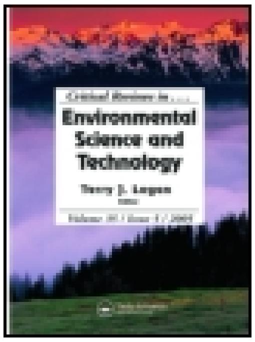

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Human urine-based fertilizers: A review

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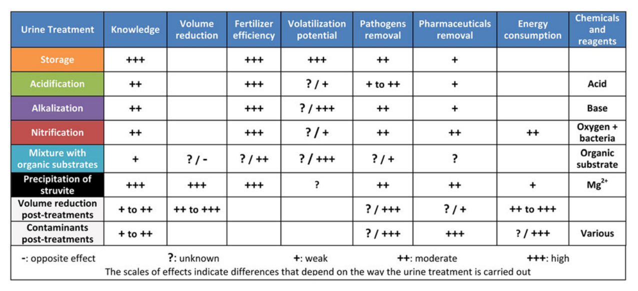
ABSTRACT



Urine Treatment	Knowledge	Volume reduction	Fertilizer efficiency	Volatilization potential	Pathogens removal	Pharmaceuticals removal	Energy consumption	Chemicals and reagents
Storage	+++		+++	+++	++	+		
Acidification	++		+++	? / +	+ to ++	+		Acid
Alkalization	++		+++	? / +++	++	+		Base
Nitrification	++		+++	? / +	++	++	++	Oxygen + bacteria
Mixture with organic substrates	+	? / -	? / ++	? / +++	? / +	?		Organic substrate
Precipitation of struvite	+++	+++	+++	?	++	++	+	Mg^{2+}
Volume reduction post-treatments	+ to ++	++ to +++			? / +++	? / +	++ to +++	
Contaminants post-treatments	+ to ++				? / +++	+++	? / +++	Various

-: opposite effect ?: unknown +: weak ++: moderate +++: high
The scales of effects indicate differences that depend on the way the urine treatment is carried out

Urine contains most of the nutrients excreted by humans. They are mainly released into the environment and contribute to the strong disruption of planetary biogeochemical cycles. These nutrients could be separated at source and used in agriculture. Once the urine is separated different treatments can be applied to stabilize nitrogen, reduce the volume, treat contaminants, or extract nutrients. These treatments and the resulting fertilizers have different characteristics and environmental impacts that must be assessed. We reviewed the characteristics of six treatments and the resulting fertilizers. The studied treatments were restricted to those that retain the most nutrients: storage, acidification, alkalinization, nitrification, and mixture with organic substrate. Phosphorus precipitation was also included, as it is one of the most developed treatments. Volume reduction posttreatments were also considered. We compared these treatments and the resulting fertilizers in terms of nutrient forms and concentrations, fertilizer efficiency, contaminant removal, energy, and chemical consumption. Although some parameters, such as fertilizer efficiency or pharmaceutical removal were in the same range, the nutrient concentrations and the energy and chemical consumption requested by fertilizer production varied widely. Holistic studies and pilot facilities implemented at a larger scale are needed in order to assess the whole value chain.

KEYWORDS Contaminants; human urine; nutrients; urine-based fertilizer; urine recycling; urine treatment

1. Introduction

1.1. Disconnection between sanitation and agriculture

Today, in high-income countries, most wastewater is collected, but treatments vary in their efficiency, leading to various environmental impacts

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such as eutrophication (Sutton et al., 2011). In many cases, the denitrification of nitrogen (N) is considered as the best sanitation practice. However, it is an energy-intensive treatment (Maurer et al., 2003) that hinders nitrogen recovery for agricultural use and emits greenhouse gases as nitrous oxides (Bollon et al., 2016). Phosphorus (P) can be recovered from sewage sludge and thus be recycled in agriculture, whereas potassium is usually not treated and only little recovered. The nutrients present in wastewater have been largely considered pollutants for water bodies since the middle of the 20th century (Barles, 2005) and are only partially recycled in agriculture. Less than 10% of phosphorus present in wastewater is currently recycled worldwide (Cordell et al., 2009), and the figures are much lower for nitrogen, e.g., for a high-income country, less than 5% in Paris (Esculier et al., 2019).

Since the 20th century, an increasing amount of the nitrogen used in agriculture comes from atmospheric nitrogen fixed by the Haber-Bosch process; this process introduces twice as much nitrogen as biological fixation today (Sutton et al., 2013) and represents 70% of the nitrogen used in agriculture. However, this reaction requires a substantial amount of energy and represents approximately 1%–2% of the current total world energy consumption (Matassa et al., 2015). Moreover, phosphorus and potassium fertilizers are fossil resources extracted from mines which are geographically unequally distributed (e.g., about three quarters of phosphate rock reserves are located in Morocco and Western Sahara, U.S. Geological Survey, 2019). This implies risks for food security and food resilience of countries. Some authors highlight that a peak production may be reached in the coming decades, especially for phosphorus (Cordell et al., 2009). Such situation may imply a higher fertilizer price and a higher trace metals content in phosphorus fertilizers due to lower quality phosphate rock (Cordell & White, 2011). The use of Haber-Bosch-produced nitrogen and mined phosphorus for crop production together with the poor recycling of nutrients from wastewater at the end of the food chain are the main disruptors of these open biogeochemical cycles and are currently transgressing the safe operating space of planetary boundaries (Steffen et al., 2015).

1.2. Human urine, the missing link

1.2.1. An essential part of wastewater

Urine is responsible for most of the nutrients present in wastewater, with 79% of the nitrogen, 47% of the phosphorus (even higher when phosphorus detergents are banned) and 71% of the potassium (Friedler et al., 2013). These nutrients from urine are concentrated in a small volume (approximately 0.5 m^3 of urine per year per person) in comparison to the whole volume of wastewater (approximately 55 m^3 of wastewater per year per person in high-income countries). Urine is about 100 times more concentrated than domestic wastewater (about 9 g N/L in urine versus less than 100 mg N/L in wastewater, calculated from Friedler et al., 2013). The nutrients present in urine represent a huge amount of the total nutrient supply worldwide. Recycling all urine-excreted nutrients would represent about 19% of the current worldwide inorganic nitrogen inputs (recalculated from Trimmer et al., 2019, assuming 88% of human-excreted N nutrients in urine) and about 11% of the current global phosphorus demand (assuming 50% P of human-excreted nutrients in urine, Mihelcic et al., 2011). For example, in the Paris region (12 million inhabitants and 569,000 ha of cultivated lands, including 82% field crops), the nutrients present in the urine of the inhabitants of the metropolis could cover all nitrogen and half of the phosphorus currently spread in the fields (calculated from Esculier et al., 2019 and Union des industries de la fertilisation [UNIFA], 2018).

1.2.2. Fresh urine

Composition. Approximately 1.3 L (1.0–1.5 L) of urine is excreted daily by the human body (Friedler et al., 2013). About 85% of the nitrogen is excreted as urea, 5% as ammoniacal nitrogen and 10% as other organic N, which is mainly composed of creatinine, uric acid and creatine (Rose et al., 2015; Udert et al., 2006). Phosphorus is mainly present in a dissolved form as phosphate (Udert et al., 2006). The pH of fresh urine is slightly acidic, approximately 6.2 (Udert et al., 2006). If the urine is not diluted, its daily mean concentration in high-income countries usually ranges between 8.8 and 9.2 g N/L for nitrogen, 0.74 and 2 g P/L for phosphorus, and 2.2 and 2.7 g K/L for potassium (Maurer et al., 2006; Udert et al., 2006). The load of nutrients is correlated with the diet (Trimmer et al., 2019). Urine also contains many micronutrients that are needed for plant growth (Etter et al., 2015).

For organic matter, the chemical oxygen demand (COD) of urine is about 10 g O_2/L and 85% of the COD is easily degradable (Udert et al., 2006). Fresh urine contains more than 3,000 different compounds (urine metabolome, Bouatra et al., 2013), actually few studied.

Contaminants. Urine is not sterile at excretion for healthy adults and it can includes approximately 240 bacterial operational taxonomic units from diverse orders (Lahr et al., 2016). Some pathogens, such as *Schistosoma haematobium*, *Salmonella typhi*, *Salmonella paratyphi*, *Leptospira interrogans* can be naturally excreted in urine (Feachem, 1983). However, contamination of urine in pathogens mostly comes from fecal cross-contamination, which can occur at the diverting toilet (Höglund et al., 1998). Usually, urine concentrations of trace metal elements are low and lower compared to manure and mineral fertilizers (especially phosphate fertilizer, Ronteltap et al., 2007). Among the different contaminants found in urine, pharmaceutical residues seem to be of greatest concern because of their ecotoxicological potentials and relatively high concentration compared to other compounds. Lienert, Bürki, et al. (2007) have done a screening of the excretion pathways of 212 pharmaceutical active ingredients. On average, 64% of active ingredients were excreted in urine, of which 42% were in metabolized forms, generally less toxic (Lienert, Güdel, et al., 2007). However, these proportions vary extremely depending on the compound. Pharmaceutical residues are present in "average urine" at concentrations from zero to several hundred micrograms per liter (Winker et al., 2008) and even reach concentrations higher than a few mg/L for individuals under medication (Bischel et al., 2015). The need for specific treatments to remove pharmaceuticals before the application of urine to land is currently a topic of debate (World Health Organization [WHO], 2012; Winker, 2009).

1.2.3. Source separation and treatment

Source separation is needed to separate urine from other types of wastewater and to avoid the dilution of nutrients or contamination with pathogens (Johansson et al., 2000). Source separation can be performed in various types of toilets and urinals. The most common are waterless male urinals and urine-diverting toilets (Rossi et al., 2009). Some dilution can occur in diverting toilets equipped with a flush (Maurer et al., 2006). After separating the urine, treatments are possible to create different fertilizers that could be used in agriculture. The name "urine-based fertilizer" (UBF) identifies these fertilizers as resulting from urine treatments. Dozens of these treatments were developed since the 2000s, with different objectives as presented in Maurer et al. (2006):

Nitrogen stabilization. Most nitrogen is excreted in urine as urea but is rapidly transformed during urine storage into ammoniacal nitrogen due to the ubiquitous presence of urease enzyme secreted by microorganisms (Friedler et al., 2013). Ammoniacal nitrogen is very sensitive to volatilization due to the high pH in hydrolyzed urine which is close to the pKa of the NH_4^+/NH_3 couple (9.2). Ammonia is responsible for the nitrogen losses after application and for a part of the malodor of urine (Rodhe et al., 2004; Hashemi & Han, 2017). Stabilization corresponds to processes that keep nitrogen into a form that limits losses during treatment and allows further volume reduction or limits losses during the other steps of the value chain such as the spreading on crops. Nitrogen can be stabilized for example, as urea, nitrates, ammonium or organic nitrogen.

Volume reduction. The water content of urine represents approximately 95% of the weight of fresh urine (Rose et al., 2015). Urine is a low-concentration fertilizer compared to typical mineral fertilizers. Volume reduction is useful for storage, transport and application in the field.

Nutrient extraction. Nutrient extraction may be used to separate nutrients from the influent and produce a concentrated fertilizer. Generally, these treatments aim to recover one specific or a few nutrients from the influent.

