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Life cycle assessment of sewage sludge treatment and disposal based on nutrient and energy recovery: A review

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Abstract

With the acceleration of urbanization, the production of urban sludge is increasing rapidly. To minimize resource input and waste output, it is crucial to execute analyses of environmental impact and assessments of sustainability on different technical strategies involving sludge disposal based on Life Cycle Assessment (LCA), which is a great potential environmental management mean adopted internationally in the 21st century. This review aims to compare the environmental sustainability of existing sludge management schemes with a purpose of nutrient recovery and energy saving, respectively, and also to include the substitution benefits of alternative sludge products. Simultaneously, LCA research regarding the emerging sludge management technologies and sludge recycling (cement, adsorbent, bricks) is analyzed. Additionally, the key aspects of the LCA process are worth noting in the context of the current limitations reviewed here. It is worth emphasizing that no technical remediation method can reduce all environmental damage simultaneously, and these schemes are typically more applicable to the assumed local conditions. Future LCA research should pay more attention to the toxic effects of different sludge treatment methods, evaluate the technical ways of adding pretreatment technology to the 'front end' of the sludge treatment process, and further explore how to markedly reduce environmental damage in order to maximize energy and nutrient recovery from the LCA perspective.

Keywords: Life cycle assessment; Sludge treatment; Energy saving; Nutrient recovery

Abbreviations

AA	Aquatic acidification
AD	Anaerobic digestion
ADP	Abiotic depletion potential
AET	Aquatic ecotoxicity
ALO	Agriculture land occupation
AP	Acidification potential
ARD	Abiotic resources depletion
BMT	Biological mechanical treatment
CC	Climate change
CED	Cumulative Energy Demand
CExD	Cumulative Exergy Demand
CG	Carcinogenic
coAD	Anaerobic co-digestion
DALY	Disability adjusted years
DM	Dry matter
DQG	Data quality goals
DS	Dry sludge
EDIP	Environmental Design of Industrial Products
EDW	Electro-dewatering
EP	Eutrophication potential
EQ	Ecosystem quality
ET	Ecotoxicity
FBC	Fluidized bed combustor
FE	Freshwater eutrophication
FET	Freshwater ecotoxicity
FFD	Fossil fuel depletion
FU	Functional unit
GHG	Greenhouse gas
GWP	Global warming potential
HAP	Hydroxyapatite
HH	Human Health
HPT	Hydrothermal-pyrolysis technology
HS-AcD	High-Solids Anaerobic Co-Digestion
HS-AD	High-Solids Anaerobic Digestion
HT	Human toxicity
HTA	Human toxic air
HTC	Hydrothermal Carbonization
ILCD	International Reference Life Cycle Data System
IR	Ionizing radiation

ISO	International Standards Organization
L-AD	Liquid-Anaerobic Digestion
LCA	Life cycle assessment
LCC	Life cycle cost
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LO	Land occupation
MAP	Magnesium ammonium phosphate
MD	Metal depletion
ME	Mineral Extraction
MEP	Marine eutrophication potential
MET	Marine ecotoxicity
MFA	Material flow analysis
MOE	Manual to Calculate and Report Emissions of Greenhouse Gases
MSW	Municipal solid waste
N	Nitrogen
NCG	Non-carcinogenic
NLT	Natural land transformation
ODP	Ozone depletion potential
OF	Organic fraction
P	Phosphorus
PAHs	Polycyclic aromatic hydrocarbons
PCAs	Polychlorinated alkanes
PCBs	Polychlorinated biphenyls
PE	Person equivalent
POF	Photochemical ozone formation
POP	Photochemical oxidation potential
PPCPs	Pharmaceutical and personal care products
PMF	Particulate matter formation
QMRA	Quantitative microbial risk assessment
R	Resources
RE	Respiratory effects
ROI	Return On Investment
SBAC	Sewage sludge-based activated carbon
SEA	Statistical Entropy Analysis
SP	Smog potential
SS	Sewage sludge
SSA	Sewage sludge ash
STW	Sludge treatment wetlands
TA	Terrestrial acidification
TE	Terrestrial eutrophication
TET	Terrestrial eco-toxicity

TS	Total solids
UASB	Upflow anaerobic sludge blanket
ULO	Urban land occupation
WDP	Water depletion Potential
WWTPs	Wastewater treatment plants

Highlights

- Recent progress in LCA research of sludge management scenario is reviewed
- Essential processes and parameters within the LCA framework are explained
- Sludge treatment technologies based on various orientations are presented
- Future LCA research directions and challenges are proposed

Content

1. Introduction

Wastewater treatment plants (WWTPs) are engineered and operated to reduce wastewater pollution originating from human activities to minimize damage done to the environment and people's health. Sewage sludge is a byproduct of the wastewater treatment process, from the primary sedimentation tank, secondary sedimentation tank and other linked processes. In order to avoid environmental damage, it is necessary to implement a series of intricate treatment and disposal procedures for wastewater sludge, such as, concentration, anaerobic digestion (AD), dewatering, thermal drying, incineration and landfill disposal. Improper management of organic waste can result in serious environmental pollution, such as, odor, disease transmission, and global warming (Singh et al., 2011). As there is not enough space for sludge treatment in sewage treatment plants, the problem of sludge treatment and disposal is becoming very serious, which is aggravated by the rapid urbanization and industrialization that has occurred over the last 30-40 years, especially in the developing world. The focus of sludge (biosolids) treatment is to minimize its mass and volume in order to cut down the expenditures of disposal, while minimizing any latent health challenges ascribing to disposal (Barry et al., 2019).

With the increasing global demand for renewable energy and organics, organic waste may become one of the most readily available resources. Sludge contains a tremendous amount of renewable organics and can be deemed a sustainable resource with economic potential (Spinosa et al., 2011), being in the form of nutrients or energy recovery, thereby producing potential value-added products based on sludge (Pradel et al., 2016). For example, phosphorus

(P) recovered in the form of magnesium ammonium phosphate (MAP) and biochar obtained through thermal treatment play the role of mineral fertilizer or soil amendment; methane from AD process or bio-oil from thermo-chemical process help to achieve energy self-sufficient, volatile fatty acids (VFAs) from sludge hydrolytic acidification is often used as a supplementary carbon source for biological nitrogen (N) and P removal process in wastewater, and bio-hydrogen from sludge fermentation is used as clean fuel. Pathogens, heavy metals and trace organic pollutants in sewage tend to accumulate in sludge, urgently desiring a blending of LCA with quantitative microbial and trace organic pollutants risk assessment (Corominas et al., 2020). However, this recognition of the value of nutritional sludge refers to stabilized sludge, which is often used as fertilizer and/or soil amendment for agricultural land (Singh and Agrawal, 2008). Sludge should be considered a renewable resource for an available energy and material regeneration (Tyagi and Lo, 2013)， as this will conform to the philosophy of “circular economy”. Controversies also exist in whether sludge should be regarded as a product. For now, sludge is a misplaced and wasted resource. Thus it’s necessary to explicitly clarify when the sludge is a waste and when it is a product, so that the environmental burden could be reasonably saved from the resulting sludge (Pradel et al., 2016).

To decide the treatment procedures suitable for each situation, not only the geographical location, socio-economic circumstances, but also the specified environmental regulatory standards and technical costs are supposed to be cogitated (Arroyo and Molinossenante, 2018; Marlow et al., 2013). To assess the environmental impact of products and services and to help achieve consistency in policy and environmental planning (ISO, 2006a; ISO, 2006b), LCA of sludge treatment and disposal has been extensively used over recent years (Guinee et al., 2011), which is reflected in the rapid increase of the number of publications and databases related to LCA (as shown in Fig. S1 and Table S1). LCA originated from the tracking and quantitative analysis of the whole process of beverage containers from raw material extraction to waste ultimate treatment processes commissioned by the Coca-Cola company in 1969 (Hunt et al., 1996). The LCA is a management tool with vitality and development prospect and is highly respected when making comprehensive assessments of complete chains of products or technologies from cradle to grave (Chen et al., 2019; Hospido et al., 2005). In the late 1990s, the intense demand for standardized LCA methodologies promoted the birth of LCA guidelines in the International Standards Organization (ISO) (Corominas et al., 2013). Since the beginning of this century, scholars at home and abroad have applied LCA methods to sludge disposal technology selection, and conducted a series of analyses and research on the environmental impact of sludge disposal. LCA of sludge disposal refers to the whole process of sludge disposal, including the input and output of all raw materials and energy from sludge generation to the collection, treatment and final disposal, followed by the identification and quantification of corresponding environmental emissions. Consequently, the assessment of the environmental impact at each stage is implemented, to select the sludge resource utilization technology that can minimize the environmental load.

Some reviews in the field of wastewater sludge treatment and LCA have come into public vision and caused tremendous repercussions. Yoshida et al. (2013) reviewed 35 studies on

LCA in respect to sludge research, and highlighted key techniques and methods, but did not investigate these studies in any great detail. In the review published by Pradel et al. (2016), they highlighted awareness on the adjustment of sludge status and subsequent LCA modeling if a transition from "waste" sludge to "product" sludge occurs. Another review briefly introduced some fundamental judging standards for the maximum realization of the circular economy "from waste to resources" concept (Kacprzak et al., 2017). Teoh and Li (2020) assessed and reviewed the breakthroughs of 67 studies published between 2000-2018 (consisting of 32 LCA-related literature reviews), using a semi-quantitative assessment methodology. In the review, the comprehensive capability in reducing sludge volume/weight and environmental impacts for various biological, chemical, thermal, and thermo-chemical sludge treatment methods were identified. In the past reviews published, much attention is paid to the model and methodology of LCA for sludge management strategies, and very few works focused on the LCA of sludge treatment and disposal based on guidelines for nutrient and energy recovery.

To this end, the current review will be structured as follows: 1) providing an overview of existing LCAs based on nutrient/energy recovery-oriented sludge treatment; 2) analyzing the development and main options of each LCA procedure (goals and scope, inventory, impact assessment methods and interpretation) to determine the common elements and differences; 3) evaluating nutrient/energy recovery-oriented sludge disposal technology and summarize development prospects; and 4) describing the main challenges and gaps identified. This paper largely centered around sludge treatment and disposal process, as we consider that this issue belongs to the waste management filed, not the field of waste water treatment.

Table 1

Fig. 1

Fig. 2

2. Review of research on LCA of sludge treatment and disposal

Fig. 1 shows the stages and framework of LCA research and the main process of sludge treatment and disposal in the studied literature. In this review, 37 studies on LCA of sludge treatment aiming at nutrient/energy recovery were subjected to statistical analysis. Keywords used for search on Web of Science include: "(life cycle assessment OR LCA) AND sludge". The literature on LCA for the production of cement, adsorbents and bricks based on sludge recycling is quite scarce and the research scope is relatively limited (see Section 5). They do not include the pretreatment process of sludge before transporting from the WWTPs, so most of them are not included in the statistical table (see Supplementary materials). The studies found were mainly published in the *Journal of Cleaner Production* (43.2%) and *Waste Management* (16.2%), during the last decade, 2010-2020. Table 1 presents the main characteristics of the research included in this review. More details can be found in the Supplementary materials. The setting distribution of each stage of LCA in these articles is presented in Fig. 2.

2.1 Definition of goals and scope

2.1.1 Functional unit (FU)

FU is a measure of the output function of the product system, and its basic function is to supply a reference benchmark for related inputs and outputs. FU is the crucial foundation that makes a simultaneous contrast and analysis of optional scenarios possible (Bonton et al., 2012; Rebitzer et al., 2004).

