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REFLOW
EUROPEAN TRAINING NETWORK

REFLOW

"Phosphorus REcovery for FertiLisers frOm dairy processing Waste"

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Literature reviews of each research area in WP 3

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LIST OF ABBREVIATION

AD	Abiotic Depletion
AP	Acidification Potential
ASP	Activated Sludge Process
$	American Dollar
NH_3	Ammonia
AD	Anaerobic Digestion
AF	Annuity Factor
ANR	Apparent N Recovery
ALCA	Attributional Life Cycle Assessment
AC sludge	Bio-chemically Treated Activated Sludge
BNRT	Biological Nutrient Removal Treatment
BOD	Biological Oxygen Demand
BOD	Biological Oxygen Demand
CAPEX	Capital Expenditure
CB	Carbamates
CB	Carbamates
COD	Chemical Oxygen Demand
ClO_2	Chlorine Dioxide
Cl_2	Chlorine Gas
CIP	Cleaning In Place
CSTR	Completely Stirred Tank Reactors
CLCA	Consequential Life Cycle Assessment
CRMs	Critical raw materials
CED	Cumulative Energy Demand
DPS	Dairy Processing Sludge
DPW	Dairy Processing Waste
DBPS	Disinfection By-products
DM	Dry Matter
EcTox	Ecotoxicity
EAC	Equivalent Annual Cost
€	Euro
EU	European Union
EP	Eutrophication Potential
FOG	Fats, Oil and Greases
FEV	Fertiliser Equivalent Value
FRV	Fertiliser Replacement Value
FD	Fossil Depletion
FEW	Fresh Water Ecotoxicity
FU	Functional Unit
GWP	Global Warming Potential
HAAs	Haloacetic Acids

Tox	Human Toxicity
HTC	Hydrothermal Carbonization
ClO^-	Hypochlorite
IDF	International Dairy Federation
ISO	International Standard Organization
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LCIA	Life Cycle Impact Assessment
LCI	Life Cycle Inventory
DAF sludge	Lime Treated Dissolved Air Flotation Processing Sludge
ME	Marine Ecotoxicity
MFA	Material Flow Analysis
MARS	Membrane Anaerobic Reactor System
MBR	Membrane Bioreactor
MBRs	Membrane Bioreactors
Mt	Million Metric Tonnes
MCA	Multi-Criteria Analysis
NPV	Net Present Value
N	Nitrogen
FEV-N	Nitrogen Fertiliser Equivalent Value
OPEX	Operational Expense
OM	Organic Matter
OC	Organochlorines
OP	Organophosphate
OD	Ozone Depletion
POPs	Persistent Organic Pollutants
P	Phosphorus
FEV-P	Phosphorus Fertiliser Equivalent Value
PUE	Phosphorus Use Efficiency
POFP	Photo-Oxidation Formation Potential
PCmtx	Physico-chemical allocation
PCBs	Polychlorinated Biphenyls
PCDD/Fs	Polychlorinated Dibenzo-p-dioxins and Dibenzofurans
PAHs	Polycyclic Aromatic Hydrocarbons
K	Potassium
SBRs	Sequencing Batch Reactors
SBR	Sequential Batch Reactor
SS	Sewage Sludge
SSA	Sewage Sludge Ash
Na	Sodium
SSA	Specific Surface Area
SFA	Substance Flow Analysis
SEK	Swedish Krona
SUB	System Expansion by Substitution
TC	Total Carbon
TN	Total Nitrogen
TP	Total Phosphorus

TSS	Total Suspended Solids
TEQ	Toxic Equivalent
THMs	Trihalomethanes
UASB	Upflow Anaerobic Sludge Blanket
VSS	Volatile Suspended Solids
WWTP	Wastewater Treatment Plant
WD	Water Depletion
MCPA	2-methyl-4-chlorophenoxyacetic acid
MCPA	2-methyl-4-chlorophenoxyacetic acid

Introduction

This document contains the literature reviews of the Early Stage Researcher (ESR) Fellows active in Work Package (WP) 3 of the REFLOW European Training Network. The overall research goals of the REFLOW project is to develop and demonstrate processes for the recovery and reuse of phosphorus (P) products from dairy processing waste (DPW). This involves establishing their fertilizer value and optimum application rates through laboratory protocols as well as to address environmental, social, food safety and economic challenges and finding market-driven solutions for the new processes and fertilizer products.

WPs 1 and 2, respectively, are focused on developing phosphorus recovery technologies and testing generated fertilizer products with regard to crop yield and soil health. This WP, WP 3, aims to guide the work in the other WPs towards economic and environmental sustainability, establish a farmer-friendly tool for application rates and timing for the fertilizers, test the fertilizers for microbial pathogens and chemical contaminants, and investigate the impacts of CE quality compliance criteria on the techno-economic feasibility.

The specific objectives of WP 3 are to:

- Establish intrinsic fertilizer equivalency value (FEV) of the fertilizer products based on chemical testing and bio-assays;
- Validate the FEVs using crop yield from field trials data
- Undertake two-stage techno economic and environmental assessment of REFLOW processes and products - to identify potential process hotspots, and for a proof of concept against benchmarks;
- Develop a technoeconomic model of the market dynamics for recycled P fertilizers
- Identify sustainability indicators, develop metrics and use these to evaluate the sustainability of the fertilizer production;
- Establish the impact on food safety arising from the potential presence of persistent organic pollutants and microbiological hazards in the fertilizers;
- Develop new market models which ensure distribution of value for all stakeholders in the process chain.

The ESRs being trained in this WP are working on:

- ESR 10: FEV of DPW and the REFLOW fertilizers and smart farming
- ESR 11: Life cycle sustainability assessment
- ESR 12: Technoeconomic assessment of the commercial viability of the REFLOW fertilizer processes and products, regulatory costs and financial incentives and deterrents
- ESR 13: Development of new financial models for a circular (bio)economy

The literature reviews provide the fellows with the essential background knowledge and current scientific state-of-the-art in order to clearly define research questions and hypotheses for testing. The fellows will prepare technical reviews for publication from this review document.

The literature review of ESR 10 looks into what is currently known when it comes to the characteristics of dairy effluents in terms of e.g. nutrients and metals. It elaborates on the definitions of the concepts of FEV and of fertilizer replacement value (FRV) and it reviews different potential emerging contaminant risks of relevance for DPW.

ESR 11 looks at earlier work with regard to life cycle assessment (LCA) of different dairy products and provides a comparison of methodologies used in the assessments, and specifically looks into how waste and wastewater treatment was covered and what could be learnt from those assessments. The literature review also covers earlier LCAs of phosphorus recovery technologies of relevance for future research work in the REFLOW project.

The report by ESR 12 covers earlier work on material flow analysis (MFA) for phosphorus and on technical and economic analysis of phosphorus recovery technologies. The review also specifically focused on how to handle uncertainties in such contexts. Although activities of the kind reviewed in this part directly related to dairies and DPW are scarce, methods and experiences related to wastewater treatment in other sectors seem relevant to use but need to be complemented with specific indicators and considerations.

Finally, the report by the ESR 13 takes a value chain perspective in looking into drivers and barriers for the adoption of new sustainability-branded alternatives in the dairy industry. By mapping actors in the value chain and their attitudes and response to external drivers, earlier work suggests that pro-sustainability action strongly depends on sustainability commitments from each actor as well as competence levels and governmental support.

Within WP3, technologies and approaches generated in WP 1 and WP2 will be put in a real context with a system focus, to ensure that the REFLOW project ends up with fertilisers that are socially acceptable, economically viable and environmentally superior to alternatives. This report summarises the first parts of the work that aims to gather earlier knowledge and set a starting point for research within the project. It shows that there are currently considerable knowledge gaps with regard to phosphorus recovery from DPW but that there are things to learn from what has been done in other sectors and that methods for filling those gaps are available or will be available after some methodology development efforts.

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Dairy processing sludge and other bio-based alternatives characterisation, fertilizer replacement value and emerging contaminant concerns: a review (ESR10)

This document contains the Literature Reviews of the Early Stage Researcher's (ESR 10) active in Work Package 3 of the REFLOW European Training Network.

Globally, the milk processing industry continues to grow and therefore generates large volumes of waste. The dairy processing sludge (DPS, including bio-chemically treated activated sludge and lime-treated dissolved air flotation sludge) are products from *in situ* wastewater treatment to meet discharge licence needs and are eventually directed to landfill, incineration, anaerobic digestion, or are spread directly onto land.

In the agricultural sector these raw wastes are perceived as having great potential, but more knowledge is needed before they can be recognised as a fertilizer and not a waste. Limited data are available in terms of the nutrient and metal content of these wastes and how this changes during the storage season before being disposed of to land. Another option for this waste is to convert it to other products such as biochar, pyrochar, or hydrochar through thermochemical treatment. All of these derivatives have uses as a bio-based fertilizer or as a potential soil conditioner to support various soil functions.

Furthermore, in-depth characterisation on these products in terms of their nutrient and metal content is needed. A new knowledge gap relates to examining other potential contaminants in this waste, and techniques and protocols in relation to ascertaining the levels in these wastes is needed. Many sources of emerging contaminants are feasible e.g. sourced from environment during storage and on-farm, or on-processing plant sources including veterinary compounds, hormones, pesticides, disinfectants, micro-plastics and persistent organic pollutants (POPs). In addition, during the thermochemical treatment process, other toxic compounds produced from high-temperature reactions could be introduced that may be then transferred to soil.

More work is needed with respect to (fertiliser equivalent value) FEV estimation which can be determined in controlled pot or field trials using different soil types. Presently, knowledge gaps in this area include the way in which the FEV is currently calculated, the effect of application rate, time and method, and the impact the type of bio-based fertilizers and crops have on FEV. In particular, phosphorus fertilizer equivalent value (FEV-P) is often difficult to ascertain and needs to be examined using pot trials, intact core incubation studies and micro plots. As DPS also may be imbalanced in terms of its nutrient status, optimisation with respect to Nitrogen (N), Phosphorus (P) and Potassium(K) needs to be further investigated.

Introduction

The scarcity and unequal global distribution of phosphorus (P) are the serious “P challenges” (Chiders et al., 2011), particularly as world P reserves may be exhausted within 50-400 years (van Dijk et al., 2016). Europe lacks natural P-rock deposits and mainly depends on imported P. Conversely, leakage and losses of P from point and diffuse sources causes environmental damage and food security problems (van Dijk et al., 2016). This is sometimes referred to as the “P paradox” (Jarvie et al., 2019). As one of the largest agricultural sectors in the European Union (EU) (Augère-Granier, 2018), the dairy industry generates a large amount of P-rich dairy processing wastes (DPW), including dairy effluents, dairy processing sludge (DPS) and struvite from dairy wastewater treatment, and thermochemical treated products using DPS as a feedstock (Ashekuzzaman et al., 2019a, Ashekuzzaman et al., 2019b, Carvalho et al., 2013, Uysal and Kuru, 2015) (Figure 1). They are widely reused as bio-based fertilizers to recycle the nutrients such as nitrogen (N), P and potassium (K) back to soil because of their high nutrients contents (Table 1).

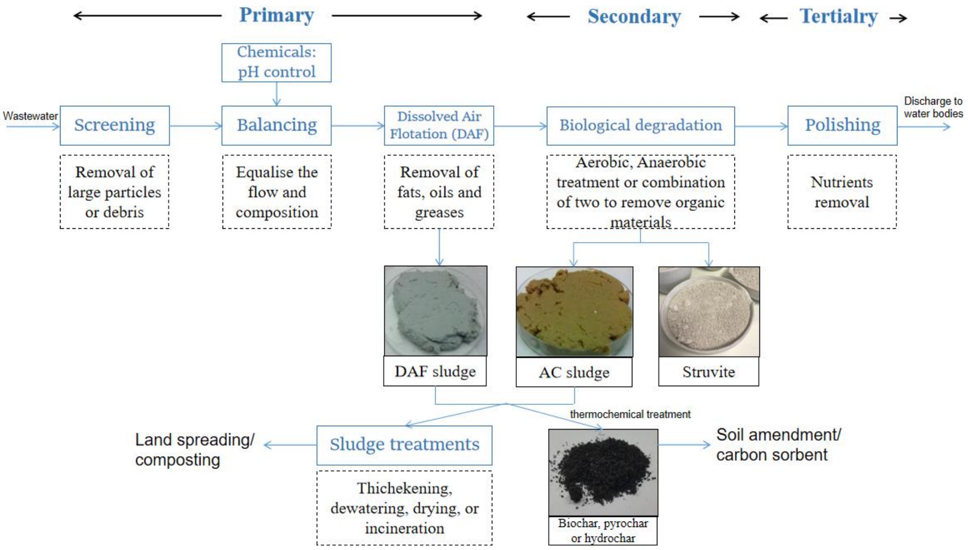

Figure 1. *Flow chart of dairy wastewater treatment process and sludge, struvite, and char generation. DAF sludge=Lime treated dissolved air flotation processing sludge; AC sludge= Bio-chemically treated activated sludge (adapted from Ashekuzzaman et al. 2019a).*

Demand for dairy products is increasing worldwide, which has led to a huge growth in the dairy sector both in terms of animal numbers to produce milk and its influence on the environment (International Dairy Federation, 2018). In 2018, global milk production increased by 1.6% to about 838 million metric tonnes (Mt), and is expected to reach 981 Mt by 2028, faster than most other main agricultural commodities (OECD/FAO, 2019). While overall per capital demand in some regions like Europe and North America is in decline (OECD/FAO, 2019), the opposite trend is the case in Africa and Asia. This will lead to an expanding global market and high export of milk powders (EC, 2018; OECD/FAO, 2019). Over the next decade, EU milk production is projected to grow at 1.1% per year (OECD/FAO, 2019). For example, in Ireland, dairy is the largest food export sector with the value of dairy exports in 2019 of € 4.4 billion, an 11% increase on the previous year (Bord

Bia, 2020). The main markets for Irish dairy products are the UK, the Netherlands, China, the US and Germany, while there is notable growth of Irish exports in other Asian countries such as Vietnam and Malaysia (Bord Bia, 2020).

As dairy processing plants continue to process greater volumes of milk, there is an additional increase in the generation of DPS. The dairy processing industry, is now considered the largest global industrial food wastewater source, especially in EU (Kolev Slavov et al., 2017). Potable water used in dairy processing is needed in every step of the technological process, including production, cleaning, disinfection, heating and cooling (Sarkar et al., 2006), and contributes to the total amount of wastewater produced. Based on different products and production lines, the scale of factory and batch, or continuous processes a factory use, the final effluent varies widely both in volume and in composition (Durham and Hourigan, 2007; Nadais et al., 2010). For example, dairy effluent has large variations in terms of pH and high concentrations of total suspended solids (TSS), biological oxygen demand (BOD), chemical oxygen demand (COD), nutrients and nitrate (Britz et al., 2006; Karadag et al., 2015; Mohan et al., 2008; Tawfik et al., 2008) (Table 3). To meet discharge limits, dairy wastewater must be treated before discharge to the receiving water-bodies at dairy processing sites. The wastewater treatment results in a large amount of DPS and struvite, which are the major solid organic wastes in the dairy industry (Figure 1). Currently, recycling or reuse of DPS and stuvite to replace other resources is the best solution for the disposal and recovery of valuable fertilizer components (Ashekuzzaman et al., 2019b; Uysal and Kuru, 2015). Alternatively, DPS may be processed further into various co-products, for example, pyrochar, biochar and hydrochar are derived from the different thermochemical treatments of DPS to gain energy and can be used as soil ameliorants and a filter medium to adsorb phosphorus (Ashekuzzaman et al., 2019b; Kwapinska et al., 2019).

The high nutrient and low heavy metal content of dairy wastewater may increase the potential for agricultural use of DPS (Table 3 and 4). Most wastewater from dairy factories is discharged after treatment and a small part is used for irrigation (Durham and Hourigan, 2007). The most important and high-nutrient effluent, whey, is usually reused as animal feed and is recovered to some marketable products like salts (Durham and Hourigan, 2007, Prasad et al., 2004). Agricultural land spreading, compost, and biofuel are some of the common pathways to utilize DPS and to add value towards circular economy (Daufin et al., 2001; Durham and Hourigan, 2007; Korsström and Lampi, 2001; Prasad et al., 2004; Ryan and Walsh, 2016) (Table 1). Such management practices have both environmental and agronomic benefits e.g. they provide a "circular economy" (EC, 2020) returning nutrients, trace elements and humus to soil, whilst removing such wastes from disposal sites e.g. landfill or incineration. Given the lack of native P in Europe, recovery of P from various agricultural, industrial and urban organic wastes may contribute to the reduction of dependency on import P from other countries.

