limited to EBPR only. Ashes from municipal biosolids of chemically removed P can also be recovered. The process, such as SEPHOS, prefers sludge ash generated from P removal using aluminum-based chemicals (Hester and Harrison, 2013).

Dissolution using wet chemical treatment can be accomplished by either acid, alkaline (Franz, 2008; Donatello et al., 2010; Xu et al., 2012; Weigand et al., 2013), or both (Petzet et al., 2012). Depending on the P content of ash, acid leaching is preferred for Fe-rich MSA (Ottosen et al., 2013). Petzet et al. (2012) further suggested the use of both acid and alkaline treatment to increase the P recovery potential up to 78% for Al-rich MSA. Besides wet chemical methods, P dissolution can be accomplished by thermochemical method (Adam et al., 2009; Mattenberger et al., 2008) such as the AshDec process. Separation can be done by electrodialytic or ion exchange mechanisms (Guedes et al., 2014; Xu et al., 2012; Franz, 2008). However, no further development

beyond a pilot plant installation is known for the AshDec approach. Derived from the production of triple superphosphate, the RecoPhos process uses phosphoric acid to dissolve P from MSA and produce readily marketable products such as magnesium or calcium phosphate (Weigand et al., 2013). The process has already been implemented on a full scale, with a production rate of 4,000 metric tons per year (Egle et al., 2016).

LEACHPHOS is another process using a sequential leaching and precipitation process to recover P from the sludge ash (P-Rex 2015). Sulfuric acid is used to dissolve phosphate, followed by a solid-liquid separation using filter presses. While the filter cake will need to be disposed of, the filtrate is then pumped into a stirred reactor where sodium hydroxide or calcium hydroxide is added for phosphate precipitation. Additional steps are needed to completely remove the heavy metals.

2.3 Economic Viability of Phosphorus Recovery

As discussed previously, P recovery at WRRFs as struvite has been widely accepted in recent years. However, simply because P recovery is the right thing to do, it does not always justify the course of action. WRRFs are responsible for many stakeholders, and therefore they need to invest wisely for the best return on investment. WRRFs always face the same question when it comes to P recovery: Is the project financially justifiable? A study by Geerts et al. (2015) showed that factors such as phosphate price, capital, and operation and maintenance costs all contribute to the feasibility of P recovery initiatives at WRRFs. The following discussion attempts to answer the question from a local perspective based on a simple payback analysis of a struvite recovery system within a WRRF, and from a global perspective based on a

comparison between current fertilizer costs and costs to recover P as struvite from a WRRF.

2.3.1 Simple Payback Analysis of P Recovery at a WRRF

Cost for struvite recovery includes both equipment (fluidized bed reactor, feed pump, and equipment, etc.) and chemicals. Struvite recovery requires magnesium addition and pH adjustment, mostly in the form of magnesium chloride and sodium hydroxide. These chemicals could be costly, depending on whether they are locally available. The end product, struvite, is used as slow-release fertilizer in agriculture.

Table 3 illustrates a simple payback example of a struvite recovery system for a 95 ML/day (25 MGD) facility. Assuming an EBPR facility recovers approximately 205 kilograms per day of phosphorus based on the following information: influent total phosphorus (TP) of 5 mg/L to 6 mg/L, target effluent TP of 1 mg/L, 40 percent solids destruction after anaerobic digestion, and 80% of the phosphorus in the recycle sidestream can be recovered (Puchongkawarin et al., 2015).

Table 3. Simple payback analysis of a struvite recovery at a 95 ML/d facility.

Simple Payback Analysis	Low End of Capital Costs	High End of Capital Costs
P recovered at a 95 ML/day (25 MGD) WRRF, kg per day	205	
<u>Capital Costs</u>		
Capital costs ¹ , US\$ in millions	7	10
<u>Annual Revenue, Savings, and Operation Costs</u>		
Struvite production at 12.6% P in struvite, kg/day	1,633	1,633
Struvite revenues, US\$/metric ton	385	385
Total annual struvite revenues, US\$ in millions	0.23	0.23
Total annual savings in operation and maintenance ² , US\$ in millions	0.59	0.59
Total annual operation costs ³ , US\$ in millions	(0.26)	(0.26)
Total annual surplus or deficit, US\$ in millions	0.56	0.56
Simple payback ⁴ , in years	12	18

Note:

1. Depending on site conditions, capital costs for a struvite recovery system ranges from \$7M to \$10M. Capital costs include equipment, building, waste activated sludge fermentation, odor control, and all appurtenances.
2. Including potential savings on sludge disposal and struvite maintenance and operation costs
3. Including operation costs such as power, labors, and chemicals
4. Simple payback (in years) = (Capital Costs) / (Total Annual Surplus or Deficit)

As shown in **Table 3**, taking into account the annual fertilizer revenues, annual O&M savings from eliminating struvite maintenance and less biosolids to be disposed of, and annual operation costs due to chemicals, labor, and power associated with the struvite recovery process, a simple payback period of 12 to 18 years is anticipated, depending on the initial capital costs of the system. As most WRRFs use a ten-year payback period as a rule of thumb to decide whether a capital investment makes economic sense, the 12 to 18 years of payback does not indicate an attractive return on investment.

Recovery of P in the form of Ca-P may be preferred to struvite since the composition of Ca-P is the same as raw phosphate rock, which has been heavily mined as the global supply of fertilizer. Ca-P can be readily accepted by phosphate industry if it is recovered in a suitable physical form (Song et al., 2006a). For Ca-P recovery, the equipment required would be comparable with that of struvite recovery, although a simple settling tank is proposed in some technologies instead of a fluidized bed reactor. From the chemical usage standpoint, less expensive and much more widely available chemicals such as calcium hydroxide (lime) are used. Calcium hydroxide is much more readily available in several parts of the world than the factory produced magnesium chloride. The end products can be used as fertilizer, in phosphate manufacturing industry, or can be stored as Ca-P in dedicated landfills (monofills) for future refining. On the surface, P recovery as Ca-P appears to cost less, and the end products allow for more applications. Nevertheless, there has not been a side-by-side comparison of the two P recovery technologies to-date, so this hypothesis is yet to be proven.

2.3.2 Cost of P Fertilizer

Understanding the costs of rock phosphate and triple superphosphate (TSP or calcium dihydrogen phosphate), a widely used fertilizer, is crucial in evaluating whether P recovery as fertilizer is economically viable. As of December 2017, triple superphosphate (TSP) costs US \$337 per metric ton, or the US \$0.33 per kilogram, and rock phosphate costs US \$97 per metric ton, or US \$0.097 per kilogram, according to World Bank (2017). **Figure 3** presents the costs for TSP and rock phosphate from 2001 to 2017. As shown in **Figure 3**, the cost for rock phosphate peaked at US\$0.53 per

kilogram in 2008. Similarly, TSP cost peaked at US\$1.37 per kilogram also in 2008. Using the same example as presented in **Table 3** and let us consider only the operation costs (which included chemicals, power, and labor) required to produce 0.454 kilograms of struvite as fertilizer. The 1,633 kilograms of struvite produced per day would require \$705 of daily operation costs, which is equivalent to \$0.43 per kilogram of struvite as fertilizer. Note that this cost does not include the transportation costs necessary to remove the fertilizer from the WRRFs. When comparing the costs of purchasing fertilizer and the costs of producing fertilizer, it is more economical to purchase fertilizer using today's market price.

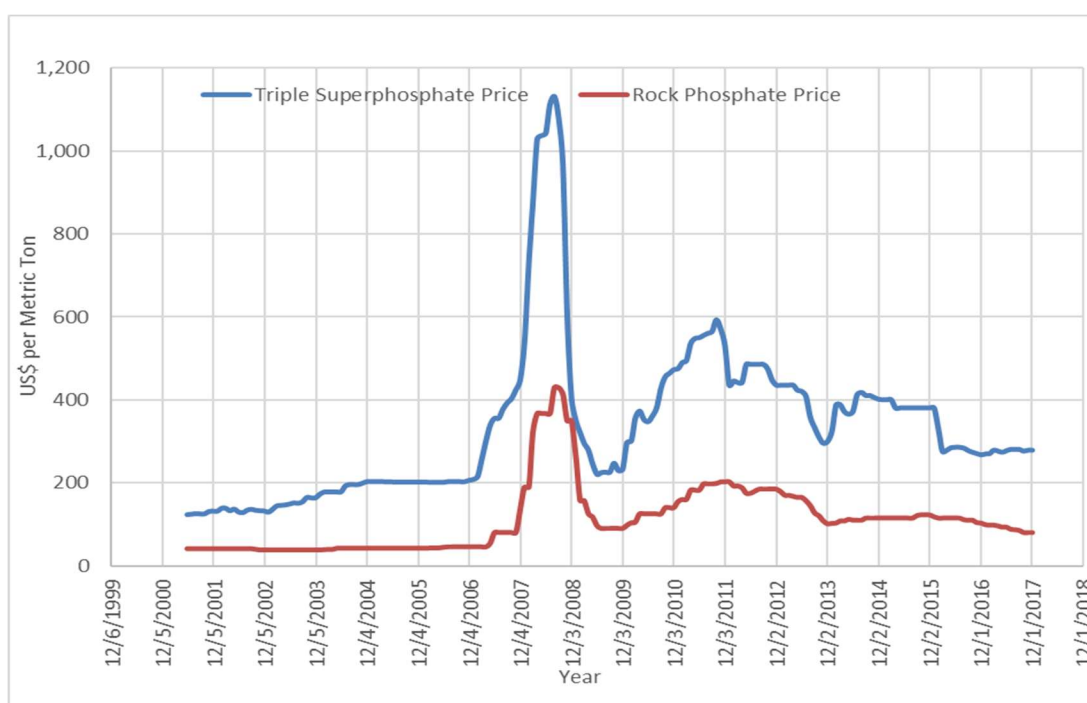


Figure 3. Costs of triple superphosphate and rock phosphate 2001 – 2017 (World Bank 2017).

Based on these simple cost analyses, phosphorus recovery as fertilizer through struvite method may not be economically justifiable. Therefore, other feasible P recovery

processes using lime to precipitate as Ca-P for P recovery for future use needs to be explored. The cost of crude Ca-based products could be significantly lower. A sustainable scenario would be to recover P in an inexpensive Ca-P form and store it for future refining into fertilizer when the need or economic considerations become attractive. Monofills or dedicated landfills of recovered crude P precipitates can be strategically located as regional reserves in urban areas to further reduce future mining and transportation costs.

2.4 Strategic P reserve

Phosphorus does not exist in gaseous form and cannot be manufactured. Unlike many other mineral commodities, there is no substitute element that can be used to replace phosphorus (USGS 2019). It is only through slow, natural weathering, and leaching processes that phosphorus is mobilized into terrestrial systems as PR for human mining (Suh and Yee, 2011). Since P is available only through PR mining, it is a non-renewable resource for all practical purposes. In the U.S., the majority of P consumed is provided for by domestic mining, and less than 24% of the P consumed relied on import in 2018 (USGS 2019). The criticality of P is, therefore, less than those minerals such as rare earth (compounds and metals) that can only be supplied by other countries. P is not listed on the Final List of Critical Minerals 2018 published by the Office of Secretary of the Department of Interior (83 FR 23295). Europe, on the other hand, relies 100% on P import. Recognizing how vital P is, the European Union (EU) included P as one of the twenty (20) critical raw materials in 2013 (European Commission, 2014). Despite the

seemingly sufficient PR reserves in the U.S. to date, P should be considered as a critical mineral and a strategic reserve is warranted for the following reasons:

- 1) The U.S. has approximately 2% of the total world PR reserves, and although not significant, the U.S. does rely on P imports. Understanding that the supply and demand relationship is dynamic and new mines are discovered to replace the exhausted mines, the situation can also turn to the opposite direction where the development of more advanced technologies to harness a higher grade of phosphate cannot keep up with the P demand. As P is essential to all life forms, a steady and dependable P supply is critical.
- 2) According to Section 12 (1) of the Stock Piling Act enacted in 1939 (50 USC § 98), strategic and critical materials are defined as those that would be needed to supply the military, industrial, and essential civilian needs of the United States during a national emergency, and are not found or produced in the United States in sufficient quantities to meet such need. If wheat and other grains are included in reserves and stockpiles, why not include P? With P being the building block for all plant growth, without P, there will be no grains, wheat, or food. The potential impacts of P shortage are unfathomable.
- 3) The U.S. Strategic Petroleum Reserve (SPR) was established as a lesson learned after the 1973 Arab Oil Embargo. Since then, the SPR has been used to offset disruptions to oil supply, such as after Hurricane Katrina in 2005 and the political upheaval in Libya in 2011. With more than 70% of the world PR reserves found in Morocco and Western Sahara (USGS 2019), geopolitical constraints are prone to happen. The 2007-2008 food crisis caused price surge of

fertilizers and PR. It is anticipated that food crises will happen again in the future (Headey and Fan, 2010). With potential geopolitical constraints and unpredictable price surges, it is critical to secure a P-independent future by stockpiling P today.

CHAPTER 3

METHODOLOGY

3.1 Overall Approach

This research took a two-step, systematic approach to demonstrate an economically viable and sustainable future P supply by recovering P from wastewater in a metropolitan area. The first step (Step 1) investigated the total amount of P that can be recovered from different sources at WRRFs and the potential costs associated with the different P recovery technologies that would produce fertilizer as final recovered products. As discussed on Chapter 2, the majority of P entering WRRFs would end up in one of these streams: sludge liquor; biosolids; and sludge ash. These are the sources for P recovery in Step 1. The overall approach for Step 1 included: 1) identifying WRRFs serving the studied area, 2) collecting data and treatment process information from the WRRFs, 3) developing P recovery scenarios and the corresponding P mass balances, and 4) estimating costs associated with the scenarios. The boundary conditions for Step 1 were set within the fence lines of the WRRFs. Therefore, transportation costs from WRRFs to the final biosolids disposal sites were not included in the evaluation. Results of Step 1 were expressed in unit costs per kilogram of P recovered so that the evaluation of the different P recovery sources and technologies were conducted on common ground. Step 1 is presented in Chapter 4 of the dissertation.

Step 1 identified the sources in WRRFs from which P can be recovered most cost-effectively. Once the sources were identified, a triple bottom line (TBL) analysis was conducted in Step 2 to compare the P recovery scenarios concluded in Step 1 and a “do nothing” (status quo) P recovery approach via land application since it represents the

majority of WRRFs' final disposal scheme of biosolids. The TBL analysis took into account criteria for a sustainability solution that considered not only the costs, but also the social and environmental factors such as health and safety impacts and the likelihood of additional pollution due to P recovery. The framework in conducting a TBL analysis followed the US EPA guidelines (2005) which consisted of 1) identifying baseline and alternative scenarios in P recovery and reuse, including development of boundary conditions and solids mass balances of each scenario, 2) determining evaluation criteria under the three pillars: economics, social, and environmental, 3) establishing the metrics and a standard scale for the selected evaluation criteria, and 4) evaluating scenarios using the metrics and the standard scale. Unlike Step 1, the boundary conditions for Step 2 extended from within the WRRFs' fence lines to the final disposal sites. Step 2 is presented in Chapter 5 of the dissertation.

Figure 4 presents a diagram representing the overall approach. Brief descriptions are presented below. More detailed approach, methodology, results, and discussions of each step are presented in Chapter 4 and Chapter 5.