Pharmaceutical residues and/or pathogens treatment. The possible pharmaceutical residues or pathogens contamination in urine can be reduced through adequate treatments such as filtration or sorption processes.

This review focused on the first two objectives. Nutrient extraction and treatments for pharmaceutical residues are briefly presented in SM 1 and SM 2.

1.3. Scope of the study

The objectives of this study were to review existing treatments for urine and to analyze the properties of the resulting UBFs in terms of nutrient content and fertilizing efficiency as well as their contaminant content and other environmental impacts. As we choose to focus on using urine as a fertilizer in agriculture, we excluded treatments related to other types of uses such as electricity generation or water reuse from urine.

Various reviews were carried out on treatments allowing the use of urine nutrients as fertilizer (e.g., Chipako & Randall, 2020; Harder et al., 2019; Maurer et al., 2006; Udert et al., 2016). The recent review by Harder et al. (2019) presented a comprehensive review on all possible ways to recover nutrients. We decided to focus on treatments that produce fertilizers specific to urine and detailed them as it was done by Maurer et al., (2006) with updated data from recent researches. We wanted to put forward technical and practical information on the treatments and the resulting fertilizers including their effectiveness as fertilizers.

Most of the treatments were developed in high-income countries. Although they can be applied in developing countries (e.g., Pronk & Koné, 2009), this review focused on the context of high-income countries. We highlighted the use of UBFs for grain crops as they use large amount of fertilizers over large areas.

The treatments were separated into two categories as done by Harder et al. (2019). Conservative treatments retain most of the nutrients in the final product. If a volume reduction is carried out, the water will be extracted and not the nutrients. On the opposite, extractive treatments aim to recover one or several nutrients present in the urine to obtain a concentrated or contaminant-free fertilizer. We focused on conservative treatments since they offer the possibility of closing all nutrient cycles and limiting waste discharge. The treatments studied were storage, acidification, alkalinization, nitrification and the mixtures with solid organic substrates. Phosphorus precipitation was also included as it is one of the most studied extractive treatments. Conservative volume reduction posttreatments to remove water were also considered.

1.4. Literature recovery

Relevant articles were collected from the Web of Science and Google Scholar databases. The search terms "urine fertiliz[s]er" and "urine agriculture"—both separated by the Boolean operator "OR"—were used. No restrictions on language, publication date or type of publication (e.g., gray literature, research project reports) were used. For literature on contaminants, the search terms "pharmaceutical," "metal," and "pathogens" were added to the search string. Finally, the articles found in the reference lists of the first identified papers were added.

2. Characteristics of the treatments

The studied treatments and their value chains are summarized in Figure 1.

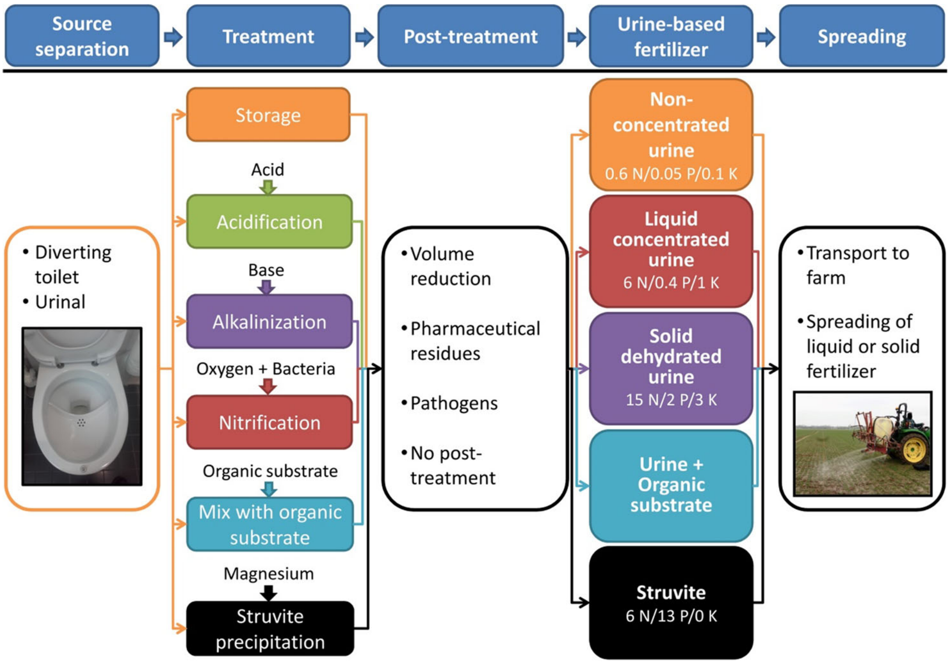

Figure 1. Summary of treatments and value chains studied. The nutrients concentrations (in % mass or g N/100 g raw material) in UBFs are based on average values from literature (see 3.1.1. Nitrogen) and theoretical pure dry struvite (Ahmed et al., 2018).

2.1. Storage

Urine storage was used since ancient times in several countries (Paulet, 1853) and more recently in high-income countries, for example, in ecovillages in Sweden or with waterless urinals during temporary events in France (e.g., Brun, 2017; Johansson et al., 2000). During storage and without additional stabilization treatment, urea is rapidly hydrolyzed to ammoniacal nitrogen. The half-life of urea ranges from 40 years without urease enzyme to a few millisecond in its presence in high concentration (Senecal & Vinnerås, 2017). Hydrolysis of urea occurs more quickly in cases of fecal cross-contamination or presence of biofilms on pipe surfaces (Vinnerås, 2002). Usually, it occurs within a few days of storage (Udert et al., 2003). Storage in this study thus refers to the hydrolysis of urea and then the storage of hydrolyzed urine. The storage time ranges from a few hours (e.g., before treatment such as nitrification) to several months before the application on crops. After urea hydrolysis, approximately 90% of the nitrogen in urine is in ammoniacal form (Udert et al., 2006). The remaining nitrogen is in organic form. The hydrolysis of urea raises the urine pH up to 9.1 (Udert et al., 2006). Ammonia volatilization can occur during storage since the pH is close to 9.2, the pKa of the NH_4^+/NH_3 couple. Ammonia exchange with air is slow, but losses can be high if the storage tank is not sealed (Udert et al., 2006). The increase in pH also promotes phosphorus precipitation (Friedler et al., 2013). Compared to fresh urine, approximately 30% of phosphorus precipitates as struvite [$MgNH_4PO_4 \cdot 6H_2O$] and hydroxyapatite [$Ca_5(PO_4)_3(OH)$] together with the calcium and magnesium present in urine, forming sludge at the tank bottom or deposits in urine pipes (Höglund et al., 2000). The sludge is usually mixed with the liquid part but it can potentially be managed separately from the liquid part as a phosphorus fertilizer, however no studies on this technique were found. Due to the presence of ammonia after urea hydrolysis and of various volatile compounds, the odor of stored urine is strong (Hashemi & Han, 2017; Troccaz et al., 2013).

Depending on the importance of nitrogen gaseous losses and phosphorus, magnesium and calcium precipitation during storage, the content in nutrients differs more or less from fresh urine. Depending on the collection method (e.g., flush toilet, waterless urinal), the concentration ranges between 1.8 and 9.2 g N/L for nitrogen, 0.1 and 0.5 g P/L for phosphorus, and 0.8 and 3.3 g K/L for potassium (Maurer et al., 2006).

2.2. Acidification

Acidification stabilizes the nitrogen in urine. Experimental pilot or laboratory experiments were carried on this treatment (e.g., Boncz et al., 2016, Antonini, Nguyen et al., 2012). When applied on fresh urine (Hellström et al., 1999), acidification to below pH 4 prevents urea hydrolysis during storage and maintains nitrogen as urea (Boncz et al., 2016). The addition of approximately 60 mEq H^+/L is recommended to inhibit urea hydrolysis for more than 100 days. Various strong or weak acids can be used such as sulfuric acid, acetic acid or citric acid (Antonini, Nguyen, et al., 2012; Boncz et al., 2016; Saetta & Boyer, 2017). For concentrated strong acid such as sulfuric acid (e.g., 96%), 60 mEq H^+/L represents about 3 g/L. A larger quantity of weak acids than of strong acids has to be added to provide the same amount of mEq (Boncz et al., 2016). Lactic fermentation can also be used to decrease the pH of fresh urine below 5 by adding easily degradable organic matter and lacto-fermenting bacteria (Andreev et al., 2017). Stored urine can also be acidified to maintain the ammoniacal nitrogen in the NH_4^+ form rather than the NH_3 form (Antonini, Nguyen et al., 2012; Kirchmann & Pettersson, 1994). The optimal pH of acidified stored urine to avoid losses during further dehydration is 4 (Jiang et al., 2017). However, 10 times more acid (600–650 meq H^+/L) is needed to acidify stored urine due to the high buffer capacity of the hydrolyzed urine caused by the ammonia and carbonates formed after urea hydrolysis (Udert et al., 2006). Acidification may lower the organic content of the urine (Antonini, Nguyen et al., 2012). As odor is linked to the NH_3 content of urine (Hashemi & Han, 2017), the odor of the acidified urine is lower than that of the stored urine.

Theoretically, phosphorus precipitates may be redissolved if the stored urine is acidified, due to the pH decrease (Wang et al., 2006). However, the kinetics for dissolution is not known. Then, most of nutrients should be found in liquid phase. Dilution can occur when a diluted weak acid such as vinegar is used. Nutrients can also be added through acidification depending on the chosen acid (e.g., phosphoric acid).

2.3. Alkalinization

Another way to stabilize nitrogen and keep it as urea form is to alkalinize the urine. Some pilot-scale prototypes are currently being field-tested in Finland, France and Sweden (e.g., Karlsson, 2019). Alkaline dehydration is moving toward the commercialization stage with a spin-off company from the Swedish University of Agricultural Sciences: Sanitiation360. An adjusted pH above 10 by adding a base inhibits the urease enzyme (Geinzer, 2017; Kabdaşlı et al., 2006). The base should be added to fresh urine (Simha et al., 2018); if added to hydrolyzed urine, the pH increase promotes NH_3 volatilization. Different types of base or alkaline media can be used, such as calcium, magnesium or potassium hydroxides, wood ash and alkalinized biochar (e.g., Dutta & Vinnerås, 2016; Simha et al., 2018). Urine can also be alkalinized using an anion exchange resin (Simha et al., 2018). An example of a readily available base is calcium hydroxide [$Ca(OH)_2$]. The addition of 10 g $Ca(OH)_2$ per liter of fresh urine represents a good safety margin for stabilization (Randall et al., 2016). This represents approximately 5 kg of $Ca(OH)_2$ per year per capita. Wood ashes should be added at rates of 5 to 10% (weight to weight). In this case, 25 to 55 kg of ash is needed per year per capita (calculation from Senecal & Vinnerås, 2017; Simha et al., 2018). When $Ca(OH)_2$ is used, 95% of the phosphorus precipitates, mostly as calcium phosphate (Flanagan & Randall, 2018). As for stored urine, the tank bottom sludge could potentially be managed separately as a phosphorus fertilizer. The odor of alkalinized urine in liquid or solid form is less strong than that of stored urine (personal test).

The concentration of nitrogen in the liquid phase of alkalinized urine is expected to be approximately the same as that in fresh urine. Since phosphorus dissolution depends on the pH, the phosphorus concentration in the liquid phase will vary accordingly.