Before DPS may be used as a fertiliser, issues such as the efficiency of nutrient recovery and the potential presence of pathogenic organisms and contaminating substances must be considered (Reijnders, 2014). The type of dairy plants from which the DPS originates may have a significant effect on its fertilizer value. For example, cheese factories generally have 50% more P than fresh milk dairies (Kwapinska et al., 2019). DPS and other co-products still have an unknown fertilizer equivalent or replacement value (FEV or FRV), which is a barrier for farmers to incorporate these options into their fertilizer application programme (Teagasc Greenbook, 2016). If these products are to be used in agriculture, their FEV needs to be quantified and the perception of these products needs to be converted from a waste to a fertilizer. Since the bio-based fertilizers from organic wastes may contain pollutants, many food companies will not use produce originating from land where they have been spread (Perkins, 2019). Although a completely different bio-based waste to DPS, there have been examples in Austria and Switzerland where land application of municipal human waste (often termed biosolids) has been banned (Kügler et al., 2004; Schweizerischer Bundesrat, 2003), whereas other European countries like Sweden and Germany advocate that this practice be phased out (Regeringskansliet, 2018; Wiechmann et al., 2013). Some studies found emerging contaminants in such waste e.g. pharmaceutical and personnel care products (Healy et al., 2017). There is therefore an impetus to fully characterise new bio-based wastes such as DPS not only in terms of their FEV but also in terms of their content. This will then enable these products to be certified as safe and usable in agricultural areas to build both fertility and soil quality (Amoah-Antwi et al., 2020). According to the new EU regulation (EC, 2016), bio-based fertilizer should be placed on the market and marked with CE only if they fulfil certain requirements including obligatory maximum contaminant levels, the use of defined component material categories, and labelling requirements to benefit from free circulation in the EU's internal market (Table 2). To reduce the risk to human health and environment, DPS and other co-products still need to be characterized in a more meaningful way which includes emerging contaminants in addition to the contaminants listed by EU regulation (Table 2) before they are put to use.

This current systematic review aims to examine DPS and the co-products of DPS in terms of their sources, treatment processes, storage and physico-chemical characterisation, FEV determination and potential emerging contaminant risk.

Table 1. *DPS generation (per unit volume/mass of processed milk) and disposal pathways in different countries.*

Country	Water consumption	Effluents loads	DPS volume	Disposal of wastes	Reference
EU	0.2-11 L/L processed milk	0.3×10^6-3×10^6 L (in a factory with capacity:10^6L milk/day)	1-3t dry matter sludge (in a factory with capacity: 10^6L milk/day)	wastewater: drained to rivers sludge: land spread	Daufin et al., 2001
EU	0.8-60 m^3/t processed milk	0.9-60 m^3/t processed milk	0.2-30 kg sludge/t processed milk	--	EC, 2006
Sweden	0.96-4.0 L/L processed milk	0.86-4.3 L/L processed milk	--	Landfill, compost, irrigation, biogas production. In Denmark, 2/3 sludge from dairies is irrigated on cultivated land and the rest is utilised in biogas production.	Korsström and Lampi, 2001
Denmark	0.60-1.9 L/L processed milk	0.75-1.5 L/L processed milk	--		
Finland	1.2-4.6 L/L processed milk	1.2-3.9 L/L processed milk	--		
Norway	2.5-6.3 L/L processed milk	2.0-3.3 L/L processed milk	--		
Ireland	--	--	--	Sludge: land spread (63%), compost (13.6%), or removed by licensed contractors (23.4%)	Ryan and Walsh, 2016
Australia	0.07-2.90 L/L milk	--	31kg organic waste/t product	Compost, fertilizer, stockfeed and recovery of marketable products.	Prasad et al., 2004
United States	--	0.10-12.4 kg /kg milk	--	Effluents: discharge into municipal sewage treatment system or irrigate on the land	Durham and Hourigan, 2007
United States	--	170-2081 m^3/d	--		Danalewich et al., 1998
UK	1.8 L/kg product	1-5 L/L processed milk	--	Sludge: landfilling	Klemes et al., 2008

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Dairy processing sludge (DPS) characterization

Present knowledge of dairy effluents nutrient and metal content

Dairy industry produces various dairy products such as sterilized and pasteurized milk, yogurt, ice cream, butter, cheese, and milk powder, with different processes such as pasteurization, coagulation, filtration, centrifugation, chilling (Carvalho et al., 2013). Dairy effluents vary significantly both in quantity and quality based on dairy factory characteristics (Janczukowicz et al., 2008) (Tables 1 and 2). The flow rates of dairy effluents vary due to scale, products, techniques, processes and equipment (Gutiérrez et al., 1991), and also vary diurnally (Danalewich et al., 1998). Milk processing rates, typically being higher in summer and lower in winter, also result in high seasonal variations in wastewater volume and properties (Janczukowicz et al., 2008). Moreover, the composition of these effluents varies greatly depending on the different types of products, system and operation methods (Carvalho et al., 2013). The effluent generally comprises dilutions of milk (or milk constituents including lactose, minerals, fat, whey and protein) lost in the technological cycles, starter cultures used in manufacturing, by-products (whey, milk and whey permeates), residues and contaminants from washing milk containers, equipment and floors, disinfectant applied in clean-in-place (CIP) processes, and other additives that may be used (Ahmad et al., 2019; Carvalho et al., 2013, Kolev Slavov, 2017). Some reported data on the characteristics of dairy processing effluent is given in Table 1. Dairy processing effluent is distinguished by high BOD, COD and nutrient concentration, and pH varying from 4-12. Such a large variation of the pH is attributed to the use of acid and alkaline detergents and sanitizers for washing (Britz et al., 2006). The residues of milk and milk by-products in the waste stream results in the significant fractions of the organic components and nutrients, especially the N and P contents are higher than those normally present in

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domestic sewage (Booker et al., 1999). Suspended solids are derived from coagulated milk, cheese curd fines or flavouring ingredients (Demirel et al., 2005). Extremely high sodium (Na) concentrations points out the large use alkaline detergents at factory (Table 4).

Table 2. *The requirements of EU bio-based fertiliser (EC, 2016)*

	Solid bio-based fertilizer		**Liquid bio-based fertilizer**	
Composition	**One macronutrient**	**More than one macronutrient**	**One macronutrient**	**More than one macronutrient**
Minimum required content				
Total N (%)	2.5	1	2	1
Total P (%)	2	1	1	1
Total K (%)	2	1	2	1
Sum of N-P-K contents	4		3	
Organic carbon (%)	15		5	
Contaminants limit values				
Cd (mg/kg dry matter)	1.5			
Cr^{6+} (mg/kg dry matter)	2			
Hg (mg/kg dry matter)	1			
Ni (mg/kg dry matter)	50			
Pb (mg/kg dry matter)	120			
As (mg/kg dry matter)	40			
$C_2H_5N_3O_2$	No present			
Micronutrients limit values				
Cu (mg/kg dry matter)	300			
Zn (mg/kg dry matter)	7800			

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Table 3. *Characteristics of dairy processing effluents*

Effluent type	**pH**	**BOD_5 (g/L)**	**COD (g/L)**	**TS (g/L)**	**TSS (g/L)**	**VS (g/L)**	**VSS (g/L)**	**FOG (g/L)**	**TN (mg/L)**	**TP (mg/L)**	**DOM (mg/L)**	**Reference**
Milk factory	5.5- 6.9	0.092-0.116	0.160-0.208	0.094-0.110							76.4-86.4	Mishra et al., 2000
Dairy plants (produce cheese)	6.2- 11.3	0.565-5.72	0.785-7.62	1.84-14.21	0.326-3.56	0.562-11.03	0.225-1.94		14.0-40.0	29-181		Danalewich et al., 1998
Mixed dairy	4-11	0.24-5.9	0.5-10.4	0.71-7	0.06-5.8			0.02-1.92	10- 660	0-600		Kolev Slavov, 2017
Milk reception	7.18	0.798	2.54		0.654			1.06				Janczukowicz et al., 2008
Butter	12.08	2.42	8.93		5.07			2.88				Janczukowicz et al., 2008
Cheese	7.90	3.46	11.75		0.940			0.331				Janczukowicz et al., 2008
Cottage cheese	7.83	2.60	17.65		3.38			0.950				Janczukowicz et al., 2008
Cheese whey	4.46	40	60	59	1.5							Gannoun et al., 2008
Cheese whey	4.0-4.6	10-12.5	8.8-25.6	7.0- 8.3	1.6-4.8			1.83-3.76	310- 356	6.6-7.2		Rivas et al., 2010
Hard cheese whey	5.80	29.48	73.45		7.15			0.994				Janczukowicz et al., 2008
Cottage cheese whey	5.35	26.77	58.55		8.13			0.492				Janczukowicz et al., 2008
Ice cream	5.2	2.45	5.2	3.9		2.6			60	14		Karadag et al., 2015

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Creamery	8-11	1.2-4	2-6	0.35-1	0.33-0.94		50-60	Demirel et al., 2005
Cleaning water	10.37	3.47	14.64	3.82		3.11		Janczukowicz et al., 2008

Table 4. *Concentrations (mg/L) of trace elements in dairy processing effluents*

Effluent type	**Cd**	**Fe**	**Cu**	**Pb**	**Zn**	**Ni**	**Na**	**K**	**Ca**	**Mg**	**Al**	**Co**	**Mn**	**Reference**
Dairy plants (mainly produce yogurt)	0.090	1.181	0.350	1.095	0.234	0.166								Afolabi et al., 2015
Creamery		2-5				0.5-1.0	170-200	35-40	35-40	5-8		0.05-0.15	0.02-0.10	Demirel et al., 2005
Cheese		0.039-4.33	0-0.03			0.012-0.071	263-1265	8.6-155.5	1.4- 58.5	6.5-46.3	0.063-0.257	0-0.007	0-0.835	Danalewich et al., 1998
Mixed dairy		0.5-6.7				0-0.13	123-2324	8-160	11-120	2-97		0	0.03-0.43	Demirel et al., 2005

Present knowledge of DPS nutrient and metal content

Dairy wastewater must be treated to meet licensed discharge limits before discharge to surface water bodies (e.g. rivers). Normally, there are three main stages of wastewater treatment (Figure 1). The primary treatments consist of sedimentation/physical screening to remove large particles or debris, flow and composition balancing to stabilize effluent, chemical addition to control pH, and dissolved air floatation (DAF) to remove fats, oil and greases (FOG) (Ryan and Walsh, 2016). Two types of biological degradation systems, aerobic and anaerobic systems, can be used in secondary treatment to remove organic materials. Large quantities of DPS are produced in this stage and pollutants can be absorbed into it. Aerobic biological techniques, including activated sludge process (ASP), sequencing batch reactors (SBRs), bio-towers or membrane bioreactors (MBRs), are carried out using dissolved oxygen (Ryan and Walsh, 2016). This is a reliable and cost-effective treatment in producing a high-quality effluent, but results in high DPS generation (0.6 kg dry DPS per kg of BOD_5 removed) and serious and costly disposal problems (Britz et al., 2006). The common anaerobic biological technologies involve anaerobic lagoon, upflow anaerobic sludge blanket (UASB), membrane anaerobic reactor system (MARS), and completely stirred tank reactors (CSTR) (Britz et al., 2006). Less DPS is generated in anaerobic digestion than the amount produced by aerobic processes (Britz et al., 2006). During dairy wastewater treatment, only one of them or a combination of the two can be used. Phosphorus is removed in the tertiary treatment through the use of chemicals like ferric sulphate and aluminium chloride, before final discharge (Britz et al., 2006; Ryan and Walsh, 2016).

The wastewater treatment processes within a dairy processing plant generates a specific DPS type, which can be predominantly categorised into (1) lime treated DAF sludge and (2) bio-chemically treated activated sludge (Ashekuzzaman et al. 2019a). The former is produced after chemical and DAF treatment of raw wastewater during primary treatment. The latter is stabilized sludge from secondary biological degradation treatment, which can be either aerobic or anaerobic, or a combination of the two. DPS contains casein, lactose, fat, valuable nutrients and organic matter (Singh et al., 2013). Pollutants, including non-biodegradable materials like heavy metals and other harmful components, can be absorbed in the DPS. However, information on the chemical composition of DPS is scarce. In a recent study by Ashekuzzaman et al. (2019a), four types of DPS were recognised across the two types: (1) AC: bio-chemically treated activated sludge, (2) DAF: lime treated dissolved air floatation sludge, (3) CM: combined treated (using both AC and DAF process), and (4) AD: anaerobically digested sludge. Nutrient concentrations were statistically different across four sludge types (Table 5).

In addition, storage is a fundamental step for the DPS application, because dairy products and sludge production is a continuous process while its utilization can be discontinuous. In many cases, sludge spreading is limited to one or two periods every year (Lue-Hing et al., 1992). In Ireland, farmers are prohibited spreading organic fertilizers from October to January (S.I. NO. 378 of 2006), which means that the sludge might be kept in the dairy plants and leads to the variation of sludge and greenhouse gas emissions during this period.

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There is no study on the variation of the properties of DPS during the storage so far and further study about this will be carried out in the future.

Table 5. Characteristics of DPS. Adapted from Ashekuzzaman et al. (2019a) and López-Mosquera et al. (2000).

Parameters	**Bio-chemically treated activated sludge "AC"***	**Lime treated DAF sludge "DAF"***	**Combined treated sludge "CM"***	**Anaerobically digested sludge "AD"**	**Dairy-plant sludge**
DM (% of wt.)	13.3	25.9	16.1	3.5±1.1	
OM (% of DM)	62.9	46.9	73.9	72.5±1.3	
pH	7.3	7.2	6.8	7.5±0.1	
TN (g/kg)	57.2	19.5	46.0	70.4±1.2	
TP (g/kg)	36.8	65.9	20.0	14.6±1.2	
TC (g/kg)	29.4	24.3	42.2	35.6±1.2	
K (g/kg)	7.2	3.9	2.9	6.1±1.1	
Mg (g/kg)	3.2	4.3	1.4	1.9±0.1	
S (g/kg)	4.8	2.1	7.6	5.3±0.7	
Na (g/kg)	5.3	3.5	3.6	19.9±3.0	
Ca (g/kg)	44.8	152.9	21.0	59.7±12.0	
Cr (mg/kg)	9.8	5.4	8.8	13.4±3.5	15.99±0.04
Cu (mg/kg)	12.6	5.3	17.3	38.2±6.7	58.55±0.08
Ni (mg/kg)	4.6	4.0	7.9	9.3±2.4	11.04±0.04
Pb (mg/kg)	<2.0	<2.0	<2.0	6.3±2.9	10.05±0.12
Zn (mg/kg)	75.2	54.7	109.8	217±46	289.74±0.67
Al (g/kg)	27.7	0.6	37.2	1.5±0.5	
Fe (g/kg)	1.5	1.1	1.8	0.7±0.1	
Co (mg/kg)	0.8	0.3	0.7	0.9±0.2	
Mo (mg/kg)	2.2	0.5	2.1	18.4±3.6	
Mn (mg/kg)	55.1	28.2	80.7	28.2±6.8	
Cd (mg/kg)	< 0.15	< 0.15	< 0.15	< 0.15	0.11±0.001
Hg (mg/kg)	n.a.	n.a.	n.a.	n.a.	0.08±0.02

DM=dry matter, OM=organic matter, TN=total nitrogen, TP=total phosphorus, TC=total carbon, n.a. = not available

**Values are presented in median value.*

Co-products of DPS and other bio-based alternatives

Several other bio-based products could be derived from dairy processing wastewater treatment, such as struvite and co-products produced from therochemical treatments of raw DPS resulting pyrochar, biochar and hydrochar. These have potential uses in agriculture as fertilizers, animal feedstocks or as soil amendments (Amoah-Antwi et al., 2020; Sadeghi et al., 2018; Uysal and Kuru, 2015).

Struvite (magnesium ammonium phosphate hexahydrate, $MgNH_4PO_4.6H_2O$) precipitate is normally formed in wastewater treatment plants (WWTP) during the anaerobic digestion process when significant levels of Mg occur in the wastewater (Booker et al., 1999). Sometimes, hundreds of tonnes of struvite may form and deposit on the walls of the digesters and connecting pipes, which results in downtime, loss of hydraulic capacity and increased maintenance costs (Booker et al., 1999). However, struvite precipitation has become a potential alternative approach of nutrient removal and recovery from wastewater and agricultural application (Uysal and Kuru, 2013). Uysal and Kuru (2015) detected high N, P and Mg contents in struvite precipitate produced from dairy industry wastewater, while heavy metal concentrations were below detection limits. The fertilizing effect of the struvite precipitate on maize was investigated by a pot trial and the results obtained show that struvite can be an effective source of fertilizer (Uysal and Kuru, 2015).