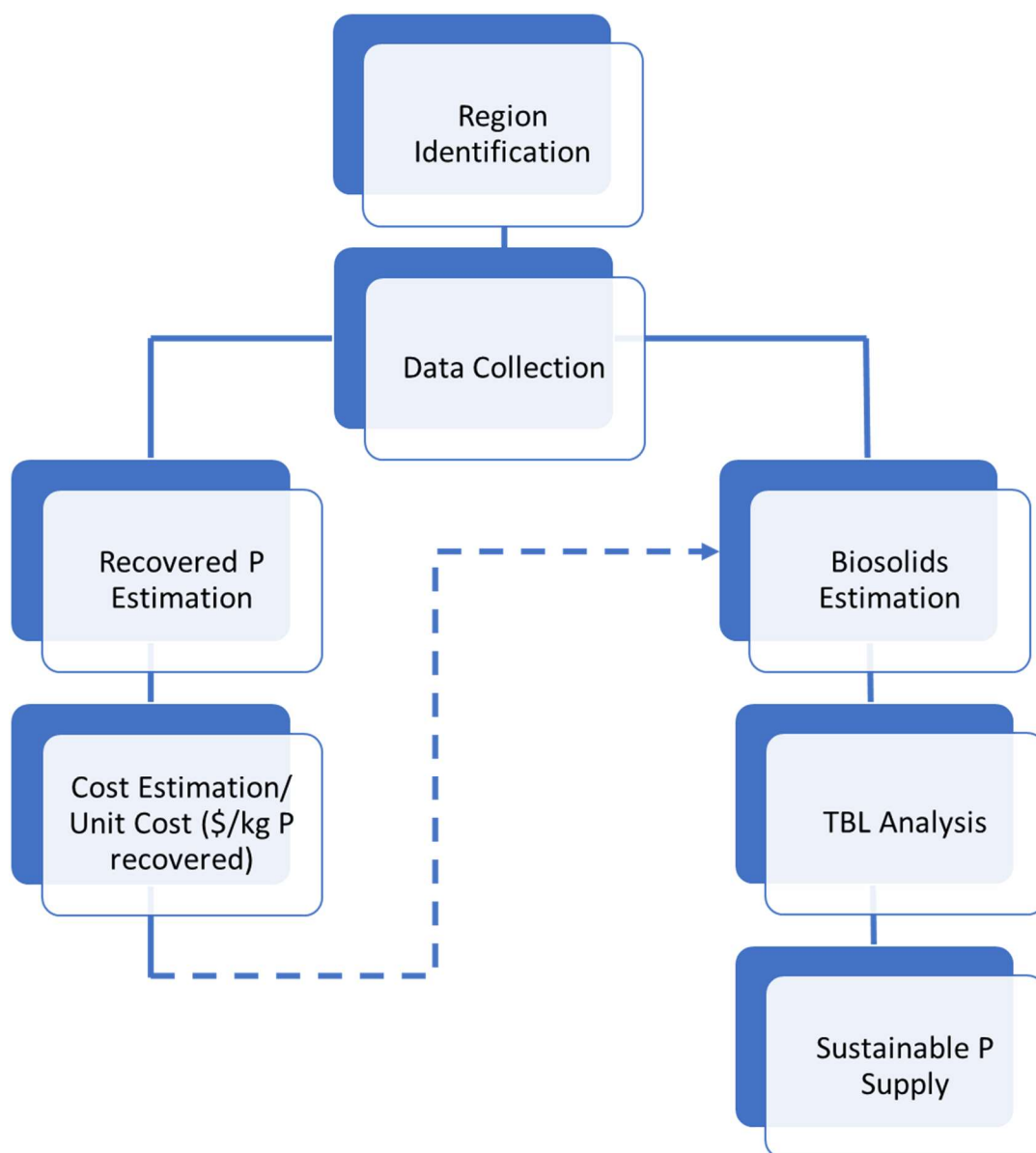


Figure 4. Overall approach for a sustainable P supply.

3.2 Region Identification and Data Collection

The metropolitan area identified for this research is the Chicagoland area which included six counties of a total population of 8.4 million (2010 Census). Data, including plant flows, influent total phosphorus (TP), and effluent TP, were collected for WRRFs for Year 2017. Data collection was conducted through access to WRRFs' websites and

USEPA Enforcement and Compliance History Online (ECHO) system website where appropriate information was posted, direct contact with WRRFs, and data regression method for facilities where information was not made available. In addition to the flow, influent and effluent TP, information regarding the types of liquid and solids handling treatment processes for all WRRFs were obtained based on the descriptions on the National Pollution Discharge Elimination System (NPDES) permits. All WRRFs larger than 3,800 cubic meter per day (1 MGD) were included in the dataset.

3.3 Development of Mass Balances

Based on the data and information such as treatment types collected, mass balances were developed to estimate the amount of P that can be recovered from sludge liquor, biosolids, and sludge ash, as well as the amount of biosolids produced collectively in the Chicagoland area. The amount of recovered P and the amount of biosolids produced were then used for cost estimation in P recovery and biosolids disposal. To develop the mass balances, assumptions such as percent P contents in sludge liquor, biosolids, and sludge ash and the percent solids content in various stages of solids handling treatment were made based on literature review.

3.4 Cost Estimation

Cost estimation included both opinion of probable construction costs (CapEx) and annual operation and maintenance costs (OpEx). Sources for both CapEx and OpEx were identified from facility reports and technology evaluation reports obtained from public domain. Where applicable, the equipment costs identified in reports published in past years were scaled up using Construction Cost Index published annually by the Engineering News-Record. In addition, the annual OpEx was extended for 20 years and

the present worth was calculated based on assumed interest and inflation rates for the Chicagoland area.

3.5 TBL Evaluation

Triple bottom line encompasses economic, social, and environmental impacts and benefits for a holistic approach in decision-making. Evaluation criteria under each category were developed based on the objective of the research which is to develop a sustainable P supply through a strategic P reserve from recycled P while protecting the public health and the environment. A scoring system was developed for both quantitative and qualitative criteria, with a common scale ranging from -3, representing the least favorable, to +3, representing the most favorable. A weighting system was also established based on priorities set forth by the Chicago Metro Agency of Planning of the Chicagoland area. An uncertainty analysis was performed to address the impacts of the underlying unknowns on the evaluation results.

CHAPTER 4

EVALUATION OF SOURCES FOR P RECOVERY

4.1 Introduction

Phosphorus can be recovered from sludge liquor, biosolids, and municipal sludge ash (MSA). As the amount of P that can be recovered from the three sources differ, so does the technologies and costs associated with it. This work was based on 2017 influent P loads data collected for all major WRRFs within a metropolitan area. The potential amount of P that can be recovered from the different sources was calculated based on P mass balances. A 20-year present worth associated with the P recovery technologies was estimated, and a final unit costs on a \$ per kg of P recovered basis was presented.

4.2 Methodology

4.2.1 Identifying WRRFs

Chicago, Illinois, is the third-largest metropolitan city in the U.S. (United States Census Bureau, 2012). From a regional perspective, the Chicagoland area includes six counties (Cook, DuPage, Kane, Lake, McHenry, and Will) and has a total population of 8.4 million (2010 census). This population represented 66% of the total population of 12.8 million in Illinois (2010 census). Since human waste contributes to the majority of the P loads entering WRRFs (Rittman et al., 2011), the Chicagoland area represents a dense population region with high potential for P recovery from wastewater. Major WRRFs in the Chicagoland area were identified using the USEPA Enforcement and Compliance History Online (ECHO) system. “Major” is defined as facilities with design average flow (DAF) of 3,800 m³/d (1 MGD) or higher. Since the amount of P that can be

recovered from the non-major WRRFs is relatively small, only major WRRFs were considered in the study.

Based on the ECHO website, there are 97 WRRFs in the Chicagoland area. Out of the 97 WRRFs, 80 of them are major WRRFs. Locations of all major WRRFs in the six (6) Chicagoland area counties are shown in **Figure 5**. Out of the major WRRFs identified, the three largest WRRFs are located in Cook County, and 61% are small facilities, ranging from 3,800 m³/d to 19,000 m³/d as shown in **Figure 6**.

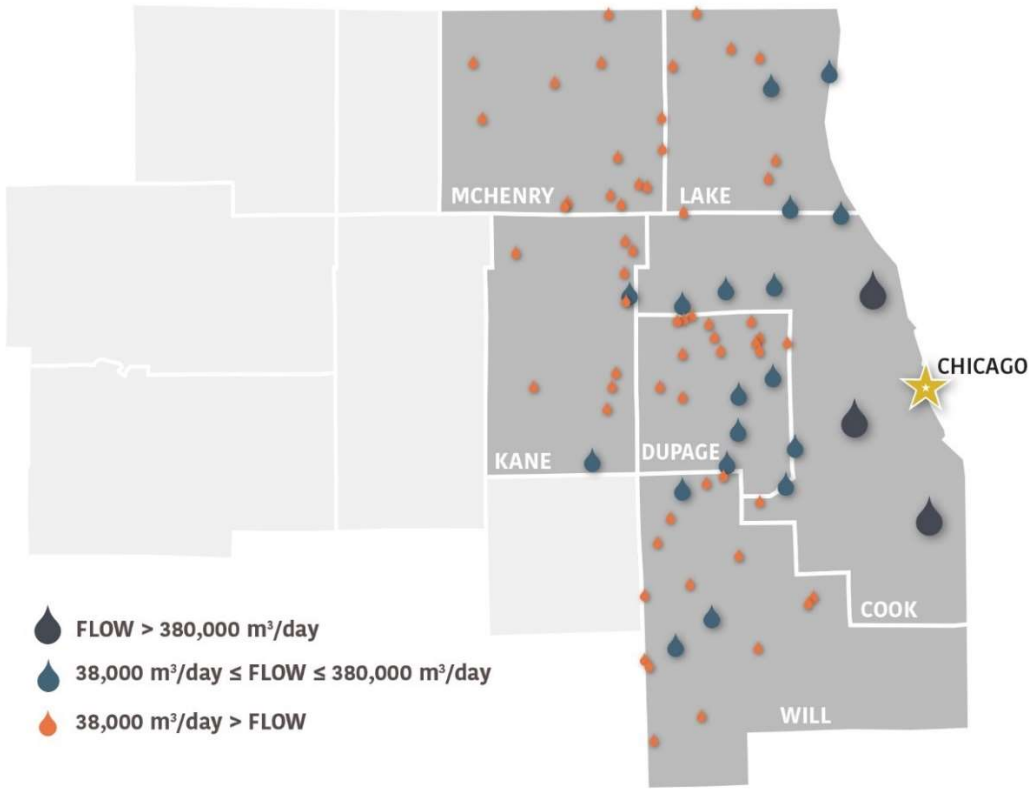


Figure 5. Major Water Resource Recovery Facilities in the Chicagoland Area.

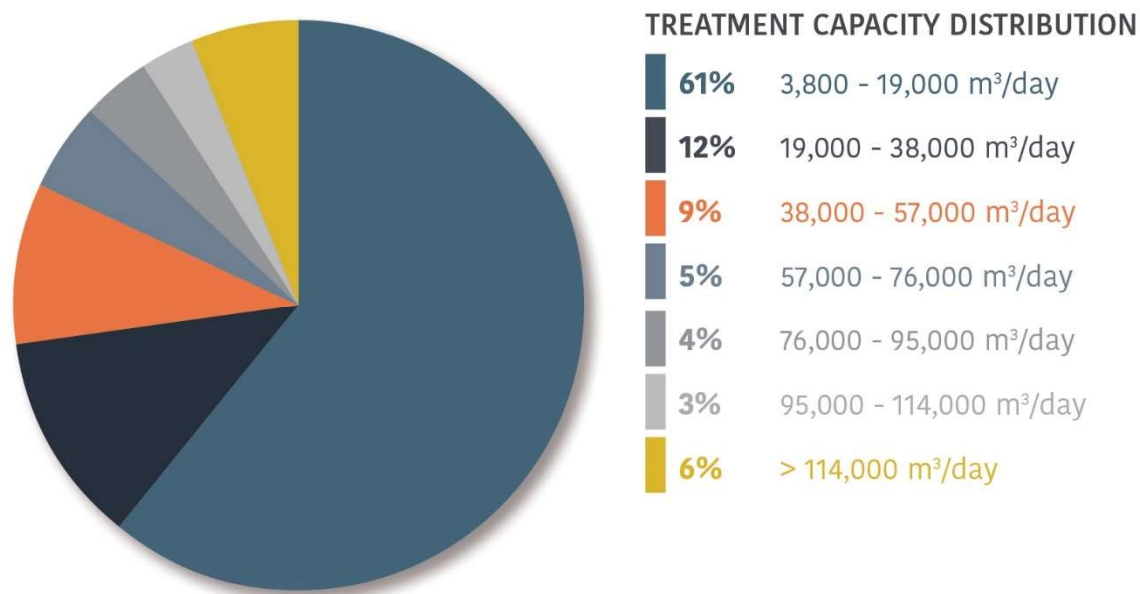


Figure 6. Treatment Capacity Distribution for the 80 WRRFs in Chicagoland Area.

4.2.2 Collecting data and treatment process information from WRRFs

Influent wastewater flow for each facility in 2017 was collected for all major WRRFs using the ECHO system. Influent wastewater TP data were obtained either from the ECHO system, through direct contact with facilities, or calculated based on regression analysis of available data. To derive P recovery scenarios that can be implemented at the WRRFs with minimal modifications to existing infrastructure, existing liquid and solids treatment schemes at each WRRF were identified based on review of the National Pollutant Discharge Elimination System (NPDES) permits for all major facilities. Each WRRF's liquid treatment process information was identified, such as suspended growth or attached growth, enhanced biological phosphorus or chemical precipitation phosphorus removal, and solids treatment processes (e.g. aerobic digestion, anaerobic digestion, or other means of solids stabilization).

Out of the 80 identified major facilities, data for 29 facilities were either retrieved from the ECHO website or provided by the facilities through direct contacts. These data sets represented the highest confidence level. Influent TP concentrations obtained before 2017, but within the past ten (10) years, were used for five (5) of the 80 major facilities. The concentrations were then multiplied by the 2017 average influent flows for the mass loadings in 2017. The communities served by these five (5) major facilities are well-developed, and the wastewater is domestic. These are separate sewer systems, and the influent characteristics to the WRRFs do not change to a large extent. Facilities without influent TP data (2017 or prior) were estimated using regression analysis of the 29 facilities with the highest level of confidence data. Additional analysis was conducted for the 29 facilities before the regression analysis. From the 29 facilities, seven (7) of them were identified as combined sewer overflow (CSO) systems. Since the facilities with unknown influent TP concentrations are all separate sewer systems (based on information provided on NPDES permits), data from the CSO systems were removed for the regression analysis. Based on the regression analysis, the influent TP loadings for the facilities were calculated. These loadings represented medium level of confidence. Treatment processes for the major WRRFs were identified based on descriptions on NPDES permits of the respective facilities. The major phosphorus removal processes, including EBPR and chemical precipitation, were included in the overall treatment train. Facilities that are yet to implement P removal processes or if the type of P removal process are unknown were identified as “unknown.” Distribution of the various liquid treatment processes, P removal processes, and the amount of P removed in 2017 based on EBPR, chemical precipitation, or an unknown method is presented in **Figure 7**.

As shown in **Figure 7b**, 49% of the facilities uses EBPR to remove phosphorus. However, collectively, this 49 % of facilities contributed to 96% of total phosphorus loads removed by EBPR as shown in **Figure 7c**. Regarding chemical precipitation for P removal, the type of chemicals used were not specified in the NPDES permits. For solids treatment (**Figure 7d**), the majority of the facilities (85%) implemented digestion, either aerobic, anaerobic, or both in some WRRFs for solids stabilization. Five (5) facilities implement heat drying. These biosolids are then disposed of either through land application or in a landfill. As indicated by NEBRA (2007), the majority of the WRRFs land apply their biosolids. As for biosolids disposal (**Figure 7e**), 62% of the WRRFs dispose of biosolids via land application; 5% listed both land application and “backup” landfill, and 7% transport their solids to other facilities for further treatment. Although not identified, it is reasonable to assume that the WRRFs which use heat drying for stabilization transport the dried product to a landfill for disposal. As shown in **Figure 7e**, currently there is no incineration facility in the Chicagoland area.