2.4. Nitrification

To reduce NH_3 volatilization and odor and make volume reduction possible, nitrogen in stored urine can be stabilized through nitrification. A pilot plant was installed at the Swiss Federal Institute of Aquatic Science and Technology (EAWAG). Nitrification followed by distillation is moving toward the commercialization stage with a spin-off company from EAWAG: Vuna (http://www.vuna.ch/). This reaction acidifies the urine. Approximately 50% of the ammonium content of urine can be converted into nitrate before approaching pH 5.5, the limit for nitrifying bacteria (Etter et al., 2015). Bases such as NaOH or Na_2CO_3 can be added to complete nitrification (e.g., Jiang et al., 2011). Due to the supply of oxygen and aerobic conditions during nitrification, heterotrophic bacteria develop and degrade approximately 90% of the organic substances contained in urine (Fumasoli et al., 2016), decreasing the odor of urine (Etter et al., 2015). This also reduces the interactions between organic matter and any optional following treatments (e.g., adsorption onto an activated carbon filter to remove pharmaceutical residues), increasing their efficiency.

As for the acidification of urine, phosphorus precipitates may be redissolved due to pH decrease during nitrification (Wang et al., 2006).

2.5. Mixture with solid organic substrates

Mixing with organic substrates is widely applied in many contexts (e.g., dry toilets in all types of countries). Urine can be mixed with various organic substrates, such as compost (Shrestha et al., 2013), wood chips (Brun, 2017), sawdust (Brun, 2017), biochar (Bai et al., 2018), or rice straw (Hashemi & Han, 2018). Processes range from a simple impregnation of the organic substrates (e.g., adding urine to wood chips in dry toilet) to a complete transformation e.g., through composting (Brun, 2017; Shrestha et al., 2013). Mixing with organic substrates could be a way to reduce odor and ammonia emissions, as it is also used for the management of animal manure with straw or sawdust (van der Weerden et al., 2014). Nitrogen stabilization may occur due to the interactions with organic matter.

The organic substrate-urine mixtures generally contain fewer mineral nutrients than the other UBFs but contain more nutrients in organic forms. Therefore, the nutrient concentration in the fertilizer depends on the organic substrate used.

2.6. Phosphorus precipitation

Phosphorus precipitation is already implemented in some wastewater treatment plants (WWTPs), and pilot facilities on urine are running (e.g., Zamora et al., 2017). Phosphorus precipitation occurs due to oversaturation resulting from high pH and the presence of magnesium (Ronteltap, 2009) or calcium ions (Randall et al., 2016). In hydrolyzed urine, approximately 30% of phosphate precipitates spontaneously due to pH increase after urea hydrolysis (Udert et al., 2006). Struvite is a salt composed of phosphate, magnesium and nitrogen that can precipitate in stored urine. Struvite precipitation can be increased by adding magnesium to the urine (Zamora et al., 2017). Then, the struvite precipitate can be filtered, washed with water and dried (Etter et al., 2011; Ganrot et al., 2008; Huang et al., 2019). Different forms of magnesium can be used, such as MgO, $Mg(OH)_2$, $MgCl_2$, and $MgSO_4$. Magnesium must be added at a ratio between 1:1 and 1.5:1 Mg:P to recover up to 99% phosphorus (Antonini et al., 2011; Etter et al., 2011; Ronteltap, 2009; Wilsenach, et al., 2007). It is also possible to use sea water, brine or even wood ash as magnesium sources for phosphorus precipitation (Liu et al., 2013; Sakthivel et al., 2012). Phosphorus precipitation can also be increased by adding calcium base in fresh urine to precipitate phosphorus as calcium phosphate but not in stored urine because it reacts with the carbonates resulting from urea hydrolysis, forming calcium carbonate (Randall et al., 2016). Potassium can also precipitate and be recovered with phosphorus as K-struvite [$MgKPO_4{\cdot}6H_2O$] once there is no more ammonium in the solution (Wilsenach et al., 2007).

The precipitation of struvite allows the recovery of phosphorus but only a small fraction of nitrogen. The N:P ratio is about 0.5 in struvite when it is more than 10 in fresh urine (Ahmed et al., 2018). Pure dry struvite theoretically contains 5.7% nitrogen and 12.5% phosphorus (Ahmed et al., 2018). Phosphorus in struvite is more than 100 times more concentrated than it is in fresh urine (Friedler et al., 2013). However, the concentration and crystal structure can vary depending on the precipitation and drying processes, e.g., from 7.2% to 18% of P and the presence of dittmarite or nesquehonite in the crystal structure in examples from Antonini, Arias et al. (2012).

2.7. Volume reduction posttreatments

The water content of urine represents approximately 95% of the weight of fresh urine (Rose et al., 2015). Strauss (1985) mentioned that nitrogen represents 14%–18% and phosphorus and potassium represent both 3.7% of the weight of dry solids in urine. The nitrogen concentration in dry urine can also be lower when chemicals are added. In the case of nitrified urine, the concentration of dry solids can go up to 24% of nitrogen and possibly higher since there is no more carbonate and less organic matter (Udert & Wächter, 2012). Simha et al. (2020), also mentioned 24% of N in the total solid of urine. Post-treatments occur after storage and other treatments mentioned previously. These volume reduction posttreatments are characterized by their concentration factors, expressed as the ratio between the output and the input nitrogen concentrations. Considering the mean urine at 8.7 g N/L (Friedler et al., 2013), the theoretical concentration factor to reach 24% of nitrogen [dehydration] (Udert & Wächter, 2012) is 28. If the urine is flush-diluted to 1.8 g N/L (Etter et al., 2015), the concentration factor is 134 times. Various methods were developed for volume reduction for which energy use is a key factor. Their concentration factors and energy consumption are presented in Table 1.

2.7.1. Evaporation

Nitrogen is stabilized before evaporation to prevent gaseous ammonia emissions and losses of nitrogen. Then, the water can be evaporated by managing the temperature, pressure and hygrometry (e.g., passive evaporation, Bethune et al., 2014, 2015). Evaporation can be a simple volume reduction process (the final product stays liquid) or can continue until dehydration (the final product is a solid). When urea hydrolysis is inhibited, the evaporation should take place below 40–60 °C to prevent chemical degradation of the urea and losses (Randall et al., 2016; Simha et al., 2018, 2020). Nitrogen losses can reach about 5 to 30% during evaporation operating at temperature up to 60 °C in a pilot experiment using alkalinized urine or solar still with acidified stored urine (Antonini, Nguyen, et al., 2012; Simha et al., 2018, 2020). Chemical urea hydrolysis and volatilization are

Table 1. Concentration factor, energy and primary energy (PE) consumption of volume reduction posttreatments and some treatment compared to the production of nitrogen fertilizer, treatment of nitrogen in WWTP and transport.

Process		Energy kWh/m^3	Primary energy		Concentration factor	Comments	Reference
			MJ/m^3	MJ/kg N			
Evaporation	Solar still		0	0	Dehydration	5.5%–8.3% N in final product. Adding acid for stabilization.	Antonini, Nguyen et al. (2012)
	Non-optimized prototype	3483–6333	40,452–73,548	3548–6452	5	Urine at 11.4 g N/L. 5.6% N in final product. Adding base for stabilization. 1100–2000 MJ/kg N*.	Karlsson (2019)
	Pilot scale	100*	1161	208**	Dehydration	13.2% N in final product from Meyer et al. (2018). Adding base for stabilization.	Swiss Federal Institute of Aquatic Science and Technology (n.d.)
Distillation	Without energy recovery	710*	8245	1198–3450	35–67	Urine at 2.4–6.9 g N/L. 99.2% water removed. 16%–24% N in final product.	Udert and Wächter (2012)
		611*	7095	1273**	Dehydration	To remove 1 m^3. 2200 MJ/m^3*. Considering electricity.	Tettenborn (2011)
		722	8387	1505**	Dehydration	To remove 1 m^3 water. 2600 MJ/m^3*. Considering electricity.	Maurer et al. (2006)
			3737	389*	10	Urine 9.6 g N/L. 9.7% N in final product.	Maurer et al. (2003)
	With energy recovery	107*	1243	302–699	20–25	Urine at 1.8–4.1 g N/L. 4.8% N in final product. Energy consumption recovered on graph.	Fumasoli et al. (2016)
		110*	1277	186–534	35–67	Urine at 2.4–6.9 g N/L. 99.2% water removed. 16%–24% N in final product.	Udert and Wächter (2012)
		80*	929	516	28	Urine at 1.8 g N/L. 5% N in final product.	Etter et al. (2015)
		77–110	894–1277	103–147**	Dehydration	96% water removed. 277–396 MJ/m^3*.	Maurer et al. (2006)***
			326	34*	10	Urine 9.6 g N/L. 9.7% N in final product.	Maurer et al. (2003)***
		42–50	184–581	87–104**	Dehydration	Large scale. To remove 1 m^3 of water. 150–180 MJ/m^3*. Considering electricity.	Maurer et al. (2006)***
		111	1290	231**	Dehydration	To remove 1 m^3 of water. 400 MJ/m^3*. Considering electricity.	Maurer et al. (2006)***
Forward osmosis		0.84*	10	0.82	2–3	Urine at 11.9 g N/L. Volume concentration factor (recovered on graph) is 2 to 4. Considering 20% N losses.	Zhang et al. (2014)
Reverse osmosis		5–10*	58–116	19–37	5	Urine at 3.1 g N/L.	Maurer et al. (2003)
		30*	348	97	5	Urine at 3.6 g N/L. Volume concentration factor is 5. Considering 5% N losses. Adding acid for stabilization.	Ek et al. (2006)
Freeze-Thaw		1200*	13,935	2500**	4–5	Laboratory scale. Volume concentration factor. No N losses.	Gulyas et al. (2004)

	244*	2834	508**	4–5	5000 cap scale. Volume concentration factor. No N losses.	Gulyas et al. (2004)***
	30*	348	62**	4–5	500,000 cap scale. Volume concentration factor. No N losses.	Gulyas et al. (2004)***
Lyophilization	2511	29,160	5231**	Dehydration	To remove 1 kg of water. 8100 kWh/m^3 PE*.	Udert and Wächter (2012)***
Nitrification	20–242	230–2809	128–685	—	Urine at 1.8–4.1 g N/L. 11–59 kWh/kg N*.	Fumasoli et al. (2016)
	50*	581	323	—	Urine at 1.8 g N/L.	Etter et al. (2015)
Struvite production			25*	—		Maurer et al. (2003)
N Fertilizer production			45*	—		Maurer et al. (2003)
Nitrification/ Denitrification WWTP			45–109*	—		Maurer et al. (2003)
Transport of non-concentrated urine	10 km	48	6–27**	—	Non-concentrated urine at 1.8–8.7 g N/L. 4.8 MJ/ton/km.	Wilsenach and van Loosdrecht (2006)
	100 km	480	55–267**			

Data recovered from literature are noted with "*." The mean influent nitrogen concentration of all reviewed experiments in Table 1 was 5.6 g N/L. This value was used when the influent nitrogen concentration was not given in order to calculate the energy consumption per kg of nitrogen. Recalculated values using urine at 5.6 g N/L are noted with "**." Conversion into primary energy is based on the European electricity mix (efficiency of 0.31, Maurer et al., 2003). Electricity was taken, if no energy source was given. References are noted with "***" when process was not applied on urine. The energy consumption was recalculated in the same units for all treatments. The concentration factor was calculated as the ratio between the nitrogen concentration in the final products and the corresponding influents. When the concentration factor is given as a volume concentration factor, it was recalculated taking into account the expected nitrogen losses during the process.

linked to the temperature and the duration of the treatment at this temperature. Reducing the temperature (Randall et al., 2016) or increasing dehydration rate (Simha et al., 2020) will reduce nitrogen losses.