The carbonaceous materials obtained from the thermochemical conversion of biomass are considered as emerging organic amendments that can be applied as energy production, agriculture, carbon sequestration, wastewater treatment, bio-refinery, etc (Kambo and Dutta, 2015). They are coal-like solid material that is more stable, more carbon rich and less toxic than the feedstock (Atallah et al., 2020, Kambo and Dutta, 2015). Different thermochemical pre-treatment processes and conditions result in different final products. Pyrolysis is a thermal decomposition technology of OM (e.g. agricultural wastes, lignocellulosic biomass and sewage sludge) to convert biomass into valuable products like biochar, bio-oil and gas components at temperatures between 350 and 1000 °C in the absence of oxygen (Nanda et al., 2016, Ashekuzzaman et al., 2019b). Pyrolysed organic matter with a carbon content higher than 50% of DM are defined as biochar, otherwise, termed as pyrochar (EBC, 2012). Hydrothermal carbonization (HTC) is a thermochemical process at the temperature range of 180-260 °C to degrade the organic content of waste in the presence of water and produce a solid product, known as hydrochar (Kambo and Dutta, 2015). The feedstock, pre-treatment method and temperature are key factors of the physicochemical characteristics of chars, and play a vital role in determining their importance and application (Amoah-Antwi et al., 2020). Typically, biochars produced by high temperature pyrolysis (>550 °C) have high specific surface area (SSA) (>400 m^2/g) and more condensed polyaromatic structures and hence are good absorbents for agricultural and wastewater treatment industries due to high adsorption capacity (Amoah-Antwi et al., 2020; Kambo and Dutta, 2015). Biochars produced by low temperature (<550 °C) normally have a higher concentration of labile OM and macronutrients, and are more suitable for amending nutrient- deficient soils (Amoah-Antwi et al., 2020). Pyrochar may contain more ash than carbon because of the mineral-rich feedstocks (EBC, 2012), and is not suitable for energy production (Kambo and Dutta, 2015). Compared with biochars, hydrochars have lower SSA, higher energy density and H/C-O/C ratios due to the different treatment

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processes, which means hydrochar can be used for energy production (an alternate to coal) (Kambo and Dutta, 2015).

DPS could be potential candidate for thermochemical treatment due to its low heavy metal content. Sadeghi et al. (2018) spread the biochar derived from air-dried DPS over the surface of the small-scale boxes filled by an erosion-prone soil and found that the biochar increased C, N, OM and C/N of the soil. In addition, they detected that biochar production significantly decreases the heavy metal, N, P and K contents, and increased the C and C/N ratio. Their study showed great potential of DPS-derived biochar to be an eco-friendly soil amendment and carbonaceous adsorbent. Ashekuzzaman et al. (2019b) produced pyrochars from two DPS types, i.e. AC sludge and DAF sludge and used them as carbonaceous adsorbent for P removal from wastewater. The type, composition and the mineral composition (i.e. availability of Ca, Mg and Si) of DPS-derived pyrochar samples were associated with P removal process (Ashekuzzaman et al., 2019b). Atallah et al. (2020) carried out batch HTC experiments using DAF sludge to investigate the effects of changing temperature, residence time and water-sludge ratio on the yield and quality of the hydrochar. They found that the production of hydrochar improved characteristics of DPS and the increase in reaction temperature, residence time and water-sludge ratio increased the hydrochar yield along with their energy and carbon content, and decreased the oxygen and volatile matter content.

However, the production costs of chars materials are too high for large-scale agricultural use. More studies should work on this to find some more economical and effective production technologies. Thermochemical treatments increases the risk of producing chars with other highly toxic compounds produced from high-temperature reactions such as polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), dioxins, furans, and polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/Fs) (Amoah-Antwi et al., 2020, Kambo and Dutta, 2015).

Fertilizer replacement or equivalent value (FRV or FEV) of bio-based fertilizers

The FRV or FEV is defined as the application rate of mineral fertilizer to which the fertilization effect of organic waste on crop yield or nutrient uptake is equivalent (Brod et al., 2012). The efficiency of most bio-based fertilizers is normally lower than chemical fertilizers because of their comparatively low nutrient content, slow nutrient release rate and highly variable nutrient composition (Chen, 2006). The FEV calculations are used to estimate the efficiency of plants to uptake nutrients from organic fertilizers. There are two methods that can be used to assess the FEV. The most common one is determining FEV by solving the yield or nutrient uptake response function of crop applied by incremental additions of mineral fertilizer (Delin, 2011; Lalor et al., 2011). Linear polynomial, quadratic polynomial or cubic polynomials can be set up to get this relationship (Figure 2). Normally, cubic polynomials can fit better than the other two. As illustrated in Figure 2, the best fitted polynomial function, describing crop yield or nutrient uptake corresponding to different mineral fertiliser application rates, is used to determine the corresponding mineral fertilizer rate (x_1) to the crop yield or nutrient uptake (y1) of a bio-based treatment and x_1

can be expressed as percentage of total nutrient applied from that bio-based treatment to estimate FRV.

Another way to assess the FRV is calculated by the apparent nutrient recovery without using a response curve. The apparent nutrient recovery represents the nutrient fraction taken up by the test crop of total applied nutrients. FEV is the ratio of the apparent nutrient recovery of bio-based fertilizer and that of mineral fertilizer (Cavalli et al., 2016, Sigurnjak et al., 2019).

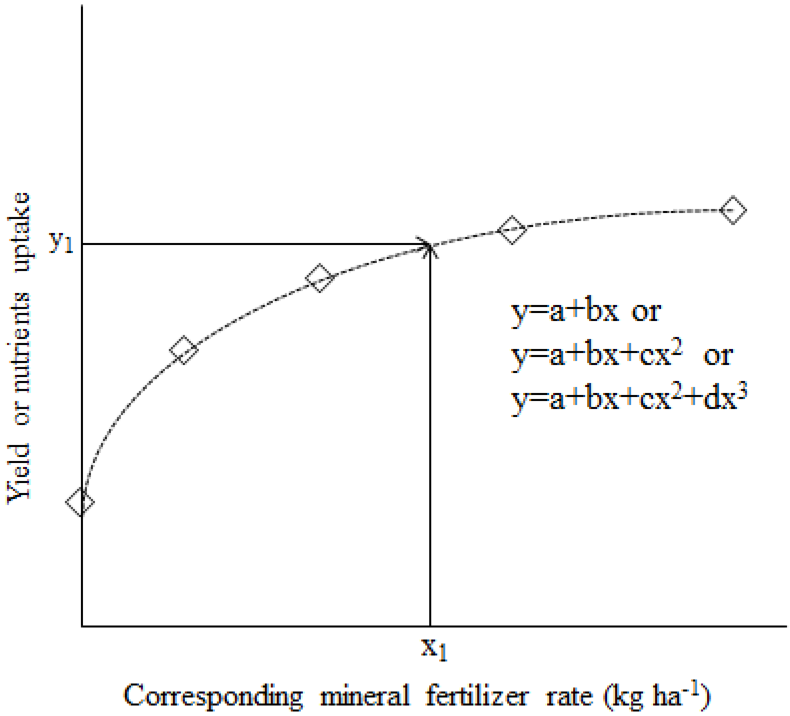

Figure 2. *Illustration of the calculation of FRV or FEV by response curve. a is the intercept (crop yield or nutrients uptake at 0 kg ha^{-1} of mineral fertilizer); b, c and d are the linear, quadratic and cubic coefficients, respectively.*

The FRV can both provide a quantitative estimate of the amount of efficient nutrients in bio-based fertilizer and theoretically estimate the actual price of it in comparison to mineral fertilizer, which can give farmers a decision support about how to use bio-based fertilizers and how much they can save by using them as part of their nutrient use efficiency plan (Ashezzuaman et al. 2019a). However, the results of FRV are not usually reliable, as FEV can not only be affected by the calculation methods, but also factors like type of bio-based fertilizers, different crops, fertilizer application time and method can have an effect on FRV (Schröder, 2005, Delin, 2011, Lalor et al., 2011, Sigurnjak et al., 2017). For example, different treatments of manure can affect the N content, composition and apparent N recovery (ANR) and nitrogen fertiliser equivalent value (FEV-N). Sigurnjak et al. (2019) used an ammonia (stripping-)scrubbing technology to recover N from pig manure. It significantly increased the total N concentration and recovered N entirely in mineral form in the end-products, which is similar as synthetic mineral N fertilizers. High concentration of available N (ammonium; NH_4-N) resulted in high crop yield, N uptake and FEV-N. Although NH_4-N is the available part for crop, increasing NH_4-N concentration may cause N losses by ammonia (NH_3) emissions (Fangueiro et al., 2015). Sigurnjak et al. (2017) supposed that acidification of animal slurry can reduce the NH_4-N concentration and, thus, may reduce NH_3 volatilization and subsequently increase FEV-N, but may be more effective on horticultural crops with a longer cultivation period. Application time is another important factor, as dynamic factors such as weather and temperature may have strong influence on available N. If fertilizer application time is too late in spring, the higher temperature and warmer temperatures increase soil and fertilizer surface temperatures, which result in greater NH_3 volatilization and reduced crop yield and NFRV (Delin, 2011). Some fertilizer application methods like trailing shoe also can reduce NH_3 emissions and increase FEV-N compared with more traditional broadcast methods such as splashplate (Lalor et al., 2011).

So far, studies of FEV mainly focused on the FEV-N of manure and slurry. As an important type of bio-based fertiliser, research on the FEV of DPS is scarce. Apart from N, there is almost no research on the FEV-P. According to the previous unpublished research from Teagasc (Ireland’s agriculture advisory service), FEV-P might be difficult to ascertain as there was no response with the corresponding mineral fertilizer rate if the P content in the soil is not low enough.

Potential Contaminants in DPS

Although DPS as a food processing industry waste might be much safer than the sewage sludge (Singh et al., 2013), a wide range of potentially harmful chemicals might be detected as some compounds may enter into the milk processing chain through various routes and ultimately accumulate in DPS (Figure 3). Lactating animals are exposed to various chemicals directly or indirectly via the agricultural and veterinary practices in the farm. The active ingredient may be absorbed by animals, subsequently excrete into the milk, and eventually enter the waste stream through residual milk in the factory. In addition, some common contaminants such as dioxins and heavy metals are likely to be found in the milk and dairy products, as they may enter and form incidentally during the production process (Fischer et al., 2011a). At present there is limited information available on emerging contaminants in dairy processes except heavy metals. In this study, we list the potential contaminants and their sources and fate in the DPS.

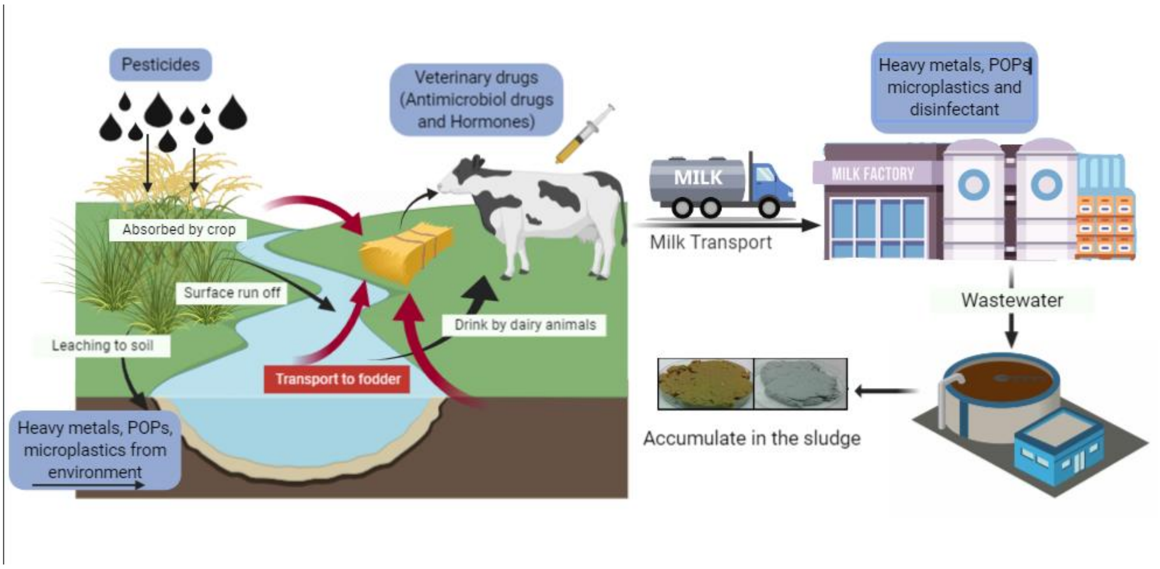

Figure 3. *The sources and fates of emerging contaminants in DPS*

Antimicrobial Drugs

Antibiotics including the β-lactams (penicillins, cephalosporins), tetracyclines, macrolides, aminoglycosides, quinolones and polymyxins are the most frequently and commonly used antimicrobial drugs in dairy cattle management (Fischer et al., 2011a). They are widely administered to treat, control and prevent spread of diseases of dairy cows such as mastitis, laminitis, respiratory diseases, and metritis, and to enhance animal growth and feed efficiency (International Dairy Federation, 1997). All the administered antibiotics could enter the milk and subsequently transfer to other dairy products to some extent, depending on their physicochemical properties and ability to interact with the fat and protein (Giraldo et al., 2017). Adetunji (2011) found streptomycin, penicillin and tetracycline residues in soft cheese and yoghurt. Rama et al. (2017) indicated that amoxicillin, penicillin G, and cloxacillin were the most frequently detected residues in the raw milk collected from six different major regions of Kosovo. Sniegocki et al. (2015) observed that chloramphenicol can be easily transferred from raw milk to the commercial butter, white cheese, sour cream and whey as this antibiotic is accumulated more in dairy products with high fat content. However, antibiotic residues in cheese generally decrease during ripening except some highly stable substances such as quinolones (Quintanilla et al. 2019).

Hormones

Endogenous hormones occur naturally in food of animal origin because animals can excrete steroid hormones. The excrete amount depends on age, state of health, diet, or pregnancy (Silva et al., 2012). Hormones are also used on to promote growth, increase food production, medical treatment and improve reproductive performance, but the use of anabolic hormones in animal production is prohibited in EU (EC, 1996; EC, 2003; IDF, 1997). 75% of Milk is produced predominantly by pregnant cows, which means that milk represents an important source of steroid hormones (Goyon et al., 2016). The natural hormone content of milk is typically between 40 and 500 µg/kg for the steroids (IDF, 1997). During the processing in the dairy plants, the residual hormones will enter the effluents through residual milk. In the WWTP, part of hormones are removed through sorption to suspended solids and degradation, followed by removal of the excess sludge (Silva et al., 2012), which means that hormones may accumulate in the DPS.

Pesticides

Pesticides including insecticides, herbicides, rodenticides and fungicides etc. applied in agriculture have been shown to transfer to dairy animal bodies through feed and fodder and accumulate in them (Rather et al., 2017). In addition, to protect the animals against disease from mites, ticks and insects, some pesticides are directly sprayed onto the animals when they are housed. Animals will absorb pesticides orally, cutaneously, or via inhalation in such closed environment (Fischer et al., 2011a). Currently, common pesticides include organophosphate (OP), pyrethoids and carbamates (CB) that can be used on both routes (from environment and directly application on animals) and lead to the bioaccumulation in the dairy products (Akhtar and Ahad, 2017). The pesticides used in the cropping system and their metabolites will be lost to the environment via volatilization, aerial drift, runoff to surface water bodies and leaching into groundwater basins (National Research Council, 1993), and can accumulate in the dairy animals or directly compromise water used in the dairy factory. The residues of organochlorines (OC) and their metabolites also need to be considered. Although they were banned in many countries since 1970s, the residues can still be found in the environment due to their persistence and thus prolonged efficacy (Fischer et al., 2011a, Akhtar and Ahad, 2017). There is a vast list of pesticides commonly used presently or in the past in agriculture with various levels of persistence in the soil, bedrock and water phases (McManus et al., 2017). This could have implications for grazing animals especially on heavily drained soils where for example 2-methyl-4-chlorophenoxyacetic acid (MCPA, high solubility and low adsorption to soil matrix) is used to clear vegetation and has been found to have a much longer residence time in anaerobic waterlogged conditions (Morton et al., 2019). High proportions of grassland areas in Ireland for example (30%) exist in such areas.

From the US Food and Drug Administration data, DDT and its metabolites DDE, and dieldrin are the most commonly detected pesticides in foodstuffs, including baked goods, fruit, vegetables, meat, poultry, and dairy products (Schafer and Kegley, 2002). The OC pesticide chlordane was found at a concentration of 1 ng/mL in one of the 35 raw milk samples (Fernandez-Alvarez et al., 2008). Golge et al. (2018) optimized a determination method for 167 pesticide residues in milk and milk products and then tested its applicability by analysing 92 real samples including raw milk, whole UHT milk, Feta cheese and cream obtained from retail markets in Turkey, but none of the 167 pesticide residues were detected.

Disinfectants

Each procedure of milk and dairy products process requires cleaning and disinfection to ensure removal of the bacteria and milk residues from all contact surfaces, including all processing equipment, transfer lines, tanks, trays, bins, blenders and conveyors (Cardador and Gallego, 2015). The most commonly used disinfectants are iodine liberating agents, chlorine-containing substances, quaternary ammonium compounds and hydrogen peroxide (Fischer et al., 2011a). A large amount of cleaning and disinfection agents enter into the dairy wastewater with the rinse-and-wash cycle of a CIP system. Furthermore, using inadequately treated water to rinse and wash can be another source of contamination (McCarthy et al., 2018). Milk and dairy products can also be contaminated with such treated water and inadequate post-rinse and drainage. In addition, the disinfectants are directly applied in the dairy wastewater to kill pathogens (e.g. fecal coliform and total coliform) during the wastewater treatment (Akhlaghi et al., 2018). The residual of disinfectants could be either in their original state or as disinfection by-products (DBPs). Iodine sanitizers (usually as iodophors) are widely used in teat and skin disinfectants, filling/packaging machines, culture processing equipment, drop hoses, and hand dipping stations (Hladik et al., 2016). Iodinated DBPs are considered as one of the most toxic DBPs but have been tested less frequently than chlorine DBPs (Postigo and Zonja, 2019). Hladik et al. (2016) found trihalomethanes (THMs), including iodinated THMs in the dairy wastewater and surface waters that receive dairy effluents (either directly from the dairy or through WWTP). Sanitation of water and equipment with chlorine-containing substances such as chlorine gas (Cl_2), chlorine dioxide (ClO_2), chlorhexidine and hypochlorite (ClO^-) remains common practice due to chlorine's bactericidal and oxidative properties (McCarthy et al., 2018). Chlorine reacts with any natural organic matter present in milk to form chlorine DBPs (Cardador and Gallego, 2015). Cardador and Gallego (2015) tested 84 milk and dairy products samples, 17 of them were found positive for haloacetic acids (HAAs), the major class of non-volatile DBPs. The HAAs found in commercial samples can be attributed to contamination within the industrial processes like the washing of packages and equipment.