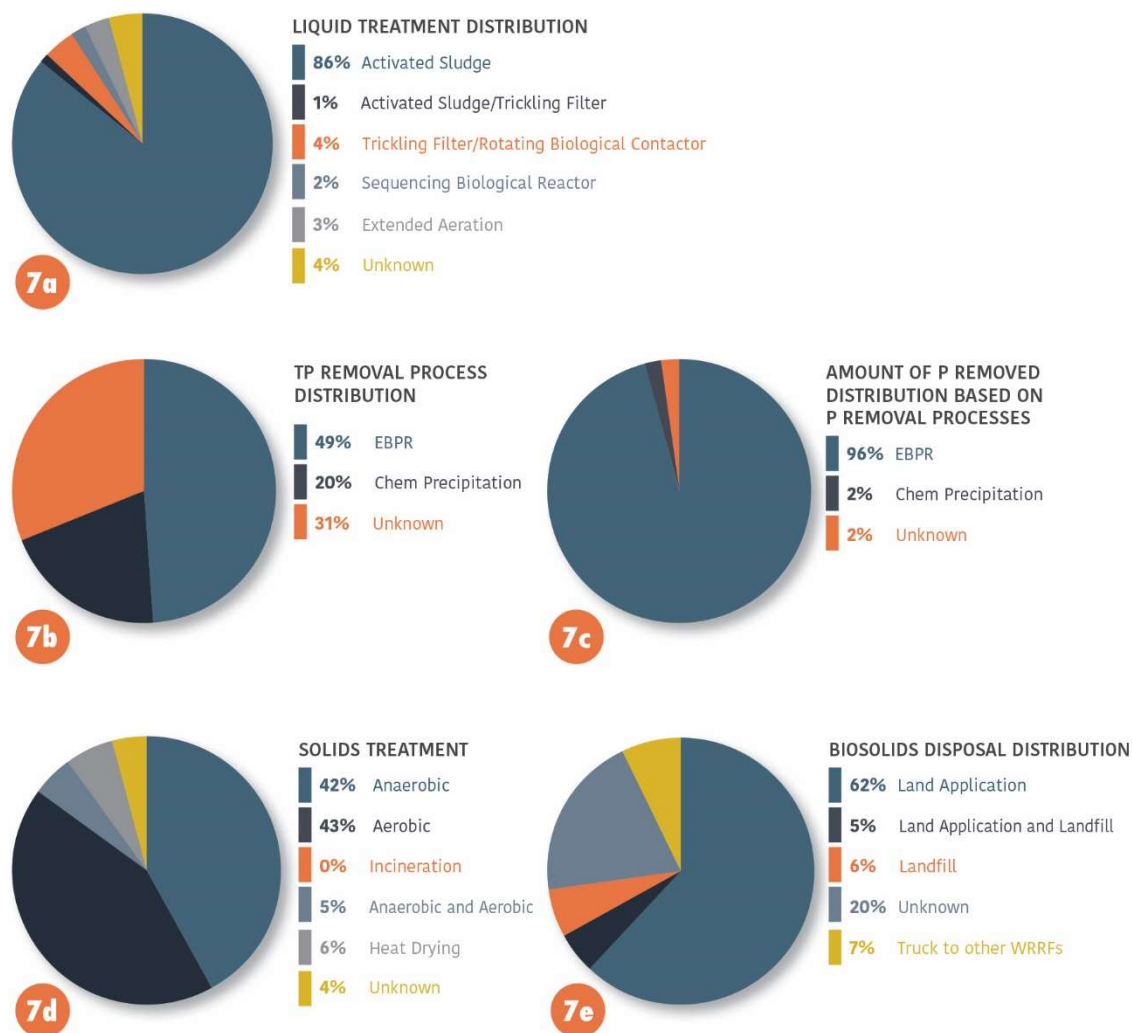


Figure 7. Distribution of liquid and solids treatment among the major WRRFs.

4.2.3 Developing P recovery scenarios and P mass balances

Phosphorus recovery scenarios were developed, taking into consideration the existing treatment configurations of the WRRFs in order to minimize necessary infrastructure modifications. According to USEPA (2013), the majority of wastewater treatment facilities implemented activated sludge (AS) systems rather than attached growth systems (such as trickling filters, TF). This is reflected in the data collected for

the major WRRFs in the Chicagoland area where 86% of the facilities implemented AS systems. A baseline treatment process was established using an AS system which consists of primary sedimentation, final sedimentation, and solids treatment, as shown in **Figure 8a**.

Since the P content in the liquid and the solids streams vary based on the treatment processes, five (5) P recovery scenarios were developed:

- Scenario 1: EBPR with anaerobic digestion
- Scenario 2: EBPR with aerobic digestion
- Scenario 3: Chemical precipitation with anaerobic digestion
- Scenario 4: Chemical precipitation with aerobic digestion
- Scenario 5: Chemical precipitation with incineration

They were first grouped according to the employed liquid treatment P removal process of either EBPR or chemical precipitation. Under each EBPR and chemical precipitation scenario, the WRRFs were sub-grouped into anaerobic digestion or aerobic digestion facilities based on the WRRF's approach for sludge treatment. This study considered only anaerobic and aerobic digestion due to the popularity of those digestion processes among the major WRRFs. The baseline flow diagram and the four (4) P recovery scenarios are presented in **Figures 8b** and **8c**. In an additional scenario, Scenario 5, using incineration as a means of biosolids disposal was also developed. Scenario 5 assumed chemical P removal to minimize infrastructure modifications and is coupled with incineration without digestion. This flow diagram is presented in **Figure 8d**.

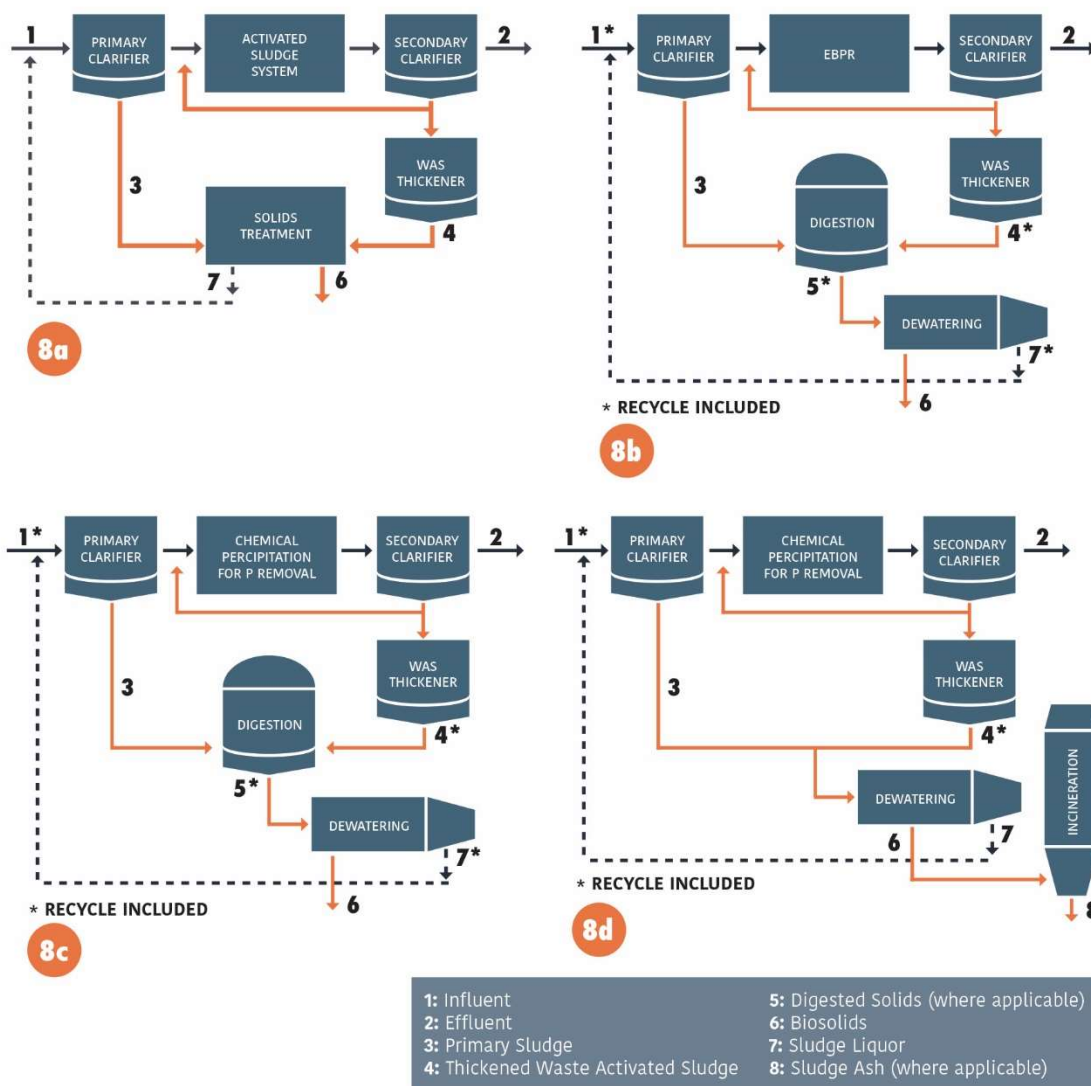


Figure 8. Flow diagrams for five (5) P recovery scenarios: 8a – Baseline; 8b – EBPR with anaerobic digestion (Scenario 1) and EBPR with aerobic digestion (Scenario 2); 8c – Chemical precipitation with anaerobic digestion (Scenario 3) and chemical precipitation with aerobic digestion (Scenario 4); 8d – Chemical precipitation with incineration (Scenario 5).

Based on the five (5) P recovery scenarios, phosphorus mass balances were developed. For each scenario, influent TP loadings were based on the 2017 annual average flows and influent TP concentrations of the WRRFs that were grouped under the

corresponding scenario. It is anticipated that, at a minimum, a 1.0 mg/L effluent TP limit will be imposed on Illinois WRRFs by the Illinois Environmental Protection Agency. Therefore, an effluent TP concentration of 1.0 mg/L has been used to develop mass balances.

Phosphorus removal starts at primary clarification where the influent particulate P is removed along with the suspended solids. Petzet and Cornel (2013a) indicated that a range of 17% to 26% of influent TP removal could be anticipated in primary clarification. Based on influent TP and primary effluent TP data available from the WRRFs, an average of 19% removal of influent TP was applied to the phosphorus mass balances for all scenarios. After primary clarification, primary effluent enters biological treatment where additional phosphorus is removed either through EBPR or chemical precipitation. Phosphorus content in biomass is impacted by the phosphorus removal mechanism (Egle et al., 2015). Studies have shown that P content in EBPR biomass ranges from 3% to 6% dry weight (Bi et al., 2013) whereas P content in conventional activated sludge biomass ranges from 2% to 3% dry weight (Menar and Jenkins, 1970). After secondary clarification, excess biomass is wasted for digestion and dewatering. Phosphorus content in dewatering sludge liquor is impacted by the type of digestion process. While there is secondary P release from EBPR waste biomass during anaerobic digestion due to the polyphosphate accumulating organisms (PAOs), the activities of PAOs are somewhat limited (Bi et al., 2013). Studies by Ju et al. (2005) and Bi et al. (2013) suggested that P release during digestion is related to biomass hydrolyses or volatile solids (VS) destruction regardless of the P removal processes. Their studies concluded that the rate of biomass hydrolysis is faster under anaerobic conditions than aerobic conditions. For

aerobic digestion, the VS destruction ranges from 38% to 50% (depending on time and temperature), while VS destruction of anaerobic digestion ranges from 56% to 65.5% (depending on time) (Tchobanoglous et al., 2003).

After digestion, biosolids are dewatered and dewatered sludge liquor is recycled back to the front end of the liquid treatment. P content in dewatered sludge liquor varies from 6% (with chemical P removal and aerobic digestion) to 26% (with EBPR and anaerobic digestion), based on the mass balances of the scenarios. Once the P mass balances were developed, the potential amount of P that can be recovered was estimated. Phosphorus recovery efficiency varies based on different technologies. Studies have shown that the efficiency can vary between 70% and 97% (Egle et al., 2016; Cohen et al., 2011; Ye et al., 2017; Peng et al., 2018). Using an 85% P recovery efficiency, the amount of P that can be recovered from different sources of the four (4) scenarios were estimated. As for the incineration scenario, primary sludge can be combined with thickened waste activated sludge and dewatered to 15% and 35% (USEPA 2003) before incineration. Since digestion would not be needed, the impacts of using EBPR versus chemical precipitation for P removal on P recovery would be minimal. The P content in the sludge liquor is anticipated to be between 2% and 3% which is similar to the P content in conventional activated sludge (Menar and Jenkins, 1970).

Phosphorus mass balances for the five (5) scenarios, as identified in **Figure 8** were developed, and the results are presented in **Table 4**. The amount of P that can be recovered per scenario is summarized in **Table 5**. Since the majority of the WRRFs implemented EBPR and anaerobic digestion, the amount of P recovered under Scenario 1 was the highest. Also, more P could be recovered from the biosolids than the sludge

liquor due to a higher P content in biosolids than in sludge liquor. This was reflected in **Table 5**. Results also indicated that collectively (Scenarios 1 through 4), approximately 31 metric tons of P can be recovered per day from biosolids when using 85% recovery efficiency. As there is no biosolids incineration facility in the Chicagoland area, the mass balance for Scenario 5 was developed using data from all 80 major WRRFs. The intent was to estimate the unit cost per kg of P recovered and compare with other P recovery scenarios on a unit cost basis.

Table 4. Phosphorus mass balances for P recovery scenarios.

Scenario	TP	Flow Stream Identification							
		1	2	3 ²	4	5	6	7 ^{3,4,5,6}	8 ⁷
1 (Fig. 7b)	kg/d ¹	51,396	5,540	7,755	38,101	45,856	35,274	10,582	NA
	% Inf TP	100	14	19	67	86	86	26	NA
2 (Fig. 7b)	kg/d	1,606	271	259	1,070	1,335	1,090	245	NA
	% Inf TP	100	20	19	61	80	80	18	NA
3 (Fig. 7c)	kg/d	1,297	257	228	811	1,039	945	94	NA
	% Inf TP	100	21	19	60	79	79	8	NA
4 (Fig. 7c)	kg/d	188	40	34	114	148	137	10	NA
	% Inf TP	100	23	19	58	77	77	6	NA
5 (Fig. 7d)	kg/d	44,303	6,108	8,275	29,920	38,195	37,446	728	36,383
	% Inf TP	100	14	19	67	86	86	2	86

Notes:

1. kg/d = kilograms per day
2. TP removal from primary treatment = 19% (2017 WRRFs data)
3. TP contents = 30% of digested sludge for Scenario 1 (Petzet and Cornel, 2013b).
4. TP contents = 22.5% of digested sludge for Scenario 2 (Tchobanoglous et al. 2003).
5. TP contents = 10% of digested sludge for Scenario 3 (Petzet and Cornel, 2013b).
6. TP contents = 7.5% of digested sludge for Scenario 4 (Tchobanoglous et al. 2003).
7. TP contents = 2% of dewatering feed for Scenario 5 (Menar and Jenkins, 1970).

Table 5. Phosphorus recovery potential for five scenarios.

Scenario	P Streams for Recovery, metric ton per day		
	Sludge Liquor	Biosolids	Sludge Ashes
1 (EBPR + AN DIG)	9.01	30.05	NA
2 (EBPR + AE DIG)	0.21	0.93	NA
3 (Chem P + AN DIG)	0.08	0.80	NA
4 (Chem P + AE DIG)	0.01	0.12	NA
5 (Chem P + Incineration)	0.64	NA	31.90

4.2.4 Estimating costs for the scenarios

Capital and annual operation and maintenance (O&M) cost analyses were conducted based on the P recovery technologies. The availability of cost information and the associated reclaimed P products were the main driving factors in technology selection. As a result, P recovery technologies used for cost estimation were as follows, with Ca-P in the form of fertilizer as the reclaimed product:

Scenarios 1, 2, 3, and 4: DHV Crystalactor for sludge liquor; Aqua Reci for biosolids.