After dehydration, the nitrogen concentration in the final product can be approximately 5%–13% (Antonini, Nguyen, et al., 2012; Meyer et al., 2018) depending on the amount of chemical added (e.g., acid sulfuric, lime) and nitrogen losses.

2.7.2. Distillation

Water distillation consists of evaporation at high temperature (e.g., 80 °C) and low pressure (e.g., 500 mbar). To perform distillation, nitrogen has to be stabilized in a form other than NH_3 (to prevent volatilization) or urea (to prevent chemical hydrolysis at high temperature) with treatments such as nitrification or acidification. Nitrogen losses during the process are less than 3% for nitrified urine but can be higher for distilled acidified urine (Etter et al., 2015; Tettenborn, 2011; Udert & Wächter, 2012). Phosphorus losses during the process are driven by the pH and the concentration factor, but phosphorus is not likely to precipitate during distillation of acidified urine (pH 4.7–7.3, up to 60 g P/L, Tettenborn, 2011). Up to 50% of sodium chloride may be removed during distillation (through sequential distillation) because it is the first salt to precipitate; this reduces the risk of soil salinization after the use of UBFs (Etter et al., 2015). Energy recovery can be performed on the distiller using vapor compression or heat exchange, which reduces the energy consumption by up to 90% (Etter et al., 2015).

The nutrient concentrations of nitrified urine after distillation can reach 4%–6% N, 0.2%–0.4% P, and 2%–4% K in liquid concentrate form (Fumasoli et al., 2016; Martin, 2018, considering a density of 1.14 in the final product, Etter B. [personal communication, November 14, 2018]) and 16%–24% N, 2% P, and 7% K in dehydrated solid form (Etter et al., 2015; Udert & Wächter, 2012).

2.7.3. Forward osmosis

For forward osmosis, due to osmotic pressure, water is separated from the urine passing through a semipermeable membrane into a highly saline solution such as sea water (Zhang et al., 2014). This process requires a low amount of energy. However, the tested membranes show permeability to urea or NH_3 (Nikiema et al., 2017; Zhang et al., 2014), implying the need to use hydrolyzed acidified urine to obtain a higher retention rate for nitrogen (Volpin et al., 2019). The recovery rates for nitrogen depend on the nitrogen form, e.g., less than 50% for urea and between 31% and 91% for charged ammonium (Zhang et al., 2014). Recovery rates may be higher than 90% for phosphorus and potassium (Nikiema et al., 2017; Zhang et al., 2014). Forward osmosis show lower fouling potential than other membranes processes. Most of experiments use synthetic urine but experiments with real urine are necessary to assess membrane fouling (Volpin et al., 2018; Zhang et al., 2014). Progress is needed to produce membranes with high solute rejection. The volume concentration factor could ranges from 2.5 to 15 depending on how diluted the influent urine is (Zhang et al., 2014).

2.7.4. Reverse osmosis

During reverse osmosis, the water from urine passes through a semipermeable membrane due to the application of a pressure higher than the osmotic pressure. As for forward osmosis, the membranes have the tendency to be permeable to NH_3, so the hydrolyzed urine can be acidified to keep the nitrogen in NH_4^+ form (Ek et al., 2006). In experiments using hydrolyzed urine, nutrient recovery ranges from 70% to more than 90% (Ek et al., 2006; Maurer et al., 2006). However, scaling problems (salt precipitation on the membrane) were reported (Maurer et al., 2006).

2.7.5. Freezing-thawing

As the solution is slowly frozen, ions and chemical compounds are excluded from the ice crystalline structure. When melted, the ions and compounds are the first to be found in solution and can be separated (Lind et al., 2001). Such a treatment enables the concentration of approximately 80% of the nutrients in 25% of the original volume of urine (Lind et al., 2001). In such a case, the volume concentration factor is between 3.2 and 5 (Gulyas et al., 2004; Lind et al., 2001).

2.7.6. Lyophilization

During lyophilization, water is sublimated at very low temperatures (less than $-40\,^{\circ}C$) and pressures (Udert & Wächter, 2012). Based on total dehydration, the concentration factor for lyophilization is the same as that for evaporation, but the process requires more energy (Udert & Wächter, 2012).

2.7.7. Membrane distillation

During membrane distillation, water is removed through a microporous hydrophobic membrane. Distillation is driven by the vapor pressure gradient induced by the temperature difference on each side of the membrane (e.g., Tun et al., 2016; Volpin et al., 2020). Operational temperature is lower than classic distillation. Then, energy consumption is expected to be lower. However fouling is an important issue and gaseous NH_3 can pass through the membrane (Derese & Verliefde, 2016; Tun et al., 2016). Treatment such as nitrification (Volpin et al., 2020), forward osmosis (Liu et al., 2016), acidification (of hydrolyzed urine) or filtration (Tun et al., 2016) aim to avoid NH_3 permeation or fouling on the membrane. Water removal can be up to 75%–95% removal (Derese & Verliefde, 2016; Volpin et al., 2020). No energy data were found for membrane distillation applied to urine but energy consumption can vary a lot depending on the different types of technology. Zuo et al.,(2011) mentioned that the energy consumption of a large scale membrane distillation desalination plant can be more than $144\,MJ/m^3$ of water removed ($40\,kWh/m^3$), but the environmental impact differs if a natural heating resource is used (e.g., solar, geothermal energy, Cabassud & Wirth, 2003).

3. Comparisons between treatments and resulting fertilizers

3.1. Nutrient concentration

3.1.1. Nitrogen

The UBFs were separated into three ranges of nitrogen concentrations, depending on how far the volume reduction was pushed after source separation of urine, and compared to mineral fertilizers and animal slurries in Figure 2.

Non-concentrated urine and animal slurries are in the same ranges, with much lower nitrogen contents than mineral fertilizers. Liquid concentrated urine is approximately 10 times more concentrated but remains below the level of mineral fertilizers. Only dehydrated urine reaches concentrations close to those of mineral fertilizers. Struvite contains nitrogen but is mainly a phosphorus fertilizer. Mixtures of urine and solid organic substrates are not presented, as the nitrogen content varies considerably depending on the mixture. Indeed, the nitrogen forms present in UBFs vary depending on the fertilizer (Figure 3).

Except in the urine-organic substrate mixtures, the nitrogen forms are similar to those found in typical mineral fertilizers (urea, ammoniacal and nitrate). Urea is the main form of nitrogen present in fresh urine or in stabilized products after acidification or alkalinization of fresh urine. Stored urine, acidified stored urine and struvite are dominated by ammoniacal nitrogen. The pH has a strong influence on the ammoniacal nitrogen form (pKa $[NH_3/NH_4^+] = 9.2$). Considering pKa, NH_3 may represent more than one third of the total nitrogen in stored urine at pH 9.1 and less than 1% of the total nitrogen in acidified urine at pH 6.5. Generally, the organic nitrogen content is lower in UBFs than in slurry, except in mixtures of

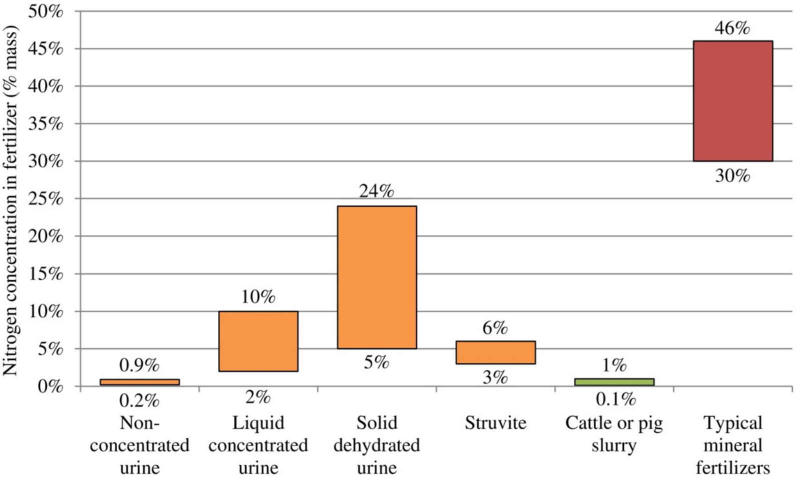

Figure 2. Nitrogen concentration in UBFs (in % mass or g N/100 g raw material), compared to nitrogen contents in animal slurries and in the most frequently used mineral fertilizers for grain crops in France: ammonium nitrate, urea ammonium nitrate solution and urea (UNIFA, 2018). Cattle and pig slurry average nitrogen concentration come from Benoît et al. (2014). Non-concentrated urine is based on fresh urine and urine diluted by flushing (Maurer et al., 2006). Liquid concentrated urine is based on reverse osmosis and distilled acidified urine (Maurer et al., 2003), density of 1 is assumed. Solid dehydrated urine is based on acidified dehydrated urine (Antonini, Nguyen, et al., 2012) and nitrified dehydrated urine (Udert & Wächter, 2012). Struvite concentrations come from Antonini, Arias, et al. (2012).

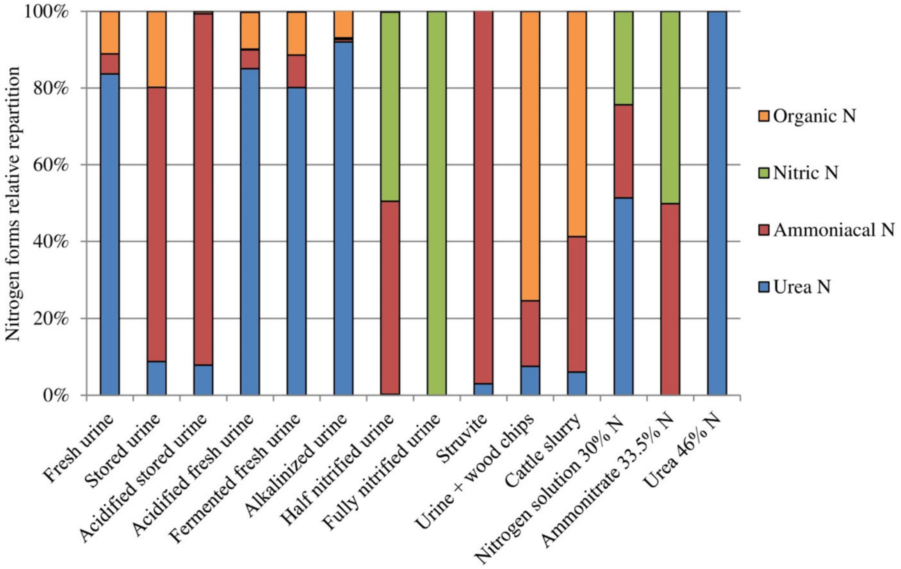

Figure 3. Nitrogen forms relative repartition in the different UBFs, cattle slurry and mineral fertilizers. Data for UBFs and cattle slurry are based on Jiang et al. (2011), Martin et al. (2020), and Udert et al. (2006). Urine was acidified using sulfuric acid and alkalinized using lime. Data for mineral fertilizers are based on the 3 most sold nitrogen fertilizers in France for grain crops (UNIFA, 2018).

urine and organic substrates. Chemicals used to stabilize urine can have an impact on the nitrogen forms repartition (e.g., base or acid which contain nitrogen).