Persistent organic pollutants (POPs)

There are thousands of persistent organic pollutants (POPs) widely spread in the environment. POPs tend to accumulate in the food chain because of their lipophilicity and low biodegradability (Jones and Voogt, 1999). Since POPs are occurring ubiquitously, the dairy animals are at danger from various sources of POPs, and these contaminants can transfer to the milk. In addition, some POPs such as PAHs, PCBs, dioxins, furans etc. are common by-products or formed accidentally in industrial processes, and subsequently enter the wastewater and sludge (Fischer et al., 2011b). PAHs are generally formed through a series of combustion processes occurring in industrial units. Boruszko. (2017) detected 16 PAHs contents in three types of DPS and found 689 μg/kg DM in excess sludge, 95μg/kg DM in post-flotation sludge, and 497.7 μg/kg DM in a mixture of excess and flotation sludge, which are considerably lower than those in municipal sewage sludge. There was a national survey carried out in April 2006 in France to assess the concentrations of PCDD/Fs as well as PCBs in 239 raw milk samples. The average PCDD/Fs and PCBs concentrations were 0.33 pg toxic equivalent (TEQ)/g fat and 0.57 pg TEQ/g fat, respectively (Durand et al., 2007). Mamontova et al. (2007) found PCBs residues in milk and obtained a good correlation between PCB levels in autumn milk and in soil. Furans can be formed from the dehydration of sugars and would be expected to be found in dairy products that have been heated. Heaven et al. (2014) found three analogues of furan in milk samples.

Microplastics

Plastic particles with a diameter ranging from 0.1μm to 5 mm are defined as microplastics and are a widespread anthropogenic pollutant in the environment with the extensive use of plastic (Phuong et al., 2016). Microplastics are mainly derived from synthetic fibres in clothing, industrial processes and personal care products, such as face cleaning soaps (Åström, 2016, Fendall and Sewell, 2009, Mahon et al., 2017). As an important food processing industry, the fate and sources of microplastics during the production process of dairy industry are largely unknown. The possible risks of milk contamination for microplastics may occur from cleanness procedure equipment, the environment, as well as water supply conditions and inadequate handling of milk (Kutralam-Muniasamy et al., 2020). In addition, plastic based packaging materials may lead to the microplastics contamination in milk. Kutralam-Muniasamy et al. (2020) collected 23 milk samples in Mexico and tested for the occurrence of microplastics. Results showed the ubiquity of microplastics in the samples with an average of 6.5 ± 2.3 particles/L. The microplastics entering the internal wastewater treatment are effectively removed and will concentrate in the DPS (van den Berg et al., 2020).

Conclusion

DPS and struvite derived from a WWTP are good alternatives to chemical fertilizers because of their high nutrients content and low heavy metal content. Thermochemically treated products such as biochar, pyrochar and hydrochar, using DPS as a feedstock, can be used as a clean source for soil amendment and as a sorbent in wastewater treatment. Although there has been a lot of studies about the technologies to produce chars materials, they are still not widely used because of the high production costs. Before putting these products into the markets and in large-scale use, their fertilizer efficiency and potential risk should be estimated, but there is very limited data available so far. One of the major challenges of using DPS in agriculture is the uncertain FEV, as studies of FEV mainly focused on the FEV-N of manure and slurry. The research on the FEV of DPS, especially FEV-P is scarce. In addition, dairy products may be contaminated with a wide range of potentially harmful chemicals like antimicrobial drugs, hormones, pesticides, disinfectants, POPs, mictoplastics. These compounds can enter dairy products through various routes and then may leave trace levels of residues in the DPS, which will definitely bring some environmental burdens. However, the studies on DPS characteristics have mainly been focused on nutrient and heavy metal content. Potential emerging contaminants in the DPS need to be explored in more detail. Other co-products from dairy factory like struvite, biochar, pyrochar and hydrochar also need to be characterised for all constituents e.g. nutrients, metal and emerging contaminants.

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A review of LCAs studies analysing dairy processing water treatment and P-recovery technologies (ESR 11)

In 2014 the European Commission declared phosphorus (P) a critical raw material. P is a precious and finite resource, important in the global agricultural food system. On the basis of this awareness, individual researchers proposed an alternative approach to extract phosphorus, which consists of its recovery from wastewaters rich in nutrients and low in contaminants, or rather the dairy processing wastewater (DPW). Many technologies exist or are under development for P-recovery or recycling from wastewater.

However, the efficiency of these technologies and their adaptation to DPW are dependent on a low environmental impact. A common method to assess the environmental impact is Life Cycle Assessment (LCA). In the study described in this report, a review of earlier LCA studies on dairy activities and P-recovery technology was undertaken. This review aims to evaluate how effectively the LCA methodology has been adapted to assess P-recovery technology and wastewater treatment (WWT) of DPW and to compile earlier results. In total forty-four LCA studies, were examined and discussed to explore the impacts of these processes.

The review of these articles led to the understanding that not all LCAs on dairy industries even include the WWTP and if it is considered, the relative environmental impact results are not consistent. Moreover, relevant P-recovery technologies had been assessed using LCA but not for DPW. Most of these studies also assessed the utilisation of the P contained in the final by-products obtained from the sludge treatment. Regarding the LCA approach, insights and methodological practices were synthesised and discussed.

However, for the possibility to use the information found in reviewed studies, several weaknesses were identified; important details on chemical suppliers, inflow wastewater, effluents and attribution of the environmental impacts is missing. It was found that the employed allocation method is crucial, and detailed guidance is missing. The lack of information, data or details make it difficult to conduct LCA on these topics. This study will contribute to the further work on LCA for P-recovery technologies implemented in DPW treatment plants.

Introduction

Phosphorus (P) is a chemical element that does not occur in nature other than in the form of phosphate, a chemical derivate of phosphoric acid. There are several forms of phosphate in nature, but the most used for human activities is derived from rock-phosphate rock. In this rock, phosphorus is present at an appreciable concentration in only a few minerals, primarily fluorapatite (Filippelli, 2008).

Phosphorus is an important nutrient because it is essential for all biological life. One example is for bones and teeth strength as a result of hydroxyapatite, which contains phosphorus; an adult human contains around three quarters of a kilogram of phosphorus. Further, in the Calvin cycle in our metabolism, adenosine triphosphate (ATP) plays an important role, and the double helix of the DNA is only possible because of a phosphate ester bridge (Filippelli, 2008). However, besides being an essential nutrient in biological systems, this element is also important for economic growth in several industrial sectors. The fertilizers and food industries are the dominating user sectors of phosphate rock and agricultural sectors are the main end users of phosphorus.

However, phosphorus is currently accompanied by various environmental problems, as the industrialization of food production in order to feed a rapidly expanding population is giving rise to serious leakage of phosphorus through the global agricultural food system; this is particularly pertinent in the dairy industry. The leakage of phosphorus to water from dairy production is considered a serious environmental problem. Phosphorus that ends up in water may give rise to eutrophication, when it promotes excessive growth of algae and cyanobacteria (Larsson, 1985). When this biomass dies and biodegrades, it also depletes oxygen, causing the death of fish and mollusks (Larsson, 1985). Losses of phosphorus from dairies are causing environmental damage and are ultimately putting food safety at risk.

Today, in addition to this environmental problem, phosphorus is considered a particularly important element with a substantial and non-replaceable role in the environment. It is a finite resource, a non-renewable element that, because of economic growth, will be a limited resource in the future. The awareness of an overall increasing demand for phosphorus and the fact that this element is not a renewable resource, brought the European Commission to declare that phosphorus, or better phosphate contained in rock, or rock phosphate, as a critical raw material in 2014 (EC, 2014).

Moreover, there are a variety of policies and Regulations focused on P-recovery across Europe. These conditions have led do to research of P-recovery technologies and nutrient recovery has been getting significant attention across Europe (Nèmethy, 2016). Today, several projects within the Horizon 2020 program are aiming to increase the European phosphorus recycling rate from wastewater, and REFLOW is one of those.

REFLOW focuses on the recovery of phosphorus from dairy processing wastewater (DPW) and by recycling of fertiliser products, the enabling of sustainable expansion of the dairy industry in Europe. The dairy sector was considered in this research largely due to the abolishment of dairy quotas in April 2015 (EUROSTAT, 2018), after which the European Union became the largest producer of cheese in the world with 9.5 million tonnes in 2015 (EUROSTAT, 2017). Moreover, phosphorus is an element present in milk and consequently also in dairy products, as well as in by-products and waste flows. A large share can be found in whey in the form of phosphate (842.7 P_2O_5 mg/L of whey) (Ercoli et al., 2007). The whey can be reused to produce cheese or can be emitted out of the dairy together with the wastewater, which increases the concentration of phosphorus in the receiving water if not properly treated.

From the perspective of the REFLOW project, the part of the work reported in this section of the literature review report focuses on providing input to environmental assessment of P-recovery from waste flows in dairy industry. Therefore, this study will answer the follow questions:

- Are earlier Life Cycle Assessments (LCAs) of dairy production looking into the wastewater treatment? Are the considered dairy industries performing wastewater treatment on site?
- What methodologies were selected in earlier LCAs of dairy production?
- When it is evaluated, which impact does the dairy wastewater treatment (WWT) have on the environment?
- What methodologies were selected in earlier LCAs of relevant P-recovery technologies?
- Which impact does these P-recovery technologies have on the environment?

It is of interest to look at both the methodologies applied in the assessments in earlier studies and to look at what the results can tell us about the environmental performance of the specific technologies. To answer the above questions, a review was first made of LCAs on dairy activities to shed light on how these flows have been considered in earlier studies and what environmental impact the treatment of these flows has. Furthermore, a review of earlier LCAs on relevant P-recovery technologies was also performed.

Methods

Literature Review methodology

In order to answer the research questions and complete this state-of-the-art review, 44 articles were selected and consulted. 24 articles are about the LCA on dairy and 20 are about LCA on P-recovery technology. These articles were selected because they studied the impacts from the production of a variety of dairy products and from P-recovery technologies applied in a wastewater treatment plant (WWTP), through use of the LCA method. During this literature review, some articles were rejected because the LCA studies focused on only the raw milk production, before its ingress into the dairy factory, or on CIP (Cleaning In Place), without considering the dairy production, or in the case of the second group of articles, because the P-recovery technologies studied were not aligned with the REFLOW project aim.

LCA methodology

LCA is a method for assessing potential environmental impacts of a product life cycle, from raw material acquisition through production, use, recycling and final disposal (ISO 14044, 2006). The LCA method is described in the international standard ISO 14044. On the basis of this standard, an LCA study consists of four phases: goal and scope definition, Life Cycle Inventory (LCI), Life Cycle Impact Assessment (LCIA) and interpretation of the results. In this literature review, important methodological choices in each of these four phases of LCA were analysed for published LCA studies on the dairy processes and of relevant P-recovery technologies.

Goal and Scope definition

In the first step of an LCA, it is important to define the goal and scope of the study. The goal is the contextual aspect of the LCA study that consists of determining the reasons for the specific research, which the audience is and the subject of analysis. The scope, however, concerns modelling aspects of the LCA study, and it consists of the establishment of the Functional Unit (FU) and system boundaries. The FU will be the subject of the analysis, the reference flow to which all the other modelled flows of the system are related, and it is expressed as a quantity (Baumann & Tillman, 2004). The system boundary defines which activities shall be included within the LCA. The selection of the system boundary shall be aligned with the goal of the study (ISO 14040, 2006). The single units in the systems of a process or service that are being analysed will here be referred to as activities.

Life Cycle Inventory (LCI) analysis

The LCI is the "*phase of life cycle assessment involving the compilation and quantification of inputs and outputs for a product throughout its life cycle*" (ISO 14044, 2006). The LCI includes the construction of a model represented by a flowchart, in line with specifications provided in the goal and scope for system boundaries, data collection and calculation of the environmental loads of the system in relation to the FU (Baumann & Tillman, 2004).

Data quality

Data quality is one of the critical points of the LCA. Data are here categorised as primary, secondary and "surrogates" [1]. The primary data are collected/measured directly by the producing company, public companies, or consumer behaviour survey, the secondary data are collected from databases and software, industry sector associations or LCA reports, and "surrogates" data sources are for example statistics, technical literature, legislation or assumptions.

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[1] M. Janssen, personal communication, lecture – LCA course

Allocation approaches

The two cases of LCA studies that were taken in consideration, the dairy process and the P-recovery technology, are complex systems that produce more than one product. These systems are therefore multioutput systems.

In this case, ISO 14044 recommends an allocation approach or the expansion of the system boundary, if subdividing the system is not possible (ISO 14044, 2006). *The allocation is the partitioning of the input or output flows of a system between the product under study and additional products (ISO 14044, 2006).* According to ISO 14044, wherever it is possible, the allocation should be avoided by dividing the unit process into sub-processes and collecting input and output data related to these sub-processes (FIL-IDF, 2015). Where allocation cannot be avoided, the standard recommends a partitioning, based on a physical relationship, of input and output between products of the system considered (FIL-IDF, 2015). Where physical allocation is not possible, the ISO 14044 suggests that input and output might be allocated between co-products in proportion to the economic value of the products (FIL-IDF, 2015).

Another option is the physico-chemical allocation matrix advised by the International Dairy Federation (IDF) (Finnegan et al., 2018).The physico-chemical allocation matrix has been developed for the dairy manufacturing industry to enable better allocation of resources to dairy products given the whole of plant information and consists of a process of subtraction/substitution to determine average resource use and wastewater emissions for individual dairy products from multi-product manufacturing plants (Feitz et al., 2007).

The allocation procedure recommended by the international standard ISO 2006 does not distinguish between different objectives of the LCA, but it has been stated that the choice of allocation method should depend on the goal and scope of the LCA study (Consoli, 1993). One way to divide LCAs into different types if´s to talk about consequential LCAs (CLCAs) and attributional LCAs (ALCAs) (Tillman, 2000). It has been suggested that in CLCA which focuses on studying how impacts change when the system is changed, it is appropriate to avoid partitioning through substitutions and in ALCA which focuses on mapping impacts related to a fixed system, it is appropriate to solve the allocation problems through partitioning (Tillman, 2000).

Life Cycle Impact Assessment (LCIA)

The LCIA aims at describing the environmental consequences of the environmental loads quantified in the inventory analysis (Baumann & Tillman, 2004). The impact assessment is the result of an environmental load “translation” from the inventory results into environmental impacts (Baumann & Tillman, 2004). The results of the LCIA are typically classified into numerous parameters and further reduced into environmental categories. In the characterization step, these can be further weighted and grouped into impact categories (ecosystem quality, human health, and resources) (Baumann & Tillman, 2004).

In this work, the life cycle impact categories used in the reviewed studies were analysed and merged into groups. As an example of inconsistencies between studies, some studies considered climate impacts using the Global Warming Potential (GWP) indicator but under the name of Climate Change (CC) or Carbon Footprint (CFP) and some. Further, energy use was typically evaluated as Cumulative Energy Demand (CED) but also under the name of Primary Energy (PE) or Energy Intensity (EI).

Interpretation of results

According to ISO 14040, the life cycle interpretation is the phase of the LCA in which the total results obtained from the LCI and LCIA are combined and discussed in accordance with the defined goal and scope, in order to form the basis for conclusions, recommendations, and decision making (ISO 14040, 2006). Often, this also leads to that the analyst needs to go back and change aspects of the goal and scope definition. Most LCAs are therefore iterative processes.

Results from review of LCAs on dairy industries

This first results section deals with the review on LCAs of dairy products. This section includes a compilation of information from the review relative to the LCA method that will be considered for future LCA studies. In Table 1 there is a summary of the dairies considered in the published LCA studies. From Table 1, it is possible to see that cheese is typically the main product of the dairies assessed with the LCA method. Considering the aim of the REFLOW project, it was particularly important to analyse if and how the WWT was included in the LCA. From *Table 2*, it is possible to see that out of the 24 studies found, 18 studies considered the WWT in the LCA and out of these, 8 studies are on dairies which perform the WWT on-site. Generally in dairy industries, when a significant amount of wastewater is produced or when the municipality´s WWTP is already running at full capacity, it is required by law that a pre-treatment is done before the wastewaters are sent into the sewer (Water Online, 2016) or to wetlands (Grundfos). The results section is broken down into the four phases of LCA, in order to discuss the main outcomes from the review in a structured way.