Most literatures (29 of 37 studies in Table 1) chose mass as FU, and 2 of the studies chose volume-based FU, as well as 2 of the other chose person equivalent (PE, the amount of sludge generated in a specific time period by one individual). The good outcomes provided by sewage sludge treatment can also be regarded as a FU, e.g. Liu et al. (2011) considered the 1 TJ of steam generated by sludge incineration as FU. In the previous LCA study of sewage treatment unit, the specified amounts of sewage or sludge mass are the most common FUs (Hospido et al., 2004; Pradel et al., 2016; Yoshida et al., 2013). Volume-based unit is the most commonly used unit in the LCA of wastewater, but it's probably unrepresentative for inability of showing the features of wastewater (Corominas et al., 2013). Some authors suggested that LCA studies should use more than one FU to analyze the results (Zang et al., 2015).

2.1.2 System boundary

Rigorous definition of system boundaries exerts a significant impact on LCA (Finnveden et al., 2009). The system boundaries of the studied literature include all the processes contained in the sludge management strategy.

Most studies do not consider the construction and demolition stages of treatment plants (31 of 37). Previous research has shown that the construction and dismantlement of WWTPs make only a negligible impact (Johansson et al., 2008; Lombardi et al., 2017; Lundin et al., 2004; Yoshida et al., 2013), and for long-term technical systems, the environmental impact due to construction is often less than the actual operation (Righi et al., 2013). Of the 37 studies, 23 considered vehicles, machinery, auxiliary equipment and the transportation process of sludge. Much research shows that the construction and transportation industries make significant contributions to environmental damage (Righi et al., 2013; Tarpani et al., 2020; Uggetti et al., 2011). Amann et al. (2018) stressed that when comprehensively analyzing the nationwide environmental impact of P recovery, further consideration should be given to the possible increasing transport demand of sludge. Considering the heavy metal and GHG emissions from transportation trucks, a large proportion of environmental impact may originate from these stages (Morero et al., 2017; Peters and Lundie, 2001). The transportation of sludge between different treatment plants produces a considerable amount of greenhouse gas (GHG), mainly in the form of CO_2. Lam et al. (2016) investigated the correlation between transportation distance assumptions and climate change (CC) impacts, and the default setting for the uniform distance between the sewage treatment plants and terminuses for disposal was validated to be lack of rationality for the environmental impact analysis. This was especially the case for multiple sewage treatment plants located in very different places. Meanwhile, the choice of transportation vehicles, engine efficiency, and fuel types for collection and transportation should also adhere to the actual conditions and policy requirements of specific locations (Gentil et al., 2010).

In most sludge LCA studies, the temporal horizon of the technical system is not clearly stated. Only 9 studies explicitly mention the temporal horizon. For example, the time horizon was stated as 30 years of facility operation in the research implemented by Lam et al. (2016).

In theory, there is a probable synergy between the change in time and the environmental impact of different schemes (for example, population size changes over time, annual government policies, and changes in the quantity of municipal solid waste (MSW) between 2020 to 2050) into the scenario storyline (Nakatsuka et al., 2020). In view of the extended life span of the plant, future uncertainties will seriously affect the sustainability of the environment and the economy; current research lacks a comprehending of which material or process might be decisive for future sustainability (Nakatsuka et al., 2020).

About half of the reviewed studies (18 of 37) included biogenic CO_2 in their inventories (see Table 2). With regard to the possibility of global warming, more accurate calculation of direct GHG emissions is essential. Indirect emissions refer to the emissions related to production and use of electricity, heat and steam, transport of secondary materials and even the administrative management for the plants, which may account for 10% of the CC, 40% of the ecosystem quality (Fallaha et al., 2009; Gentil et al., 2009). The physical and chemical forms of carbon often change during the metabolic process, causing carbon to migrate through environmental media such as air, water, and soil. Those consequent emissions may comprise various forms of carbon-containing compounds, i.e. CO, CO_2, hydrocarbons, volatile organic compounds and other organic compounds, which will cause a sequence of environmental problems such as CC (Cox et al., 2000). In the process like decomposition in landfills and combustion of biogas in cogeneration plants, sewage sludge is assumed to consist entirely of biomaterials, and CO_2 in biomass is considered climate neutral (Houillon and Jolliet, 2005; Lombardi et al., 2017; Meisel et al., 2019; Pawelzik et al., 2013). However, it was pointed out that not all GHG emissions should be regarded as biological emissions, because as many as 20% of the total organic carbon in wastewater might come from fossils (such as detergents, cosmetics and pharmaceuticals). Nonetheless, it is unspecified in the LCI which leads to an underestimation of the relative impact of indicator expression (Rodriguezgarcia et al., 2012; Zang et al., 2015). Comparing the GHG emissions results in the composting, AD, or incineration process, it emerged that the bio-based emissions play a momentous role (Piippo et al., 2018). The absorption and transformation of CO_2 helps to improve the GHG balance of the new process (Salomoni et al., 2011).

Eight studies did not expand the system boundary. Expanding the system boundary, means that the alternative products after sludge disposal, such as chemical fertilizer, electricity, etc., are included in the overall evaluation, considering the avoidance of impact of alternative products and positive benefits. System expansion, also identified as system replacement/substitution, is usually accomplished by treating the co-product as a replacement for other products in the market (Cao and Pawlowski, 2013; Righi et al., 2013). Such a type of sludge reuse is economically practicable, and its economic benefit is reflected in counteracting the expenses associated with conventional sludge management technologies, reducing the health expenses of waste disposal, reducing the energy requirement expenses by using biomass energy resources (which can partially replace traditional fossil fuels), and even sales revenue from excess heat, electricity, or fertilizer (Lee et al., 2020). The results of studies by Linderholm et al. (2012), Yoshida et al. (2013), Vadenbo et al. (2014), Lombardi et al. (2017), Yoshida et al. (2018), indicated that the system boundaries and substitution systems are the main source of sensitivity for the treatment schemes optimization. For the avoided burdens, methodological choices and assumptions for substitution production is vital in investigating thoroughly any potential benefits (Heijungs and Guinee, 2007; Weidema, 2000). There are challenges encompassing determining the character and amount of replaced raw materials or energy and the impressionable treatment processes. Researchers in this field usually convert the sludge amount to the mineral fertilizer amount with the same fertility efficiency based on the N and P content in 1kg sludge. However, different substituted fertilizer (such as N fertilizer classification: ammonium-N, nitrate-N and amide-N) may lead to different outcomes (Yoshida et al., 2018). For energy substitution, it is common and reasonable to adopt the national power mix data for specific study sites while the recognition of impressionable treatment processes is generally informed through relative contribution of life cycle steps (subsystems) to different impact categories.

The choice of sludge end use is also reflected in the objectives and scope of the study, but there are three studies that lack a description of subsequent disposal, for example disposal or reuse of incineration ash and the disposal of sludge after dehydration.

2.2 Life cycle inventory (LCI)

Inventory analysis sets out to analyze the energy and material requirements, pollutant emissions and environmental hazards produced by raw materials mining, refining, product manufacture, transportation, sales, consumption and disposal. Inventory analysis needs to process huge data, and must use LCA software or build a programming algorithm for calculation. It is important to collect, analyze the data for the manufacturing, use and waste of the products. These are generally referred to as the foreground data, and this is followed by collecting the raw material data used for the products. It includes the data concerning the amount of power and fuel used in mining raw materials resources. This phase is generally referred to as obtaining the background data. Since it is difficult to gather this type of data, it is usually stored on a LCI database, e.g. Ecoinvent (Corominas et al., 2013).

2.2.1 Inventory data analysis

The sources of inventory data in the literature can be classified into 7 categories: field collection; reference; database; statistical yearbook or report (government or business), calculation/simulation/calculation, experiment and interviews with experts (see Table 1). When referring to official documents or other sources for list data, it is necessary to follow up on the latest updates on relevant process parameters in a timely manner. This should be followed by the geographical selection involving similar regions as far as possible, due to the large differences in environmental parameters caused by the different geographical locations of some larger countries. This is with respect to the selection of national or regional official environmental rules/legislation/regulations that refer to the relevant documents of the next level administrative region as far as possible.

LCI generally plays an essential role in LCA analysis. Restrictions on the source, region and time of inventory data affect the quality of data, while the quality of data affects the uncertainty of LCA results. The geographically represented data and temporal definition for whole procedures is crucial for effectiveness of LCA (Peereboom et al., 1998; Su and Zhang, 2016; Xu et al., 2014). Directly measured values are often used in conjunction with references and official reports as surrogate data for materials, chemicals, and energy requirements without collecting adequate information and assessment of their underlying incompatibilities (Gallego-Schmid and Tarpani, 2019). Average data is frequently applied to the background system as well, e.g. the product systems like fossil fuel originally estimated to have no effect on the investigated system, have been proved that they are imperative for its normal running and comprehensiveness of the evaluation. This mix of methods means that the LCA consequences will not be an accurate measure of how each method impacts on the global environment (Heimersson et al., 2019).

As shown in Table 1, 15 studies did not explicitly indicate the databases used, while 19 studies used the Ecoinvent database. The current LCA research background data in many countries is still based on databases developed under conditions in Europe and North America. The insufficient specific background data that reflects local conditions is bound to severely impair the accuracy of the evaluations. Fernandez-Gonzalez et al. (2017) remarked that the choice of multiple data sources, along with the fluctuation in the market value of the raw materials and energy structure, would alter the prioritization of scenarios across diverse studies. Previous studies regarding agricultural products stated that results showed variations in HT (0.02-0.18%), FET (89-99%) and TET (8006-26177%) between those models with and without regional emission information (Kim et al., 2015). There is little attention paid to the numerical and methodological differences in existing databases associated with sludge management so far, which should be future hotspots. Improving the integrity of site-specific databases, characterization parameters or normalization and weight values will greatly improve the accuracy of WWTP-related LCA (Gallego-Schmid and Tarpani, 2019).

A total of 20 papers have provided relatively complete sludge components (such as water content, dry matter, volatile solids, calorific values, etc.). In recent years, much research has focused its attention on the environmental impact and energy efficiency of sludge-energy pathways. They have done so by investigating and evaluating different types of feed sludge organic matter content (Li et al., 2017a; Li et al., 2017b).

A clear understanding of the accurate physicochemical property of the sludge helps to increase the possibility of using digester or compost and the final product safely and effectively (Ahmad et al., 2016). The physical and chemical parameters of sludge play the main roles that lead to different evaluation outcomes. In the process of converting sludge to energy, such effect is even more pronounced. The energy conversion capability of sludge highly depends on its organic content, while its content also presents a state of fluctuating between 30% -80% (Li et al., 2017a). With the increase of organic matter in sludge, the C/N ratio increased, which effectively improved the degradation rate of organic matter and methane production in AD process. High N content results in the transformation of excess N into free ammonia-N, and high C/N caused by low content of N can easily prompt VFAs accumulation, both of which inhibit the activity of methanogens (Chen et al., 2008; Rajagopal et al., 2013). In terms of thermal treatment like incineration, dewaterability and biodegradation of sludge is limited to organic content, and the self-sustained combustibility and calorific value of MSW is inversely proportional to moisture while proportional to organic matter (Komilis et al., 2014), which cause the distinction in energy recovery efficiency and GHG emissions.

Less GHG emissions and human toxicity (HT) could be realized by the handling with a low moisture of sludge. However, the results revealed that when the toxicity arising from the dewatering process was considered, the total toxicity of the treatment with 60% water content is higher than that of the treatment with 80% water content (Lishan et al., 2018). Therefore, the dewatering method and water content of the sewage treatment plant shall be determined according to the environmental, social and economic conditions involved in the subsequent treatment process. It was emphasized by Lee et al. (2020) that the reliability of LCA consequences is significantly affected by the real waste features, since the findings for the environmental impacts of two different compositions were diverse. Rostami et al. (2020) also asserted that the sludge characteristics appear to be crucial for discrepancy between the LCA environmental impacts results for the two different WWTPs.