3.1.2. Phosphorus and other nutrients

As for nitrogen, the contents and forms of phosphorus differ among UBFs. Considering an N:P ratio of 11.8 (Friedler et al., 2013), the phosphorus concentration is between 0.02% and 0.08% for non-concentrated urine, 0.3% and 0.5% for liquid concentrated urine, and 0.4% and 3.7% for solid dehydrated urine (considering dry solid urine with the upper value of Strauss, 1985). In the case of stored or alkalinized urine, a large part of the phosphorus precipitates and may be managed separately.

Considering a 4.2 N:K ratio (Friedler et al., 2013), the potassium concentration is between 0.05% and 0.2% for non-concentrated urine, 0.7% and 1.4% for liquid concentrated urine, and 1.2% and 5.7% for solid dehydrated urine. Most often, potassium is not affected during the treatments, except in a few cases, e.g., the addition of sulfuric acid followed by distillation may make potassium precipitate as potassium sulfate (Tettenborn, 2011) or precipitation of K-struvite when there is no more nitrogen in solution. Some micronutrients can come from the different chemicals used, such as calcium from lime, phosphorus or potassium from ash or sulfur from sulfuric acid; this may explain the values sometimes higher than those given by Strauss (1985). For pure struvite, the phosphorus concentration is approximately 12.6% (Ahmed et al., 2018), which is in the same range as that in common phosphorus mineral fertilizers (e.g., simple superphosphate, 8% P or 18% P_2O_5, or triple superphosphate 20% P or 45% P_2O_5).

3.2. Fertilizing efficiency and related impacts

3.2.1. Nitrogen and phosphorus use efficiencies

UBFs and especially stored urine were tested on various crops (e.g., Karak & Bhattacharyya, 2011; Pandorf et al., 2019). Table 2 provides data about the nutrient use efficiency of different UBFs. Different methods can be used to calculate these efficiencies. When nutrient use efficiency (NUE) is not directly given, it was calculated as the ratio between the additional nutrient uptake by the fertilized crop compared to the uptake by the control crop without fertilization and the nutrient added by the fertilizer (Equation 1):

$$\text{NUE (\%)} = \frac{\text{Nutrient uptake fertilized crop} - \text{Nutrient uptake control crop}}{\text{Nutrient added by fertilizer}} \times 100 \tag{1}$$

Table 2. Examples of pot and field agronomical trials using UBFs on various crops.

Product	Crop	Type	Element	MFE	Other results and comments	Reference
Stored urine	Barley	Pot	N P	79%** 128%**	Recalculated from N and P exportations. Compared to K_2PO_4 (soluble phosphate) for phosphorus. Soil pH: 6.2	Kirchmann and Pettersson (1994)
Stored urine	(1) Winter wheat (2) Oats	Field	Yield	60%–80%* 50%–60%*	Organic farming on 4 sites. Calculated as the yield ratio between urine and mineral fertilizer (tested on another conventional site).	Lindén (1997)
Stored urine	Spring barley	Field	Yield	80%–90%*	MFE using grain yield. 3 years of experiment.	Johansson (2000)
Stored urine	(1) Spring wheat, (2) winter rye, (3) maize. Also tested on oats, oil rape seed and hemp.	Field	N	(1) 107%** (2) 86%** (3) 64%**	Calculated using N exportation from all plant part. When different doses of nitrogen are given, the slope of the response curve of N exportation according to N input was use as NUE.	Muskolus (2007)
Stored urine	(1) Maize, (2) tomato. Also tested on carrot and beetroot.	Pot	N	(1) 104%** (2) 70%**	Calculated using nutrient exportation from all crop part (e.g., fruit, leave) and the slope of the response curve of N as NUE.	Mnkeni et al. (2008)
Stored urine	(1)Winter barley (2) Italian ryegrass	(1) Field (2) Pot	N	(1) 96%–101%** (2) 91%**	Recalculated using N exportation and N use efficiency (recovered on graph). 2 years of experiments.	Simons (2008)
Stored urine	Spring oats	Field	N	64%–82%*	2 years of experiment.	Gómez-Muñoz et al. (2017)
Stored urine	English ryegrass	Pot	N	86%–87%*	2 doses of nitrogen.	Martin (2018)
Stored urine	English ryegrass	Pot	N P	103%** 74%**	Ratio between nitrogen exportation of urine and mineral fertilizer treatments. Exportation calculated from yield (recovered on graph) and nutrients concentrations in tissues. No 0N0P control. Mineral fertilizer: 2:3:2 NPK. Acidic soil (pH 4).	Mchunu et al. (2018)
Stored urine	Spring barley	Field	Yield	74%–114%**	MFE calculated using grain yield (recovered on graph). Mineral fertilizer: N + Mg and 23:3:8 NPK. 2 doses of nitrogen.	Viskari et al. (2018)
Acidified urine	Italian ryegrass	Pot	N	102%*	Efficiency higher than stored urine (MFE 91%)	Simons (2008)
Alkalinized urine	Italian ryegrass	Pot	P	83%*	Compared to soluble P: $Ca(H_2PO_4)^{\cdot}H_2O$. Alkaline soil (pH 7.7)	Meyer et al. (2018)
Nitrified urine	Italian ryegrass	Pot	N	97%**	Calculated as ratio of N fertilizer recovery in plants from urine and mineral fertilizer. N input not equals. Acidic soil (pH 5.4).	Bonvin et al. (2015)
Nitrified urine	English ryegrass	Pot	N	94%–99%*	2 doses of nitrogen.	Martin (2018)

(*continued*)

Table 2. Continued.

Product	Crop	Type	Element	MFE	Other results and comments	Reference
Nitrified urine	English ryegrass	Pot	N P	127%** 104%**	Ratio between nitrogen exportation of urine and mineral fertilizer treatments. Exportation calculated from yield (recovered on graph) and nutrients concentrations in tissues. No 0N0P control. Mineral fertilizer: 2:3:2 NPK. Acidic soil (pH 4).	Mchunu et al. (2018)
Compost impregnated with stored urine	English ryegrass	Pot	N	72%–73%*	Calculated only with N from urine. Efficiency lower than stored urine alone (MFE 86%–87%). 2 doses of nitrogen.	Martin (2018)
Struvite	Italian ryegrass	Pot	P	97%*	Compared to soluble P: KH_2PO_4. Acidic soil (pH 5.4).	Bonvin et al. (2015)
Struvite	English ryegrass	Pot	P	90%–109%*	9 struvite not from urine. Efficiencies recovered from “apparent effectiveness” compared to $Ca(H_2PO_4)_2$. Soil pH: 6.6–7.1. Efficiency better than iron and calcium phosphate from WWTP.	Johnston and Richards (2003)
Struvite	Italian ryegrass	Pot	P	(1) 66%–75%* (2) 104%*	Recovered on graph. Compared to soluble P: $Ca(H_2PO_4)·H_2O$. (1) Alkaline soil (pH 7.7). (2) Acidic soil (pH 5.1).	Meyer et al. (2018)
Struvite	(1) Italian ryegrass, (2) red clover. Also tested on grassland.	Pot	P	(1) 150%–175%* (2) 98%–144%*	Compared to superphosphate. Soil pH: 6. Considering struvite powder from urine. Efficiency generally higher than precipitated P from WWTP. 2 year of experiment.	Simons (2008)

Efficiency is characterized by the nitrogen (N) and phosphorus (P) mineral fertilizer equivalence (MFE). Values recovered from literature are noted with “*.” Calculated values are noted with “**.” Details of the calculations are given in SM 3.

If no data were available about nutrient exportation, NUE was calculated using the additional yield increase in the fertilized crop compared to the yield of the control crop without fertilization.

The nutrient use efficiency of UBF can be compared to that of mineral fertilizer by calculating the mineral fertilizer equivalence (MFE) as the ratio between the NUE of UBF and that of a reference mineral fertilizer as ammonium nitrate (Equation 2):

$$\text{MFE}\ (\%) = \frac{\text{NUE Urine-based fertilizer}}{\text{NUE Mineral fertilizer}} \times 100 \tag{2}$$

Nitrogen. The nitrogen NUE of UBFs is similar or slightly lower than that of mineral fertilizer and the MFE is high (generally higher than 75% and close to 100%). The differences among acidified, nitrified and stored urine (no nitrogen MFE data for alkalinized urine) are low. However, MFE seems to be slightly higher for nitrified urine or acidified urine compared to stored urine. Alkalinized urine was little studied, however tests performed with alkalinized urine have shown positive impacts on crop yield (Filling, 2018). The MFE of urine-organic substrate mixtures may be lower due to the possible fixation of the nitrogen from urine with the organic substrate (e.g., Martin, 2018). Other mixtures such as compost supplemented with urine have a positive short-term impact on yield (Fatunbi, 2009; Shrestha et al., 2013). The nitrogen efficiency is generally higher than for animal slurries and manure (Martin, 2018; Pradhan et al., 2011).

Phosphorus. Phosphorus availability depends on many parameters. UBFs are generally compared to calcium superphosphate or other soluble phosphate. However, these two mineral fertilizers have different NUE, which can impact the MFE values of UBFs. Because of its high phosphorus content, struvite is mainly used as a phosphorus fertilizer. Different tests were performed with different crops (e.g., Meyer et al., 2018 for dehydrated alkalinized urine). Phosphorous availability also strongly depends on the soil type. In the case of struvite, its efficiency is much higher in acidic soils than in alkaline soils. The MFE varies more for phosphorous than for nitrogen but also remains close to 100% compare to soluble phosphate. Some of the UBFs as struvite may act as a slow-release fertilizer (Everaert et al., 2017).

To summarize, the efficiencies of UBFs are close to those of mineral fertilizers for both phosphorus and nitrogen. However, to allow a clear comparison of the different UBFs, an experiment comparing all UBFs under the same conditions to avoid pedo-climatic and cropping system variations is needed.

Table 3. Examples of product application rates to provide 170 kg N/ha.

	Fertilizer nitrogen concentration (% N and g N/L)		Quantity spread to bring 170 kg N/ha (ton/ha)		
Fertilizer	min	max	min	max	Spreading machinery
Non-concentrated urine	0.2% (1.8 g N/L)	0.9% (8.7 g N/L)	20	94	Slurry spreader
Liquid concentrated urine	2% (15.5 g N/L)	10% (96.5 g N/L)	1.7	11	Liquid sprayer/slurry spreader
Solid dehydrated urine	5% (54.6 g N/kg)	24%	0.7	3.1	Solid spreader
Struvite	3% (29.3 g N/kg)	6% (60.0 g N/kg)	2.8	5.8	Solid spreader
Cattle slurry	0.1% (0.9 g N/kg)	1% (5.4 g N/kg)	31	189	Slurry spreader
Mineral fertilizer	30%	46%	0.37	0.57	Liquid sprayer/solid spreader

This is the highest dose authorized for an organic fertilizer in the European nitrate directive (Council Directive 91/676/EEC, 1991) and also represents an average nitrogen dose for grain crop in conventional agriculture in temperate climate such as France. Fertilizers concentrations are taken from Figure 2. The exact concentrations (in g N/L or g N/kg) were added when it was given in the references.