Goal and Scope

Goal

The main goal of all these studies is to assess the environmental impacts of dairy products. These studies were performed either to compare two or more dairy processes, to assess which product of a single dairy has the highest environmental impact or to identify the hotspots in a dairy during the production process.

Functional Unit

Once the goal has been decided, that is to compare or assess the environmental impact of dairy processes, the FU and system boundaries are established. The unit advised by IDF

(Industrial Dairy Federation) for the case of an LCA on a dairy process is one kilogram of product, packaged and ready to be distributed *(FIL- IDF, 2010).* From *Table 2*, it is possible to see that most of the studies indeed used 1 kg of the final product as FU. This FU unit is recommended for solid products. The Eide (2002) study, however, which focuses on a dairy that produces only drinking milk, uses 1000 litres of drinking milk produced as FU. Palmieri et al, (2017), Nilson et al., (2010) and Cannellada et al., (2018) all deviate from recommendations in that other masses than one kilogram are used: 123 g, 500 g and 4770 kg of the final product. Another important exception is the FU used by *Vergé (2013),* since it focuses instead on 1 kg of protein.

System Boundaries

For analysing the system boundaries in terms of which parts of the life cycle are covered, a model suggested by Finnegan was further developed, see *Figure 1*. The letters A to H represent the stages in system boundaries of the 24 studies. The overall results relative to the system boundary are presented in *Table 2*. Analysing the system boundaries of these LCA studies, it is notable that processes before the dairy industry, such as feed and raw milk production, are included in most of the studies. Other relevant activities that could be included within the system, besides the dairy process itself, are the transport of the raw milk, from the farm gate to the dairy, freshwater usage and wastewater treatment, usage of energy, consumption of energy, waste management, and release resulting from processes, production, delivery, and consumption of operating materials *(FIL- IDF, 2010).* These reviewed studies could be classified into cradle-to-grave (A-H), from the raw milk production, including also feed production, to the commercialization and disposal of the final product, cradle-to-gate (A-E) from the dairy farm to the end of the dairy production and packaging, or gate-to-gate (C-E) from the delivery of the raw milk to the packaging stage. An important aspect that should be considered is that WWT in some cases is located in the dairy processing stage (D), inside the dairy industry. 18 LCA studies include WWT in the system boundaries and 8 of the dairy processes perform the WWT on site. The DPW can be treated on-site with treatment that consists on anaerobic digestion and activated sludge systems with denitrification (González-García, Castanheira, et al., 2013) or it can be purified in a municipal WWTP followed by mechanical, biological and chemical treatments, including the sludge digestion (Palmieri et al., 2017).

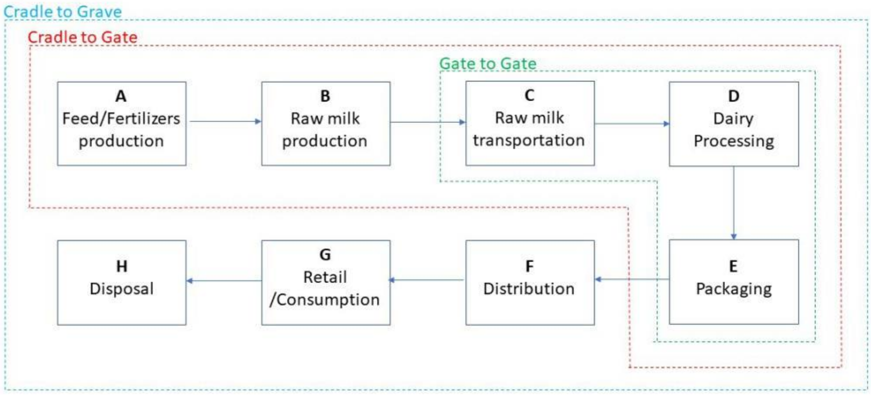

Figure 1. *System boundaries and processes in the life cycle of dairies assessed with the LCA method. The letters A to H indicate the stages used in the Table 2. This figure was adapted from Finnegan et al., (2018)*

Table 1. *Main characteristics of the dairies that were evaluated with the LCA.*

Study	Location of study	Dairy Products								
		Fluid milk	Cheese	Butter	Milk pwd	Whey Pwd	Cream	Yoghurt	UHT milk	Other
Finnegan et al., 2017	Ireland			X	X					
Djekic et al., 2014	Serbia	X	X	X			X	X	X	
Mondello et al., 2018	Italy		X							
Palmieri et al., 2017	Italy		X							
Broekema & Kramer, 2014	Netherland	X	X							
González-García, Hospido, et al., 2013	Spain		X							
González-García, Castanheira, et al., 2013	Portugal		X							
Kim et al., 2013	USA		X			X				
Dalla Riva et al., 2018	Italy		X							
Eide, 2002	Norway	X								
Santos et al., 2017	Brazil		X							
Flysjö et al., 2014	Denmark	X	X	X	X	X	X	X		X
Aguirre-Villegas et al., 2012	USA		X							
Van Middelaar et al., 2011	Netherland		X							
Nilsson et al., 2010	UK Germany France			X						X
Mahath et al., 2019	India	X		X			X			X
Canellada et al., 2018	Spain	X								
Doublet et al., 2013	Romania	X	X	X			X	X		X
Vergé et al., 2013	Canada	X	X	X	X	X	X	X		X
Flysjö, 2011	Denmark			X						X
Berlin, 2002	Sweden			X						
Yan & Holden, 2018	Ireland			X	X					
Yan & Holden, 2019	Ireland			X	X					
Dalla Riva et al., 2017	Italy		X							

Table 2*. Goal and scope characteristics in earlier LCAs of dairy production.*

Study	Functional Unit	System Boundary	WWT in the LCA	WWWT on site
Finnegan et al., 2017	1 kg of packaged milk powder 1 kg of packaged butter	C - E	X	X
Djekic et al., 2014	1 kg of final product	B – E	X	
Mondello et al., 2018	1 kg of packaged Pecorino cheese	A – E	X	
Palmieri et al., 2017	123 g of mozzarella cheese made from 1l of cow milk	A – D	X	
Broekema & Kramer, 2014	1 kg packed semi-skimmed milk 1 kg packed semi-cured gouda cheese	A – E	X	
González-García, Hospido, et al., 2013	1 kg of cheese	A – E	X	
González-García, Castanheira, et al., 2013	1 kg of mature cheese	A – E	X	X
Kim et al., 2013	1 tonnes of cheddar cheese consumed 1 tonnes of mozzarella cheese consumed 1 tonnes of mozzarella cheese consumed	A – H	X	X
Dalla Riva et al., 2018	1 kg of Asiago cheese	A – D	X[d]	
Eide, 2002	1000 l of drinking milk	A – G	X[d]	X
Santos et al., 2017	1 kg of cheese	D – E		
Flysjö et al., 2014	1 kg of product [a]	B – F		
Aguirre-Villegas et al., 2012	1 kg of cheddar cheese	B – E		
Van Middelaar et al., 2011	1 kg of semi-hard cheese	A – G		
Kim et al., 2013	500 g of packaged butter/margarine	A – F [b]		
Mahath et al., 2019	1 kg of product	D – E [c]		X
Canellada et al., 2018	4770 kg of cheese	B – G	X[d]	
Doublet et al., 2013	1 kg of product	A – D	X	
Vergé et al., 2013	1 kg of product 1 kg of protein	A – D	X	
Flysjö, 2011	1 kg of packaged butter or blend provided at the costumer 1 kg of packaged butter or blend consumed	B – G	X	X
Berlin, 2002	1kg of Ångsgården cheese wrapped in plastic	A – G [e]	X[d]	
Yan & Holden, 2018	1 kg of final product	C – D [e]	X	X
Yan & Holden, 2019	1 kg of solids in the final product	B – F	X	X
Dalla Riva et al., 2017	1 kg of mozzarella produced	B – H	X	

a. *The study does not specify the functional unit but shows the results of carbon footprint per kg final product.*

b. *The starting point of the studied system is extraction of raw material for the production of ingredients, materials and fuels required to produce butter and margarine.*

c. *The WWT in this case is called Effluent Treatment Plant (ETP): the dairy wastewater is treated in the ETP before release into the aquatic environment.*

d. *The waste management considered in this LCA study include all the general waste treatment, without to specify WWT*

e. *The phase D, other than the WWT, could include also the production of other ingredients and the waste management (e.g. the use of effluents, sludge or other by-products on land).*

LCI

Data quality

The main elements of the LCI considered in the studies are raw milk production and transportation, energy usage, water consumption, wastewater treatment, packaging of products, chemicals and additional ingredients consumption. Table 3 Table 3 provides more detail about the inventory data. Since the aim of this report is to assess the WWT process and its possible impacts on the environment, among all these elements cited above, the data considered for this literature review are relative to raw milk used, water consumption, water treatment, energy consumption, and additives, cleaning agents and refrigerants used. Both primary, secondary and "surrogates" data had been used, as specified in Table 3 (FIL- IDF, 2010). Some of the data in Table 3 were converted from the publication through several calculation to provide comparable results per 1 kg of average product output.

Allocation

The choice of allocation method is important in LCAs on dairy, influencing final results per FU. A comparison of results using different allocation strategies can be a useful approach if LCA is to be used as a decision support tool. All the allocation approaches that were used are among the ones recommended by ISO (FIL- IDF, 2010). An overview of the allocation approach adopted in the reviewed studies is presented in Table 4. Generally, to distribute the environmental impacts between co-products in multi-output processes, economic allocation is employed and, in some cases, physical allocation. Physical allocation is sometimes also performed to check the robustness of the results from economic allocation. Physical allocation is based on physical characteristics of products and in the case of these LCA studies, mass and energy are the two physical characteristics mainly used. In the Finnegan et al., (2017), Mondello et al., (2018), Nilsson et al., (2010), Vergé et al., (2013) studies, mass and energy allocation were preferred because they allow flow analyses that refer to a physical unit of a product. Mass allocation is adopted by Aguirre-Villegas et al., (2012), Dalla Riva et al., (2017), Djekic et al., (2014), Eide, (2002), Flysjö et al., (2014), while energy allocation is used by Yan & Holden, (2019).

Table 3. *LCI summary for the manufacturer of 1 kg of final product of the dairy processes assessed with the LCA.*

Study	Inputs					Outputs	Source
	Raw milk	Water (l/kg)	Energy consumption (kWh)	Additives (g)	Cleaning agents (g)	WW (l/kg)	
Finnegan et al., 2017[a]	6.695	7.6	2.25		15.15	9.6	Primary data
Djekic et al., 2014 [a]	5.67	6.82	0.88		15.84	6.44	Secondary data
Mondello et al., 2018 [b]	5.52	82.04	7.22	4		81.45	Primary & Secondary data
Palmieri et al., 2017		18.13	0.23				Primary & Secondary data
Broekema & Kramer, 2014							surrogate data
González-García, Hospido, et al., 2013 [b]	11	51.04	0.7154	23.27	12.82	21.86	Secondary
González-García, Castanheira, et al., 2013 [c]	7.98	23	1.08	138			Primary & Secondary data
Kim et al., 2013 [d]	42.25	44.26	14.23	2.88	455	75.07	Secondary data
Dalla Riva et al., 2018 [b]	11.24	146.47	24.45		19.24	34.8	Primary data
Eide, 2002							
Santos et al., 2017	7.5	13	0.51	89.87	10.99	13	Primary data
Flysjö et al., 2014 [i]	846996240.1		2058055555.56	$143*10^{9}$			Primary data
Aguirre-Villegas et al., 2012	9.98		3.53	19	19.7		Secondary data
Van Middelaar et al., 2011				40.8	17.94		Secondary data
Nilsson et al., 2010 [e]	17.76	14.07	2.08			14.07	Secondary & surrogates data
Mahath et al., 2019	2.92	5.25	0.015			4.67	Primary & surrogates data

a. *These data are an average of the inputs and outputs used for all the products manufactured.*
b. *These data inventories are related to both farm and dairy activity.*
c. *These data inventories are related just to the dairy activity.*
d. *These data are an average of all input and output for the mozzarella and cheddar production.*
e. *This data is referred just to the butter production.*
f. *All these data, except Energy consumption, are referred to 1 kg of raw milk.*
g. *These data are an average of input and output for one ton of processed milk.*
h. *These data are referred to 1 kg of processed milk in the dairy farm.*
i. *These data are collected on an annual basis.*

Magdalena

Table 3. *(continued)*

Study	Inputs					Outputs	Source
	Raw milk	Water (l/kg)	Energy consumption (kWh)	Additives (g)	Cleaning agents (g)	WW (l/kg)	
Canellada et al., 2018[c]	9.35	72.32	2.91	14.32	1564.57	72.32	Primary data
Doublet et al., 2013[h]	0.97	1.2	0.12		2	1.2	Primary data
Vergé et al., 2013[f]		1.5	2822115778	1.68		1.7	Secondary & surrogates data
Flysjö, 2011							Secondary
Berlin, 2002	9.79	1.2	1.86	19.65	18.2		Primary & surrogates data
Yan & Holden, 2018			447.85		4830	1.88	Secondary data
Yan & Holden, 2019[g]							
Dalla Riva et al., 2017	25	1.42	10.11	19.8	30		Secondary data

a. These data are an average of the inputs and outputs used for all the products manufactured.
b. These data inventories are related to both farm and dairy activity.
c. These data inventories are related just to the dairy activity.
d. These data are an average of all input and output for the mozzarella and cheddar production.
e. This data is referred just to the butter production.
f. All these data, except Energy consumption, are referred to 1 kg of raw milk.
g. These data are an average of input and output for one ton of processed milk.
h. These data are referred to 1 kg of processed milk in the dairy farm.
i. These data are collected on an annual basis.

Table 4. *Summary of the four approaches used to handle the co-products in dairy LCA studies. PCmtx (Physico-chemical allocation), SUB (system expansion by substitution).*

Study	**Allocation**			
	PCmtx	**Economic**	**Physical**	**SUB**
Finnegan et al., 2017	X		X	
Djekic et al., 2014	X		X	
Mondello et al., 2018		X	X	
Palmieri et al., 2017			X	
Broekema & Kramer, 2014		X	X	
González-García, Hospido, et al., 2013		X		
González-García, Castanheira, et al., 2013		X		
Kim et al., 2013		X		
Dalla Riva et al., 2018		X		
Eide, 2002		X	X	
Santos et al., 2017		X		
Flysjö et al., 2014		X	X	
Aguirre-Villegas et al., 2012		X	X	X
Van Middelaar et al., 2011		X		
Nilsson et al., 2010	X	X	X	
Mahath et al., 2019	X			
Canellada et al., 2018				
Doublet et al., 2013	X		X	
Vergé et al., 2013			X	
Flysjö, 2011		X	X	
Berlin, 2002		X		
Yan & Holden, 2018				X
Yan & Holden, 2019			X	
Dalla Riva et al., 2017			X	

LCIA

LCIA methodology

In the reviewed studies, the employed life cycle impact categories could be grouped into: Global Warming Potential (GWP), Eutrophication Potential (EP), Acidification Potential (AP), Water Depletion (WD), Cumulative Energy Demand (CED), Ecotoxicity (EcTox) and Toxicity (Tox) and other, see Table 5. It is important to note that there were variations in which specific impact assessment methods were used and that, for example, EP could include terrestrial, freshwater or marine eutrophication and EcTox is terrestrial or marine ecotoxicity. AP is terrestrial acidification and Tox is human toxicity. All of the 24 studies evaluated GWP, and all the studies which include the WWT in the LCA investigated EP and AP.

***Table 5**. Environmental impact categories chosen for the LCA studies – GWP (Global warming Potential), EP (Eutrophication Potential), AP (Acidification Potential), WD (Water Depletion), CED (Cumulative Energy Demand), EcTox (Ecotoxicity), Tox (Toxicity), Others (Ozone depletion, Photochemical Oxidation, Particulate Matter Formation, Ionising Radiation, Fossil Depletion, Abiotic Depletion, Agricultural land occupation, Photo-Oxidation Formation Potential , Land Competition).*

Study	**WWT in the LCA**	**WWWT on site**	**Environmental impact Indicators**							
			GWP	**EP**	**AP**	**WD**	**CED**	**EcTox**	**Tox**	**Other**
Finnegan et al., 2017	X	X	X	X	X	X	X	X		
Djekic et al., 2014	X		X	X	X					
Mondello et al., 2018	X		X	X	X	X		X	X	X
Palmieri et al., 2017	X		X	X	X			X	X	X
Broekema & Kramer, 2014	X		X	X	X					X
González-García, Hospido, et al., 2013	X		X	X	X		X			X
González-García, Castanheira, et al., 2013	X	X	X	X	X		X			X
Kim et al., 2013	X	X	X	X		X	X	X	X	X
Dalla Riva et al., 2018	X		X	X	X	X	X		X	X
Eide, 2002	X	X	X	X	X			X		X
Santos et al., 2017			X	X	X	X				X
Flysjö et al., 2014			X							
Aguirre-Villegas et al., 2012			X				X[a]			
Van Middelaar et al., 2011			X				X			X
Nilsson et al., 2010			X	X	X		X			X
Mahath et al., 2019		X	X	X	X			X		X
Canellada et al., 2018	X		X	X	X	X		X	X	X
Doublet et al., 2013	X		X	X	X	X		X	X	X
Vergé et al., 2013	X		X							
Flysjö, 2011	X	X	X							
Berlin, 2002	X		X	X	X					X
Yan & Holden, 2018	X	X	X				X			
Yan & Holden, 2019	X	X		X						
Dalla Riva et al., 2017	X		X	X	X	X	X	X	X	X

LCIA results

A detailed study about the impacts due to the WWT is shown in Table 6. Not all the LCA studies on dairies took into consideration the WWT. Moreover, some of this assessment considers the wastewater as general waste, so in these cases the LCIA results relative to the WWT cannot be specified. Therefore, the LCIA results in Table 6 are mainly due to the WWT and general waste treatment. A detailed analysis of these results is described in the ANNEX I.