Five processes encompassing mesophilic and thermophilic AD (CAD and TAD), mesophilic and thermophilic high-solids AD (HSAD and THSAD) and AD with thermal hydrolysis pretreatment (THPAD) were analyzed. This was executed by adopting LCA in terms of environment and economy in the study of Li et al. (2017b). Their results showed that thermophilic processes such as THSAD and TAD have the least environmental impact on common high-organic-content sludge, while THSAD and THPAD are the most economical. For sludge with low-organic-content, high-solids processes such as THSAD and HSAD are superior to other processes. The explanation for this lies in their lower heat energy consumption. Energy output touches the most sensitive nerve of the assessment results (Li et al., 2017b).

2.2.2 Data quality goals (DQG)

DQG refers to the specific data target according to the data characteristics required by the research object. In the stage of target and boundary determination, data quality target needs to be determined, and DQG can guide data collection. Data quality should meet the following criteria:

(1) Time span: the required data should not be too far from now, e.g. 4 years.

(2) Geographic scope: geographic coverage of unit process data, such as local, regional, national, continental, global.

(3) Accuracy: the degree of variation in values in each data type (such as variance).

(4) Repeatability: a qualitative assessment of the possibility that other researchers or institutions engaged in LCA can obtain the same research results based on the reported data and methods.

2.3 Life cycle impact assessment (LCIA)

LCIA is the stage of processing the LCI results and generalizing them as environmental impacts. The main purpose of LCIA is to measure the extent to which different products, processes or activities have a comparative impact on the environment or human, not to investigate the absolute damage. This stage also enhances the relatedness and interpretability of LCI for project participant.

Fig. 3 presents the impact categories used in 37 reviewed papers and the number of studies that included each impact category. The most common impact category considered was the global warming potential (GWP) in kilogram CO_2 equivalent, which was included in 35 reports. Acidification potential (AP), human toxicity (HT), photochemical oxidation potential (POP) and ozone depletion potential (ODP) were also common impact categories, reflecting the public concerns about sludge treatment. Most researches centered around one or several predetermined midpoint impact categories, and only 5 of these studies considered assessing the impact of end point damage.

Midpoint and endpoint methods are the specialized approaches applied to LCIA. Endpoint approach stipulates more data integrality, weighting, modeling and value choices to execute a comprehensive environmental assessment, thus owning less reliability than the midpoint method (Goedkoop et al., 2009; Van Hoof et al., 2013). However, considering a better comprehension and assessment of the end impact in various schemes for decision-makers and stakeholders, the endpoint method is more preferable. Practitioners can quantify damages/effects with the most relevant damage indicators at the terminal of the causal chain, i.e. Human Health (HH), Ecosystem quality (EQ), Resource (R), which are the highly aggregated consequences of all midpoint impact categories (Bare and Gloria, 2006; Corominas et al., 2020). It's more advantageous for regulators to use endpoint methods to determine those ultimate impact of regulations and explain this to the wider society (Bare et al., 2000).

Even if the midpoint approaches are effortless to implement, they have complexity in assessing impact, e.g. reasonable selection of impact categories. Some bias would emerge during midpoint characterization and normalization, thus bringing about more uncertainties to midpoint method (White and Carty, 2010). It's also claimed that midpoint method has advantages of greater comprehensiveness and modeling certainty, considering the precautionary principle and giving an extra weight to uncertain aspects (Bare, 2009; Finnveden et al., 2009).

2.4 Resourcization of output

As shown in Fig. 1, the output of sludge treatment and disposal includes not only the discharge of various pollutants, but also nutrients and energy with recovery value, and those solid waste can also be used in the production of building materials. 15 of the 37 articles evaluated the various technologies related to sludge nutrients or energy recycling. The number of LCA articles based on sludge value-added by-products (fertilizer, heat, electricity, bio-char, adsorbent, cement, brick) has been increasing with year, implying great social interest in this area, and the comparative advantages and disadvantages revealed in them are non-negligible.

Fig. 3.

3. Nutrient recovery orientation

The main characteristics of the literature related to the LCA on sludge nutrient recovery are shown in Table 2. More details can be found in the Supplementary materials section. There have been 19 studies that extend boundaries to include the avoided environmental impacts by sludge products substituted for chemical fertilizers. Technical systems for each reference study are described in the Supplementary materials.

3.1 Technology systems

3.1.1 Soil application

One option for carbon and nutrient cycling closure is to apply sewage sludge to agricultural land. Landfilling, incineration, agricultural application and substitutive fuel in industrial processes have been the most frequently applicable disposal scenarios for sludge management over the past decade, while considering the waste hierarchy and the valuable N, P and organic matter in sludge, the use in agriculture is preferred (Rovira et al., 2011). Although there are some decrease in GWP and AP categories for less equipment production, electricity consumption, landfill schemes (sludge of 80% water content) performed a more prominent harm to economic and environmental impact than the agricultural application routes (sludge of 10% water content) (Hong et al., 2009). As reported by Jeffery et al. (2011), land use of biological solids is commonly evaluated as a worthy disposal way, attributing to its ability of recycling materials and increasing crop productivity by enhancing nutrient availability and water holding capacity together with lime effect.

Recycling organic substance from waste residues during production and living into the farmland will benefit the sustainable development of agriculture over the long-term operation (ÖZYAZICI, 2013). Composting of municipal sludge and organic matter from other industrial chains, such as MSW (Liu et al., 2009), woodchips (Zhao et al., 2015) and kitchen waste (Righi et al., 2013), has long-term development prospects due to their complementary strengths. A percentage of the organic carbon (C) in sewage sludge could resist biodegradability, resulting in the deposition of C in the land, so that the application of sewage sludge in farmland not only provides essential nutrients for plant growth but assists in the mitigation of CC. Meanwhile it improves soil quality and reduces environmental impacts, so the economic potential is evident (Alvarenga et al., 2015; Woolf et al., 2010).

Yoshida et al. (2018) highlighted that site-specific climate and geology have a significant impact on the assessment results of sludge on land use. This is due to the reason that the emission coefficient depends on sludge characteristic and local conditions. Although the land use of sludge has certain economic and biological outcome, a multitude of treatment technologies engender various sludge products with different composition, characteristics and quality, thus it's imperative to simulate and provide the long-term effect before applying sludge to soil (Bruun et al., 2016).

3.1.2 Phosphorus recovery

In the developed countries with sewage and wastewater treatment infrastructure, sludge treatment has evolved into the primary approach of P recovery. The P content in sewage sludge is usually utilized by distributing sludge directly to farmland. Research results showed that the DALY value of recycled products (HAP, MAP, calcium phosphate, ash fertilizer) was smaller compared with inorganic fertilizer (phosphate rock fertilizer) (Lederer and Rechberger, 2010). The direct use of P-rich sludge after stabilization involves reductionist recycling techniques, but because of the possible existence of heavy metals, persistent organic pollutants and pathogenic bacterium, it is prohibited in many countries (Harrison et al., 2006).

Previous work has shown the further benefits of P recovery, namely reducing the possibility of eutrophication (reducing phosphate mining), thereby reducing the discharge of phosphate water from mining (Remy and Jossa, 2015), curtailing cadmium and uranium introduction into farmlands (Bigalke et al., 2017), minimizing heavy metal pollution compared to traditional agricultural sewage sludge applications (Lederer and Rechberger, 2010), and reducing N discharge from N recovery processes (Johansson et al., 2008).

Among the optional schemes of sewage sludge and food waste resource utilization technologies for P recovery, the pyrolysis gasification of sludge and kitchen waste mixed digestate, combined with MAP method and alkaline extraction from ash command an advantage over the others in terms of GHG emissions, P recovery and health risks (Nakakubo et al., 2012). According to the study of ten Hoeve et al. (2017) using PLCI model to evaluate the LCI factors of waste products applied to farmland, the initial P content of the soil largely determines the P fate of waste products and mineral fertilizers. Different types of fertilizers applied to soils with low P content had little effect on P fate. LCA-based P substitution and loss studies have demonstrated that agricultural applications of waste are significantly associated with impact categories such as freshwater eutrophication (FE), CC, and reserve-based abiotic resources depletion (ARD). Amann et al. (2018) analyzed the primary technical approaches aiming at P recovery from liquid phase, sewage sludge or sewage sludge ash (SSA). They pointed out that recovery from SSA seems to have brightest prospects from environmental impact perspective, because it can achieve a high recovery rate, possible heavy metal purification, non-existing organic micro pollutants, and positive results in reducing gas emissions and energy demand. It is proven that compared with the widely used post-digestion recovery technology, any proposed configuration to improve sludge management prior to AD is economically and environmentally feasible. Efficiency is also noted with diminished uncontrolled P-precipitation, life cycle cost (LCC) and global warming impact (Roldan et al., 2020). Meanwhile, there is a standpoint that the positive benefit of P extraction from sludge cannot offset the environmental burden. Thus, the optimization of P extraction technology should be explored (Oliver-Tomas et al., 2019).

3.1.3 Nitrogen recovery

Steffen et al. (2015) emphasized the large possibility of environmental damage from natural N cycle abruption due to the input of industrial fixed N into the ecosystem, which enormously go beyond the limitation of the Earth. Reutilization of N which has been fixed into an active state and imported into biological N cycle, could alleviate the redundant N fixation (Deviatkin et al., 2019). Thermal drying followed by incineration of sewage sludge leads to a certain removal of the vital nutrient element accelerating the eutrophication of water– N. Thus Deviatkin et al. (2019) proposed the reclamation of N separated in the process of thermal drying of MSW, of which the GWP diminished up to 28% contrasted with chemical fertilizer manufacturing with similar features. It is noted that N mineralization in soil not only offers the opportunity to be absorbed by plants but also incurs risks of leaching to hydrosphere (Basso and Ritchie, 2005).

3.2 Human health and environment impacts

The diffusion effect of sludge on land wields a great influence on the toxicity index of human and ecosystems, as well as acidification and eutrophication. Composting the sludge before it is used on agricultural land can slightly reduce these impacts, but of course, this option will increase resources consumption (Lombardi et al., 2017). The significant environmental risks source of composting largely lie in eutrophication and acidification caused by ammonia-N release and the diffusion of heavy metals into ecosystems (Zhao et al., 2015). Considering compost emissions, heavy metals changed almost all toxicity-related categories and dramatically accelerated the eutrophication of fresh water. Therefore, before using compost produced from agriculture, it's imperative to be informed about the accurate amounts of nutrients and heavy metals which can be efficiently absorbed by vegetations or diffused in land and water (Rostami et al., 2020). From an environmental point of view, the composting scheme may be a reasonable choice for sludge treatment, but eutrophication is its research focus. Application of advanced processing techniques to separate phosphorous and nitrogenous compounds could mitigate this adverse impact (Gallego et al., 2008). It's also feasible to apply the compost for the reclamation in the greenbelt of cityscapes (Zhao et al., 2015).

Nonetheless, the continuous employment of sludge with simplistic treatment on soil have not been attractive to various countries in the world because of the high environmental impacts attributed to the anticipated existence of heavy metals, causative agents, newly discovered microplastics (Sun et al., 2019) and pharmaceuticals (Carballa et al., 2004; Petrie et al., 2015). Sludge contains a lot of nutrient elements, which can be used as plant fertilizer. However, sludge also contains a range of pollutants, including inorganic pollutants (heavy metals etc.) and organic pollutants (PAHs, PCBs, absorbable organic halogens pesticides, surfactants, hormones, drugs, nanoparticles and many other pollutants etc.) (Kacprzak et al., 2017). Niero et al. (2014) investigated 460 WWTPs in Denmark and observed significant HT and ecotoxicity (ET) due to the land use of sludge. The plant intake of heavy metals and organic pollutants in sludge land use is toxic to human. For example, chromium (Cr), mercury (Hg), lead (Pb) and zinc (Zn) were found to be the most dangerous metal elements with the most impact on whole population (Harder et al., 2016). The carcinogens posing the most risk to humans in terms of toxicity are predominantly Hg and Pb (Yoshida et al., 2018). In the environmental effects review of sludge toxins, the increase of persistent toxins in soil, wild fauna and flora as well as the decrease of biodiversity can be found during sludge soil applications (Manzetti and van der Spoel, 2015).