Scenario 5: Mono-incineration of dewatered but undigested sludge, followed by DHV Crystalactor for sludge liquor and LEACHPHOS for sludge ash.

Since the P recovery technologies originated from around the world, and some are still in their infancy, the capital costs, chemical dosages, and energy requirements were based on literature information. Details are presented in **Table 6**. It should be noted that the operation costs considered included only costs within the boundaries of WRRFs. Therefore, transportation and final disposal costs were not included. Capital and cost constants are summarized in **Table 7**. Unless otherwise noted, cost constants are based on local information.

Table 6. Operational parameters (chemical dosage and energy requirements)

Chemicals	Chemical Dosage (kg/kg P recovered)	kWh/kg P recovered	Reference
DHV Crystalactor²			
Acetic acid (99%)	1.1	0.5 (electricity)	Egle et al. 2015
Calcium hydroxide (100%)	6.8		
Sulfuric acid (98%)	1.3		
Sodium hydroxide (100%)	0.9		
Aqua Reci			
Oxygen (100%)	32.5	11.5 (electricity) 119 (natural gas)	Egle et al. 2015
Calcium carbonate (100%)	7.4		
Sodium hydroxide (100%)	0.5		
LEACHPHOS			
Sulfuric acid (98%)	5.6	1.6 (electricity)	P-Rex 2015
Calcium oxide (100%)	3.9		
Sodium hydroxide (100%)	0.6		
Mono-incineration	NA	See Note 1	Haghi 2015

Note:

1. Annual O&M (in million) = $0.0744 \times C^{0.8594}$ where C = 1,000 MT of biosolids per year
2. Impacts of dewaterability were not accounted for.

Table 7. Capital costs and unit costs

	Capital Costs
DHV Crystalactor	\$79.2/kg P recovered/year
AqauReci	\$946/mt biosolids/year ¹
Mono-incineration	See Note 2
LEACHPHOS	\$79.2/kg P recovered/year ³
Chemicals	Cost Constants
DHV Crystalactor	
Acetic acid (99%)	\$1.5/kg
Calcium hydroxide (100%)	\$150/mt
Sulfuric acid (98%)	\$125/mt
Sodium hydroxide (100%)	\$882/mt
Aqua Reci	
Oxygen (100%)	\$3/kg
Calcium carbonate (100%)	\$73/mt
Sodium hydroxide (100%)	\$882/mt
LEACHPHOS	
Sulfuric acid (98%)	\$125/mt
Calcium oxide (100%)	\$77/mt
Sodium hydroxide (100%)	\$882/mt
Energy Unit Costs	
Electricity	\$0.06/kWh ⁵
Natural Gas	\$0.02/kWh ⁵
Labor	\$60,000 per year
Sales (as Ca-P)	\$276/mt ⁴
Annual Parts and Replacements	2% of capital costs

Note:

1. Biosolids quantities were calculated based on 5% P content from EBPR and 2% P content from chemical P removal.
2. Based on formula by Haghi 2015, Capital Costs (in million) = $2.3507 \times C^{0.7753}$ where C = 1,000 MT of biosolids per year
3. Capital costs for LEACHPHOS have been assumed to be the same as DHV Crystalactor
4. Gable, 2017.
5. Utility costs paid by local municipalities.

Present worth (PW) analyses were used for comparing technologies since the method takes into account the initial capital costs as well as the annual recurring O&M costs that are brought forth to today's values using a 20-year life span, an interest rate of

2.75%, and an inflation rate of 1.70%. A present worth factor (PF) of 17.95 was calculated using the following formula:

$$PF = \frac{(1+d')^{n-1}}{d'(1+d')^n}$$

Where d' = adjusted interest rate = $\frac{d-i}{1+i}$

n = no. of years

i = inflation rate

d = interest rate

Therefore, $PW = \text{Capital costs} + PF \times \text{annual O\&M costs}$

4.3 Results

The total present worth (PW) for each P recovery scenario and the unit cost per kilogram of P recovered are summarized in **Tables 8** and **9**, respectively. Costs for all scenarios took into account the potential Ca-P fertilizer revenues at US\$276/mt (Gable, 2017). As shown in **Table 8**, it would cost the US \$21 billion to recover P from sludge liquor and biosolids into fertilizer for the entire Chicagoland area on a 20-year PW basis. This is represented by the total PW of Scenarios 1 through 4. The PW would reduce to US \$2.1 billion should sludges from all WRRFs be incinerated in a central location followed by P recovery. This is represented by Scenario 5, a hypothetical scenario since currently no incineration is practiced in the Chicagoland area. On a unit cost basis, P recovery from sludge liquor is ten times less than P recovery from biosolids, even though more P can be recovered from biosolids. However, the unit cost of P recovery from sludge ashes would be further reduced when compared with P recovery from sludge

liquor. The amount of P recovered from sludge ashes would be comparable with P recovered from biosolids.

Table 8. Present worth for P recovery scenarios in the Chicagoland area.

Scenario	Present Worth, in millions		
	Sludge Liquor	Biosolids	Sludge Ashes
1 (EBPR + AN DIG)	473	19,610	NA
2 (EBPR + AE DIG)	11	607	NA
3 (Chem P + AN DIG)	5.7	538	NA
4 (Chem P + AE DIG)	0.7	78	NA
5 (Chem P + Incineration)	67	NA	2,004 ¹

Note:

1. Out of the total PW of Scenario 5, 32% of the PW is associated with incineration, and 68% of the PW is associated with the LEACHPHOs process for P recovery.

Table 9. Present worth in unit cost for P recovery scenarios in the Chicagoland area.

Scenario	Present Worth, \$/kg P recovered		
	Sludge Liquor	Biosolids	Sludge Ashes
1 (EBPR + AN DIG)	7.2	89	NA
2 (EBPR + AE DIG)	7.1	90	NA
3 (Chem P + AN DIG)	9.7	92	NA
4 (Chem P + AE DIG)	11	92	NA
5 (Chem P + Incineration)	14	NA	8.6

4.4 Discussion

A phosphorus recovery (as fertilizer) PW analysis was completed for all 80 major WRRFs in the Chicagoland area based on minimal modifications to existing treatment processes at the WRRFs. A total PW of \$21 billion would be needed should all WRRFs recover P as fertilizer. In particular, results indicated that recovering P as fertilizer from sludge liquor costs approximately US \$7.1 to US \$14 per kilogram (kg) P recovered while P recovery from biosolids cost approximately US \$92 per kg P recovered. When

compared with the current price of triple superphosphate (TSP, a commercial fertilizer) of \$326/metric ton (mt) (or \$1.66/kg as P) and rock phosphate of \$103/mt (or \$0.74/kg as P) (World Bank, May 2018), purchasing fertilizer or rock phosphate is more economical than intentionally recovering phosphorus from WRRFs. Based on the current prices of TSP and rock phosphate, it is challenging to justify P recovery from an economic standpoint. Municipalities will need to identify other justifications, such as water quality improvements, in order to move forward with the initiative.

This PW analysis also indicated that it would be much more expensive to recover P from biosolids than from sludge liquor. On a unit cost (\$ per kg P recovered) basis, P recovery from biosolids is ten times the costs to recover P from sludge liquor, as shown in **Table 9**. Phosphorus recovery from biosolids requires either a wet chemical or a thermal process, and either process is energy-intensive (Yoshida et al., 2013). Much of the cost associated with biosolids P recovery is related to energy costs (Tyagi and Lo, 2016). Most, if not all, biosolids P recovery technologies are still not widely adopted for municipal applications due to the complexity and a low maturity level of the technologies. It is impractical and not economically viable to recover P from biosolids.

As for P recovery from sludge liquor, the amount of P discharging into lakes and streams can be reduced, which in turn will also improve water quality. Since only a limited amount of P (up to 30% of influent P load) can be recovered, the remaining P in the biosolids can be used as soil supplements through land application, a status-quo approach in recovering P. From a cost perspective, O'Dette (1996) indicated that the cost for land application ranges from US \$60 to US \$290 per dry ton (DT), which is equivalent to a 20-year PW of US \$1.19 per kg to US \$5.75 per kg of biosolids. Reasons

for the full range of costs are due to the differences in dewatering technologies, farmland availability, and the hauling distances between WRRFs to farm fields where biosolids are applied (USEPA 2000). Regardless, the PW unit costs for land application can be comparable with the costs for P recovery from sludge liquor. However, as discussed earlier, P recovery through land application may cause excess P runoff and the regulation for this age-old practice will only become more stringent due to concerns with emerging contaminants. Therefore, a more practical and cost-effective approach to recover P is needed. This leads to an investigation of P recovery from sludge ashes.

Understanding that the PW analyses represented a high level of cost estimation, which consists of many assumptions and unknowns, a cost sensitivity analysis was performed using a range of accuracy levels for the estimates. The accuracy level of the cost estimation falls under Class 5 (Concept Screening) of the *Recommended Practice No. 18R-97 - Cost Estimate Classification System – As Applied in Engineering, Procurement, and Construction for the Process Industries* published by AACE International (2019). Under Class 5 definition, the expected accuracy ranges from the low end of -20% to -50% to a high end of +30% to +100%. Using the extreme low and high end, -50% and +100%, the PW unit cost for the low end of the biosolids recovery costs is still three times higher than the high end of the sludge ash recovery costs. The recovery potential from the sources is not impacted by the cost of recovery.

The results of this work align with a cost assessment study conducted by Nattorp et al. (2017). Nattorp's study concluded that the costs per capita or costs per kg of P recovered are in the order of sludge leaching (i.e., P recovery from biosolids) > ash-based processes (i.e., P recovery from sludge ash) > precipitation processes (i.e., P recovery

from sludge liquor) and the magnitude of costs (per kg of P recovered) associated with the various P recovery technologies are 230% to 600% higher than the current TSP costs. Drivers for P recovery will need to include environmental impacts in addition to costs.

4.5 Future work

As discussed previously, currently, there is no biosolids incineration facility in the Chicagoland area. A hypothetical scenario (Scenario 5) was developed to estimate the PW to recover P from sludge ashes by disposing all solids generated through incineration. Results from the analysis indicated that a PW unit costs of US \$7.1 per kg P recovered would be incurred to recover P from sludge ashes. The unit cost is comparable with land application and P recovery from sludge liquor as shown in **Table 9**. However, a large quantity of P can be recovered from sludge ashes. Results from the PW analysis of all scenarios were plotted in **Figure 9** to depict the potential amount of P that can be recovered from different streams and the corresponding unit costs. The size of the circle is proportional to the amount of P that can be recovered. As shown in **Figure 9**, the unit cost for P recovery from sludge ash is comparable with that from sludge liquor, but with a larger reclaimed P quantity. On the other hand, although a large quantity of P can be recovered from biosolids, the unit costs associated with the process are very high.

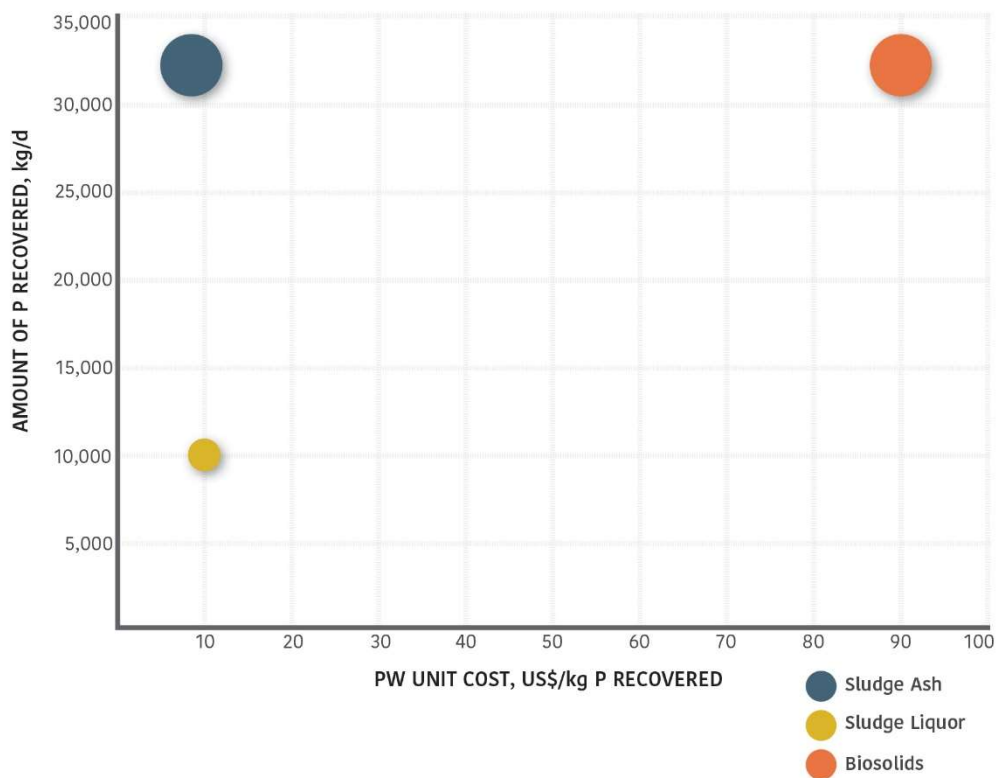


Figure 9. The amount of P recovered from different streams and the corresponding unit costs in the Chicagoland area.

Based on the magnitude of costs of the PW analyses, it can be concluded that the unit cost to recover P from sludge ashes is comparable with the unit cost to recover P from sludge liquor, as well as the cost for land application as a soil supplement, a status-quo approach to recover P for beneficial reuse. An additional advantage of P recovery from sludge ashes includes reduced disposal costs since the volume of the biosolids to be disposed of would be substantially reduced after incineration. Regulations for land application would be less of a concern. However, more importantly, it offers an opportunity to centralize P recovery for all the small WRRFs who otherwise cannot afford P recovery implementation at their facilities. WRRFs, regardless of their size, can

transport their biosolids to a centralized P recovery facility where biosolids can be incinerated, and P can be recovered from the sludge ashes. Also, the final product of P recovery does not need to be in fertilizer form for immediate use. P can be recovered as Ca-P which has a composition close to mined PR. The recovered Ca-P can be stored in a monofill as a commodity reserved for future use and consequently reduce the dependency of PR from other countries. The costs to recover Ca-P may be lower than recovering P as fertilizer.

The overall findings of this work are in alignment with Amana et al. (2018), where the authors reported that P recovery from sludge liquor might have a lower impact on energy demand and gas emissions compared to recovery from biosolids and sludge ashes. P recovery from sludge liquor also has the lowest P recovery potential. Comparatively, P recovery from biosolids has the highest gas emissions and highest energy demand, while P recovery from sludge ashes offers the highest P recovery potential and heavy metal decontamination but has varying energy demands. This work proves a practical and viable future path to recover P from sludge ashes and reserve P on a regional scale. However, much work is yet to be completed for a well, thought-out plan in order to execute the P commodity reserve concept. Future work such as a more detailed cost estimates associated with a centralized P recovery facility, which includes onsite incineration, P recovery processes from sludge ashes, and a monofill is needed. Additional costs for biosolids transportation from the WRRFs to the centralized P recovery facility will require further research. From an administrative perspective, entities or private companies to oversee the centralized facility will need to be considered. An idea of privatizing the centralized facility could be explored for as long

as the costs for biosolids disposal at the centralized facility are lower than the costs that WRRFs are paying now. Onsite sludge stabilization could be eliminated, and therefore operating costs could be reduced. On the other hand, there would no biogas generation for beneficial reuse. A Triple Bottom Line (TBL) evaluation including not only economical but also environmental and social aspects, will need to be performed for a holistic view of a regional P commodity reserve.