3.2.2. Application

Based on the different forms of UBFs (liquid concentrated, solid, etc.), the quantities required to provide 170 kg N/ha are presented in Table 3.

The quantities range from 0.7 t/ha for dehydrated urine to 94 t/ha for flush-diluted urine. Non-concentrated urine may be spread using a slurry spreader. Liquid concentrated urine has a nitrogen concentration that can be suitable for both liquid mineral fertilizer sprayers and slurry spreaders, depending on the dose needed. Dehydrated products and struvite are more similar to typical mineral fertilizers and may be applied with the same type of equipment if they are transformed into an adapted form (e.g., granular, pellet). It seems possible to replace all mineral fertilizers with dehydrated urine, highly concentrated urine or struvite at similar applied volumes and working times. However, complete fertilization with non-concentrated urine in an intensive cropping system would require approximately 30 to 250 more quantities to be spread compared to the amount of mineral fertilizer required, which represents a considerable amount of time with identical equipment and increased risks of soil compaction.

3.2.3. Gaseous emissions

Ammonia volatilization. Ammonia volatilization causes air pollution (e.g., acidification and particulate matter). Volatilization depends highly on the product characteristics (e.g., pH and NH_3 concentration), application techniques and pedo-climatic conditions (Huijsmans, 2003). For stored urine, using trailing hoses followed by harrowing 4 hours later, ammonia emissions from open soil range from 2% to 10% of the nitrogen applied and only from 0.3% to 1.1% with trailing shoes. Volatilization is very low for both techniques when urine is applied on growing crops (0.2%–0.4% of nitrogen applied)

(Rodhe et al., 2004). Ammonia emissions from other products were not studied, and great diversity is expected due to the various characteristics of the products. NH_3 volatilization is expected to be lower for nitrified and acidified urine than for other products due to the lower ammonia concentration and low pH. Alkalinized urine may have a volatilization rate as high as that of granular urea since increasing pH near the granule (due to urea hydrolysis) can promote volatilization (Fenn & Hossner, 1985). The other volatile organic compounds (VOCs) were barely studied but may imply some environmental impacts. For example, VOCs from slurry may contribute to the formation of tropospheric ozone (Liu et al., 2018).

N_2O emissions. Nitrous oxide is a greenhouse gas with a strong global warming potential (265 times higher than that of CO_2). N_2O emissions after UBF application were little studied. In an incubation experiment, Gómez-Muñoz et al. (2017) observed significantly lower emissions from stored urine (and cattle slurry) than from mineral fertilizer or other organic products. Simons (2008) measured N_2O emissions ranging from 0.1 to 0.8% of nitrogen applied in a pot experiment. This rate was higher than that for mineral fertilizer (0.1%–0.3% of nitrogen applied). N_2O emissions were slightly lower for acidified urine than for stored urine in this experiment. Further studies on N_2O field emissions are needed, as well as N_2O emissions during nitrification.

3.2.4. Other impacts linked to field application

Impacts on earthworms. Cattle slurry and animal urine have toxic effects on earthworms because they contain ammonia, benzoic acid and sodium sulfide (Muskolus, 2007). Stored urine has a strong impact on earthworm abundance after application and was qualified as "toxic for earthworms" immediately after application (Muskolus, 2007). The effects decrease with time but can last more than six months under certain conditions. The liquid form of urine infiltrates earthworm holes, and the direct contact with earthworms may induce toxicity. Tillage shortly before or just after application may limit this impact (Muskolus, 2007). The toxicity may also be lower with a solid product. To our knowledge, the impacts of other UBFs on earthworms were not studied.

Seed germination. The application of stored urine may reduce the germination of some plants depending on the experimental conditions, e.g., on hemp in pot conditions or direct application on wheat or sunflower in Petri dishes (Muskolus, 2007; Winker et al., 2013). No effect was observed in the field, as the urine is buffered by the soil (Viskari et al., 2018; Winker et al., 2013). The high salt concentration in "pure urine" seems to have a strong negative impact on seed germination (Simons, 2008). Lacto-fermentation of urine before application increases the germination of radish seeds compared to that after the application of stored urine in Petri dish experiments, probably due to the lower ammoniacal nitrogen content in the fermented urine. The pH may also play a role in the germination process (Andreev et al., 2017). No study has shown negative effects on seed germination after struvite application (Gell et al., 2011). The impacts of other UBFs were not studied.

Soil salinization. Fresh urine contains a lot of different salts (Beler-Baykal et al., 2011). The main ions involved in soil salinization are sodium, boron, chloride and sulfates (WHO, 2012). Salinity can affect soil structure and plant growth (Friedler et al., 2013). The concentration in salt and especially in sodium (linked to high levels of table salt in most diets) may be higher than in animal slurry with for example 2.6 g/L in urine (Udert et al., 2006) versus 0.9 g/L in pig slurry (Moral et al., 2008). Salinization generally occurs in semi-arid and arid areas where salts are not lixiviated by the rain. In areas where there are soil salinization problems, caution should be taken before using urine as fertilizer because of its high salt content (Jönsson et al., 2004). High rate of fertilization using urine may imply higher sodium uptake by the crops and may have a negative impact on yield. Then urine may not be use with salt-sensitive crops especially when the soil is sensitive to salinization (Mnkeni et al., 2005, 2008). To our knowledge, no problems were reported in other areas.

Soil acidification. Soil acidification involves complex physico-chemical processes (Bolan & Hedley, 2003). The use of animal urine or mineral nitrogen fertilizers acidifies the soil after application (Black, 1992). Therefore, the application of human urine can be expected to acidify the soil as well. Acid added for the treatments may increase acidification while in areas where the soil is acidic and needs to be limed, the high pH of the alkalinized urine could be beneficial.

More broadly, the long-term impacts of urine application were barely studied. To our knowledge, only one long-term trial is documented (Crucial field test in Denmark, Gómez-Muñoz et al., 2017).

3.3. Contaminants

3.3.1. Trace metals

There is little data on the behavior of trace metals during the treatments. However, trace metal contents are generally low in UBFs, most often lower than in mineral fertilizers such as phosphate rock or cattle slurry, although a fraction of the metals contained in urine may precipitates in struvite (Ronteltap et al., 2007). These contents generally meet regulations for fertilizers or the use of sewage sludge in agriculture (Antonini et al., 2011; Gell et al., 2011, Fumasoli et al., 2016). However, contamination can potentially come from the chemicals used for treatment (e.g., ash for alkalinization, Senecal & Vinnerås, 2017; Simha et al., 2020).

3.3.2. Pharmaceutical residues

The panel of pharmaceutical residues studied is very diverse, including more than 100 substances and metabolites. The effect of storage on pharmaceutical

Table 4. Efficiency of treatments on pharmaceutical residues.

Treatment	Efficiency	Examples	Reference
Storage	+	Reduction from 0% (e.g., bezafibrate) to 35% (e.g., tetracycline) of 10 compounds during 11 months storage.	Butzen et al. (2005)
		Reduction from 79% to 96% of 2 hormones during 6 weeks storage.	Zanchetta et al. (2015)
		Reduction from 24% (e.g., trimethoprim) to more than 99% (e.g., rifampicin) of 8 compounds during 6 months storage.	Jaatinen et al. (2016)
Acidification	+	Reduction from 0% (e.g., bezafibrate) to 30% (e.g., tetracycline) of 10 compounds during 11 months storage at pH 4. Near 100% reduction (e.g., diclofenac) for some compound at pH 2.	Butzen et al. (2005)
		No significant reduction of 4 compounds during 1 year storage at pH 4.	Gajurel et al. (2007)
		Reduction from less than 10% (e.g., tramadol) up to more than 80% (e.g., diclofenac) of 8 compounds during 6 months storage at pH 3.	Schürmann et al. (2012)
		Reduction from 30% to 42% of 2 hormones during 6 weeks storage at pH 2.	Zanchetta et al. (2015)
Alkalinization	+	Reduction from less than 10% (e.g., ibuprofen) up to more than 80% (e.g., sulfadimidine) of 8 compounds during 6 months storage at pH 11.	Schürmann et al. (2012)
		No significant reduction of 4 pharmaceutical during 1 year storage at pH 10.	Gajurel et al. (2007)
Nitrification	++	Estimated half-lives of 10 compounds during nitrification range from 40 min (e.g., atazanavir) to more than 48 h (e.g., diclofenac).	Etter et al. (2015)
Mix with organic substrate	Unknown	Not studied.	No reference found
Precipitation of struvite	++	More than 98% of the 7 compounds studied remained in solution.	Ronteltap et al. (2007)
		From 10 compounds, only two were found in struvite at low concentration (less than 5% of the initial amount) and one at high amount (up to 98% of the initial amount). The 7 other were not found in struvite.	Kemacheevakul et al. (2012)
Volume reduction	Unknown to +	Reduction from 88% to 97% of 2 hormones during evaporation of stored and acidified urine (pH 2).	Zanchetta et al. (2015)
Post-treatment		No significant reduction (e.g., phenazone) to complete reduction (e.g., ß-Sitosterol) for the distillation of acidified urine.	Tettenborn (2011)
		Not studied for other volume reduction treatments on urine.	
Pharmaceutical Post-treatment	+++	90% of pharmaceutical residues reduction during batch test using unactivated biochar.	Solanki and Boyer (2017)
		Over 98% pharmaceuticals removal for 11 compounds in nitrified urine using granular activated carbon.	Köpping et al. (2020)

Weak (+): Low impact on pharmaceutical residues. Removal can be high for some compounds and low for other. Moderate (++): Significant removal for most pharmaceutical residues. Strong (+++): Strong impact on short term for most pharmaceutical residues.

residues is poorly studied. Treatment efficiencies for reducing pharmaceutical residues were classified on a 3-step scale in Table 4. Since no reduction guidelines were found, it is difficult to define which concentration reductions are suitable for agricultural use.

All treatments and posttreatments have an impact on pharmaceutical residues, but the impact varies depending on the compound considered (e.g., Schürmann et al., 2012). However, none of the treatments allows high pharmaceutical removal, except specific pharmaceutical posttreatments. Storage is not sufficient to remove pharmaceutical residues and some authors mentioned that the urine of people under medication should not be used in agriculture (Jaatinen et al., 2016; Winker, 2009). The impact of acidification and alkalinization also appears to be small. Nitrification may increase the degradation of pharmaceutical residues compared to storage. The behavior of pharmaceuticals during struvite precipitation is highly dependent on the compound (Kemacheevakul et al., 2012). However, the application of struvite (derived from swine wastewater) can increase the abundance and diversity of antibiotic resistance genes in soil (Chen et al., 2017). We highlight that there is high variability between studies on the same treatment. For example, Gajurel et al. (2007) found no significant reduction in diclofenac during a one-year storage experiment at pH 4, while Butzen et al. (2005) and Schürmann et al. (2012) found over 80% degradation of diclofenac at pH 2–3 within 6–11 months.