Potential negative impacts of sludge land application on the environment include: heavy metal (Tarpani et al., 2020); chemical pollutants (Mattara et al., 2014); pharmaceuticals and personal care products (Verlicchi and Zambello, 2015); excessive nitrate leaching (Urbaniak et al., 2016); impacts on soil biodiversity (Manzetti and van der Spoel, 2015); and GHG emissions (Yoshida et al., 2018). Landfill or agricultural land use can cause CC, principally because of methane and N_2O emissions during field nitrification, and denitrification (Lederer and Rechberger, 2010; Willen et al., 2016; Yoshida et al., 2018). The possible HT of compost in agricultural soil and forestry soil revealed that the human health risk of using compost for landscaping is smaller and more acceptable (Zhao et al., 2015).

3.3 Resource saving (chemical fertilizer substitution)

The assumptions about the surrogate products systems replaced by recovered byproducts were essential for the improved results, based on the results of the sensitivity analysis which were reported by Vadenbo et al. (2014). Rapid expansion of industrial manufacture for chemical fertilizers has a negative impact on CC, acidification and eutrophication due to rising emissions of pollutants (N_2O, NO_x, NH_3, PO_4-P) (Skowronska and Filipek, 2014). The positive impact of the recovery of organic components in sludge lies in the fact that certain products are avoided and can outweigh the problems associated with the sludge management system. Fertilizer replacement can counteract FE caused by soil use of sludge, and dehydrated digested sludge can reduce the release of active N into the environment (Tarpani et al., 2020).

The environmental benefits for the avoided use of chemical fertilizer, which is replaced by sludge or digestate, originate from avoided manufacture of mineral fertilizer and emissions from the use of chemical fertilizer on soil (Yoshida et al., 2018). In general for phosphate fertilizers, triple superphosphate or urea are assumed as being replaceable, whereas for N fertilizers, ammonium nitrate is assumed to be replaced (ten Hoeve et al., 2017). The magnitude of the avoidance effect depends on the N content of the composting material, and the N content determines the amount of synthetic fertilizer substitution: the higher the N content in the compost, the higher the savings in pollutants generated by the avoided fertilizer (Righi et al., 2013). Environmentally friendly sludge P recovery technology is feasible. As long as the price of primary P can be increased and energy efficiency can be improved, it is possible to achieve the economic realization of this technology (Lederer and Rechberger, 2010). The use of final products not only makes the circular economy possible, but also facilitates the environment due to the avoidance of emissions from the production, transportation and application of chemical fertilizers and enhanced soil fertility (Thomsen et al., 2017). The saved N_2O emissions from the utilization of mineral fertilizer and avoided CO_2 from the manufacture of mineral fertilizer contribute to reduction in GHG emissions. This means the digested sludge may have the preferred lowest eutrophication level and highest resource recovery rate (Yoshida et al., 2018).

There is still some uncertainty about whether artificial fertilizers and biosolids produce the same amount of methane and N_2O emissions, especially when the application rate of biosolids is different (Chiaradia et al., 2009). Different choices concerning avoided fertilizer lead to different outcomes. For example, other type of chemical N fertilizers present minor environmental impacts (apart from the ionizing radiation impact of urea), compared to the calcium ammonium nitrate for 1 kg N production (Gourdet et al., 2017). Furthermore, the uptake, runoff, stripping and seepage rate for nutrient in the sludge-based fertilizer should also be taken into account (Corominas et al., 2020; ten Hoeve et al., 2017).

Emphasis of future research should be placed on the flow and form of heavy metals in sludge applied to farmlands, as well as figure out the definite resource recycling and utilization potentiality based on the technical restrictions of management approaches, and the effect of environmental protection (Tarpani et al., 2020).

4. Energy recovery orientation

The main characteristics of the literature related to the LCA of sludge energy recovery are shown in Table 2. More details can be found in the Supplementary materials. There have been 25 studies that extend these boundaries to include avoided environmental impacts through the processes of targeted energy recovery.

4.1 Technology systems

The conversion of sludge into energy is a crucial part of a sustainable sludge management strategy. As a promising biomass energy, sludge help to enlarge the proportion of renewable energy in the energy structure, as well as diminish GHG emissions by reducing dependence on imported fossil fuels. An investigation into MSW treatment technologies in 13 Spanish municipalities, concluded that any MSW-to-Energy technologies bring more prominent benefits than Biological Mechanical Treatment (BMT) from the environmental and economic standpoint (Fernandez-Gonzalez et al., 2017). Scores of environmental benefits can be obtained by energy produced from sludge, including reduction of GHG emissions, acid gas emissions, depletion of natural resources (fossil fuels and raw materials), water and soil pollution. In comparison with landfill involving energy recovery, the research convinced that landfill scheme of MSW has substantial negative effects on ecosystem, CC, human health and resource damage, which prominently beyond other energy-friendly schemes, attributing to disorder emission of landfill gas, formation of acid-forming compounds and disadvantages of avoided production (Fernández-Nava et al., 2014).

4.1.1 Incineration

Incineration technology makes it possible for a great number of sewage sludge to be reduced and effectively converted into energy. Collaborative plans between WWTPs and other industrial production sectors like incineration plant or power station are feasible and prospective. However, the level of energy recovery may be conclusive for its environmental performance.

Xu et al. (2014) investigated the technologies encompassing landfill, incineration and AD of SS and the results confirmed that incineration creates the least environmental damage.

The results of LCA by Liu et al. (2011) revealed that co-combustion of coal, municipal sludge and refined oil was beneficial in reducing GWP. If sludge drying process can be achieved by solely utilizing the heat from the fuel, with the recovery of sludge energy, its EP will continue to decline, however AP and HTA will continue to deteriorate. It appears that when the replacement ratio of coal and sludge is 14%, the overall environmental impact potential attained by weighting the four indicators (GWP, AP, EP, HTA) become smallest (Liu et al., 2011). Research results demonstrate that the complementary synergetic effects exist in the co-incineration process of MSW and sewage sludge, led to the most positive outcomes for CC and resources, energy efficiency and profit (Chen et al., 2019). The co-incineration process could regenerate plentiful electricity and heat, meanwhile consuming fewer non-renewable energy, and fully take advantage of the surplus heat to dry the sludge. In turn, utilizing the bottom recycled slag has helped to realize the production of building materials (Chen et al., 2019).

Nakakubo et al. (2017) conducted a comparative evaluation of the separate treatment or co-incineration of MSW and AD sludge. The results confirm that the co-incineration plant decreased CO_2 emissions by 18% in comparison with the separate treatment plant, mainly due to the skipping of some sludge pre-treatment technologies. The outcome is identical to the conclusion illustrated by Nakatsuka et al. (2020), who have identified the positive potential of integrating WWTPs and incineration plants to reduce CO_2 emissions (35%). These authors also determined that the disposal amount of MSWs and efficiency of electricity generating act as decisive parts in the sustainable utilization of municipal biosolids.

With reference to reducing the impacts on high-water-content sewage sludge, incineration confronts technical and economic challenges. Dehydration of the sewage sludge normally consumes more heat than incineration can produce, which means that the process of sewage sludge incineration cannot realize energy self-sufficiency. As recognized by Syed-Hassan et al. (2017), in the case that sewage sludge was dried to 40%, its performance was close to that of 10%, with minor economic and environmental loads.

4.1.2 Anaerobic digestion (AD)

It has been widely reported that AD exhibits good environmental performance. The integrated treatment process of AD and agricultural use also present better environmental adaptability due to their less emissions and energy depletion (Suh and Rousseaux, 2002). Houillon and Jolliet (2005), Dong et al. (2014) and Li et al. (2017a) also discovered that AD is able to lessen the environmental damages of succeeding incineration and cement production process.

Chiu and Lo (2018) found that anaerobic co-digestion (coAD) treatment of sewage sludge and food waste seemed to be most environmental friendly but landfilling brought about the highest environmental burdens. Morero et al. (2020) also reported that the coAD of sewage sludge (SS) and the organic fraction (OF) of MSW with an original mixing ratio (40% OF - 60% SS) generates the smallest effect on the environment in nine impact categories.

There was research that compared High-Solids Anaerobic Digestion (HS-AD) of biosolids with Liquid-Anaerobic Digestion (L-AD) and revealed that HS-AD brought more environmental and economic optimization than L-AD (Li et al., 2017b). The study also showed that coAD system had the least environmental impact when compared to the separate disposal system of food waste and sewage sludge, as the benefits of generating more bioenergy and diverting waste from landfill exceeded the effects associated with additional collection and pretreatment processes (Edwards et al., 2017). Lee et al. (2020) examined the environmental and economic disadvantages and advantages of High-Solids Anaerobic Co-Digestion (HS-AcD) of biosolids, food waste, and yard waste. The results showed that HS-AcD caused the least environmental pollution in each investigatory category (GWP, AP, EP, and ET) and the minimum LCC, whether or not the cost of land expropriation was accounted for. The study of do Amaral et al. (2018) regarding an upflow anaerobic sludge blanket (UASB) which can effectively separate sewage gas and sludge, identified that employing biogas produced in UASB reactors to dry sludge provides an effective approach to curtail environmental burdens in all selected categories.

Nevertheless, there is huge energy consumption for giant biogas plants operation, breeding enormous potential for AD integrated with thermal technologies to actualize higher recovery rate (Dussan and Monaghan, 2017). Simultaneously, it's worthy to keep track of state-of-art sludge pretreatment technologies (chemical, mechanical, thermal, biological and electrical (Wang et al., 2017)) with capacity of elevating biogas yield in AD (Carrere et al., 2016).

4.1.3 Pyrolysis

Pyrolysis is a leading-edge process equipped with superiority of converting sludge into biofuels (bio-char, biogas or bio-oil) (Roy and Dias, 2017). Additionally, pyrolysis process at

350°C–800°C stimulates the thermal degradation of hazardous substances into gases and liquids without an oxidizing agent (Jahirul et al., 2012), thus ensuring the sanitary safety of the sludge (Marazza et al., 2019). It has been proven that the biochar produced from pyrolysis and used as soil amendment possess a double-edged sword effect of efficient nutrients, less available heavy metals and peril of excessive specific heavy metals deposited in plants (Faria et al., 2018).

Compared with the simplified system without the pyrolysis process, the sludge-energy system of combined AD and pyrolysis can not only generate enormous net energy, but also reduce GHG emissions and ET (Mohammadi et al., 2019a). While Li and Feng (2018) identified that the pyrolysis process produced greater environmental damage than the AD technologies. And even when pyrolysis combined with AD, the total environmental impact of the integrated system barely made any improvement due to the lessened benefit from the excessive heat or electricity production, and increasing demand for energy and auxiliary reagents during thermal drying and pyrolysis process. Recent research by Marazza et al. (2019) listed that the main differences between their model and the model proposed by Cao and Pawlowski (2013), Mills et al. (2014) and Li and Feng (2018), originated from VS/TS ratio, moisture of the input sludge, the productivity of the AD system, AD heat losses, moisture of the feedstock entering dehydration and dry equipment. They recommended to consistently mention the data list in order to make more meaningful comparisons between different studies.

4.1.4 Hydrothermal Carbonization (HTC)

Recently, considering the low cost and high resource recovery rate of HTC process, it has been considered as a quite hopeful technology for processing wet biomass for value-added products. HTC is defined as a thermochemical process used to convert organic biomass into carbonaceous product (hydrochar) in the presence of water under moderate temperatures (180–350℃) and pressures (2–10 MPa) (Mumme et al., 2011). Like sewage sludge, hydrochars can be applied for energetic use and soil amelioration because of the increasing dewaterability of SS (Kim et al., 2014) and advantages of biological sterilization. Regarding the subsequent disposal of hydrochar, compared with the agricultural use, the energetic use are usually more favorable for less additional GHG emissions (Meisel et al., 2019). However, it was observed in some studies that HTC-char decomposed rapidly and accelerated emissions of GHG to the air in the process of land application (Andert and Mumme, 2015; Schimmelpfennig et al., 2014).