CHAPTER 5

TRIPLE BOTTOM LINE ANALYSIS FOR A SUSTAINABLE P SUPPLY

5.1 Introduction

With the objectives of developing a sustainable P supply while protecting the public health and environment, a TBL analysis was conducted, taking into consideration the three pillars: economic, social, and environmental. This work is a continuation of previous work conducted by Law and Pagilla (2019) (Chapter 4), which concluded that on a unit cost basis, the magnitude of cost to recover P from sludge liquor and MSA are comparable. This results in additional analyses that took into account social and environmental factors for a sustainable future P supply. This work addressed unanswered questions from the previous study related to the cost of transportation of biosolids and the construction of a monofill for recovered P storage. Also, the cost for incineration has been refined based on a recent 2018 facility plan report for the Metro Plant owned and operated by the Metropolitan Council in Minneapolis, Minnesota.

The framework in conducting a TBL analysis included these steps (USEPA 2015): 1) Identifying baseline and alternative approaches in P recovery and reuse, including development of boundary conditions and mass balances for each alternative, 2) Determining evaluation criteria under the three pillars: economics, social, and environmental, 3) Establishing the metrics and a standard scale for the selected evaluation criteria and 4) Evaluating scenarios using the metrics and the standard scale established.

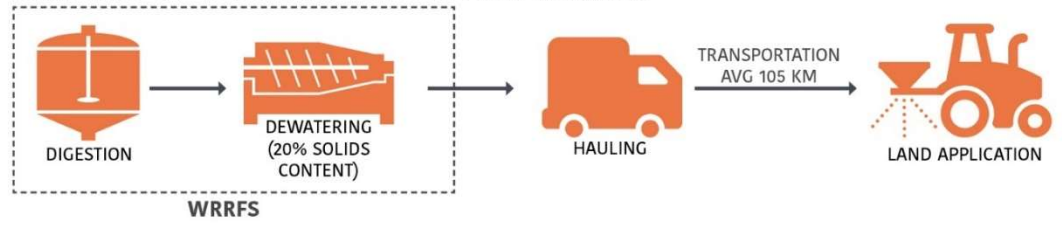
5.2 Methodology

5.2.1 Identifying baseline and alternative scenarios, boundary conditions, and developing solids mass balances

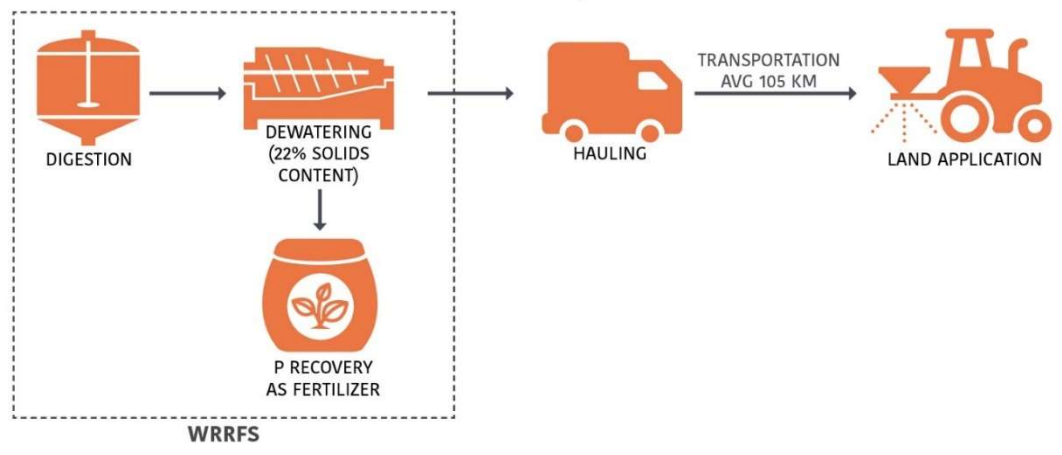
Currently, the most commonly practiced P recovery approach in the U.S. is via direct land application (Blocher et al. 2012). This could be considered a status-quo scenario in all regions where land application of biosolids/sludge is allowed. Biosolids from WRRFs are rich in nitrogen and phosphorus. Upon meeting USEPA Class A or B biosolids classification for pathogen reduction, WRRFs typically dispose of their biosolids to nearby farmland as a source of N and P and as soil supplement. The baseline scenario, P recovery via land application (Scenario 1), includes anaerobic digestion and dewatering of sludge at each WRRF, and transportation (via truck hauling) of dewatered biosolids to farmland for land application. A flow diagram representing Scenario 1 is shown in **Figure 10**. Scenario 2 represents P recovery as N and P fertilizer for immediate use. P recovery from sludge liquor is the most commonly implemented; active P recovery approach at WRRFs due to the relative ease of implementation and potential revenue of fertilizer sales. This scenario included anaerobic digestion, dewatering, P recovery as fertilizer via chemical precipitation of dewatered sludge liquor (based on the DHV Crystalactor process to produce calcium phosphate), and transportation (via truck hauling) of dewatered biosolids to farmland for land application. A flow diagram representing Scenario 2 is shown in **Figure 10**. Scenario 3 represents a recovery of P into calcium phosphate (CaP) form from all sludge generated at a WRRF followed by CaP storage in a dedicated P reserve. This scenario included undigested sludge dewatering at each WRRF, followed by transportation (via truck hauling) to the

centralized P recovery facility. Undigested sludge was used due to higher sludge calorific value than the digested sludge (Tchobanoglous et al., 2003). With undigested sludge, the energy required for the mono-incineration process can be self-sustained with external energy needed only for process start-up (USEPA 2003, Donatello et al., 2013). At the centralized facility, mono-incineration would be implemented for the undigested, dewatered sludge to produce sludge ash as the final product from which phosphorus would be extracted from and precipitated as Ca-P based on the LeachPhos process (Herzel et al., 2015). Both the Ca-P and the residue ash would be stored in separate monofills at sites nearby the centralized P recovery facility. A sophisticated air pollution control system would be required for the flue gas before it is dispersed into the atmosphere. A flow diagram representing Scenario 3 is shown in **Figure 10**. A proposed location for the centralized P recovery facility and P reserve monofill for the case study of Chicago (IL, USA) area WRRFs is presented in **Figure 11**. It has been determined that the facility would be located in an existing quarry that is centrally located near the city for easy access and reduced transportation costs from each of the WRRFs in the region.

SCENARIO 1: P RECOVERY VIA LAND APPLICATION



SCENARIO 2: P RECOVERY FROM SLUDGE LIQUOR AS FERTILIZER FOR IMMEDIATE USE



SCENARIO 3: P RECOVERY FROM SLUDGE ASH AS CaP (P RESERVE) FOR FUTURE USE

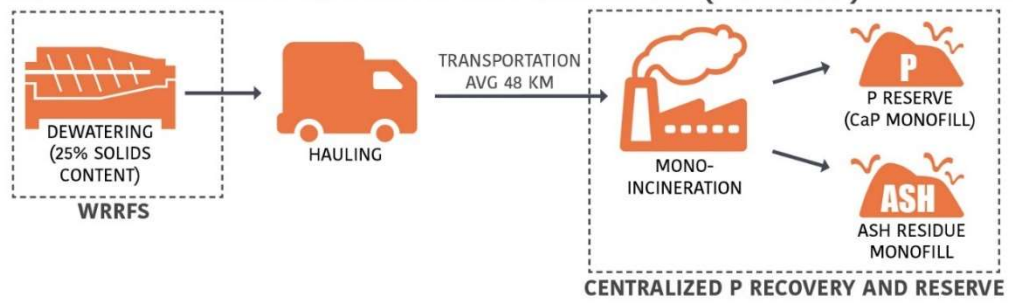


Figure 10. Flow diagrams for P recovery scenarios.



Figure 11. Proposed locations for a centralized P recovery facility including mono-incineration, P recovery, and monofills for Scenario 3.

With the scenarios identified, boundary conditions and mass balances were developed accordingly. A previous study conducted by Law and Pagilla (2019) indicated that the concept of building a regional P reserve through a centralized P recovery facility is economically justifiable compared to the status-quo practice. During the study, 2017 influent wastewater flow rate and phosphorus concentration data were collected for the WRRFs in six counties surrounding Chicago, IL: Cook, DuPage, Kane, Lake, McHenry, and Will County. Using the same data set, total solids (TS) production was estimated using the 2017 Stickney Water Reclamation Plant (SWRP) data which represented 42%

of the total flow of the six counties. The solids production was then used to develop mass balances for the three scenarios. In addition to the solids production, a 3.4% TS concentration in the sludge prior to digestion and 3.0% TS in the sludge after digestion was also considered for the scenarios based on the 2017 SWRP data. These percent TS values fall within a typical range of TS for pre- and post-digestion sludge in typical municipal WRRFs (Tchobanoglous et al., 2003). For percent solids after dewatering, a 20%, 22% and 25% solids content were used for Scenario 1, 2, and 3, respectively. These percent solids represent the solids characteristics and the ease of dewatering. For Scenario 1, a portion of the solids are from enhanced biological P removal (EBPR) processes and the dewaterability would be the worst among the scenarios (Mangrum and Jenkins, 2019). Scenario 2 represents solids production with P recovery and the dewaterability is expected to be better when compared with Scenario 1. For Scenario 3, the sludge dewaterability would be the easiest because digestion is not included. In Scenario 3, it has been reported that mono-incineration can reduce solids volume of undigested-dewatered sludge by 90% (Oladejo et al., 2019). For the mass balance, an 85% solids reduction was used for Scenario 3.

For sludge hauling impacts, a specific gravity of 1.22, 1.35, and 1.55 was used for Scenarios 1, 2, and 3, respectively (O’Kelly, 2005). Considering the amount of solids that needs to be hauled from the mass balances, the specific gravity, and a standard hauling truck volume of 15.3-cubic meter, the number of trucks for each scenario was estimated. A summary of the mass balances for each Scenario is presented in **Figure 12**.

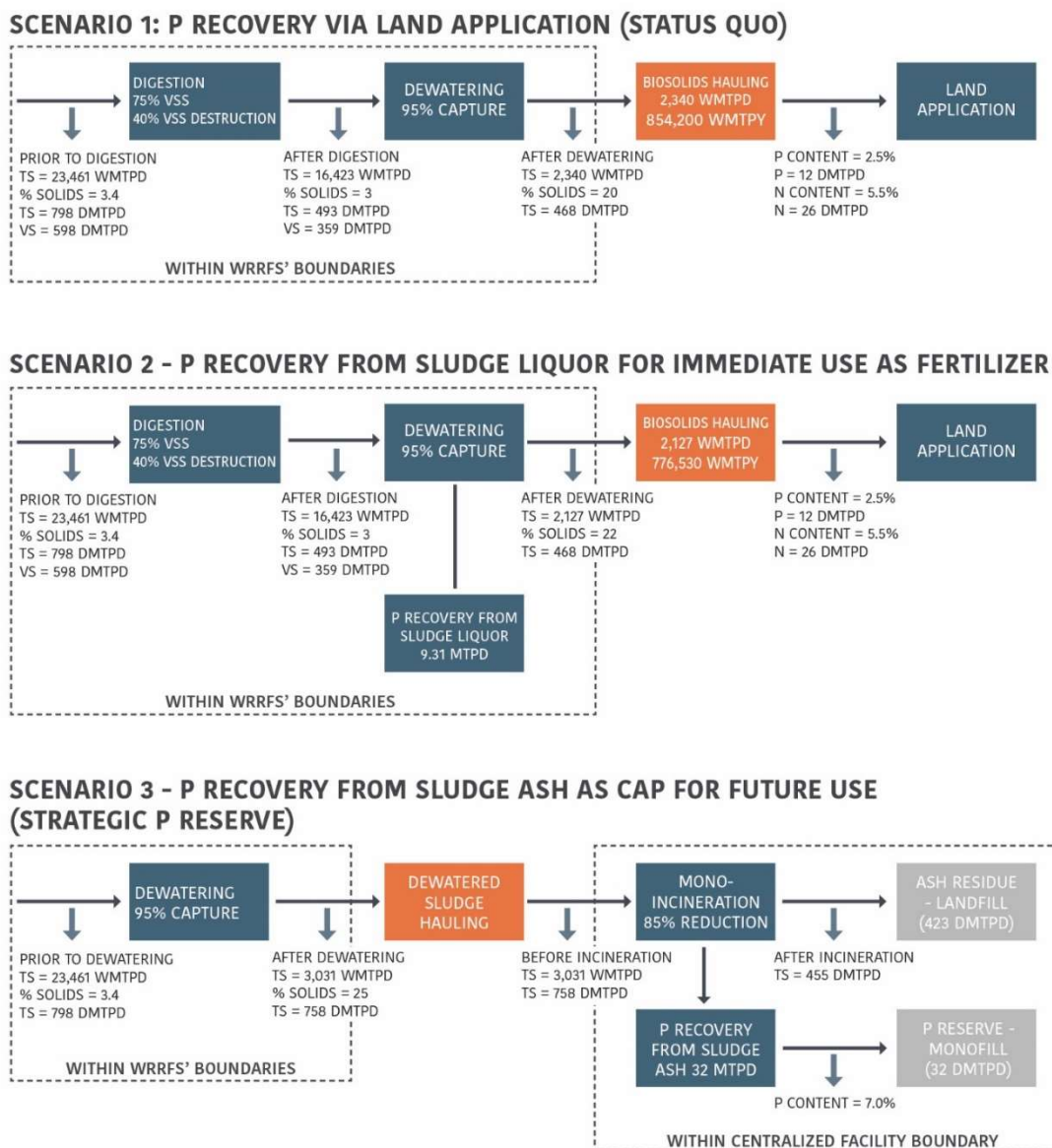


Figure 12. Mass balances for Scenarios 1, 2, and 3 of the P recovery alternatives.

5.2.2 Identifying evaluation criteria for the three TBL component pillars: economic, social, and environmental categories

When determining the evaluation criteria, similar research studies on P recovery using life cycle assessment (LCA) were reviewed and considered. The common criteria used include abiotic depletion (mineral and energy resources), eutrophication, human toxicity, and global warming potential (Linderholm et al., 2012; Kalmykova et al., 2015;

Fang et al., 2016; Sena and Hicks, 2018; Pradel and Aissani, 2019). Taking into consideration the criteria used in previous studies and the objective of this study which is to develop a practical approach to secure a sustainable P supply while protecting the health of the public and the environment, the criteria selected for this study are: Economic category including capital costs (CapEx) and operating costs (OpEx); Social category including public health and sustainable P supply; and Environmental category including future regulations, air quality impacts, water quality impacts, and greenhouse gas (GHG) emissions.

Under the economic criteria, expenses such as capital investment and recurring operation and maintenance costs were considered. Capital investment includes infrastructure, process equipment, auxiliary equipment such as piping and valving, electrical, instrumentation and controls, heating, ventilation, and air conditioning, installation, and land acquisition (Scenario 3 only). For a planning level evaluation, the accuracy of the capital costs is anticipated to meet Class 5 (Concept Screening) of the *Recommended Practice No. 18R-97 - Cost Estimate Classification System – As Applied in Engineering, Procurement, and Construction for the Process Industries* published by AACE International (2019). Under Class 5 definition, the expected accuracy ranges from the low end of -20% to -50% to a high end of +30% to +100%. Capital costs, unit costs, and cost constants used in the study are summarized in **Table 10**. Operation and maintenance costs included energy (electricity and natural gas), chemicals, and labor associated with the digestion (Scenarios 1 and 2), dewatering (all scenarios), incineration (Scenario 3), and P recovery processes (all scenarios). In addition, biosolids transportation costs include truck hauling to and from the final disposal sites (Scenarios 1

and 2) or the centralized facility (Scenario 3). Where applicable, land application costs were included (for Scenarios 1 and 2). Potential revenue of recovered products, operational savings such as use of biogas to offset natural gas consumption or reduced polymer usage due to improved dewaterability with P recovery were also included in the OpEx for all scenarios. Chemical dosages and energy requirements for each process are summarized in **Table 11**.