Table 6. *Environmental impact categories chosen for WWT assessment - GWP (Global warming Potential), EP (Eutrophication Potential), AP (Acidification Potential), WD (Water Depletion), CED (Cumulative Energy Demand), EcTox (Ecotoxicity), Tox (Toxicity), OD (Ozone depletion), FD (Fossil Depletion), AD (Abiotic Depletion), Agricultural land occupation, POFP (Photo-Oxidation Formation Potential), FEW (Fresh Water Ecotoxicity), ME (Marine Ecotoxicity), MEP (Marine Eutrophication) and FEP (Fresh Water Eutrophication).*

Study	**Environmental Impact Indicators**												
	GWP	**FWE**	**ME**	**MEP**	**FEP**	**EP**	**AP**	**WD**	**FD**	**AD**	**CED**	**OD**	**POFP**
Finnegan et al., 2017	X	X	X								X		
Djekic et al., 2014													
Mondello et al., 2018	X							X	X				
Palmieri et al., 2017			X										
Broekema & Kramer, 2014	X			X									
González-García, Hospido, et al., 2013	X					X	X			X	X	X	X
González-García, Castanheira, et al., 2013						X							
Kim et al., 2013				X	X								
Dalla Riva et al., 2018					X								
Eide, 2002													
Canellada et al., 2018								X					
Doublet et al., 2013								X					
Vergé et al., 2013	X												
Flysjö, 2011	X												
Berlin, 2002													
Yan & Holden, 2018													
Yan & Holden, 2019						X							
Dalla Riva et al., 2017								X					

Result from review on LCAs of P-recovery technology

This section provides results from the second part of the literature review, which focused on P-recovery technologies of relevance for the REFLOW project. The overall results of the literature review are presented in Table 7. In total 20 papers were selected as relevant with regard to the REFLOW project. Considering the aim of the REFLOW project, it was particularly important to analyse some specific technologies: Biological nutrient removal technology (BNRT), which involves the removal of nutrients, such as P and (nitrogen) N, using proliferation and selection of microbiological populations (Curtin et al., 2011); membrane bioreactor (MBR) systems which involve a combination of a biological-activated sludge process with a membrane process like microfiltration or ultrafiltration (Radjenovic et al., 2007); and sewage sludge combustion with subsequent extraction of phosphorus from the ashes (SSA). Through the implementation of these technologies (Table 7) in the WWTPs, products such as struvite, calcium phosphate (CaP) or other P-products can be recovered. Considering the aim of the REFLOW project and the scope of the technologies, the literature review was extended to consider also LCAs of technologies used to recover materials (sludge, compost or ash) or waters rich of nutrients, to be used in agriculture to avoid mineral fertiliser application (Table 7). The LCA studies describe how these P-recovery technologies have been implemented at conventional WWTPs, for the supernatant and for sludge treatment processes. The sludge treatment consists on different treatments, such as anaerobic digestion (AD), pyrolysis, stabilization, composting or incineration. No studies are available that describe LCAs of any of the P recovery technologies implemented at dairies. This indicates a gap and motivates this part of the REFLOW project.

Goal and Scope

Goal

According to Remy & Kraus (2019), different perspectives can be chosen for LCAs and consequently different goals, FU and system boundaries. According to Remy & Kraus (2019), the “system change perspective” and “product perspective” are common (see Table 8). The first type quantifies all changes in environmental impacts that can be associated with adoptions of P-recovery technology in the defined system and the second system describes the environmental impacts that are associated with the amount of phosphorus recovered (Remy & Kraus, 2019).

Functional Unit

In the case of an LCA study of a P-recovery process, Remy & Kraus, (2019) advise *“phosphorus recovered from WWTP serving 1 Mio population equivalents per year”* or *“mass of P-recovered”* as FUs for the system change perspective and the product perspective case, respectively. Table 8 shows the scope of all the LCA studies considered and, it can be seen that also other than the FUs suggested by Remy & Kraus (2019) are used: the volume of treated water, the mass of total waste or the mass of sludge (biosolid).

System Boundary

Regarding the system boundaries, all reviewed studies have considered at least one pathway through which nutrients could be recycled or used from wastewater (*Figure 2*). Some of them also include agricultural land application in their system boundary (see *Table 8*). Analysing the system boundaries of these LCA studies, it was possible to see that most of the P-recovery technologies are an implementation of the supernatant (C) or sludge (D) treatment. All studies assessed the P-recovery implementation in the sludge treatment process, but Bradford-Hartke et al., (2015), Linderholm et al., (2012) and Longo et al., (2015) assessed implementation in the supernatant treatment process in addition to the sludge treatment. A system model with the various stages in the life cycle of the WWT is shown in *Figure 2*. The letters A to F represent different parts of the life cycle. Process A, supply, include all the processes considered to produce energy or other supply of raw material, including transport. The process F, waste management, includes solid and liquid waste management considering also the flow of the treated water into the hydrosphere.

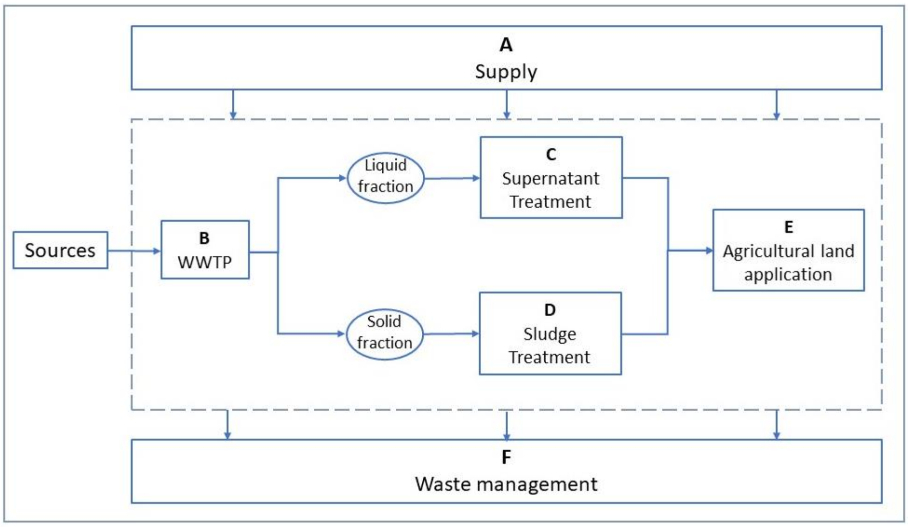

Figure 2. *System boundary and processes in the life cycle of wastewater-based nutrient recycling pathways assessed with the LCA method. The letters A to F indicate the stages used in the Table 8.*

Table 7. *Main characteristics of the P-products recovery technologies that were evaluated with the LCA.*

Study	**Location of study**	**P-recovery technology**		**Agriculture use**
		Process	**Products**	
Amann et al., 2018	Austria	Crystallization, Precipitation, Extraction (from ash)	CaP, Struvite, Phosphoric Acid, P_4	
Bradford-Hartke et al., 2015 [a]	Australia	Precipitation	Struvite	
Linderholm et al., 2012	Sweden	Precipitation and Extraction from sludge ash	Struvite	
Rodriguez-Garcia et al., 2014 [b]	Europe	AD + Crystallization	Struvite	
Kjerstadius et al., 2017	Sweden	AD + Crystallization	Struvite	
Pausta et al., 2018	Philippine	BNRT [c]	Struvite	
Dunkel et al., 2016	Florida	N-E-W [d]	Iron Phosphate	
Sfez et al., 2019	Netherland	Precipitation and Extraction from sludge ash	Struvite and Phosphoric Acid	
Zhang et al., 2020	Europe China	AD + Crystallization	Struvite	
Nèmethy, 2016	Sweden	Precipitation	Struvite	
Longo et al., 2017	Italy	AD + Dewatering, EBPR[f], SCENA[g]	Chem-P and Bio-P biosolid [e]	
Morelli et al., 2018	USA	AD [h]	P for compost	
Malila et al., 2019	Finland	Composting	Compost	X
Cornejo et al., 2016	Florida	ATU [g] WRF	P (water and biosolid reuse) P (water and biosolid reuse)	X
Pretel et al., 2013	Spain	MBR (SAnMBR)[h]	P (water and biosolid reuse)	X

a. *This information is referred to the four centralized systems.*
b. *This article is comparing three anaerobic digestion supernatant treatments. These data are referred to the Struvite Crystallization*
c. *Biological nutrient removal treatment (BNRT) leaves high concentration of nutrients in the sludge; it consists on the removal of P using proliferation and selection of microbiological populations.*
d. *The Nutrient-Energy-Water technology (N-E-W), other to use of $FeCl_3$ for Phosphorus precipitations, consists on the use of Catalytic Oxidation with O_3 for the sterilization of water and Biochar for Phosphorus removal. N-E-W treats wastewater source.*
e. *These are the products to compose the compost that is applicate on the land.*
f. *Enhanced biological phosphorus removal (EBPR) is a sewage treatment applied to activated sludge systems for the removal of phosphate in alternate aerobic and anaerobic conditions, with the presence of polyphosphate-accumulating organisms.*
g. *Septic tank with aerobic treatment unit (ATU) consist on an aerobic system that injects oxygen into the treatment tank; advanced water reclamation facility (WRF) consist on a series of skimmers, screens, and sedimentation tanks and microbial breakdown.*
h. *Submerged Anaerobic MBR (SAnMBR).*

Table 7. *(Continued)*

Study	Location of study	P-recovery technology		Agriculture use
		Process	Products	
Hospido et al., 2005	Spain	AD Incineration	Sludge (biosolid) Ash Char	
Tomei et al., 2016	Europe	Stabilization Stabilization + Incineration	Sludge (biosolid) Ash	X
Svanström et al., 2017	Sweden	Thermo/Mesophilic digestion Mesophilic digestion + Incineration	Sludge (biosolid) Ash	X
Ontiveros & Campanella, 2013	Argentina	BNRT + Digester BNRT + Digester + Incineration BNRT + Digester + Composting	Sludge (biosolid) Ash Compost	X
Niero et al., 2014	Denmark	Stabilization AD + Incineration Precipitation	Sludge (biosolid) Ash Phosphorus[i]	X

i. Phosphorus obtained through chemical precipitation is not collected to produce fertilizers

Table 8. *Goal and scope characteristics in earlier LCAs of P-recovery technologies.*

Study	LCA			
	Functional Unit	System Boundary	System change perspective	Product perspective
Amann et al., 2018	1 kg of recovered P	B-D-F		X
Bradford-Hartke et al., 2015	1 kg of plant available P able to offset synthetic fertilizers	B-C-D-E		X
Linderholm et al., 2012	11 kg of pure P	C-D-E		X
Rodriguez-Garcia et al., 2014	Reduction of EP (1kg PO_4^{3-} eq. removed)	A-B-C-D-E-F	X	
Kjerstadius et al., 2017	The management of 1 capita yearly load of food waste, black water and grey water	B-C-D-E-F	X	
Pausta et al., 2018	$12779*10^3$ kg of food consumed by the population served by the STP	B-C-D-E[a]	X	

a. In the System Boundary of this LCA study, fertiliser production, mining extraction, Haber-Bosch process, and food consumption are included.

Table 8. *(Continued)*

Study	LCA			
	Functional Unit	**System Boundary**	**System change perspective**	**Product perspective**
Dunkel et al., 2016	1000 gallons of water processed by the system	C[b]	X	
Sfez et al., 2019	kg/year of P products	B-D		X
Zhang et al., 2020	1 m^3 of treated wastewater 1kg of COD removed	B-C-D	X	
Nèmethy, 2016	1 kg recycled plant-available P	A-B-C-D		X
Longo et al., 2017	1 kg PO_4^{3-} eq. removed	A-B-C-D-E-F	X	
Morelli et al., 2018	1 m^3 of treated wastewater	A-B-C-D-E-F	X	
Malila et al., 2019	Amount of nutrient produced by one person per year	A-B-D-E-F[c]		X
Cornejo et al., 2016	1 m^3 of treated water over a 20-year lifespan	A-B-C-D-E	X	
Pretel et al., 2013	Volume of treated water	B-D-E	X	
Hospido et al., 2005	1 ton of thickened mixed sludge in dry basis	D-E	X	
Tomei et al., 2016	Treatment to the required level of the wastewater and sludge during one day for a plant of the capacity 70000PE	B-C-D	X	
Svanström et al., 2017	Environmental impact per (metric) tonne of dry solids of undigested sludge	A-B-D-E	X	
Ontiveros & Campanella, 2013	Annual flow of WW treated was used	B-C-D	X	
Niero et al., 2014	1 m^3 of treated water	B-C-D	X	

a. In this case, the treatment considered is implemented after the process C, before waters are emitted in the hydrosphere.

b. In this system boundary is also included the source separation treatment before of the WWT process.

LCI

Data quality

Quality of data is an important aspect to consider for the validity of the LCA results. A minimum set of data for an LCA study on P-recovery processes should include information on product yield, product quality, suppliers, such as electricity, heat, fuel and chemicals, and sludge property (Remy & Kraus, 2019). Since the research reported in the studies aimed to assess P-recovery technologies in the context of WWT, in this report, inventory data on wastewater, sludge, ash, energy consumption, chemicals, and products were compiled in order to provide information for a comparative evaluation of the productivity and sustainability of these technologies. Table 9 provides details about the inventory data and Table 10 about the products. Some of the data were converted to provide consistent and comparative results. Most of the data relative to the wastewater input are missing because some of these studies started from the sludge or ash management (see Table 8 above for the system boundary) and some of the studies presented the bio-chemical characteristics of the wastewater, sludge or supernatant instead of the amount that is treated.

Allocation

As stated earlier, a WWTP is a system that often performs multiple functions, such as sludge management and resource recovery, in addition to treating wastewater (Heimersson et al., 2019). Table 11below describes which kind of allocation method that was used, showing that system expansion by substitution is mainly used in this field. It has been suggested that system expansion by substitution is a more relevant approach in CLCAs than in ALCAs (reference) indicating a potential mismatch between the often attributional scope of the studies and the use of consequential elements in the applied methodology in studies in this field. Most of the LCA studies on WWT and sludge management published between 2004 and 2016 do not specify which of the two kinds of LCA was adopted (Heimersson et al., 2019).

LCIA

The environmental impact indicators used to evaluate the phosphorus recovery processes are shown in Table 12. In general, the ISO standard does not recommend specific indicators that have to be chosen for this specific LCA study (Remy & Kraus, 2019). For an adequate selection of environmental impact indicators, two aspects have to be taken into consideration: first, the environmental impact category should be affected by the process considered under study; and second, suitable data should be available to allow for a meaningful characterization of this impact in the LCA model (Remy & Kraus, 2019) The most common indicators that were been considered in these studies are GWP, EP and AP, see Table 12.

Table 9. *LCI summary for the wastewater and sludge treatment assessed with the LCA*

Study	Inputs					Source
	WW	Sludge	Ash	Energy consumption	Chemicals	
Amann et al., 2018[a]	200 l/PE*y					Secondary and Surrogates data
Bradford-Hartke et al., 2015	10 ML/d[b]				15 mg Fe/l	Secondary data
					100 m^3	
Linderholm et al., 2012[C]				300 kWh	450 kg	Surrogates Data
		181176 tonnes		340 MJ/FU		
				6.84 MJ/kg		
				0.65 MWh	240 kg	
Rodriguez-Garcia et al., 2014 [d]				4660 kWh		Secondary data
				3026 kWh	1.91 kg	
				2568 kWh		
Kjerstadius et al., 2017						
Pausta et al., 2018				300 kWh	450 kg	Secondary data
Dunkel et al., 2016				7.19 kWh[e]		Surrogates data
Sfez et al., 2019[f]	4,10E+10 kg/y			11000000 kWh/y	4800000 kg/y	Secondary and Surrogates data
		490000000 kg/y		240000 kWh/y		
		40000000 kg/y		2500000 kWh/y		
			52000 kg/y	1600 kWh/y	47000 kg/y	
	26000000 kg/y	40000000 kg/y		248201.7 kWh/y	88000 kg/y	
Zhang et al., 2020				0.79 kW/kg		Secondary and Surrogates data
Nèmethy, 2016	501600 kg/kgFU			143.46 kWh	7.662 kg/kgFU	Primary & Secondary data

a. These study asses 18 P-recovery technologies divided in three groups: recover from liquid phase, from sewage sludge and from sewage sludge ash. More details relative to the inventory are in ANNEX II.

b. Equivalenvalent population (PE) is 50000.

c. Data relative to the Struvite production, they are refereed to 500kg of Struvite produced. Regarding the Sewage sludge production, the three values of consumed energy, are referred to the energy used to produce sludge in the WWTP, to the incineration and heating process.

d. These data are referred to the three possible technologies adopted. These data are per m^3 of PO_4^{3-} eq. removed.

e. This is the consumed energy to treat 1000 gallons of water.

f. These are the input and output for the five phases useful to produce Phosphoric Acid and Struvite (WWTP, dewatering plant, Incineration plant, EcoPhos process and Struvite Production).

g. These data are referred to a m^3 of treated water.

h. These data are referred to the liquid effluent subsystem and to the solid effluent subsystem in one year of operation. These data are an average of the selected scenarios.