4.2 Human health and environmental impacts

4.2.1 Incineration

Incineration is a common technology nowadays, but it’s not easy to operate in regions with small populations because it relies on plenty of waste biosolid (100,000 tons per year) (Fernandez-Gonzalez et al., 2017). Noticeable advantages of incineration processes could be observed. Incineration process can significantly reduce the total volume of sludge, of which the regenerated energy can decrease the environmental impacts with high treatment efficiencies and small floor areas (Xu et al., 2014). At the same time, however, the direct emissions of heavy metals from the incineration process make an important contribution to ET.

The most significant hazards of incineration contribute to resource depletion and human health (Liu et al., 2011; Lombardi et al., 2017). These effects mainly derive from the utilization of electricity, fossil fuel and combustion emissions. To avoid the secondary pollution during incineration process, pollutants in flue gas and related waste water require strict cleaning technics, otherwise would prominently diminish the pollution removal outcomes (Wen et al., 2019). Therefore, it's imperative to analysis the secondary pollution in sludge incineration process. In the incineration process, N_2O emissions caused by the combustion of sewage sludge high in N shall be mainly responsible for gaseous pollution (Chiu et al., 2016). Reducing the consumption of fossil fuels during the incineration process, and the combined treatment of sludge and other wastes with higher energy content will facilitate overcome or minimize damage done to the environment (Lombardi et al., 2017).

GWP or CC potential of incineration process is mostly proposed in the literature as one of the paramount impact categories, while it will be significantly reduced when electricity is generated by recycling afterheat (Abuşoğlu et al., 2017; Piao et al., 2016). It should be noted that the impact of FE potential will not be improved even if power generation is considered in the assessment (Buonocore et al., 2018). The comparison of the results in the study by Rostami et al. (2020) confirmed the large contribution from heavy metals in a majority of toxicity-related impact categories, which showed the importance of ash management to the pollution control of the incineration technologies. In addition, energy consumption and direct discharge of processes are the main factors influencing the performance of sludge treatment devices using incineration (Alyaseri and Zhou, 2017).

Reducing the overall environmental impact of sludge co-incineration power plants requires improvements in net coal consumption efficiency, ash recycling rate, dust removal system efficiency and sludge moisture content (Hong et al., 2013). The reusability of ash is one of the keys to reduce the overall environmental impact. To avoid any potential health risks, Hong et al. (2013) recommended using incineration ash in road construction, especially ash-containing sludge due to its high heavy metal content.

4.2.2 Anaerobic digestion (AD)

The widespread use of AD is driven by its energy-related benefits, i.e. the formation of biogas with high calorific value. Major components of biogas are methane and CO_2, which lead to the realization of combined heat and power production. Consequently, the generated energy is normally employed in the power treatment facilities or delivered to a regional power grid (Gourdet et al., 2017). The positive impact for AD include all 18 categories in ReCiPe midpoint method except for CC due to direct GHG emission and credit for by-product, which was in stark contrast to the scenarios without AD (Xu et al., 2014).

AD process is mostly carried out in fermentation tanks, which can effectively degrade organic waste and avoid secondary pollution to a large extent. It has a relatively low environmental impact in terms of whole ecosystems, but the purification technologies for AD sewage serve as a pivotal role to guarantee the pollution control effects (Wen et al., 2019). Hospido et al. (2010) demonstrated that regardless of operating conditions, sludge digestion can reduce the toxic effects on human health and land by one-third. Owing to the biogas power generation, the AD scheme helped to mitigate environmental damage and functioned much better than other methods from the perspective of fossil fuel and metal consumption. However, some research indicated that considering the expected existence of heavy metals in sludge, the harm of introducing heavy metals into farmland would exceed the benefits of avoiding fertilization (Alyaseri and Zhou, 2017).

4.2.3 Sludge pyrolysis

During pyrolysis, the sludge is converted into pyrolysis gas, oil and char to play a role as internal energy source. Although the sludge loses the benefit of avoided fertilizer production for agricultural application, but also avoids the toxic impact of heavy metal accumulated in soil, plants and animals. The research pointed out negative impact for pyrolysis on EP, GWP and AP and positive impact on HT and TET, compared with AD sludge used for land spread (Hospido et al., 2005). However, Tarpani et al. (2020) were convinced that only when high resource recovery rate is attainable, is the application of thermochemical processes (pyrolysis and wet air oxidation) beneficial to the environment. Utilizing pyrolysis products as a fuel/material substitute can reduce GHG emissions. It should be noting that when these products replace fossil fuel, the contaminants stored in the bio-oil could escape during the combustion process (Teoh and Li, 2020).

In the research done by Alyaseri and Zhou (2017), which compared the fluidized bed incineration with AD, the authors concluded that if the focus is on human health, fluidized bed incineration is recommended. If the focus is on resource depletion or general sustainability without specific emphasis then AD would be recommended.

4.2.4 Hydrothermal Carbonization (HTC)

The research by Mohammadi et al. (2019b) revealed that applying the hydrochar produced through HTC to soil may lead to a more significant mitigation of CC, AP, EP, TE, compared with paper mill sludge utilization for energy (incineration). Hydrochar used as soil conditioner also indicates a potential to bring environmental benefit for GWP, but stability of hydrochar in the soil is closely related to climate tipping (Owsianiak et al., 2018). Simultaneously, emphasis on composition of biowaste plays a role in the assessment for HTC. The toxic compound in ash and exhaust gases of HTC are decisive for toxicity-related impact and overall environmental performance, which are often underestimated or neglected (Busch et al., 2013; Owsianiak et al., 2016).

4.3 Energy saving

In high energy-consuming thermochemical processes such as drying, AD, incineration and pyrolysis, energy or fuel substitution is essential for decreasing disorganized emissions of GHGs. Clean energy such as electricity, hot water, and steam produced during incineration process not only can be sent to the grid, but also achieve adequate energy supply for other treatment operations, e.g. sludge drying technology. Alternatively, the heat could be supplied to nearby residents or factories. Iqbal et al. (2019) assessed the overall effectiveness of energy production from municipal biosolids by deducting the CO_2 emissions which were avoided by using municipal biomass for power generation. Vadenbo et al. (2014) compared several alternative energy recovery systems, and subscribed to the view that the substitution assumptions performed a crucial part in selecting the preferred plan for the thermal treatment of sewage sludge.

The substitution effect of heat or electricity recovery can be evaluated with reference to a mix of natural gas, local grid power and national electricity. Here, the prevented impact depends largely on the local electricity mix, proportion of fossil fuels, hydropower, renewable sources, etc. (Buonocore et al., 2018). If this mix changes, the consequences will change simultaneously (Zhang et al., 2019). The avoided impact associated with the industry structure of national power system: the higher the amount of fossil fuels used; the higher the savings on pollutants because of biogas power generation (Righi et al., 2013). Conversely, as the boundary of the environmental assessment was expanded to cover the upstream and downstream production and consumption, the reasonability of adopting average parameters for energy and chemicals might be challenged. For example, this can include the generation of electricity, which may be open to more research (Sablayrolles et al., 2010).

The direct comparison between existing LCA studies of sludge to energy is inextricably linked to different assumptions about input data and byproduct allocation. During those thermochemical processes with high energy demand such as drying, incineration and pyrolysis, and AD, energy saving or fuel substitution plays an important role in reducing GHG emissions and the priority rating of different schemes (Teoh and Li, 2020).

5. Resource utilization of sludge

Sludge resources recycling promotes the reutilization of waste and stimulates industrial symbiosis networks, whose starting point is recycling and processing one industry’s byproduct then transforming into another industry’s raw material. The main characteristics of the literature related to the LCA of sludge resource utilization are summarized in Table 3.

Table 3

5.1 Cement

The composite characteristics of the cement industry are high demand of raw materials and heat, as well as high compatibility and receptivity for the wastes from other industries (Nakic, 2018). Employing waste bio-solids as raw material or supplementary energy source for the adjustment of cement industry structure has become a kind of widespread use due to the maturity and performance of related technologies (Aranda Usón et al., 2013; Donatello and Cheeseman, 2013). Cement manufacture results in approximately 5-8% of overall artificial CO_2 spreads, primarily on account of two aspects. Firstly, the decomposition processes of $CaCO_3$ (limestone) into CaO when heat is added (Andrew, 2018); and secondly, the incineration process of fossil fuels aiming at heating to the melting point temperature of raw clinker ingredients (Ishak and Hashim, 2015).

According to reports, the sludge contains abundant organic substances especially refractory organics such as cellulose, hemicellulose, and lignin, and is also called biomass considering its characteristics (Champagne and Li, 2009; Mottet et al., 2010). The incorporation of biomass ash into cement could bring about two added benefits: ⅰ) reduced dissipation of energy and raw materials; ⅱ) less direct landfill of the biomass incineration remnants (Tosti et al., 2020).

The study described that the employment of deinking sludge (largely composed of ink, plastics, filler and short fibers) in a cement production plant rather than its landfilling, to replace fractional petroleum coke and limestone presented best effect in all the impact categories studied (Deviatkin et al., 2016). Co-incineration in cement kilns provides a more environmental friendly processing route for low-organic content sludge by virtue of lower fossil resource demand (Li et al., 2017a). Environmental LCA conducted by Barry et al. (2019) indicated that the application of biochar in a cement kiln revealed the largest mitigation of global warming trends and freshwater ecotoxicity (FET). The outcome primarily owes to the preponderance characteristics of the biochar, such as low leachability of heavy metals, carbon stability, and potential as a solid fuel.

As reported by Pavlík et al. (2019), there were less CO_2 production and energy consumption with the increasing proportion of SSA in mortar mix. It exhibited the functional properties of SSA mortars close to the control material. Similar research demonstrated that when all SSA discarded by the studied WWTP was reutilized in concrete manufacturing industry, there might be annual reduction in GHG emissions by as much as 10 million kg CO_2eq (Nakic, 2018).

In terms of human health, the digested sewage sludge incineration in cement kiln scenarios was better than the fluidized bed combustor (FBC) scheme, because the residual materials left in cement kiln were fixed into clinker products, while for the FBC system, the residuals were landfilled (Abuşoğlu et al., 2017). In the cement production process, Deviatkin et al. (2016) demonstrated that the substitution of fossil fuels displayed more conspicuous decline trends in environmental damage compared to the substitution of raw materials like limestone and clay. Their finding agrees with that of Valderrama et al. (2013), and in addition, the environment optimization effect is closely related to the substitution rate.

5.2 Adsorbent

Based on the low cost and comparable adsorption capacity, waste sludge can be applied as a burgeoning, safe as well as economically feasible adsorbent. Contrarily, it would be defined and treated simply as a castoff. The indirect benefits of such kind employment are the avoidance of environmental and economic costs from the production of adsorbent and disposition of the sludge residue redirected to landfill (Devi and Saroha, 2016). On this theme, sewage sludge-based activated carbon (SBAC) has emerged as a promising, economical, effective, and environmentally friendly technology aiming to eliminate phenolic compounds from water (Mu'azu et al., 2017). Nevertheless, LCA of the utilization of SBAC to remove pollutants from water is rarely found in published articles (Devi and Saroha, 2016; Devi and Saroha, 2017).

From a comprehensive perspective, the reuse of papermill residuals not only has a positive performance of efficiency and economic affordability, but also provides a sustainable alternative. One study that examined the conversion of papermill sludge into adsorbing material, resulted in dramatical reduction of carbon footprint and water ecological footprint compared to synthetic absorbents. Life cycle of paper byproduct can be prolonged as long as two cycles through the production of hydrophilic sorbent material taking used sorbent as raw materials (Likon and Saarela, 2012). However, Thompson et al. (2016) subscribed to the view that conversion into biosolids biochar adsorbent affects environmental quality at the highest level, attributing to energy consumption for biosolids drying, manufacture of mineral fertilizer to substitute biosolids applied for soil amendment, and the need for supplemental adsorbent.