Table 10. Capital costs and unit costs for TBL analysis

Technology	Capital Costs	Reference
DHV Crystalactor	\$79.2/kg P recovered/year	Egle et al. 2015
Mono-Incineration	\$826,000/dry mt of solids per day	Note 1
LeachPhos	\$1,700/dry mt of ash per year	Nattorp and Remmen, 2015 Note 2
Chemicals	Unit Cost/Cost Constant	
DHV Crystalactor		Egle et al. 2015
Acetic acid (99%)	\$1.5/kg	
Calcium hydroxide (100%)	\$150/mt	
Sulfuric acid (98%)	\$125/mt	
Sodium hydroxide (100%)	\$882/mt	
LEACHPHOS		Herzel et al., 2015
Sulfuric acid (98%)	\$125/mt	
Calcium oxide (100%)	\$77/mt	
Sodium hydroxide (100%)	\$882/mt	
Polymer (for dewatering)	\$6.5/kg	Note 1
Energy Unit Costs		
Electricity	\$0.06/kWh	Note 3
Natural Gas	\$0.02/kWh	Note 3
Transportation (less than 50 miles)	\$16.5/wet mt	Note 3
Transportation (more than 50 miles)	\$38.6/wet mt	Note 3
Land Application	\$16.5/wet mt	Note 3
Labor	\$60,000 per year	
Sales (as Ca-P)	\$276/mt	Gable, 2017
Annual Parts and Replacements	2% of capital costs	

Note:

1. Metro Plant Solids Management Improvements Facility Plan, July 2018.
2. Estimated CapEx = $(S/S_0)^{0.5}$ where S = targeted amount of ash and S_0 = amount of ash in study (Nattorp and Remmen, 2015).
3. Utility costs paid by local municipalities.

Table 11. Operational parameters for TBL analysis

Chemicals	Chemical Dosage (kg/kg P recovered)	kWh/kg P recovered	Reference
DHV Crystalactor²			
Acetic acid (99%)	1.1	0.5 (electricity)	Egle et al. 2015
Calcium hydroxide (100%)	6.8		
Sulfuric acid (98%)	1.3		
Sodium hydroxide (100%)	0.9		
LEACHPHOS			
Sulfuric acid (98%)	5.6	1.6 (electricity)	Herzel et al., 2015
Calcium oxide (100%)	3.9		
Sodium hydroxide (100%)	0.6		
Mono-incineration	NA		Note 1

Note:

1. Metro Plant Solids Management Improvements Facility Plan, July 2018.
2. Impacts of dewaterability was not accounted for.

Present worth (PW) analyses were used for comparing scenarios since PW considers the initial capital costs as well as the annual recurring O&M costs that are brought forth to today's values using a 20-year life span, an interest rate of 2.75%, and an inflation rate of 1.70%. A present worth factor (PF) of 17.95 was calculated using the following formula:

$$PF = \frac{(1+d')^n - 1}{d'(1+d')^n}$$

$$\text{Where } d' = \text{adjusted interest rate} = \frac{d-i}{1+i}$$

n = no. of years

i = inflation rate

d = interest rate

Therefore, $PW = \text{Capital costs} + PF \times \text{annual O\&M costs}$

For the social criteria, two criteria that were most applicable to the objectives of the TBL analysis were used: sustainable P supply and public health. Sustainable P supply relates to the potential amount of recovered P and how the recovered P may impact the P supply in the future within the region. Public health relates to potential health and safety impacts on human beings due to the contaminants in biosolids that cannot be easily removed. Quality of life was considered as a criterion but it was not used due to the minimal change in land use for all scenarios. Scenario 1 does not change the current land application landscape. Scenario 2 includes P recovery onsite at the WRRFs and land applying the remaining biosolids. For Scenario 3, since an existing quarry was proposed to be the ash residue landfill and the P reserve monofill, no significant change related to quality of life such as aesthetics, odor, noise, traffic, etc. is anticipated.

For the environmental criteria, potential future regulations, air quality impacts, water quality impacts, and greenhouse gas (GHG) emissions were all considered under the environmental category. Future regulations referred to the likelihood of more stringent regulations related to biosolids management. Air quality impacts relates to potential impairment to existing air quality, and water quality impacts refers to further deterioration of water quality in receiving waterbodies. As for GHG emissions, a Biosolids Emissions Assessment Model (BEAM) developed by the Canadian Council of Ministers of the Environment (CCME) (CCME, 2011) was used. BEAM includes biosolids handling processes from removal of sludge from clarifiers to final biosolids disposal and utilization. The model considers three emission scopes: Scope 1 includes direct emissions such as fugitive CH₄ from anaerobic digestion or emissions from

burning natural gas to heat digesters if applicable; Scope 2 includes indirect emissions such as energy consumption; and Scope 3 includes all other indirect emissions that are not covered in Scope 2. Examples for Scope 3 includes manufacture and transport of polymer for use in dewatering or other biosolids conditioning purposes. Scope 3 also included fertilizer offset credits when land application is used. It should be noted that GHG emissions accounted for operations of the scenarios and did not account for emissions due to construction activities. Unit processes used from BEAM based on each scenario are listed in **Table 12**.

Table 12. Unit processes of BEAM applicable to Scenario 1, 2, and 3.

BEAM Unit Process	Scenario 1	Scenario 2	Scenario 3
Aerobic Digestion ¹	x	x	
Anaerobic Digestion ^{1,2}	x	x	
Dewatering	x	x	x
Combustion ³			x
Land Application ⁴	x	x	
Transportation ⁵	x	x	x

Note:

1. Based on the 2017 dataset from the six counties of the Chicagoland area (Law and Pagilla 2019), it has been assumed that 96% of the total solids produced is anaerobically digested and 4% of the total solids is aerobically digested.
2. Methane gas utilization has been assumed at 75% as heat and 25% as electricity.
3. The combustion process has been assumed to be self-sustainable for Scenario 3 due to a high raw sludge fuel value.
4. Land application included fertilizer offset credit. Credit for Scenario 2 was discounted due to P recovery from sludge liquor.
5. Truck hauling distance for Scenarios 1 and 2 has been assumed at 105 km one way whereas for Scenario 3, 48 km one way was used.

While BEAM included most of the biosolids handling processes, P recovery processes were not included. Therefore, additional GHG emissions were calculated separately in terms of CO₂e. These calculations included CO₂e due to the manufacturing of the chemicals used for the P recovery processes (for Scenarios 2 and 3), electricity

consumed due to the P recovery processes (for Scenarios 2 and 3), and fertilizer (or phosphate rock, PR) offset credit (for Scenario 3). The fertilizer or PR offset credit represents the offset of GHG due to the reduction of P mining for fertilizer, fertilizer manufacturing, and transportation from PR mine (e.g. from Florida) to the study area (i.e., Chicago IL). Using an emission factor of 1.0835 kg of CO₂e per kg of P₂O₅ (Wood and Cowie, 2004) and the potential amount of P that can be recovered from Scenario 3, CO₂e was calculated. A list of the emission factors used to calculate the CO₂e is summarized in **Table 13**.

Table 13. Emission factors summary (in CO₂e)

Chemical	Emission factor	Unit	Reference
Ca(OH) ₂	0.15	kg CO ₂ e /kg	Note 1
H ₂ SO ₄	0.14	kg CO ₂ e /kg	Note 1
NaOH	1.12	kg CO ₂ e /kg	Note 1
Acetic acid	0.61	kg CO ₂ e /kg	Adom and Dunn, 2015
Polymer (LEACHPHOS)	2.2	kg CO ₂ e /kg	Note 1
Electricity	0.6955	mt CO ₂ e /MWh	Note 2
P ₂ O ₅	1.0835	kg CO ₂ e /kg P ₂ O ₅	Wood and Cowie, 2004

Note:

1. Winnipeg Sewage Treatment Program – South End Plant Process Selection Report (2012).
2. City of Chicago Greenhouse Gas Inventory Report (2017).

5.2.3 Establishing the metrics and a common scale for the selected evaluation criteria

A scoring system was established for both quantitative and qualitative criteria (USEPA 2015). Quantitative criteria are the criteria where direct measure is possible such as cost estimates or GHG emissions. For qualitative criteria that are difficult to quantify due to constraints, such as public health, a constructed measurement was formed

which included a range of -3 to +3 scale, with negative values meaning less favorable or more negative impacts and positive values meaning more favorable or more positive impacts when compared among scenarios. A score of zero means the scenario is impartial to the criteria. With the scale of -3 to +3, the quantitative criteria were then normalized to fit into the scale based on the direct measure. To bring all criteria onto a common scale, the following formula was used to normalize the quantitative measurements (USEPA 2015):

$$\left(\frac{\text{Alternative's performance value} - \text{lowest actual raw performance value}}{\text{Highest actual raw performance value} - \text{lowest actual raw performance value}} \right) \times \text{Highest constructed value}$$

where the “lowest actual raw performance value is equal to zero for the scale of -3 to +3. A summary of the common scale and the corresponding interpretation for the non-quantitative criteria is presented in **Table 14**.

Table 14. Common scale and scale interpretation for non-quantitative criteria.

Scale	-3	-1.5	0	1.5	3
Public Health Risks/Benefits	The scenario will adversely impact public health the most	The scenario will adversely impact public health to some extent	The scenario has no impact on public health	The scenario will improve public health to some extent	The scenario will improve public health the most
Sustainable P supply (dependency on PR)	The scenario will not recover P. All mined PR in the future supply.	The scenario will recover > 20% recoverable P. Mined PR \geq Recovered P in the future.	The scenario will recover > 40% recoverable P. Mined PR > Recovered P in the future.	The scenario will recover > 60% recoverable P. Mined PR < Recovered P in the future.	The scenario will recover > 80% recoverable P. Mined PR \leq Recovered P in the future.
Future biosolids regulations	The scenario will subject to new biosolids regulations	The scenario may subject to new regulations	The scenario will not subject to new biosolids regulation	The scenario may deregulate regulation	The scenario will deregulate biosolids regulation
Air quality	The scenario will adversely impact air quality the most	The scenario will adversely impact air quality to some extent	The scenario will not impact existing air quality	The scenario will improve the existing air quality to some extent	The scenario will improve the existing air quality the most
Water quality due to P runoff	The scenario will promote the worst P runoff	The scenario will promote P runoff to a lesser extent	The scenario will not impact existing water quality due to P runoff	The scenario will reduce P runoff to some extent	The scenario will reduce P runoff the most

With the scoring system defined, a weighting system was established and assigned to the categories and sub-categories. The weighting system is intended to differentiate the relative importance among the evaluation criteria and the system can be subjective depending on the priorities of the region or community. For the three categories, the weight was distributed evenly among the main categories: economic, social, and environmental. For the subcategories, the weight was distributed based on a planning document, *On To 2050*, published by the Chicago Metro Agency of Planning (CMAP) which detailed the principles governing the future planning of the Chicagoland areas including the surrounding counties (CMAP, 2018). The relative percent weights of the subcategories were inferred from the guiding principles. For example, CMAP emphasized the importance of “... *development that supports local and regional economic strength (and) collaborating at all levels of government such as consolidation of services for greater efficiency and cost savings.*” As a result, the OpEx was given a higher percent weight than the CapEx since consolidation of services through regionalization would reduce the overall operating costs. The resulting weighting factors were then applied to the base score under each evaluation criterion. The weighting factors are summarized in **Table 15**.

Table 15. Weighting factors for the categories and subcategories of the TBL analysis.

Category	% Weight	Sub-category	% Weight	Weighting Factor, %
Economic	33.33	CapEx	40	13.33
		OpEx	60	20.00
Social	33.33	Public Health Risks/Benefits	40	13.33
		Sustainable P Supply	60	20.00
Environmental	33.33	Future Biosolids Regulation	20	6.67
		Air Quality	20	6.67
		Water Quality	30	10.00
		GHG Emissions	30	10.00
Total	100			100

5.3 Results

Economic Category. Scenario 1 represents a status quo scenario and therefore the capital costs for Scenario 1 would be zero. As anticipated, Scenario 3 which represents a centralized mono-incineration and P recovery would cost the most out of the three Scenarios. This is mainly due to the additional land acquisition costs and the need for completely new infrastructure of the centralized facility. From the O&M perspective, however, Scenario 3 incurred the lowest costs mainly due to a shorter truck hauling distance. As indicated in Figure 9, most farmland for land application is more 105 km away from the metro Chicago area. The final disposal costs which included land application and truck hauling for Scenario 1 contributed to approximately 60% of the annual O&M costs. On the other hand, about 30% of the annual O&M costs for Scenario 3 was due to the truck hauling. No land application fee was anticipated for Scenario 3. Another major cost component of the O&M is energy. For Scenarios 1 and 2, an assumption was made that 75% of biogas generated would be recovered for heating the

anaerobic digesters onsite, 20% of the biogas would be recovered for generating electricity that would be consumed onsite, and 5% of biogas would be flared. For Scenario 3, it has been assumed that the sludge fuel value is sufficient to sustain the incineration after process startup and the heat of the flue gas would be recovered to generate electricity. A summary of the PW analysis is presented in **Table 16**.

Table 16. Present worth for the three scenarios.

	Present Worth, in millions		
	Scenario 1	Scenario 2	Scenario 3
CapEx ¹	\$ -	\$ 270	\$ 760
Annual OpEx	\$ 80	\$ 89	\$ 62 ²
20-year OpEx	\$ 1,444	\$ 1,600	\$ 1,107
20-year Present Worth	\$ 1,444	\$ 1,870	\$ 1,867

Note:

1. According to AACE Class 5, the expected accuracy ranges from the low end of -20% to -50% to a high end of +30% to +100%.
2. Dewatering would be conducted onsite at each WRRF and therefore the OpEx would be borne by WRRFs.

After normalization of the CapEx and the annual OpEx, the base and weighted performance scales of the scenarios are summarized in **Table 17**.

Table 17. Base and weighted performance scale for the economic category.

Performance Scale (Base)	Scenario 1	Scenario 2	Scenario 3
CapEx	0	-1.1	-3
OpEx	-2.7	-3.0	-2.1
Performance Scale (Weighted)			
CapEx	0.00	-14.67	-40.00
OpEx	-54.14	-59.99	-41.76

Social Category. The social criteria are non-quantitative which included public health, as it relates to trace contaminants in biosolids such as pharmaceutical and personal care products (PPCPs), per- and polyfluoroalkyl substances (PFAS), and heavy metals,

and sustainable P supply (or the less dependency from existing PR sources). From the public health perspective, PPCP, PFAS, and heavy metals found in the biosolids remain in the soil through land application. PPCPs such as ibuprofen, triclosan, bisphenol A, are extensively used in humans to improve the quality of daily life. Since they are designed to maximize their biological activities at low dose use in humans, they are difficult to remove in conventional water and wastewater treatment (Ebele et al., 2017). PFAS is widely used in coatings, cookware, and firefighting foams due to its water and oil repellency and the ability to resist high temperature (ITRC, 2018). Similarly, PFAS persists in soil and is bioaccumulative, and cannot be efficiently removed via conventional water and wastewater treatment (Ahrens 2011). Heavy metals in biosolids may release and accumulate in the soil. When biosolids are applied on land where crops are planted, heavy metals end up in the food chain and ultimately cause metabolic disorders and chronic diseases in humans (Pathak et al., 2009). Studies have shown that at a temperature of above 1,000 °C, sludge incineration can remove PPCP and PFAS to below detection limits (Taylor et al. 2014), although other studies presented inconclusive results regarding PFAS removal by heat treatment (Lazcano et al., 2019). Therefore, Scenario 3 was given a score of +1.5 while Scenarios 1 and 2 were given a score of -3.