No pharmaceutical residues were detected in crops after the application of unspiked stored urine (Mullen, 2018; Viskari et al., 2018; Winker et al., 2010) and the negative impacts on crop yield or quality are assumed to be negligible by some authors (WHO, 2012). In the example of struvite, the bioaccumulation of pharmaceuticals in tomato was shown to be low and the pharmaceutical contamination of tomato was lower than the acceptable daily intake (de Boer et al., 2018). The comparison with animal manures is difficult because the compounds used in veterinary medicine are not always the same as those used for humans. The use of antibiotics is frequent in animal breeding, but humans may use other impacting substances, such as anticancer drugs. Hammer and Clemens (2007) have calculated that the fluxes of tetracycline (one of the most-sold types of antibiotic for humans and animals) are much lower when a field is fertilized with human urine than with cattle or pig slurry (at least 1 $\log_{10}$ difference). The fluxes of 3 hormones were also lower using stored urine. Post-treatments such as filtration on activated carbon can be performed (e.g., Köpping et al., 2020; Solanki & Boyer, 2017). They are efficient in terms of the reduction of pharmaceutical residues but it could hinder the implementation of urine value chains in some contexts.

Further studies are needed on the behavior of pharmaceuticals in soils after application and their potential ecotoxicological impacts. The management of pharmaceutical residues should be included in a holistic approach. For example, the “benign by design” approach aims to design “green” biodegradable chemicals (Kümmerer, 2013).

3.3.3. Pathogens

The pathogens contamination of UBFs is much lower than that in wastewater. The effects of treatments on pathogens are more documented than those on pharmaceutical residues. We classified all treatments on a 3-step scale in Table 5.

Most treatments have a moderate impact on pathogens. During storage, the effect depends on the NH_3 concentration, temperature, and storage time (Nordin, 2010; Decrey & Kohn, 2017). Most pathogenic bacteria studied are generally inactivated within a few days (Höglund et al., 1998;

Table 5. Efficiency of treatments on pathogenic organisms.

Treatment	Efficiency	Examples	Reference
Storage	++	Even at low NH_3 concentration and 4 °C, the time to inactivate 90% of *Salmonella* spp. and *E. coli* is 5 days. *Ascaris* egg inactivation is more difficult and depends on temperature and NH_3 concentration.	Nordin (2010)
Acidification	+ to ++	Strong reductions of bacteria at pH lower than 2. At pH between 2 to 7 (acidified fresh urine), bacteria concentration may be higher compared to non-acidified urine.	Hellström et al. (1999)
Alkalinization	++	Bacteria and bacteriophages are quickly inactivated in alkalinized dehydrated urine. Alkaline pH alone did not inactivate *Ascaris* eggs.	Senecal et al. (2018) Senecal et al. (2020)
Nitrification	++	Nitrification causes inactivation of the bacteria *Salmonella typhimerium*, *Enterococcus* spp. but has no influence on bacteriophages *ΦX147, MS2, Qbeta*.	Schertenleib (2014)
Mixture with organic substrate	Unknown for most mixture to +	Urine may have a positive impact if the organic substrate is contaminated. *Ascaris* egg inactivation is improved when urine is added to toilet compost.	McKinley et al. (2012)
Precipitation of struvite	++	*Ascaris* eggs and bacteria can accumulate in struvite during filtration. Bacteria, bacteriophage *ΦX174* and *Ascaris* eggs are inactivated at high temperature and low moisture content during drying of struvite. High salinity after dehydration can also play a role in the inactivation of pathogens.	Decrey et al. (2011); Bischel et al. (2016) Gell et al. (2011)
Volume reduction Post-treatment	Unknown to +++	Dehydration promotes the inactivation of bacteria and bacteriophages in alkalinized urine. *Ascaris* eggs are resistant to low moisture and temperature is a key factor in their inactivation. Distillation at 80 °C has a strong impact on pathogens. Not studied for other processes.	Senecal et al. (2018, 2020) Etter et al. (2015)
Pathogens Post-treatment	Unknown to +++	Pasteurization at a temperature of 70 °C for 30 minutes or 80 °C for 1.2 minutes can be used to sanitize urine. Oxidation, filtration or ultraviolet processes that are used to remove pathogens in other liquid streams or to treat pharmaceuticals in urine could be efficient on pathogens.	Noe-Hays et al. (n.d.) No reference found on urine

Weak (+): Weak inactivation of pathogens. Moderate (++): Partial inactivation or possible under certain conditions (e.g., storage time, NH_3 concentration). Strong (+++): Complete inactivation on short term.

Nordin, 2010), and most viruses and oocysts studied within a few weeks (Chandran et al., 2009; Goetsch et al., 2018). Other pathogens, such as *Ascaris* eggs or Clostridia, can still be present after several months of storage (Höglund et al., 2000; Nordin, 2010). The WHO recommends 1 to 6 months of storage (for stored urine) depending on the temperature and the crops on which urine will be applied (WHO, 2012). Urine storage does not lead to complete sterilization but offers a "selective environment" with a bacterial community dominated by *Lactobacillales* and *Clostridiales*, well adapted to this environment (Lahr et al., 2016). Hydrolyzed urine shows faster inactivation of bacteria than non-hydrolyzed. In hydrolyzed urine, osmotic pressure and concentration factor were found as important factors to reduce *E. coli* contamination (Oishi et al., 2020). These authors highlighted the necessity that storage recommendations to reduce pathogens should be given according to the concentration factor of the treatment and if urine was hydrolyzed or not. Furthermore, storage period may be lowered for certain storage conditions (Vinnerås et al., 2008) or in some special cases (e.g., in waterless urinal, which prevent fecal cross-contamination). Acidification is little documented but very low pH enhances bacteria inactivation, however acidification to higher pH decreases the inactivation of bacteria compared to that in non-acidified urine (Hellström et al., 1999). This may be due to the lower content of ammonia in solution compared to non-acidified urine. Alkalinization permits the inactivation of bacteria and bacteriophages even if the ammonia content is low due to nitrogen stabilization in urea form (Senecal et al., 2018). Nitrification may enhance bacteria inactivation compared to storage but not of bacteriophages (Schertenleib, 2014). Mixtures with organic matter are little documented but high pH and the presence of ammonia from urine may enhance inactivation (McKinley et al., 2012). Dried struvite is in most cases lightly contaminated with pathogens, but the contamination level differs by pathogens. However, the pathogen content in struvite is lower than required in the Dutch fertilizer regulations (Gell et al., 2011). Only some of the volume reduction treatments at high temperatures, such as distillation or posttreatments dedicated to pathogen have a strong impact on pathogens.

There is a need for further research to characterize whether some treatments (e.g., acidification) lead to exposure below the admissible risks proposed by the WHO (2012).

3.4. Treatment inputs

3.4.1. Energy

Energy is particularly needed when volume reduction is conducted or for the transport of urine. The energy consumption greatly varies among the treatments (Table 1), ranging from almost no energy to several thousand megajoules per kilogram per Newton. Energy consumption can vary greatly within the same family of treatments depending on the scale of the process and its optimization. The influent nitrogen concentration also varies greatly and has a strong impact on the energy consumption required to reach the same final nitrogen concentration. It also impacts the nitrogen concentration factor. Caution should be taken in the direct comparison of energy consumption per kilogram of nitrogen, as the concentration factor and final product concentrations also have to be considered. An experiment comparing the energy consumption of processes with the same influent would be interesting and would provide a better comparison.

Most treatments, except some of the osmosis processes and highly optimized distillation processes, require more energy than the production of nitrogen mineral fertilizer (Haber-Bosch process) or treatment in WWTPs. Treatments as nitrification or struvite precipitation require more energy than the other stabilization treatments. However, some treatments can use local renewable energy, such as passive solar stills, which makes direct comparisons of energy consumption less relevant.

The relationship between values of primary energy consumption found in the literature and the nitrogen concentration in the final product after the treatment is presented in Figure 4.

Osmosis processes (reverse and forward) do not lead to a high concentration factor (less than 4% of nitrogen in the final product), but they are the least energy-consuming processes. Freeze-thawing increases the concentration factor but the energy consumption is strongly correlated with the scale of the process (a factor of 40 between laboratory scale and large scale). Evaporation is a very heterogeneous category that ranges from solar stills to electrically powered evaporation. If the processes are not optimized, the energy consumption can be very high (Karlsson, 2019) but it can also be easily reduced. Chemicals must be added for nitrogen stabilization before evaporation (e.g., lime and acid), which can reduce the potential nitrogen concentration in the final product. The energy consumption of distillation without energy recovery is high, but the volume reduction can be brought to dehydration. Distillation using energy recovery (e.g., heat exchange and vapor compression) permits a high energy savings of approximately 85%–90% compared to that required for basic distillation (Etter et al., 2015; Udert & Wächter, 2012). Generally, volume reduction systems are more energy efficient when they have larger infrastructure due to scale effects. However urine is hardly transportable in pipes on long distance (precipitation and the need for a new collection network) preventing the development of centralized infrastructure and many recent processes are in a decentralized perspective (e.g., Randall & Naidoo, 2018). However,

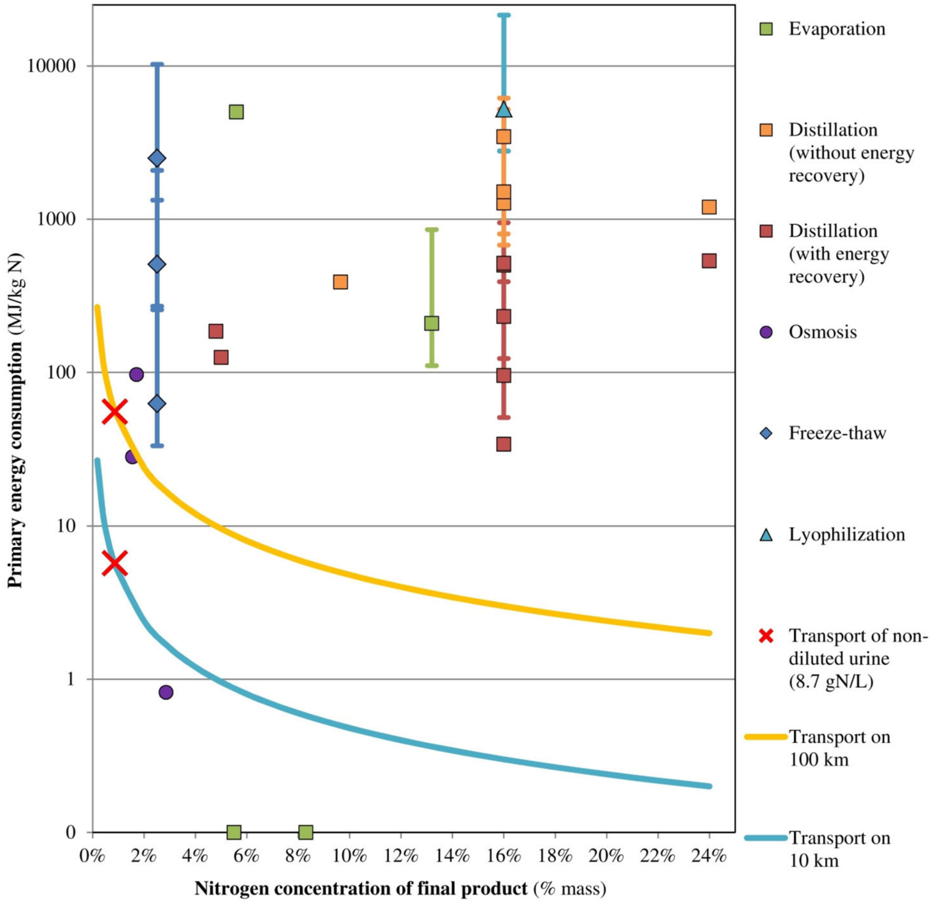

Figure 4. Primary energy consumption (MJ/kg N in influent) and nitrogen concentration in end products (in % mass or g N/100 g of raw material) for the volume reduction treatments. The average energy consumption and final product nitrogen concentration were taken from Table 1. For processes that reach dehydration, if the final product concentration is not given, a standard value of 16% N was used (mean value of Strauss, 1985). The mean influent nitrogen concentration of all reviewed experiments in Table 1 was 5.6 g N/L. This value was used when the influent nitrogen concentration was not given in order to calculate the energy consumption per kg of nitrogen. The least and most concentrated influents in Table 1 (1.8 and 11.9 g N/L) were used to plot the uncertainty of the energy consumption when urine at 5.6 g/L was used for calculation. Energy consumption for transport of urine on 10 or 100 km according to its concentration is also represented considering 4.8 MJ/ton/km (Wilsenach & van Loosdrecht, 2006). The nitrogen concentration of non-diluted urine by flushing is assumed to be 8.7 g N/L (Friedler et al., 2013).