5.3 Brick

Due to the favorable chemical and mineral composition of the sludge, SSA could function as a component of bricks (Smol et al., 2015). Adding biosolids to the production of bricks is a promising method to reduce the land demand for biosolids storage. There are two main processes for the utilization of sludge for brick: firstly, brick manufactured from dried sludge; or secondly, incinerated SSA. Lin et al. (2006) examined the compressive strength, permeability and water absorption rate of permeable bricks produced by sludge from sewage treatment plants and bottom ash from garbage incinerators. They found that this sintered product can meet most pavement brick standards. It has been explicitly indicated that the incorporation of bio-solids in raw materials for fired-clay bricks at a certain proportion is a sustainable strategic management scheme for bio-solids with the advantage of shorter transportation distance and less environmental threats arising from the storage stage (Mohajerani et al., 2018; Ukwatta Pitige and Mohajerani, 2016).

A comparative LCA study proved that contrasted with the bricks of comparison group, brick with incorporation of biosolids effectively diminished the majority of impacts on environment excluding water depletion (WD) impact. However, due to the considerable calcium oxide feeding to the sludge, consumers may suspect that the brick will lose its strength due to the reaction with CO_2 over time. It is therefore a challenge for the brick and tile industry to develop a biosolids-type material that is acceptable for the local market and has convincing levels of durability.

6. Discussion

6.1 Emerging sludge treatment technologies

There is no ultimate disposal scheme after sludge treatment, which can be called the optimal scheme without any environmental damage. Each scheme is more or less suitable for specific situations (Campbell, 2000). The combined application of AD, dehydration and pyrolysis has attracted a lot of interest due to its excellent ability of energy recovery and sludge reduction (Lacroix et al., 2014; Syed-Hassan et al., 2017). Assessment results of the impacts of AD and pyrolysis compound system in their life cycle exhibited that they can not only generate significant net energy, but also relieve CC and WD (Li and Feng, 2018; Opatokun et al., 2017). The combination of AD and composting also demonstrates good environmental performance in MSW treatment, and the major environmental deteriorations lie in HT, FET and MET (Mancini et al., 2019).

Buonocore et al. (2018) evaluated a circular and benign zoology chain consisting of three essential aspects: a) electricity and heat generated from sludge incineration; b) auxiliary heat source comes from refined waste cooking oil and c) wastewater is applied to irrigate wood-based plants for bioenergy regeneration. These processes greatly reduce the environmental burdens of the WWTP (FE and HT). The plant achieved sludge reduction, complete energy self-sufficiency and better efficiency of energy utilization through multiple recycling of available waste resources.

The research conducted by Zhang et al. (2019) stated that it is advantageous to implement the electro-dewatering (EDW) upgrade combined with incineration, when considering the GWP impact for the incineration route. The effect of sludge reduction brought about by EDW helped to reduce GWP during the transportation phase, and disposal stage (replaced fertilizer/heat). In addition to this, EDW helped to reduce other impact indicators, such as Photochemical Ozone Formation (POF), terrestrial and marine eutrophication (EP) with a profitable Return On Investment (ROI) (Zhang et al., 2019).

As reported by Meisel et al. (2019), the excessive energy generation and crop yields failed to cancel out the higher consumptions of HTC process. Whereas, coupled process of sludge digestion, process water recirculation and HTC cause effective relief of global warming. If the agricultural application of hydrochar is to further proceed, the scenario A-D+HTC 1+PR (Digestion, HTC, recirculation of process water for agricultural use) will be the optimal treatment process in terms of GWP. In contrast, if the availability of hydrochar is not permitted, the energetic valorization scenarios E-SS (sewage sludge after dewatering are directly used for energetic use) and E-D+HTC 1+PR (digestion, HTC, recirculation of process water, mono-combustion for energetic use) are the optimal scheme for reutilization of residual sludge.

Lishan et al. (2018) assessed the environmental and economic trade-offs in the operation of hydrothermal-pyrolysis technology (HPT), which is a novel technology for sludge synthesize utilization. HPT involves six processes, namely dewatering, hydrothermal treatment, mechanical separation, granulating and modeling, and pyrolysis carbonization. HPT can not only improve dehydration efficiency, but also produce biochar - a valuable medium for soil amelioration. In particular, biochar in soil can increase pH, content of organic carbon and source of nutrients to sustain the microbes' and plant’s metabolism, while minimizing bioavailable toxic metals like As, Cr and Pb (Khan et al., 2013; Waqas et al., 2014). For HPT, CC, FFD and HT contributed greatest among all considered impact, but CC, HT and LO were still lower than those of landfill and incineration scenarios. Compared with conventional treatments, HPT demonstrated a multi-functional preponderance of leveraging best balance between environment, society and economic for the sludge industry (Lishan et al., 2018).

Sludge treatment wetlands (STW) utilizes plants, sludge microorganisms and natural forces to dewater and stabilize the waste sludge. STW and direct land application is specifically tailored for sludge disposal within decentralized small communities (Uggetti et al., 2010). The treated sludge was validated that as long as there is adequate resting time, byproducts can add agricultural value without post-treatment for composting (Stefanakis et al., 2011; Uggetti et al., 2011). The whole process displayed the smallest impact potential on ADP, AP, EP and GWP (Uggetti et al., 2011).

Roldan et al. (2020) assessed two sludge line configurations aiming to enhance P recovery, and minimize uncontrolled P-precipitation during AD. The first configuration C1 was based on the production of a PO_4-enriched stream from sludge via elutriation in the primary thickeners. The second configuration of C2 was based on the WASSTRIP® process (Cullen et al., 2013) and its PO_4-enriched stream was mechanically obtained through dynamic thickeners. The results of LCA confirmed that C1 configuration presented the smallest GWP impact and both configurations of P recovery before AD are economically feasible and highly efficient.

Apparently, innovative and promising processes and technologies stimulate further energy and material recovery and reutilization from sludge, driving the concept of circular economy forward.

6.2 Future challenges of sludge recycling and LCA

It should be emphasized that under the assumed local conditions, there is no such a strategy that can simultaneously reduce all environmental impacts (Lombardi et al., 2017). These results are strictly related to assumptions and inventory data, but the benefit is that they are closely related to real plant operations or based on prudent assumptions. The most important criterion when choosing sludge management routes is the routes aligned with the local conditions (Campbell, 2000). Without prejudice, the environmental impact of comparative studies remains challenging. Differences in LCA methods, system boundaries, inventory analysis, and other parameters from different studies can affect the significance of potential influencing factors, even for the same sludge treatment process (Teoh and Li, 2020). For the environmental impact study of the waste treatment process, it was pointed out that no process can completely remove carbon pollutants.

Through landfill, composting, and incineration, most carbon pollutants are disorganized discharged into the air, while during AD into the water. Wen et al. (2019) stressed the fact that a single environmental medium may arouse more serious pollution to other media and the transfer of environmental threats. For this reason, the cross-media migration of pollutants and their impact on multimedia are worth investigating in more detail. Up till now, most LCAs of sludge disposal regarded sludge as a kind of waste, and the upstream process of sludge production was not considered. This standpoint has been criticized by some authors, who have begun to call the "zero burden assumption" into question (Cleary, 2010; Oldfield et al., 2018; Pradel et al., 2016). When sludge is regarded as a product, it denotes that the water treatment strategy is a multi-functional process, which produces two byproducts, namely sludge and "clean water". Accordingly, the environmental risks of the water treatment process should be assigned between the two byproducts (Pradel and Aissani, 2019).

Future research can further explore the toxicity effects of various pollutants (especially emerging pollutants, such as perfluorinated chemicals, PCAs, PPCPs, bisphenol A, etc.) and pathogens (like through QMRA) generated by different sludge treatment methods from the life cycle perspective (Clarke and Smith, 2011; Harder et al., 2015; Teoh and Li, 2020). The migration and transformation of those unrecognized pollutants in LCIA should be identified and quantitatively analyzed with the help of material flow analysis (MFA) or toxicological analysis. It should be considered that pretreatment technologies such as thermolysis and ultrasonic treatment may improve the degradation of organics in sludge (Nakakubo et al., 2012), and include the treatment of landfill leachate, sludge return liquors from thickening and dewatering processes, etc. (Ding et al., 2014; Guo et al., 2015).

When treating sludge in sewage treatment plants, a circular economy method is required and adjusted according to local conditions. Large-scale municipal WWTPs engender largest mass of sewage sludge, and their primary target is best lying in maximizing energy conversion efficiency; while in medium-sized WWTPs, the focus should be on improving the energy recovery efficiency, while at the same time obtaining high-quality products for further treatment (such as composting treatment). Small rural facilities should concentrate on the recycling of materials (Kacprzak et al., 2017).

With regard to the future challenges of sludge recycling, the current review summarizes the main points emerging from the analysis:

(1) At the social level, in view of the potential threat of pathogens and pollutant migration, many countries have severe regimented control of sewage sludge used for cultivating food crops. It is important to protect public health and safety.

(2) At the economic level, the local constraints of future policies, geography factors, economy, climate, etc., should be taken into consideration, and the balance between the applicability and costs of traditional and emerging treating technologies in the local area should be evaluated.

(3) At the environmental level, improve sludge cyclical utilization benefits, reduce secondary waste generation, and find a good compromise scenario between recycling efficiency and resources required by recycling technology.

7. Conclusion

This study reviewed 37 articles and provided key points to facilitate researchers develop and manage sludge treatment through LCA practices. Most sludge management strategies depend on the physiochemical properties of the sludge, the particular conditions of the ecosystem, the associated expenses, and the environmental damage anticipated to be reduced. In fact, there are discrepancies in LCIA results reported in different studies, which weaken the results comparability of various studies, therefore most research focuses on comparative scenarios.

To date, landfill is the prevalent guiding terminal for sludge disposal, but brings high transport costs and a waste of nutrients and energy potential of dewatered sludge. In most research concerning sludge management scenarios based on LCA, energy or nutrient recovery and comparison of traditional or promising alternatives normally play a role in the main driving factors. Sludge treatment system can achieve sludge reduction and harmless disposal, guard against pollution of the environment, and has two added values of greatest relevance to the project. One is to generate thermal energy and electric energy through incineration or biogas recovery; the other is to produce organic fertilizer attributed to sludge rich in N, P and potassium, which are advantageous sources of agricultural fertilizers. The common research direction in the future should be to develop strategies that minimize the disposal of sludge into landfills and utilize it as a nutrient/energy source.

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Fig. 1. Steps and frameworkof LCA research and main process of sludge treatment and disposal.

Fig. 2. Life cycle assessment phase coverage in the 37 studies reviewed.

Fig. 3 Impacts categories included in the reviewed sludge LCA: GWP, global warming /climate change /GHG emission; AP, acidification potential; HT, human Toxicity; POP, photochemical oxidation potential; ODP, ozone depletion potential; EP, eutrophication; FE, freshwater eutrophication; TET, terrestrial eco-toxicity; IR, ionizing radiation; LO, land occupation; PMF, particulate matter formation.

Tables

Table 1 Main characteristics of the studied references included in the literature review.