For the sustainable P supply criterion, the total amount of P that can be recovered in 2017 was estimated at 38,195 kg/day within the region (Law and Pagilla, 2019). For Scenarios 2 and 3, the potential amounts of P that can be recovered were estimated at 9,223 kg per day (or 24%) and 31,896 kg per day (or 84%), respectively. As for Scenario 1, a study conducted by Sullivan et al. (2015) indicated that a 2% P content with 40% plant available P is typical when estimating biosolids P impacts on farming. Based on the

mass balances of 468 mt per day of dry biosolids and assuming a 50% nutrient loss due to runoff, approximately 2,064 kg per day (or 5%) of P can be recovered through land application. Beyond recovery for immediate use, Scenario 3 can also supply for future P demand since the P recovered would be in the form of Ca-P which can be used for diverse applications with minimal needs for beneficiation. While it is difficult to estimate the P demand for the region, P demand for an average adult is about 2,000 milligrams per day (Anderson, 2018). From a dietary demand perspective, the total P demand for the region with a population of 8.4 million (Law and Pagilla, 2019) would require about 6,077 mt per year of P. To put it in perspective, Scenarios 2 would recover 3,053 mt per year in fertilizer form and Scenario 3 would recover 10,559 mt per year of P in Ca-P form that can be used for different applications. For Scenario 1, the amount of P that can be “recovered” only through land application would be 683 mt per year. The future P supply would still be heavily dependent on PR mining as raw materials from other states or countries for Scenarios 1 and 2. With a high recovery rate and the least dependency on PR mining in the future, Scenario 3 was given a +3 scale whereas Scenarios 1 and 2 were given a -3 and -1.5 respectively. A summary of the base and weighted performance scale for the social criteria is presented in **Table 18**.

Table 18. Base and weighted performance scales for the social category.

Performance Scale (Base)	Scenario 1	Scenario 2	Scenario 3
Public Health Risks/Benefits	-3.0	-3.0	+1.5
Sustainable P supply	-3.0	-1.5	+3.0
Performance Scale (Weighted)			
Public health due to PPCPs, PFAS, and heavy metals	-40.00	-40.00	20.00
Sustainable P supply	-59.99	-30.00	59.99

Environmental Category. The biosolids regulations are anticipated to become more stringent in the future for two main reasons. First, biosolids land application rates may change from N-based, as in current practice in Illinois, to P-based limits. The current N-based application rate is because N is the limiting factor for plant growth. With most crops require an 8:1 ratio of N:P and a typical biosolids N:P ratio is 2:1 (Sullivan 2015), N-based application rate increases P application by four times. Excess P runoff due to storm events exacerbates eutrophication. This change in application rate would increase the costs of land application of biosolids. Second, public concern over heavy metals, PPCPs, and PFAS in biosolids used for farming has been on the rise. These chemicals can be found in treated effluent as well as biosolids. Adverse health effects due to the persistence of these chemicals is causing a rising concern (USEPA 2019). In Europe, land application has been restricted due to contaminants in biosolids (Scholz and Wellmer, 2018). For Scenario 3, where final disposal of biosolids is through mono-incineration, there is no concern over future biosolids regulations. Therefore, Scenario 3 was given a value of zero while the other two scenarios were given a -1.5 on the performance scale under this criterion.

Air quality impacts, neither positive nor negative, are not anticipated for Scenarios 1 and 2. Both scenarios were given a performance scale of zero. For Scenario 3, the flue gas generated from the mono-incineration process will have to be treated prior to discharge. Mono-incineration is regulated under the Clean Air Act for heavy metals, particulate, gases (such as carbon monoxide, nitrogen oxides, and sulfur oxides), and visible emissions (USPEA 2003). In 2011, Maximum Achievable Control Technology (MACT) standards were adopted by the USEPA which included stricter limits on

emissions for new sewage sludge incineration unit construction. Even though air pollution control equipment must be implemented, the air quality may be subject to deterioration. The air emissions may also trigger operating permits review under Title V of the Clean Air Act. Therefore, Scenario 3 was given a -1.5 on the performance scale.

As discussed above, excess P runoff from farm fields causes eutrophication in the receiving waters. For the water quality criterion, it affects Scenarios 1 and 2 due to land application disposal scheme and it does not affect Scenario 3. For Scenario 3, a monofill would be required to store the ash residue. A typical monofill consists of a leachate collection system and an impermeable geomembrane and compacted clay to prevent leachate from leaking into the environment. Although the monofill is designed for leachate containment, there is still a potential of membrane damage and leakage. Scenario 3 was given a value of +1.5 for its water quality improvements by minimizing the potential of excess P runoff through land application. Scenario 2 included P recovery from the sludge liquor which reduces the P load recycle back to the WRRFs. The extent of the excess P runoff is therefore less severe than the excess P runoff from Scenario 1 which is essentially the “do-nothing” approach. Scenario 1 was given a -3 while Scenario 2 was given a -1.5.

GHG emissions were estimated using BEAM and emission factors for P recovery processes. A summary of the GHG emissions is shown in **Table 19**. Scenario 3 has the highest GHG emission potential. Despite a self-sustained incineration process, energy required to recover P from the sludge ashes and the associated chemicals increased the CO_{2e} emissions. However, fertilizer independency credits were taken and the CO_{2e} impacts were reduced by approximately 35%. The credit was calculated based on the

total P that can be recovered from sludge ashes, converted into P₂O₅ as fertilizer and multiplied by P₂O₅ emission factor as indicated in **Table 13**. Similar credits were calculated for Scenarios 1 and 2 based on their P recovery potential. In addition, Scenarios 1 and 2 included biogas utilization credit assuming 75% of the biogas would be recovered for use in digester heating, 20% would be used to generate electricity, and 5% would be flared as excess biogas. Similar to Scenario 3, Scenario 2 has a higher CO_{2e} potential due to the chemicals and electricity requirements for the P recovery process. Using the normalization approach, the performance scale for Scenario 1, 2, and 3 are -0.9, -1.7, and -3.0 respectively. A summary of the base and weighted performance scales for the environmental criteria is presented in **Table 20**.

Table 19. GHG emissions for Scenarios 1, 2, and 3.

GHG Emissions	Scenario 1	Scenario 2	Scenario 3
Biosolids management (from BEAM)	11,181	16,078	21,538
Chemicals manufacturing (for P recovery)	-	9,790	27,314
P recovery (electricity)	-	1,182	12,263
Fertilizer ("PR independence") credit	(1,906)	(8,366)	(28,947)
Total, mtCO _{2e} /year	9,275	18,683	32,168

Table 20. Performance scales (base and weighted) for the environmental category.

Performance Scale (Base)	Scenario 1	Scenario 2	Scenario 3
Future biosolids regulations	-1.5	-1.5	0
Air quality impacts	0	0	-1.5
Water quality impacts	-3	-1.5	+1.5
GHG emissions	-0.9	-1.7	-3.0
Performance Scale (Weighted)			
Future biosolids regulations	-10.00	-10.00	0.00
Air quality impacts	0.00	0.00	-10.00
Water quality impacts	-30.00	-15.00	15.00
GHG emissions	-9.00	-17.00	-30.00

The overall TBL results are presented in **Figure 13**. The solids bars represent the total weighted performance scales of the scenarios. Bars with positive or smaller negative size reflect desirable options. Despite a high capital cost and high GHG emissions, Scenario 3 has the least negative scale among the three scenarios. This is mainly due to a sustainable P supply in the future as well as a safer final biosolids disposal option compared with land application. The annual OpEx is also more attractive than the other scenarios because of a shorter distance of truck hauling and an improved efficiency by centralizing the operation.

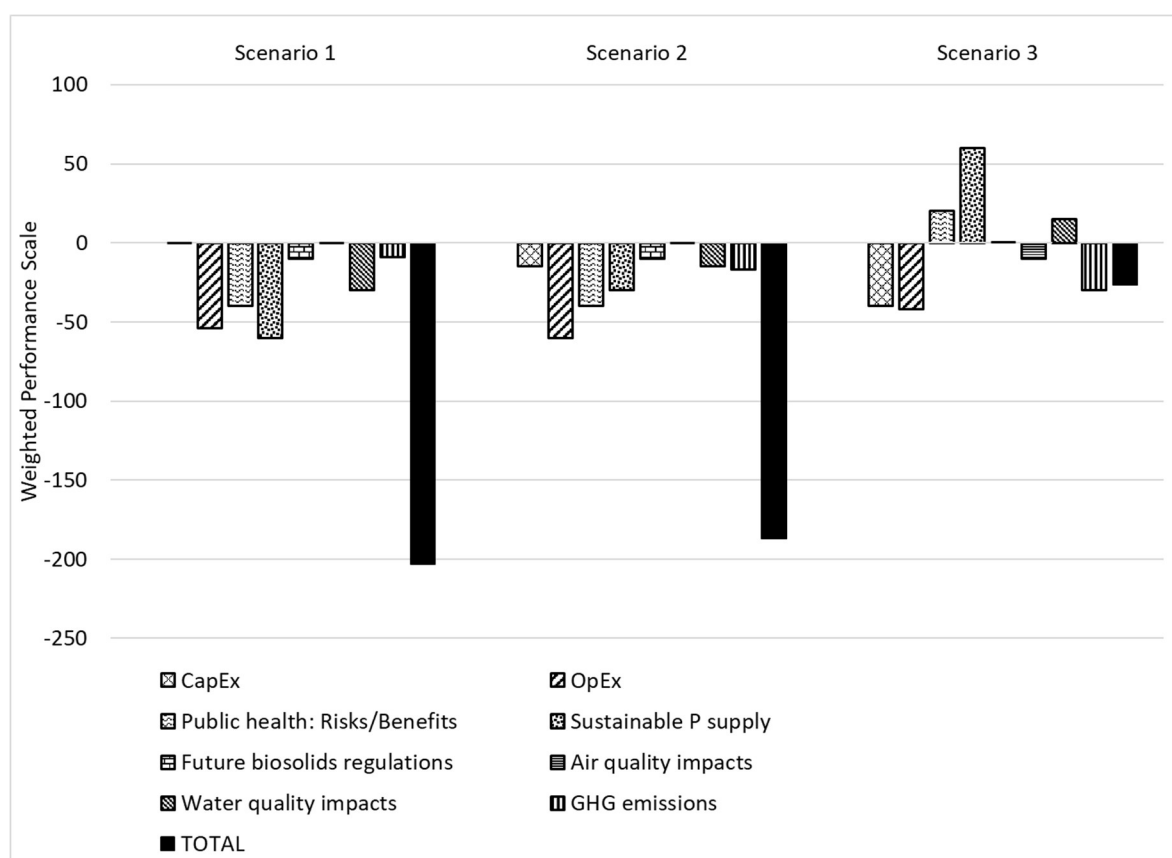


Figure 13. Weighted performance scale for the TBL analysis for all Scenarios.

5.3.1 Uncertainty analysis

Results of the TBL analysis indicated that Scenario 3 is the best approach to secure a sustainable, regional P supply while protecting public health. When quantifying some of the evaluation criteria such as costs and GHG emissions, assumptions were made which may potentially skew the final performance scales. Therefore, an uncertainty analysis was performed to identify the criteria that are most sensitive and susceptible to change in the final outcome of the TBL analysis. The uncertainty analysis was conducted by changing the performance scale of each criterion one at a time using the maximum scale of “+3” and the minimum scale of “-3” for each scenario. For example, to evaluate the impacts of CapEx of Scenario 1, a maximum of “+3” and a minimum of “-3” were assigned to the matrix (instead of the original scale of “0” from the analysis) without changing the rest of the performance scales. The total performance scales were recorded and weighting factors were applied. Similarly, to evaluate the impacts of OpEx of Scenario 1, a maximum of “+3” and a minimum of “-3” were assigned to the matrix (instead of the original scale of “-2.7”) without changing the rest of the performance scales. This procedure was repeated for all evaluation criteria of each scenario using the maximum “+3” and minimum “-3” scales. The total performance scale of each scenario with the maximum and minimum adjustments are presented in **Figure 14**.

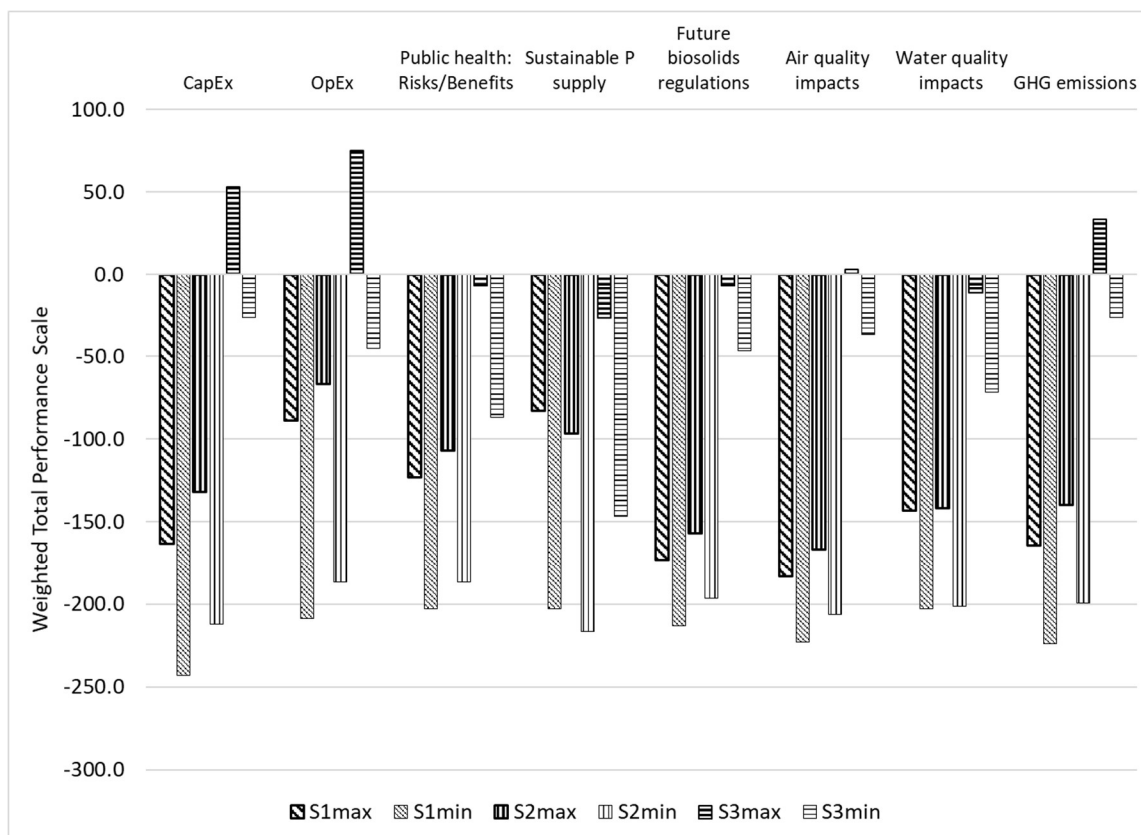


Figure 14. Uncertainty analysis of TBL results.

The diagonal, vertical, and horizontal pattern schemes represent the maximum and minimum performance scales for Scenarios 1, 2, and 3 respectively. Bars with thicker patterns represent the total performance scale of the scenario when the respective criterion was assigned with a “+3”, a maximum scale that the criterion can be assigned. Bars with thinner patterns, on the other hand, represent the total performance scale of the scenario when the respective criterion was assigned with a “-3”, a minimum scale that the criterion can be assigned. As shown in **Figure 14**, the total weighted performance scales for Scenario 3 (horizontal pattern bars) are consistently the highest (the least negative) regardless of the maximum or minimum scale, except for one criterion when the total performance scales for Scenarios 1 and 2 when assigning a +3 (thicker diagonal and thicker vertical bars) would result in more favorable scales than Scenario 3 when

assigning a -3 (thinner horizontal bar). This criterion is sustainable P supply.

Considering this criterion, it is clear that Scenario 3 will be the most favorable option than Scenarios 1 and 2 because Scenario 3 will always recover the highest amount of P than the other scenarios. Therefore, it is impossible to justify that Scenario 1 or 2 would have a higher performance scale than Scenario 3 under the sustainable P supply criterion. The implication of the uncertainty analysis is that even when the maximum scale was assigned to Scenarios 1 and 2 and the minimum scale to Scenario 3, Scenario 3 would still be the most favorable scenario for a sustainable P supply without jeopardizing the public health.