Table 9. *(Continued)*

Study	Inputs					Source
	WW	Sludge	Ash	Energy consumption	Chemicals	
Longo et al., 2017						Primary & Secondary data
Morelli et al., 2018				0.9 kWh/m^3 0.71 MJ/FU	 0.55 kg/FU	
Malila et al., 2019				0.7 kWh/m^3 in		Primary & Secondary data
Cornejo et al., 2016[g]				1.08 kWh/m^3	0.3 kg/m^3	Primary & Secondary data
Pretel et al., 2013						Secondary data
Hospido et al., 2005				196.15 kWh/FU		Secondary data
Tomei et al., 2016						Primary, Secondary and Surrogates data
Svanström et al., 2017						Secondary data
Ontiveros & Campanella, 2013[h]	729553.25 m^3			46790.5 kWh 17186.5 kWh		Primary & Secondary data
Niero et al., 2014	0.663 kg/m^3 12.108333 kg/m^3			0.313 kWh/m^3 0.365333333 kWh/m^3		Secondary data

a. *These study asses 18 P-recovery technologies divided in three groups: recover from liquid phase, from sewage sludge and from sewage sludge ash. More details relative to the inventory are in ANNEX II.*

b. *Equivalent population (PE) is 50000.*

c. *Data relative to the Struvite production, they are refereed to 500kg of Struvite produced. Regarding the Sewage sludge production, the three values of consumed energy, are referred to the energy used to produce sludge in the WWTP, to the incineration and heating process.*

d. *These data are referred to the three possible technologies adopted. These data are per m3 of PO43- eq. removed.*

e. *This is the consumed energy to treat 1000 gallons of water.*

f. *These are the input and output for the five phases useful to produce Phosphoric Acid and Struvite (WWTP, dewatering plant, Incineration plant, EcoPhos process and Struvite Production).*

g. *These data are referred to a m3 of treated water.*

h. *These data are referred to the liquid effluent subsystem and to the solid effluent subsystem in one year of operation. These data are an average of the selected scenarios.*

Table 10. *Products obtained through the application of P-recovery technologies assessed with the LCA.*

Study	**Products**		
Amann et al., 2018			
Bradford-Hartke et al., 2015	8	%	Struvite reactor P recovery
Linderholm et al., 2012	500	kg/d	Struvite
	2	tonnes	Sewage Sludge
Rodriguez-Garcia et al., 2014			
Kjerstadius et al., 2017			
Pausta et al., 2018	500	kg	
Dunkel et al., 2016			
Sfez et al., 2019	480000000	kg/y	Sludge
	40000000	kg/y	Sludge cake
	52000	kg/y	Ash
	22000	kg/y	H_3PO_4
	380000	kg/y	Struvite
Zhang et al., 2020			
Nèmethy, 2016	0.84	kg/kgFU	Struvite
Longo et al., 2017			
Morelli et al., 2018			
Malila et al., 2019			
Cornejo et al., 2016		g/m^3	P fertilizer avoided- water reuse
Pretel et al., 2013		g/m^3	P fertilizer avoided- biosolids
Hospido et al., 2005			
Tomei et al., 2016			
Svanström et al., 2017			
Ontiveros & Campanella, 2013	96633.75	kg	Sludge
	8160.25	kg	Ash
Niero et al., 2014			

Table 11. *Summary of the two LCA approaches used to handle the co-products in these studies.*

Study	**Allocation**	
	Physical	**SUB**
Linderholm et al., 2012		X
Rodriguez-Garcia et al., 2014		X
Kjerstadius et al., 2017		X
Sfez et al., 2019	X	
Zhang et al., 2020		X
Nèmethy, 2016		X
Longo et al., 2017		X
Morelli et al., 2018		X
Malila et al., 2019	X	
Cornejo et al., 2016		X

Table 12. *Environmental impact categories chosen for the LCA studies – GWP (Global warming Potential), EP (Eutrophication Potential), AP (Acidification Potential), WD (Water Depletion), CED (Cumulative Energy Demand), EcTox (Ecotoxicity), Tox (Toxicity), Others (Ozone depletion, Photochemical Oxidation, Particulate Matter Formation, Ionising Radiation, Fossil Depletion, Abiotic Depletion, Agricultural land occupation, Photo-Oxidation Formation Potential , Land Competition).*

Study	**Environmental Impact Indicators**							
	GWP	**EP**	**AP**	**WD**	**CED**	**EcTox**	**Tox**	**Other**
Amann et al., 2018	X		X		X			X
Bradford-Hartke et al., 2015	X	X				X	X	X
Linderholm et al., 2012	X	X						X
Rodriguez-Garcia et al., 2014	X	X	X			X	X	
Kjerstadius et al., 2017	X							
Pausta et al., 2018	X	X	X					X
Dunkel et al., 2016	X							
Sfez et al., 2019				X				X
Zhang et al., 2020	X	X	X			X		X
Nèmethy, 2016	X	X	X				X	
Longo et al., 2017	X	X	X			X	X	
Morelli et al., 2018					X			
Malila et al., 2019	X	X	X					
Cornejo et al., 2016	X	X			X			
Pretel et al., 2013	X	X	X			X	X	X
Hospido et al., 2005	X	X	X				X	X
Tomei et al., 2016	X	X	X					X
Svanström et al., 2017	X[a]	X	X					X
Ontiveros & Campanella, 2013	X	X				X	X	X
Niero et al., 2014	X	X				X	X	X

a. *Climate Impact*

Discussion

The reviewed studies reported some significantly different conclusions, depending on the choice of system boundaries, the final products, and the assumptions made. With this work, we intended to identify assessment methodologies used as well as the environmental impacts of dairy processes and P-recovery technologies, to get input to future assessments and technology development activities.

Referring to the initial research questions and considering the consulted LCA studies, it was possible to discover that some of the LCAs on dairies include WWT and few of these dairies have the WWT located in the industry itself. As a result of the high volume of DPW that cannot be managed by public WWTP or the lack of connection between dairies and the public WWTP, it is often necessary to clean the wastewater generated from the dairies in the industries' own WWTP. Regarding the LCAs on the P-recovery technologies, it was seen that there are two principally different ways to recover P-products, treating the supernatant (liquid phase) or the sludge (solid phase). Most of the P-recovery technologies assessed in these studies are sludge treatment methods. The sludge could be exposed to various treatments, such as incineration, stabilization, anaerobic digestion, composting or pyrolysis. Through these treatment processes, several by-products are obtained, like compost, biochar, or ash. These products are the main resources to recover phosphorus from. There were the technologies that we found LCAs for. However, undertaking a review of P recovery or P removal technologies could identify other potential applicable technologies.

Through a deep analysis of these studies, it could be possible to evaluate typical environmental impacts related to these treatment methods. Fresh water ecotoxicity and marine ecotoxicity are due to emissions of chemical elements, such as beryllium that is used for the disposal treatment of cheese whey in WWTP. The, global warming potential increase is due to the transport of the wastewater from the dairy to the WWTP. This is because some dairies are not connected to the wastewater treatment plant, thus requiring transportation of the wastewater. This results in an additional energy requirement associated with the wastewater treatment. Emissions of effluents rich in phosphorus and chemical oxygen demand (COD) are responsible for the eutrophication potential and toxicity. The pasteurization process, during the milk production, notably gives rise to considerable water depletion.

However, on the basis of reviewed studies, several weaknesses or limitations were found with regard to the LCA method. Very little information relating to WWT was reported and this leads to underestimated or inaccurate environmental impact modelling relating to these aspects of the dairy. Regarding the P-recovery technologies, there are lacks in information. The majority of the studies focused on the application of the final by-products in agriculture, due to the high nutrient content, however, the nutrient recovery is not considered. There is a lack of detailed information on the LCI data which hampers its use in new studies: no details on the chemical suppliers used, data were not always referred to the FU, and information relative to the amount of inflow wastewater in the WWTP are missing, as are also the effluents. Information relative to the LCIA are lacking details, especially for the LCAs on dairy. Some of the studies include the WWT in the LCA without specifying the final environmental impact that should be related to it but instead lumping it together with treatment of all other waste produced throughout the process. Data related to the type of treatment that is applied to the DPW are missing. The allocation method, which has effects on the final LCA results, is lacking details. The choice of the allocation method in a multifunctional system is always discussed, but in these kinds of studies, considering that most of the treatment under analysis is an implementation at a conventional WWTP, the system expansion by substitution, was the preferred allocation method. In the LCAs on dairy, economic and physical allocation were mainly chosen.

This lack of information, data and details makes it challenging, and much needed in the context of the REFLOW project, to conduct LCA on these topics. In this report, a summary of LCI and LCIA have been drawn up to be a benchmark for the future LCA studies in the REFLOW project, and it is clear that some further methodological development work will also be needed to alleviate some of the weaknesses observed in the review.

Conclusion

Phosphorus, being a precious and scarce raw material, needs to be recovered to reduce the pressure on mining. Today, most of the considered options to recover P-products are through the treatment of domestic wastewater. But with the recovery from these sources, there are contamination risks and it is complicated to separate the nutrients from other potentially harmful elements such as heavy metals, micro-pollutants, and pathogens. Considering that and considering also that after the removal of the European Commission milk quota with a 50% milk production increase in 2015, the REFLOW project focuses on DPW as a source to recover phosphorus. The DPW contains only small amounts of contaminants and it is rich in nutrients, and by recovering phosphorus, it will also be possible to avoid environmental impact, like eutrophication, which is one of the main problems related to poor management of DPW.

The purpose of this research was to review earlier LCA studies on dairy activities and P-recovery technologies to compile information about assessment methodology and of assessment results in order to facilitate future LCA studies on these topics.

From the literature review reported here, it is clear that most of the LCA methodology selected for LCAs of dairy products follow the recommendations from the International Dairy Federation. Regarding LCA methodologies for P-recovery technologies, there is no defined method to adopt, and cross-comparisons are more difficult.

On the basis of the reviewed studies, there are gaps in the knowledge of DPW management and P-recovery from industrial wastewater and that consequently, the LCA methods attributed to these systems are still missing of details that are needed to the REFLOW aim.

This work will be used as input starting the work on the development of an LCA for P-recovery technologies implemented in conventional dairies.

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A review of techno-economic analysis studies on dairy wastewater and phosphorus recovery technologies (ESR 12)

This document contains a literature review of the Early Stage Researcher (ESR 12) active in Work Package 3 of the REFLOW European Training Network.

Phosphorus is one of the fundamental and irreplaceable nutrients for all forms of life on the planet. Today, phosphorus is primarily obtained from raw phosphate rock. However, the increasing demand and use of phosphorus, especially in agriculture production, has posed challenges depletion unequal distribution over the world. Phosphorus security has become a big concern for Europe and other regions without or with little reserves. In 2014, The European Commission listed phosphate rock as one of the critical raw materials (CRMs).

Dairy wastewater has a high concentration of phosphorus, ranging from 9 to 280 mg/l while in municipal wastewater the concentration is approximately 10 mg/l, which indicates a high potential of phosphorus recovery. Various technologies have been tested and implemented for phosphorus recovery from municipal wastewater. However, limited attention has been paid to dairy wastewater.

There are different types of dairy products and operation methods used in factories, which leads to the difference of dairy wastewater quality and may further influences the selection of phosphorus technologies. It is important to evaluate both dairy wastewater characteristics and phosphorus technologies. Moreover, economic criteria affect the decision making greatly. A holistic assessment approach is proposed to compare phosphorus recovery technologies considering techno-economic aspects. This literature review is to provide an overview of previous research related to techno-economic analysis (TEA) regarding dairy wastewater treatment and phosphorus recovery from wastewater as well as incorporate the uncertainty concept.

Introduction

Phosphorus (P) is an essential element for all life (Cordell et al., 2011). It is a fundamental nutrient in agricultural production while about 90% of current phosphorus resources is used as fertilizer (Brunner, 2010; European Commission, 2017). Today, the predominant source of phosphorus deployed in agriculture is raw phosphate rock, which is a finite resource and becomes depleted at an increasing rate (Cordell et al., 2011; Desmidt et al., 2015; Egle et al., 2015). To be noted, there is no substitute for phosphate rock in natural resources and there is no alternative for phosphate fertilizers in agriculture (Cook et al., 2005; Butusov & Jernelöv, 2013). In 2008, the price spike in phosphate rock and phosphate fertilizers triggered the attention of global phosphorus scarcity (Cordell & White, 2011). Some researchers suggested that phosphate rock reserves are expected to disappear in the 21st century (Cordell et al., 2009; Vaccari, 2009) while others believed that phosphorus shortage is not likely to happen within 100 years (van Kauwenbergh, 2010; van Vuuren et al., 2010).

Another phosphorus challenge is its uneven geographical distribution throughout the world (van Dijk et al., 2016). Statistics of European Commission (2017) shows that from 2010 to 2014, approximately 70% of phosphate rock was mined in three countries, i.e. China (44%), USA (13%) and Morocco (including Western Sahara, 13%). According to the U.S. Geological Survey (2019), 71.4% of phosphate rock deposits are sited in Morocco (including Western Sahara). As a consequence, the phosphorus fertilizer global market will likely remain and be even more dominated by Morocco (including Western Sahara) and other few players (Reijnders, 2014). As for Europe, the latter is endowed with a limited amount of phosphate rock reserves. The European union (EU) strongly relies on phosphate rock imports, which accounts for 88% of total EU phosphorus supply over the 2010-2014 period (European Commission, 2017). In 2014, the European Commission listed phosphate rock as one of the critical raw materials (CRMs) due to its high economic importance and supply risk, which indicates phosphorus security has been recognized as a sustainability challenge (European Commission, 2017).

Current phosphorus use is inefficient and losses exist throughout all sectors: crop production, animal production, food processing, non-food production and consumption (van Dijk et al., 2016). When effluents are discharged to surface water, phosphorus in effluents aggravates eutrophication in water bodies, which results in a decline in water quality and aquatic biodiversity. To protect the water environment, the European Council (1991) adopted the European Union Urban Waste Water Directive according to which phosphorus is designated to be removed from domestic and industrial wastewater with a threshold of 2 mg/l.

The dairy industry is a major wastewater source of industrial effluents in Europe; in average, a dairy manufacturer produces a volume of 500 m^3 wastewater per day (Demirel et al., 2005; Britz et al., 2006; Rivas et al., 2010). This industry processes and manufactures raw milk into dairy products (cheese, butter, yoghurt and various types of desserts) by means of different techniques, for example, pasteurization, coagulation, and chilling (Rivas et al., 2010). Depending upon the types of final products and operation methods used in factories, the characteristics of processing wastewater may differ greatly (Carvalho et al., 2013). In general, these effluents have high organic content with 40 to 48,000 mg biological oxygen demand (BOD) and 80 to 95,000 mg chemical oxygen demand (COD) in 1 l effluent (Gutiérrez et al., 1991), which has to be removed before being discharged. Furthermore, dairy wastewater has a high concentration of phosphorus, ranging from 9 to 280 mg total phosphorus (TP)/l effluent; phosphorus occurs in dairy wastewater mainly as inorganic phosphate, such as orthophosphate (PO_4^{3-}), polyphosphate ($P_2O_7^{4}$-), as well as organic forms (Gutiérrez et al., 1991; Demirel et al., 2005). Meanwhile, in raw municipal wastewater, the phosphorus concentration is around 10 mg TP/l (Egle et al., 2015). Thus, there is a high phosphorus recycling potential in dairy wastewater.

Many studies have addressed the importance of recycling phosphorus from waste streams (Cordell et al., 2012; Schoumans et al., 2015). We believe that with a sustainable phosphorus recovery from dairy wastewater, a great number of chemical phosphorus fertilizers applied in agriculture can be substituted. Therefore, particularly for Europe, the phosphorus security challenge can be mitigated.

There are various phosphorus recovery technologies, however, to meet the objectives of different stakeholders and local conditions of dairy facilities, the technologies used should be carefully chosen. Technical feasibility of technologies is needed to address and evaluate, such as mass consumption and production, levels of development (reliability) and operating complexity. Moreover, economic criteria affect the decision making greatly, including costs (operating expenditures and capital costs) and revenues. To support the selection of phosphorus technologies, a holistic techno-economic analysis (TEA) should be designed to analyse different criteria and make trade-offs between different objectives of stakeholders.

The design of TEA is structured as follows: (1) the quality of dairy wastewater should be studied according to monitored data in dairy facilities; (2) based on literature review, the characteristics of phosphorus recovery technologies then need be researched; (3) important variables which should be included in the TEA should be chosen based on literature review and the preference of specific stakeholders; (4) in the end, to improve the model robustness, uncertainty concept should be incorporated in the model development. Thus, a literature review is essential to support TEA in terms of understanding the content and selecting methods which will be further used in the research.