Short reference	Country or Continent	Functional unit (FU)	Inventory type [a]	Database	LCIA methodology [b]	Normalization	Weighting	Sensitivity analysis
J. Lederer et.al (2010)	Europe	1 t raw sludge	FD+QR+D+ES	Probas Datenbank	CML, IMPACT 2002+	N	N	N
Q. Liu et.al (2011)	China	1 TJ-steam	QR+LE	N	-	Y	Y	N
E. Uggetti et.al (2011)	Spain	1 t SS (wet weight)	FD+QR+D	Ecoinvent	CML 2 baseline method	N	N	N
T. Nakakubo et.al (2012)	Japan	100,000 people receiving disposal services	QR+YR	N	-	N	N	N
Y. Cao et.al (2013)	Poland	500 m^3 liquid raw SS (5% solids content) per day	QR+D	Ecoinvent 2.1	-	N	N	Y
J. Hong et.al (2013)	China	1 GJ of net energy for electricity and steam production	FD+QR+YR	N	ReCiPe, IMPACT2002+, TRACI, CML, EPD2007	Y	N	Y
S. Righi et.al (2013)	Italy	3000 t of biodegradable waste fractions	FD+QR+D+LE+IE	GaBi Professional database 2006, Ecoinvent	CML midpoint	N	N	N

C. Valderrama et.al (2013)	Spain	1 kg clinker	FD+QR+D	Ecoinvent 2.1	CML 2000, Eco-indicator 99, IPCC	N	N	Y
C. Xu et.al (2014)	China	1 t DS	QR+YR	N	ReCiPe midpoint, IMPACT 2002+	Y	N	Y
Y. Zhao et.al (2015)	China	PE	LE	N	-	Y	Y	N
C.-M. Lam et.al (2016)	China	1 t dry solids in raw SS	FD+QR	N	-	Y	N	Y
A. Abuşoğlu et.al (2017)	Turkey	1 kg of digested SS	FD+D+ES	Ecoinvent 2.2	IMPACT 2002+	Y	N	N
I. Alyaseri et.al (2017)	America	1 kg of mixed sludge in dry basis (1 Dry kg)	FD+QR+YR+ES	Ecoinvent	ReCipe endpoint methodology	Y	Y	N

(continued)

Short reference	Country or Continent	Functional unit (FU)	Inventory type [a]	Database	LCIA methodology [b]	Normalization	Weighting	Sensitivity analysis
C. Gourdet et.al (2017)	France	1 t TS of sludge	QR+ES+IE	Ecoinvent	ReCiPe E v1.08/Europe baseline method.	N	N	Y
H. Li et.al (2017a)	China	1 t dry solid	FD+QR	N	CML (baseline) method	Y	N	Y
H. Li et.al (2017b)	China	1 t TS of the input sludge	QR+ES	N	CML (baseline) method	Y	N	Y
L. Lombardi et.al (2017)	Italy	1 t DM of SS	QR+D+YR	Ecoinvent 3.0	CML-IA baseline method	N	N	Y
M. ten Hoeve et.al (2017)	Denmark	1000 kg mixed sludge	FD+QR+D	Ecoinvent 2.2	-	N	N	Y

S.L.H. Chiu et.al (2018)	China	350 t/d of SS and 105 t/d of food waste based on a 10:3 wet weight mixing ratio	QR+YR	N	ReCipe midpoint	N	N	Y
K.C. do Amaral et.al (2018)	Brazil	1000 m^3 of sewage	FD+QR+D+LE	Ecoinvent 3.0	ReCiPe 2016 Midpoint (H)	N	N	N
H. Li et.al (2018)	China	1 t TS of thickened sludge	QR+D+ES+YR	Ecoinvent 2.2	CML (baseline) method	Y	N	Y
X. Lishan et.al (2018)	China	1 t DS	FD+QR+YR	N	ReCiPe	N	N	Y
S. Piippo et.al (2018)	Northern Finland	N	FD+QR+YR+IE	N	IPCC (2006b-c)	N	N	N
H. Yoshida et.al (2018)	Denmark	1000 kg of mixed sludge	FD+QR	Ecoinvent 2.2	LCIA method based on the recommendations in the International Reference Life Cycle Data System (ILCD) handbook.	Y	N	Y
D. Barry et.al (2019)	Canada	9918 kg of dewatered SS or 2777 kg of SS on a dry basis	LE+YR	European reference Life Cycle Database	CML Baseline 2015	N	N	N

(continued)

Short reference	Country or Continent	Functional unit (FU)	Inventory type [a]	Database	LCIA methodology [b]	Normalization	Weighting	Sensitivity analysis

G. Chen et.al (2019)	China	1 t MSW or SS	FD+D	Ecoinvent 3.1	Swedish EPD (Environmental product declarations), Impact 2002+	Y	N	Y
I. Deviatkin et.al (2019)	China	1 kg of nitrogen available in fertilizer	FD+QR+D+LE	Database embedded into the Gabi ts software	IPCC	N	N	Y
D. Marazza et.al (2019)	Italy	40,000 kg/h of sludge	FD+QR+ES	N	-	N	N	Y
K. Meisel et.al (2019)	Germany	N	FD+QR+ES+LE	Ecoinvent database v2.2 and v3.2, Gemis database v4.9	IPCC	N	N	N
Z. Wen et.al (2019)	China	1 t MSW	FD+QR+YR+ES	N	-	Y	Y	Y
H. Zhang et.al (2019)	Italy	1 dry ton of sludge	FD+QR+D+LE	Ecoinvent V3	IPCC 2013, ReCiPe 2008 V1.05	Y	N	Y
E. Lee et.al (2020)	America	1 t wet waste	QR+YR+ES+LE	N	TRACI 2.1 v1.01	N	N	N
B. Morero et.al (2020)	Argentine	1 t waste treated	QR+D+ES	Ecoinvent V3	ReCiPe midpoint	N	N	N
N. Nakatsuka et.al (2020)	Japan	The amount of urban biomass/year	QR+IE	N	-	N	N	Y
M. Roldan et.al (2020)	Spain	1 t of PO_4-P, 1 hm^3 treated wastewater inflow	ES+LE	Ecoinvent V3	Hierarchist ReCiPe(H) v 1.02 midpoint	N	N	N
F. Rostami et.al (2020)	Iran	1 kg of DS	QR+D+LE	Ecoinvent 3.3	ReCiPe midpoint, ReCiPe endpoint, CML	Y	N	N

R.R.Z. Tarpani et.al (2020)	UK	1000 kg of thickened sludge on a DM basis	QR	Ecointent 2.2	ReCiPe 1.08	N	N	Y

Y- included; N-not included or documented. More details can be found in the supplementary materials.

[a] Abbreviation: Field data (FD); Quoted reference (QR); Database (D); Statistical yearbook and report (YR); Estimation, simulation or calculation (ES); Lab/pilot experiment(LE); Interviews with experts (IE).

[b] In this column, "-" means that the conventional LCA methods were not used, but other models, methods, or tools were adopted, which could be confirmed in the supplementary materials.

Table 2 Main characteristics of the literature related to the LCA of sludge nutrient/energy recovery (Y- included; N-not included or documented).

Short reference	GHG emissions	Energy recovery of biogas/bio-oil/heat ?	Avoided energy/ fuel included?	Avoided chemical fertilizers included?	End use of sludge
J. Lederer et.al (2010)	direct & indirect	Y (heat)	Y	Y	Fertilizer, landfill
Q. Liu et.al (2011)	direct & indirect	Y (heat)	N	N	Landfill
E. Uggetti et.al (2011)	direct & indirect	N	N	N	Land application
T. Nakakubo et.al (2012)	direct & indirect	Y (biogas, pyrolysis gas)	N	N	Building materials
Y. Cao et.al (2013)	direct & indirect	Y (biochar, bio-oil, biogas)	Y	Y	Biogas and bio-oil: Burning on-site, heat and electricity cogeneration, transportation fuel use (for biogas). Biochar: land application, landfill .
J. Hong et.al (2013)	direct	Y (heat)	N	N	Landfill, building materials
S. Righi et.al (2013)	direct & indirect (biogenic CO_2)	Y (biogas)	Y	Y	Landfill, fertilizer
C. Valderrama et.al (2013)	direct & indirect (biogenic CO_2)	N	Y	N	Clinker storage
C. Xu et.al (2014)	direct & indirect (biogenic CO_2)	Y (biogas, heat, landfill gas)	Y	N	Landfill, agricultural use
Y. Zhao et.al (2015)	direct	N	N	Y	Compost for urban landscaping.
C.-M. Lam et.al (2016)	direct & indirect	Y (methane, heat)	Y	Y	Landfill, cement production
A. Abuşoğlu et.al (2017)	direct & indirect	Y (heat)	Y	N	Landfill, incineration of sludge in a cement kiln.
I. Alyaseri et.al (2017)	direct (biogenic CO_2)	Y (heat, biogas)	N	Y	Landfilling of ash, landfilling of digestate, land farming application.
C. Gourdet et.al (2017)	direct & indirect	Y (biogas)	Y	Y	Agricultural application

(continued)

Short reference	GHG emissions	Energy recovery of biogas/bio-oil/heat ?	Avoided energy/ fuel included?	Avoided chemical fertilizers included?	End use of sludge
H. Li et.al (2017a)	direct & indirect (biogenic CO_2)	Y (biogas)	Y	N	Land use, landfill. Subsequent application for sludge co-incineration in cement kilns scenario were not included.
H. Li et.al (2017b)	direct & indirect (biogenic CO_2)	Y (biogas)	Y	N	Land use
L. Lombardi et.al (2017)	direct & indirect (biogenic CO_2)	Y (biogas, landfill gas, heat)	Y	Y	Agricultural application, landfill, land spreading
M. ten Hoeve et.al (2017)	N	Y (biogas and heat)	Y	Y	Land use
S.L.H. Chiu et.al (2018)	direct & indirect	Y (biogas)	Y	N	Landfill
K.C. do Amaral et.al (2018)	direct (biogenic CO_2)	Y (biogas and heat)	N	Y	Agricultural application/ Landfill
H. Li et.al (2018)	direct & indirect (biogenic CO_2)	Y (biogas, bio-oil, and pyrolysis-gas)	Y	N	Land use
X. Lishan et.al (2018)	direct & indirect	Y (methane produced from pyrolysis liquid)	N	N	Resource Utilzation
S. Piippo et.al (2018)	direct & indirect (biogenic CO_2)	Y (biogas)	Y	Y	Soil or fertilizer
H. Yoshida et.al (2018)	direct & indirect (biogenic CO_2)	Y (biogas)	N	Y	Land application, Landfilling of ash
D. Barry et.al (2019)	direct & indirect	Y (biochar)	Y	Y	Landfilling of ash, Agricultural application, coal substitute in cement kiln (ash used as a

cement filler).