5.4 Discussion

Despite sustainable justifications for a regionalized P reserve through P recovery from wastewater, challenges and obstacles remain. As shown in the 20-year present worth, the regional facility which included mono-incineration, P recovery, and P reserve was estimated at US \$760 million. However, on a unit cost basis (\$ per kg P recovered) and for the same amount of P that can be recovered from the three scenarios as discussed in previous section, Scenario 3 costs the least. The 20-year present worth unit costs for Scenarios 1, 2, and 3 are approximately \$98 per kg P recovered, \$28 per kg P recovered, and \$8 per kg P recovered, respectively. When regionalized or centralized operation is considered, the operating costs will be lower than individual operations due to the economy of scale. As an example, the Sludge Treatment Facility in Hong Kong, a city with a population of 7.4 million, is the largest mono-incineration facility in the world and it was designed to treat 2,000 mt per day of sludge generated from all of the eleven WRRFs throughout the city. The construction was completed in 2015 and it cost

approximately HK\$ 5.5 billion (US\$ 702 million) (Swann, 2017). The facility is self-sufficient by converting all waste products into resources: heat recovered from flue gas is converted into electricity to be used onsite for a complete off-the-grid operation with the potential to export surplus power to local grid and zero effluent discharge of wastewater by reclaiming for onsite use. Currently, the sludge ash is transferred to an onsite ash lagoon for disposal. It is understood that the facility does not recover P, but it sets an example that mega projects of this magnitude can be accomplished when the future of landfilling unincinerated biosolids in Hong Kong is not practical or feasible. Another centralized, waste-to-energy facility serving a population of 5.5 million, combining both waste and wastewater treatment including sludge incineration is currently underway in Singapore (State of Green, 2019). If P recovery from the sludge ash and the reserve were included with centralized incineration facilities around the world, P supply would be far less dependent on a handful of countries and current supply chain routes. For this work, the P recovery process and the monofill for storing Ca-P as reserve was estimated at approximately 13% of the total CapEx. With the incineration facility contributing to most of the costs, the additional costs seem to be a small price to pay for a more stable P supply in the future. To further reduce costs, repurposing existing quarries to house the centralized facilities could be considered since stones and rocks would have been extracted for construction materials. There are many examples of repurposing abandoned quarries into wetlands or even recreational water parks (Tan et al., 2019). In addition to reduced costs, repurposing a quarry would add yet another sustainable element to the work.

In addition to costs, the negative perception of incineration will need to be overcome. The general public still associates incineration with air pollution and lung diseases regardless of the advancement of the combustion and emissions control technologies available. As described above, the incineration facilities in Hong Kong and Singapore are prime examples of the urgency that the cities are facing with the ever-increasing amount of biosolids that must be disposed of. In the U.S., about 55% of all biosolids generated are land applied (Lu et al., 2012) with only 15% are incinerated (NEBRA 2007), and the majority of sludge incineration facilities are located in the Northeast where land is limited and cities are more densely populated (USEPA 1995). It is apparent that only when sludge incineration is the last remaining disposal solution due to the lack of land, the public will be more open to the combustion process. In addition to limited land availability, regulations on land application are becoming stricter due to the public health concerns which results in more incineration application in Europe. In parts of Austria and Switzerland, land application has been banned (Mattenberger et al., 2008) due to heavy metals found in biosolids. A public outreach campaign on incineration is needed to promote the benefits of the process and how the products of incineration, ashes and flue gas, are valuable resources waiting to be recovered rather than wasted. In addition to P recovery, sludge ash has been used in cement manufacturing (Cyr et al., 2007). Incineration and proper flue gas control can remove all the organics and pathogens in the biosolids, as well as some heavy metals (USEPA 2003). More studies have been conducted on PFAS removal by high-temperature incineration and results are promising (ITRC, 2018). Heat and electricity recovery from the flue gas can sustain the energy demand of the facility, as demonstrated in the facility in Hong Kong. As for the

air emissions, combustion technology such as the fluidized bed incinerator (FBI) is proven to produce lower pollutant emissions than other technologies (Oladejo et al., 2018). Based on air emissions data published in the *Metro Plant Solids Management Improvements Facility Plan (2018)*, the Metro Plant in Minneapolis which has been operating FBIs for decades are consistently emitting less than the maximum allowable control technology (MACT) limits set forth in 2011 by the USEPA. The sustainability and safety benefits of incineration will need to be promoted to reverse public perception.

Since the 1970s, biosolids disposal via land application has been on the rise since biosolids are a good source for plant nutrients (Pepper and Gerba, 2006). Typically, WRRFs pay for haulers to truck their biosolids to farmlands to be land applied. Therefore, another obstacle of promoting alternative biosolids disposal approach could come from the agricultural communities where they receive biosolids as fertilizers for free. It is difficult to estimate the additional fertilizer expenses for the farmers since typically the application rate is based on the nitrogen content of the biosolids. With the N-based application rates, excess P is applied which can accumulate in the soil over time. Available P as a nutrient depends on soil pH, frequency of biosolids application, method of incorporation, P saturation in soil, and P soil storage capacity (Sullivan et al., 2015, Lu et al., 2012). A study by Sullivan et al. (2015) indicated an approximate first-year fertilizer replacement value of anaerobically digested sludge for P is approximately \$24.0 per mt, assuming 2% P content and 40% available nutrient. For this work, the fertilizer replacement value based on 468 mt per day from the mass balance would be \$11,249 per day, or \$4.1 million per year. While chemical fertilizers would be needed to replace the

biosolids, at the same time, the amount of fertilizer applied would be tailored to what is actually needed and excess P runoff could be minimized.

In a recent survey aimed to identify barriers and opportunities for closing the P loop in the Baltic Sea region, Barquet et al. (2020) found that the main obstacles in preventing P recovery and reuse from WRRFs are the lack of policy steering from the government towards nutrient reuse and performance information of P recovery technologies. In addition, the survey found that the current focus of a few high cost P recovery technologies may slow down the progress to achieve a circular economy of P. The study re-emphasized the need to think beyond fertilizers as the only means to recover P.

More work and pilot studies will be needed to better understand the LeachPhos P recovery technology so that it can be scaled up economically. A demonstration facility treating 15,000 tonnes per year of sludge ash using LeachPhos was conducted successfully in Switzerland in 2013 with a high P recovery rate of 80 to 90% (Herzel et al., 2015). In general, the process involves P leaching with sulfuric acid, followed by solids/liquid separation where the P containing filtrate is then precipitated with lime to form Ca-P. A secondary leaching can be used for the filtered cake in order to reduce the heavy metal content. The remaining inert material can be recycled to be used in manufacturing cement (Herzel et al., 2015). However, with the secondary leaching, the heavy metal containing liquid will need to be treated for potential acid reuse. Although the demonstration facility was a success, it is unclear how well the process can be scaled up to handle 166,000 tonnes per year of sludge ash as it is in this work.

5.6 Conclusion

A regional P recovery study conducted by Law and Pagilla (2019) using the Chicagoland area of six counties concluded that P recovery from sludge liquor and sludge ash is more cost-effective than P recovery from biosolids. As a continuation of the 2019 study, this research extended the evaluation by looking beyond economic impacts. A TBL analysis was conducted for three scenarios: a status quo scenario of recovering P through biosolids land application; P recovery from sludge liquor at each WRRF with biosolids land application; and a centralized facility with mono-incineration, P recovery from sludge ash into Ca-P, and storing Ca-P as reserve. Despite a high capital expenditure, P recovery from sludge ash with mono-incineration is the most sustainable scenario to secure future P supply while protecting the public and environmental health. The centralized P recovery and P reserve concept can be applied in other metropolitan regions where land is scarce and densely populated. As regulations become more stringent, incineration is an economical approach to dispose of biosolids. Incineration eliminates pathogens, reduces the amount of truck traffic, and offers resource recovery potential such as heat for electricity, sludge ash for cement and more importantly, sludge ash into Ca-P as a reserve for future use.

CHAPTER 6

DISCUSSION AND CONCLUSIONS

6.1 Discussion

The current focus of P recovery in the form of struvite fertilizer is not economically viable for all WRRFs of all sizes due to high costs. A simple payback analysis indicated that for a typical 95 ML per day (25 MGD) WRRF with EBPR, the payback of the capital investment would be between 12 to 18 years, taking into account the fertilizer revenue and operational savings. Without the economy of scale, small WRRFs are not able to justify a long payback to recover P. To promote P recovery from all WRRFs of all sizes, the costs will need to be sufficiently low. This work systematically evaluated sources within WRRFs from which P can be recovered in a form that would be the most cost-effective. With the currently low prices of PR and chemical fertilizer, the drivers for P recovery as fertilizer are challenging to justify, and therefore it benefits most when P is recovered in Ca-P forms that can be deposited as a reserve and can be used in the future for diverse applications beyond fertilizers.

First, an analysis is presented of the magnitude of costs to recover P as fertilizer from three sources in WRRFs. Sludge liquor, biosolids, and sludge ash P recovery costs were estimated for all 80 major WRRFs in the Chicagoland area. Based on a 20-year present worth analysis, it would cost about US\$ 21 billion to recover P to the extent possible from sludge liquor and biosolids. To recover a similar amount of P from sludge ashes, it would cost about US\$ 2.1 billion. Results indicate that P recovery from biosolids alone would cost US\$ 89 to US\$ 92 per kg of P recovered, while P recovery from sludge liquor and sludge ash would cost US\$ 7.1 to US\$ 14 and US\$ 8.6 per kg of P recovered

respectively. The magnitude of present worth unit costs to recover P from sludge liquor, and sludge ash is comparable, even though the amount of P that can be recovered from sludge ash was estimated at 31.9 metric tons per day, three times higher than the amount of P that can be recovered from sludge liquor. Results also indicate that the cost to recover P from biosolids is cost-prohibitive due to the high energy demand of the currently available technologies.

Second, the evaluation was taken one step further by comparing P recovery from sludge liquor and sludge ash on a TBL basis, which took into consideration economic, social, and environmental factors. The analysis also included a status quo scenario where P is recovered through a “passive” approach of land application, and biosolids are used as a soil supplement or fertilizer for plant growth. While the first evaluation was confined to within the boundaries of WRRFs, the second evaluation was extended to include the final disposal sites. The scenario that represented P recovery from sludge liquor included digestion, biosolids dewatering, and P recovery to fertilizer within each major WRRF utilized the same data set that was used in the first evaluation. Biosolids are trucked offsite for land application for final disposal. The scenario that represented P recovery from sludge ashes included a centralized mono-incineration facility where all dewatered but undigested sludge is hauled to the centralized facility for mono-incineration. The sludge ash generated is then used to recover P in Ca-P forms and finally “disposed of” as P reserve in a monofill.

The TBL analysis results indicated that the centralized P recovery facility using sludge ashes has the highest overall performance scale despite the highest capital cost and the highest GHG emissions. The scenario eliminated public health concern over the land

application of biosolids with pathogens, heavy metals, PPCP, and PFAS. Concerns about these chemicals will likely prompt more stringent biosolids regulation shortly which may potentially ban land application altogether. For example, in Europe, Switzerland has already banned agricultural biosolids use in 2006. In 2017, Germany promulgated a sewage sludge disposal plan with details on prohibiting land application by 2032 for facilities that have capacities for more than 50,000 population equivalence and which contains a mandatory P recovery requirement. It is only a matter of time when the U.S. will follow suit.

In addition to eliminating a potential public health concern, the scenario offers a shorter sludge hauling distance, which translates to lower transportation costs and consequently reduces GHG emissions. Should all sludge be incinerated, excess P runoff due to land application of biosolids will no longer be a cause for eutrophication. Also, after the P is extracted (leached) from the sludge ash, the remaining ash can be further processed to be used for cement manufacturing. However, most importantly, it provides a sustainable future P supply. With more than 70% of the world's PR reserve located in Morocco and West Sahara, building our P reserve can reduce our dependence on PR supply from other countries. In this work, an existing quarry is proposed to be repurposed as a monofill for the P reserve. This reduces the capital investment on excavation for monofill construction.

Should biosolids no longer be land applied, farmers will need to purchase chemical fertilizer for the loss of nutrients that would have been provided by the biosolids that are "paid for" by the WRRFs. However, the impacts of the varying soil characteristics on the actual N and P availability and uptake by crops are difficult to

estimate. From the water quality perspective, using chemical fertilizer may reduce the chances of excess P runoff since the current N-based application makes P typically more than the requirements that are needed for plant growth.

The centralized mono-incineration facility would eliminate the requirement of sludge stabilization at each WRRF. While the operating costs would be reduced, one negative impact is that biogas would no longer be generated. Currently most WRRFs use the biogas as fuels for boilers, which generate heat for anaerobic digester heating. The biogas generated typically can be utilized entirely during winter. In summer when the heat demand is low, excess biogas is flared. While many WRRFs are considering the use of combined heat and power (CHP) engine for electricity generation, the energy costs are low enough, particularly in the Midwest, to justify not performing CHP implementation. Nonetheless, biogas as a resource would no longer be available should a centralized incineration facility be in place. On the other hand, steam could be generated from heat recovery of the flue gas to power a turbine for electricity, or it could be sold to local communities or industries for potential revenue, turning the centralized facility into a real “waste-to-energy” facility.

6.2 Conclusions

Phosphorus is a limited resource with no substitution for it. The current focus of P recovery as struvite fertilizer for immediate use is not economically viable for WRRFs of all sizes. Given the fact that the current prices of PR and commercial fertilizer are sufficiently low, it is challenging to justify P recovery as fertilizer. Using the Chicagoland area population of 8.4 million, this work systemically demonstrated that in order to recover P cost-effectively for WRRFs of all sizes, the novel concept of a

centralized facility where mono-incineration, P recovery from sludge ash into Ca-P, and a P reserve are located is a promising approach for a sustainable P supply in the future.

With the facility centrally located within the region, WRRFs would haul dewatered but undigested sludge to the facility for further processing. The regionalization model of P recovery can be applied to other metropolitan areas where land is scarce, and population is dense. This model can provide easily accessible, stable, and sustainable P supply in the future with less dependency on imported PR from foreign countries.

6.3 Future work

Although the centralized P recovery concept is new, centralized incineration facilities have been implemented throughout the world in places such as Hong Kong and Singapore. Due to the high sludge fuel value, the combustion process can be self-sustained with the provision of energy recovery from the flue gas and the potential for exporting excess power. On the other hand, additional work will be needed to evaluate the P recovery technology. In this work, LeachPhos was used as the P recovery technology from sludge ash. LeachPhos is yet to be implemented in full scale, although a demonstration facility indicated promising results. The demonstration facility was constructed in Switzerland in 2013 and was designed to treat 15,000 mt per year of sludge ash. A high P recovery rate of 80 to 90% was achieved. The process includes P leaching with sulfuric acid, followed by solid-liquid separation in which the P-containing filtrate is precipitated with lime to form Ca-P. Secondary leaching can be used for the filtered cake in order to reduce the heavy metal content. The remaining inert material can be recycled to be used as a component in cement (Herzel et al., 2015). However, with the secondary leaching, the heavy metal-containing liquid will need to be treated for potential

acid reuse. Although the demonstration facility was a success, it is unclear how well the technology can be scaled up to handle 166,000 mt per year of sludge ash as it is in this work. More demonstrations will be needed to better understand and improve the process so that it can be scaled up and implemented economically. Alternatively, other sludge ash P recovery technologies can be explored for operational complexity and cost-effectiveness. If it is determined that none of the sludge ash P recovery technologies are ready to be scaled up, the sludge ash can still be deposited in a monofill, and it will be readily accessible when the P recovery technology is matured for full-scale implementation. Future work and research needs include but are not limited to 1) Technologies evaluation for P recovery from sludge ashes, which may include bench-scale studies for performance comparison and scalability. As shown in Table 2, there are other P recovery technologies available, which will require further investigation of the most cost-effective technology. 2) Determine the final form of Ca-P from which it would store as reserve in a monofill. 3) Identify location, funding mechanisms, and governance for the construction and operation of the centralized facility. 4) Public outreach and education to promote the benefits of incineration. 5) Stakeholders engagement, including regulatory agency, WRRFs, agricultural communities, and non-governmental agencies.

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