for some treatments such as nitrification or reverse osmosis, semi-centralized treatment plant allowing energy saving can be proposed. Nevertheless, urine would have to be transported to the treatment plant causing transport cost and impacts. For distillation processes, the nitrogen concentration of the final product highly varies without clear relationship with energy consumption which depends strongly on optimization and scale of the process.

Additionally, the energy consumption corresponding to two transportation distances between the toilet and the farm is also considered in Figure 4 and Table 5. Ten kilometers corresponds to the case of a suburban district near a farm. One hundred kilometers corresponds to the case of urine recovered in the center of a megalopolis. The impacts related to transportation vary greatly due to the different available transport modalities and the transport impact could be reduced by using more efficient way of transportation (e.g., larger capacity truck). Depending of the geographical context, nutrient and urine from cities has to be transported more or less far before to be applied on crops (e.g., Trimmer & Guest, 2018). The energy consumption required for transport is high when urine is not concentrated (up to 267 MJ/kg N for 100 km with urine at 1.8 g N/L), but decreases quickly when urine is concentrated. However, whether urine is concentrated or not, the energy used for transport remains lower than the amount of energy used for treatment. Only osmosis processes permit energy savings in some cases compared to the energy required for the transport of non-concentrated urine. Finally, the energy consumption of non-optimized evaporation, distillation without energy recovery or lyophilization is several orders of magnitude higher than that of the transport of non-concentrated urine.

Calculations were performed to compare energy consumption in simplified scenarios of a value chain that considered only transport and treatment (Table 6).

We would like to highlight the impact of the volume reduction of stored urine on the energy consumption required for its transport, especially when the urine has to be transported over a long distance. Except in the scenario of dilute stored urine at 1.8 g N/L transported for 100 km, the energy used in the stored urine scenarios was less than in the reference system (mineral fertilizer + WWTP). However, in the case of nitrified concentrated urine, the treatment has important implications for energy consumption, probably because the process is only at the plant pilot scale

Table 6. Primary energy consumption in simplified scenarios of value chains to produce 1 kg of nitrogen fertilizer.

Scenario	Fertilizer production (MJ/kg N)	Avoided treatment at WWTP (MJ/kg N)	Transport on 10 km (MJ/kg N)	Transport on 100 km (MJ/kg N)	Total (MJ/kg N)
Reference (Mineral fertilizer + WWTP)	45	0	neglected	neglected	45
Stored (diluted) urine (1.8 g N/L)	0	−76	27	267	−49/191
Stored urine (8.7 g N/L)	0	−76	6	55	−21/−70
Nitrified concentrated urine (50 g N/L)	839	−76	1	10	764/773

Data is extracted from Table 1. We consider the use of methanol at the WWTP and 70% of nitrogen treated (Maurer et al., 2003). Energy consumption of the WWTP is considered as avoided. Nitrified concentrated urine scenario is based on Etter et al. (2015). Stored urine at 1.8 g N/L (Etter et al., 2015) and 8.7 g N/L (Friedler et al., 2013) were assumed. To simplify, mineral fertilizer equivalence is 100% for UBFs.

and is not fully optimized (diluted urine, etc.). Nevertheless, many other constraints and impacts have to be taken into account (e.g., the high volume of urine at 1.8 g N/L to spread on crops) to evaluate the value chain. The scale and context of implementation are also critical factors: the Haber-Bosch nitrogen plants that produce mineral fertilizers are large-scale installations connected to the worldwide market, whereas the value chains for urine source separation should be smaller-scale and adapted to local contexts.

3.4.2. Chemicals and organic matter inputs

Many different compounds are used during treatments, such as acids, bases or organic substrates. The quantities used vary greatly depending on the treatment, from a few grams per liter (magnesium for struvite precipitation, sulfuric acid to stabilized urine) to larger quantities (addition of organic matter). The characteristics of the compounds are diverse; some are renewable (e.g., wood chips), whereas other chemicals are not (e.g., magnesium from mines). The impacts of chemical use should be taken into account. However, directly comparing their environmental impacts is not easy. Indicators such as the carbon footprint can be used (Table 7).

Mineral fertilizers generally have a strong impact on the carbon footprints of farms (e.g., Yan et al., 2015). However, the carbon footprint of

Table 7. Examples of carbon emissions for the production of chemicals compared to mineral fertilizer and transport.

	Quantity					
Chemical	(kg/ton)	(kg/kg N)	Chemical carbon emissions rate (kg CO_2eq/kg)	Carbon emissions (kg CO_2eq/kg N)	Carbon emissions compared to ammonium nitrate	Ecoinvent dataset
Ammonium nitrate	—	—	—	8.17	—	Ammonium nitrate, as N {GLO}| market for | Cutoff, U
Lime	10	1.15	0.973	1.12	14%	Lime, hydrated, packed {GLO}| market for | Cutoff, U
Sulfuric acid	2.94	0.34	0.163	0.06	1%	Sulfuric acid {GLO}| market for| Cutoff, U
Transportation of urine on 10 km (8.7/1.8 g N/L)	—	—	—	0.19/0.93	2%/11%	Transport, freight, lorry 16–32 metric ton, EURO5 {GLO}| market for | Cutoff, U
Transportation of urine on 100 km (8.7/1.8 g N/L)	—	—	—	1.9/9.3	24%/114%	

We assumed a urine sample of 8.7 g N/L (Friedler et al., 2013). Data are extracted from Ecoinvent data base version 3 (Wernet et al., 2016). Urine is transported in a 16–32 tons freight lorry. We also assumed a case with non-concentrated urine of 1.8 g N/L (Etter et al., 2015) for transport. Sulfuric acid dose is 60 mEq H^+/L (Hellström et al., 1999) and 10 g/L for lime (Randall et al., 2016). Ammonium nitrate production (using the Haber-Bosch process) was taken as an example of mineral fertilizers (other fertilizers may have lower carbon footprint). Carbon emissions were assessed with the IPCC (2013) global warming potential values for 100-year time horizon method (IPCC, 2013; GWP 100a version 1.0.3) and using the Simapro software version 8.5.4.0 (PRé Consultants, 2017).

chemicals used to treat urine is not always negligible compared to that of mineral fertilizer. For example, adding 10 g of lime per liter of urine to stabilize it represents approximately 14% of the carbon footprint of ammonium nitrate with urine at 8.7 g N/L (the carbon footprint is much higher if 10 g of lime per liter are added to flush diluted urine). As for energy consumption, chemical consumption can potentially be a "hotspot" of environmental concern. All impacts associated with the chemicals have to be assessed, not only those related to climate change (e.g., toxicity to humans).

4. Synthesis and perspectives

4.1. Treatments

As summarized in the graphical abstract, the state of knowledge and technology readiness differs among the treatments ranging from laboratory pilot (e.g., acidification of urine or osmosis processes on urine) to already implemented value chains (e.g., neighborhood with source separation and storage). In order to more accurately assess the environmental impacts and the feasibility of value chains, there is a need for implementation at higher scales in order to increase the technology readiness level. Volume reduction posttreatments are less studied and a large number of them were not tested on urine in real conditions.

If urine is source separated, most of the treatments mentioned in this review have good potential for implementation depending on the local resources available (e.g., wood ashes for alkalinization) and the geographical context (e.g., distance between the toilet and the field). Most of them are adapted to different scales of implementation ranging from a single household to neighborhood (with the exception of some high-tech treatments such as nitrification). Volume reduction posttreatments appear to be necessary for large-scale implementations in dense urban areas and for the use of UBFs in conventional farming systems over a large area. However, their energy consumption is high. Efforts to reduce energy consumption should be a specific focus of research.

4.2. Fertilizer efficiency and associated impacts

UBFs have high fertilizer efficiencies, similar to mineral fertilizers, without high variability between them. However, none of the UBFs was largely studied and they need to be tested in different conditions and on different crops. Associated impacts and especially ammonia volatilization after application are expected to vary among UBFs and should to be investigated.

4.3. Contaminant removals

The trace metal content of UBFs is low, and the pathogens they contain can be effectively treated. The issue of pharmaceutical residues in UBFs is still under debate, and the health and environmental risks linked with their use in agriculture have to be assessed. Except the treatments specifically dedicated to pharmaceutical residues removal, the other treatments are not able to ensure high removal.

4.4. Environmental impacts assessment

All impacts of the value chains must be taken into account. Some environmental impact assessments were carried out with methodologies such as life cycle assessment (e.g., Remy & Jekel, 2008; Spångberg et al., 2014). In general, the environmental impacts of the use of UBFs are better than those of the use of mineral fertilizers and of the treatment of urine at the WWTP (e.g., Spångberg et al., 2014; Tervahauta et al., 2013). A few studies focused on the impacts linked to their use in agriculture (e.g., impact of spreading and ammonia volatilization). The insertion of UBFs in the technical cropping practices is also little studied (e.g., Tidåker et al., 2007). Furthermore, most environmental impact studies focus on stored urine and struvite and other UBFs were barely assessed.

4.5. Other issues

Some issues were not considered in this review but remain important for the implementation of value chains. For example, the acceptability of the use of UBFs by farmers or by consumers should be considered (e.g., Brun, 2018; Lienert & Larsen, 2010; Segrè Cohen et al., 2020).

The economic point of view is also important and difficult to take into account. A wide variety of economic system can be considered. An option could be the “waste” approach where the cost of treatment is supported by the “producer” of urine as for the wastewater treatment. Then UBFs may be given free of charge to the farmers such as sewage sludge. A second option is the product approach. Considering that some treatments may permit to reduce the contaminant concentration and guarantee the nutrient content, UBFs may be sold to farmers as a standardized fertilizer.

In conclusion, UBFs are interesting alternative fertilizers derived from a resource that is currently released to the environment. They could help to close the biogeochemical cycles and to reduce the impacts of agricultural systems and human excreta management systems on the environment.

Declaration of interest

No potential conflict of interest was reported by the authors.

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