(continued)

Short reference	GHG emissions	Energy recovery of biogas/bio-oil/heat ?	Avoided energy/ fuel included?	Avoided chemical fertilizers included?	End use of sludge
G. Chen et.al (2019)	direct (biogenic CO_2)	Y (heat)	Y	N	Bottom slag to make building materials, Landfilling of solidified fly ash.
I. Deviatkin et.al (2019)	direct & indirect	Y (heat)	N	N	Field application
D. Marazza et.al (2019)	direct & indirect	Y (biochar, bio-oil, syngas)	Y	N	Agronomic use, Carbon sequestratio n.
K. Meisel et.al (2019)	direct & indirect	Y (biogas, heat)	Y	Y	Agricultural Use, Energetic Use
Z. Wen et.al (2019)	direct (biogenic CO_2)	N	N	N	Sludge drying and incineration , sludge landfill, co-processi

H. Zhang et.al (2019)	direct & indirect	Y (heat)	Y	Y	ng of sludge in cement kiln. Landfilling of ash, Agricultural application
E. Lee et.al (2020)	direct & indirect (biogenic CO_2)	Y (biogas, heat)	Y	Y	Landfill, composting. HS-AcD and incineration scenarios were not clear.
B. Morero et.al (2020)	direct & indirect (biogenic CO_2)	Y (biogas)	Y	Y	Landfill
N. Nakatsuka et.al (2020)	direct & indirect (biogenic CO_2)	Y (heat)	Y	N	Landfill
M. Roldan et.al (2020)	direct & indirect (biogenic CO_2)	N	N	N	N
F. Rostami et.al (2020)	direct (biogenic CO_2)	Y (biogas)	Y	Y	Compost application, landfill. Subsequent application

					for sludge incineration scenario were not included.
R.R.Z. Tarpani et.al (2020)	direct & indirect (biogenic methane)	Y (biogas, heat, syngas)	Y	Y	Fertilizer, landfill

Table 3 Main characteristics of the literature related to the LCA of sludge resource utilization

Short reference	Resource type	Fuel or raw material substitution [a]	Functional unit (FU)	GHG emissions	Impact categories or indicators	Avoided impact included [c]	Sensitivity/ uncertainty analysis [b]
C. Valderrama et.al (2013)	Substitute in cement production.	F (dried sludge); RM (lime stabilized sludge)	1 kg of clinker	direct	8 (CC, AD, EP, ADP, ODP, FAETP, POP, TET)	N	Y
I. Deviatkin et.al (2016)	Substitute in cement production.	F (dry deinking sludge); RM (deinking sludge ash)	54,750 t deinking sludge	direct (biogenic CO_2)	6 (GWP, ODP, TET, AP, EP, ADP)	Y (avoided materials)	Y
D. Nakic et.al (2018)	Substitute for cement in concrete production.	RM (sewage sludge ash)	1 m^3 of ready-mixed concrete	direct & indirect	8 (GWP, ADP, ODP, HT, TET, POP, AD, EP)	Y (avoided landfill)	N
Z. Pavlík et.al (2019)	Substitute for cement in blended mortars production.	RM (sewage sludge ash)	1 t cement and 1t SSA	direct & indirect	2 (carbon footprint and the amount of consumed energy)	N	N
L. Tosti et.al (2020)	Substitute in cement mortars production.	RM (fly ash from biomass combustion)	FU_{20}, FU_{40}	direct & indirect	13 (CC, ODP, HT-cancer, HT-noncancer, PMF, TA, TE, FE, MEP, FET, ADP)	Y (avoided landfill)	Y
M. Likon et.al (2012)	Conversion of papermill sludge into absorbent.	RM (dry papermill sludge)	1,000 kg oil spill	direct & indirect (biogenic CO_2)	1 (carbon footprint)	Y (avoided landfill, electricity production)	N

(continued)

K.A. Thompson et.al (2016)	Biochar adsorbents application.	RM (biochar)	75% removal of sulfamethoxazole (SMX) from 47 300 m3/day of secondary wastewater effluent over 40 years.	direct & indirect (biogenic CO_2)	10 (EP, HT-cancer, HT-noncancer, ET, AP, ODP, FFD, SP, GWP, RE)	N	Y
A. Mohajerani et.al (2018)	Substitute in fired-clay bricks prodcution.	RM (biosolids)	1,000 units of fired bricks	direct	8 (CC, ODP, AP, HT, MET, ULO, WDP, Embodied Energy)	N	Y

[a] In this column, the supplement in brackets is the substitute material. Abbreviation: Fuel substitution (F); Raw material substitution (RM).

[b] Y- included; N-not included or documented.

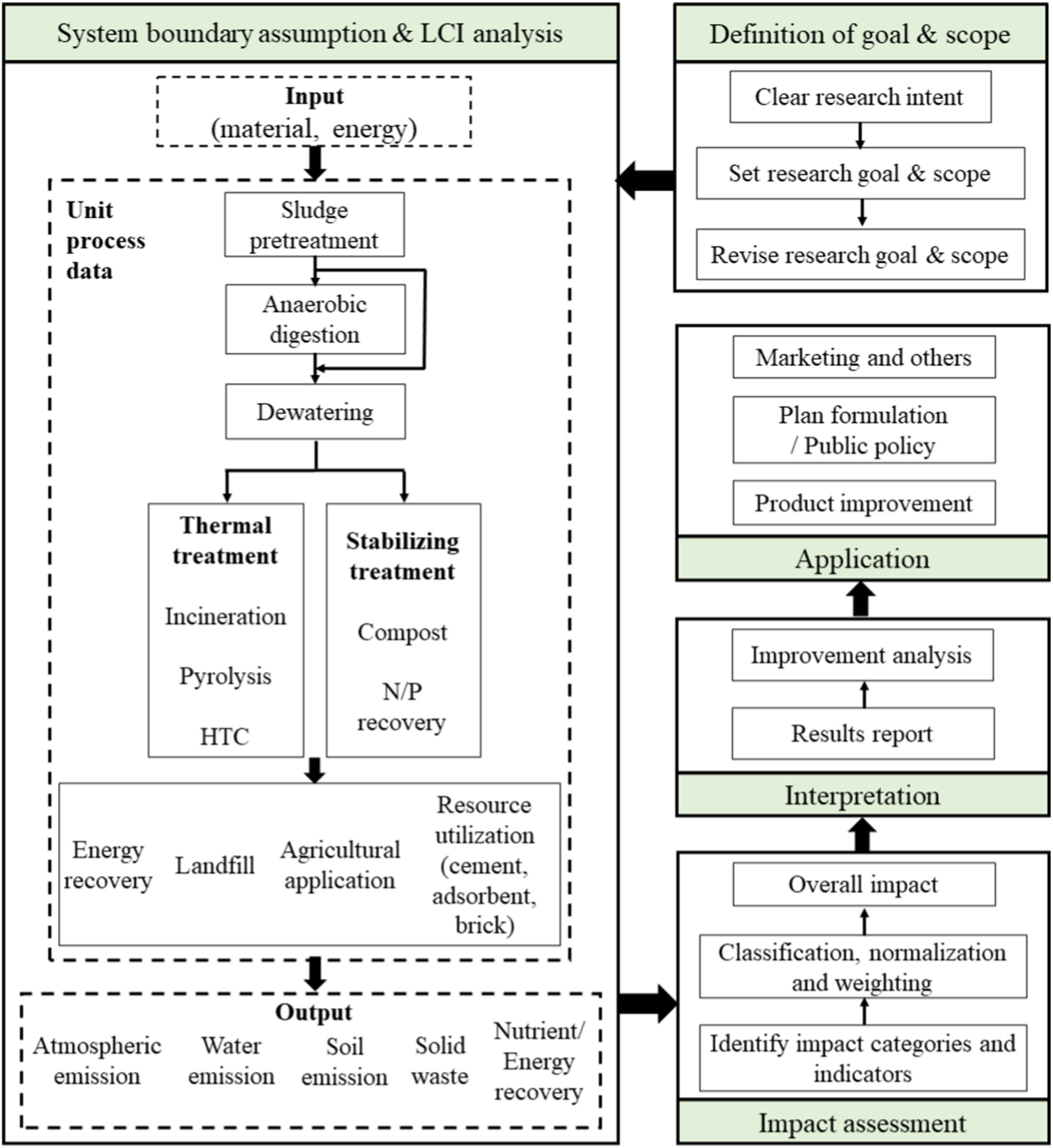

System boundary assumption & LCI analysis
Input
(material, energy)
Unit process data
Sludge pretreatment
Anaerobic digestion
Dewatering
Thermal treatment
Incineration
Pyrolysis
HTC
Stabilizing treatment
Compost
N/P recovery
Energy recovery
Landfill
Agricultural application
Resource utilization (cement, adsorbent, brick)
Output
Atmospheric emission
Water emission
Soil emission
Solid waste
Nutrient/ Energy recovery
Definition of goal & scope
Clear research intent
Set research goal & scope
Revise research goal & scope
Marketing and others
Plan formulation / Public policy
Product improvement
Application
Improvement analysis
Results report
Interpretation
Overall impact
Classification, normalization and weighting
Identify impact categories and indicators
Impact assessment

Figure 1

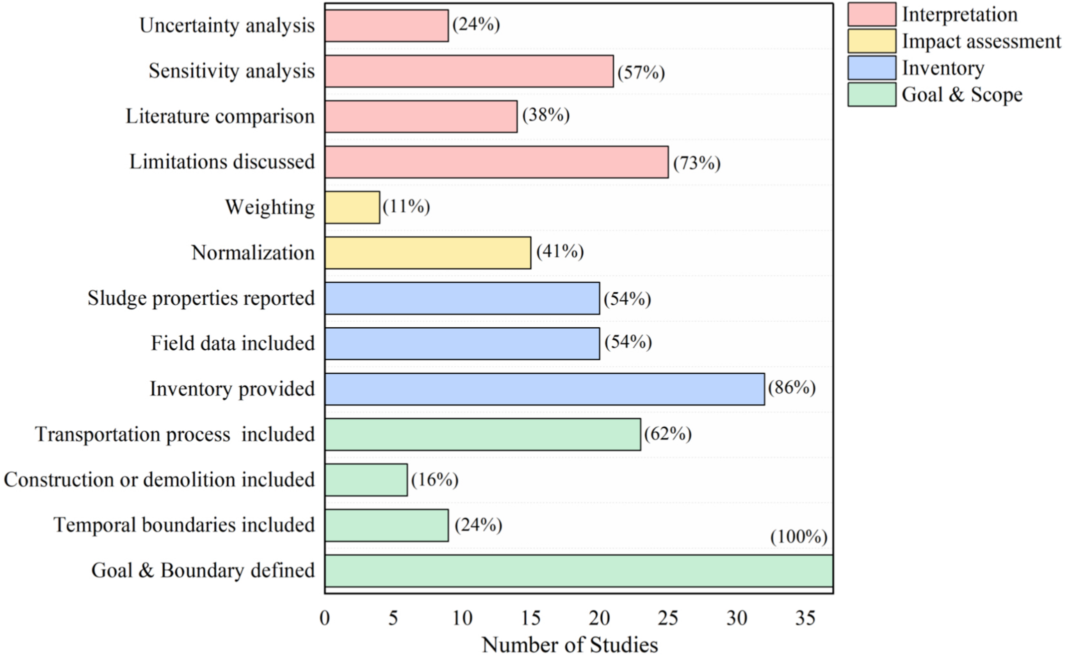
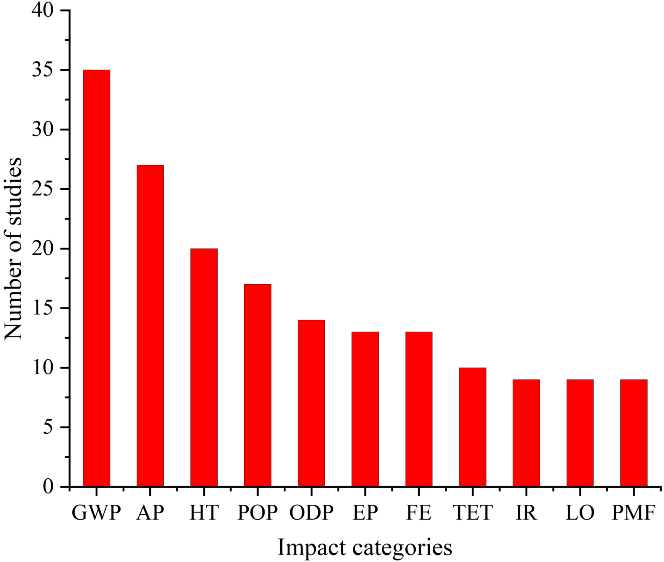

Uncertainty analysis
(24%)
Sensitivity analysis
(57%)
Literature comparison
(38%)
Limitations discussed
(73%)
Weighting
(11%)
Normalization
(41%)
Sludge properties reported
(54%)
Field data included
(54%)
Inventory provided
(86%)
Transportation process included
(62%)
Construction or demolition included
(16%)
Temporal boundaries included
(24%)
Goal & Boundary defined
(100%)
0
5
10
15
20
25
30
35
Number of Studies
Interpretation
Impact assessment
Inventory
Goal & Scope

Figure 2



40
35
30
25
20
15
10
5
0
Number of studies
GWP
AP
HT
POP
ODP
EP
FE
TET
IR
LO
PMF
Impact categories

Figure 3