The purpose of this literature review is to provide an overview of previous research related to TEA of phosphorus recovery. The first part of this chapter introduces the general condition of dairy wastewater and wastewater treatment process. Next, phosphorus recovery technologies are briefly introduced and variables which appeared in TEA are collected. Furthermore, this chapter discusses associated uncertainty issues and proposes stochastic modelling in TEA as a solution to uncertainty.

Dairy Wastewater

Wastewater content

Dairy wastewater comprises of cleaning water, sanitizers, spillages, rejected milk and milk waste. Around 2% of processed milk is discharged to sewers, which leads to high COD, BOD, nutrient, organic and inorganic contents of dairy effluents (Munavalli & Saler, 2009; Kushwaha et al., 2011). Use of cleaners and sanitizers and operation techniques and types of final products influence the characteristics of dairy wastewater (Demirel et al., 2005; Carvalho et al., 2013). COD concentration differs primarily depending on the volume of milk, cream and whey in wastewater (Kushwaha et al., 2011). Nitrogen mainly comes from milk protein, which is found as either organic nitrogen (proteins, urea and nucleic acids) or ions (NH_4^+, NO_2^-, NO_3^-; Demirel et al., 2005). Phosphorus occurs in dairy wastewater mainly as inorganic phosphate, such as orthophosphate (PO_4^{3-}), polyphosphate ($P_2O_7^{4}$-), as well as organic forms (Demirel et al., 2005; Britz et al., 2006). Detergents and sanitizers, which could be alkaline or acidic, can also be found in dairy wastewater and results in a high variation of pH (Demirel et al., 2005). Noticeable amounts of Na, Cl, K, Ca, Mg, Fe, Co, Ni and Mn also exist in dairy wastewater (Kushwaha et al., 2011). The characteristics of raw dairy wastewater is given in *Table 13*.

Table 13. *Characteristics of raw dairy wastewater (unit in mg/ l, expect for pH)*

Parameter	Value	Reference
COD	80-95,000	(Gutiérrez et al., 1991)
BOD	40-48,000	(Gutiérrez et al., 1991)
pH	4.7-11	(Munavalli & Saler, 2009)
Total Suspended Solids (TS)	135-85,000	(Gutiérrez et al., 1991)
Volatile Suspended Solids (VSS)	24–4,500	(Gutiérrez et al., 1991)
Total Nitrogen (TN)	15-180	(Gutiérrez et al., 1991)
Total Phosphorus (TP)	9-280	(Gavala et al., 1999)

Wastewater treatment and products

The process of dairy wastewater treatment is more or less same to municipal wastewater treatment process (

Figure 3). After preliminary treatment removing large objectives, such as solids and grease, dairy wastewater is sent to primary treatment tank where particles sink to the bottom and form sludge (Kushwaha et al., 2011). For the reduction of organic matters and removal of nutrients, both aerobic and anaerobic treatment can be implemented in secondary treatment with the development of various technologies. One promising aerobic technology is sequential batch reactor (SBR) while Upflow Anaerobic Sludge Blanket (UASB) is one of the widely-used anaerobic technologies for dairy wastewater treatment (Gutiérrez et al., 1991; Gavala et al., 1999; Kushwaha et al., 2011). If a higher quality of dairy wastewater is needed for industrial uses, tertiary treatment is required for higher removal rates; reverse osmosis and nanofiltration are tested at bench and plant scale, separately (Sarkar et al., 2006; Vourch et al., 2008; Andrade et al., 2014). Sludge, a semi-solid material, is produced during primary and secondary treatment, which is collected in the thickening tank to separate from water (digester supernatant). Different materials can be recovered and produced during dairy wastewater treatment: protein (Selmer-Olsen et al., 1996), lactose (Chollangi & Hossain, 2007), nutrient recovery (Adhikari et al., 2015), biogas (Ince, 1998) and water (Sarkar et al., 2006; Vourch et al., 2008; Andrade et al., 2014). Phosphorus removal and recovery mainly occurs in secondary treated effluent, anaerobic sludge and digester supernatant (Egle et al., 2015). Little known is about phosphorus recovery from dairy wastewater.

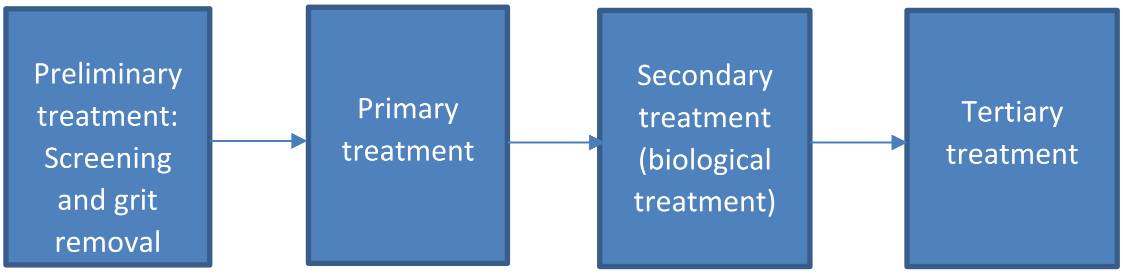

Figure 3. *Basic diagram of dairy wastewater treatment*

Phosphorus Recovery

Phosphorus removal and recovery technologies can be mainly categorized into chemical precipitation, biological phosphorus removal, wet-chemical approach, filtration through membrane or media, thermal treatment and crystallisation (Morse et al., 1998; Egle et al., 2015). The location of applying phosphorus recovery technologies during wastewater treatment processes varies, which is possible in the liquid phase (wastewater) and in the solid phase (sewage sludge or sewage sludge ash), separately (Cornel & Schaum, 2009). Egle et al. (2015) designed a flow diagram for showing potential access points for phosphorus recovery within municipal wastewater and sludge treatment processes, for example, from source-separated human urine, secondary treated effluent, sewage sludge and sewage sludge ash, which leads to the differentiation in corresponding technologies and final products, where blue lines represent the recovery coming from liquid phase, green lines represent from sewage sludge (SS) and orange lines represent from sewage sludge ash (SSA) (*Figure 4*).

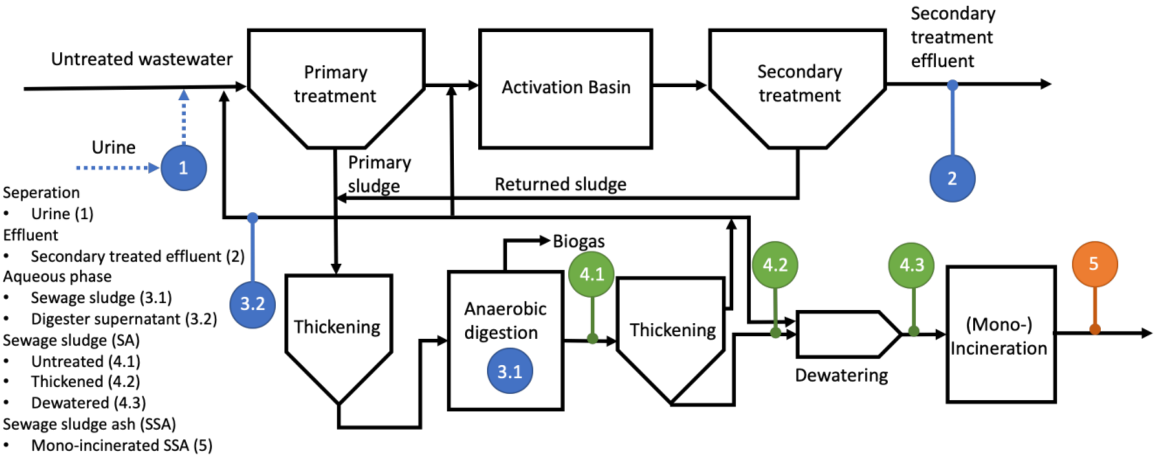

Figure 4. *Potential access points for recovering phosphorus during municipal wastewater and sludge treatment, adapted from (Egle et al., 2015)*

From the 1950s, phosphorus removal technologies have been investigated in response to eutrophication of water bodies and excess phosphorus in effluents; starting from then, chemical precipitation has gradually developed as a commonly used method for phosphorus removal (Morse et al., 1998). In spite of its simplicity and flexibility, chemical precipitation, by adding iron or aluminium salts, has a relatively low recovery potential from aqueous phase and sewage sludge because of the low dissolution of metal-phosphate compounds, and also leads to increasing sludge production (Egle et al., 2015).

Alternative solutions have been developed in recent years, such as biological removal, crystallization and other novel approaches, at different levels of development (laboratory, pilot, full, and commercial scale, (Morse et al., 1998); as demonstrated by Egle et al. (2015), a great number of phosphorus recovery technologies displayed potential for full-scale application. However, one question remains: among all potential phosphorus recovery technologies, which ones are the ideal technology that meets all criteria (more technically feasible, lower economic costs and environmental impacts)?

Technical analysis

Several researchers have conducted technical assessment to compare phosphorus recovery technologies. Morse et al. (1998) performed a review of phosphorus removal and recovery technologies and summarized their mechanisms, levels of development status, and applicability. However, Morse et al. (1998) did not compare these technologies in a quantitative way regarding operational parameters, such as temperature, molar ratio and retention time, therefore, it can hardly provide practical support.

Egle et al. (2015) reviewed technologies for recovering phosphorus from municipal wastewater and related waste flows. Technical principles, process parameters, recovery potential, resource demand, possible effects on later treatment processes, and the fate of pollutants of 50 technologies are presented. Egle et al. (2015) did not consider economic assessment in the study. Later, Egle et al. (2016) performed a more comprehensive study, integrating technical, environmental and economic assessment to compare 19 technologies (*Table 14*).

Table 14. *Phosphorus recovery technologies from different steps of WWTP, adapted from Egle et al. (2016)*

Aqueous phase	Sewage sludge	Sewage sludge ash
REM-NUT® [2; ion exchange, precipitation]	Gifhorn process [4.1; wet-chemical leaching]	AshDec® depollution [5; thermo-chemical, ash depollution, Cl-source (e.g. MgCl2)]
AirPrex® [3.1; precipitation/ crystallization]	Stuttgart process [4.1; wet-chemical leaching]	AshDec® Rhenania [5; thermo-chemical, Rhenaniaphosphate, Na2SO4]
Ostara Pearl Reactor® [3.2; crystallization]	PHOXNAN [4.2; wet-oxidation]	PASCH [5; acidic wet-chemical, leaching]
DHV Crystalactor® [3.2; crystallization]	Aqua Reci® [4.2; super critical water oxidation]	LEACHPHOS® [5; acidic wet-chemical, leaching]
P-RoC® [3.2; crystallization]	MEPHREC® [4.2; metallurgic melt-gassing]	EcoPhos® [5; acidic wet-chemical, leaching, P-acid production]
PRISA [3.2; precipitation/ crystallization]		RecoPhos® [5; acidic wet-chemical, extraction]
		Fertilizer Industry [5; acidic wet-chemical, extraction]
		Thermphos (P4) [5; thermo-electrical]

Although intensive discussion of phosphorus recovery has been made, it is acknowledged that there is no perfect technology for all circumstances and ideal considering technical, economic and environmental criteria (Egle et al., 2015). Phosphorus recovery methods are suggested to be designed individually for each treatment plant (Cieślik & Konieczka, 2017). The suitable technology should be chosen as case-specific, based on local conditions as well as different criteria, such as local environmental regulations and investment decisions of stakeholders.

Moreover, a review of previous studies reveals that the main focus of phosphorus recovery assessment is from municipal wastewater treatment. There is still a research gap regarding technical investigation of phosphorus recovery from dairy industrial water. Even the knowledge of physicochemical composition of dairy industrial wastewater and associated waste flows is limited (Ashekuzzaman et al., 2019). In order to improve dairy wastewater sludge recycling in agriculture, Ashekuzzaman (2019) analysed four types of sludge (bio-chemically treated activated sludge, lime treated dissolved air flotation processing sludge, a combined treatment sludge and anaerobically digested sludge) and the results showed significant differences of nutrient content ratio (nitrogen, phosphorus and potassium) in different sludge types, leading to different fertilizer values.

Based on a municipal wastewater treatment plant (WWTP) in Italy, Bertanza et al. (2017) compared membrane bioreactor and conventional activated sludge from techno-economic, environmental and social aspects. For conducting technical analysis, mass balances were calculated to track the flows of water, COD, TS, VSS, TN and TP; energy balance was also developed to estimate the energy consumption and biogas production. The method Bertanza et al. (2017) applied to compare two systems were multi-criteria analysis (MCA), which addressed different objectives of stakeholders and scored performances of two options in a quantitative approach. Within technical category, three subcategories were taken into account: reliability, flexibility/ modularity and complexity. To be noted, the administrative aspect (normative constraints) was also evaluated.

Material flow analysis

To calculate mass balances for a chemical process, process flowsheeting is frequently used. To understand a bigger picture of phosphorus use and recovery in the dairy industry at a country level or European level, material flow analysis might be needed. MFA, also referred to as substance flow analysis (SFA), is a method, which applies mass balance principles and assesses the flows and stocks of a substance within a defined system (Brunner & Rechberger, 2004). *Figure 5* shows an example of a material flow scheme, where: blue lines represent inputs entering the system (dotted line); red lines represent outputs; blue boxes represent processes; black arrows represent phosphorus flows within the system and numbers represent the value of material flows.

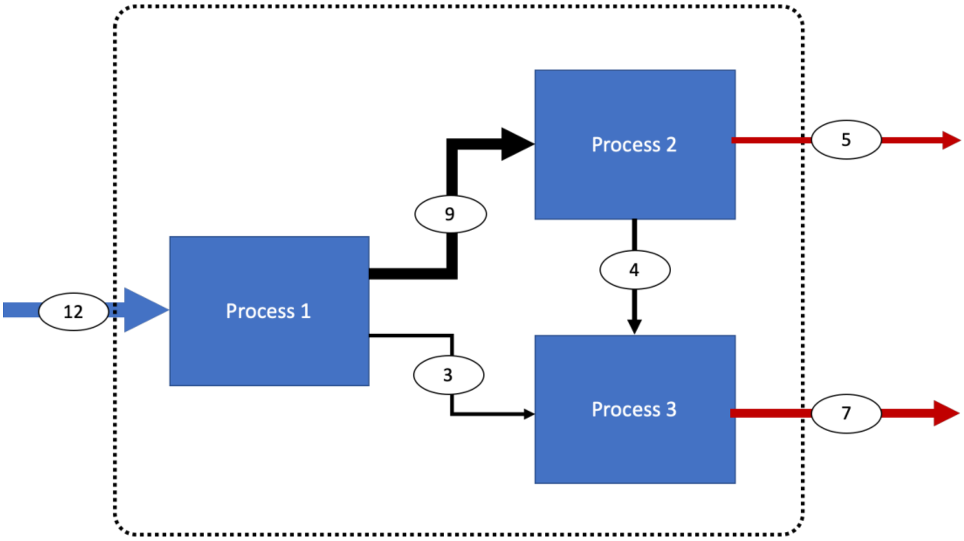

Figure 5. *An example of material flow*

The steps of MFA are as follows (Montangero, 2007):

1. System assessment (selection of the related goods, processes, indicator substances, and system boundaries)
2. Quantification of mass flows of goods and of indicator substances
3. Schematic diagram and interpretation of the results.

Phosphorus MFAs

A holistic picture of phosphorus flows enhances the understanding of the current status of phosphorus use and recovery, as well as track pathways of phosphorus and potential pollutants (Antikainen et al., 2005). Many studies have conducted MFA related to phosphorus flows. Based on the summary of recent phosphorus SFA studies by Cordell et al. (2012), a table consisting of different literature is presented to illustrate the systems researchers have investigated (*Table 15*).

Among all sectors, mining & fertilizer production have been the least studied. One explication could be that most countries import phosphate rock instead of mining, except from China (Li et al., 2016) and the U.S. (Suh & Yee, 2011). Agricultural and food production are two key sectors, which many researchers have highlighted (Antikainen et al., 2005; Suh & Yee, 2011; Senthilkumar et al., 2012b). Regarding geographical scales, Chowdhury et al. (2014) identified that, among all spatial scales, along with regional scales, multi-national scales MFA have been the least discussed despite the importance of understanding agricultural and food trades. Based on our literature review, one can note that at city scales, the main scope of MFA studies is pollution and wastewater management as municipal waste and wastewater are crucial sources for recovering phosphorus within cities (Montangero, 2007; Schmid Neset et al., 2008). Different from other studies, the scope of Yoshida et al. (2015) was a municipal wastewater treatment plant in Denmark; wastewater and sludge treatment processes are described in detail for tracking the pathways of 32 elements and 4 groups of organic pollutants at unit process level.

Ashekuzzaman et al. (2019) noted that comprehensive data related to the context and flows of dairy wastewater and waste streams is rarely available in the literature. To the best of the author's knowledge, no quantification of water flows has been made for the dairy industry, not to mention phosphorus flow in dairy wastewater.

Table 15. *Review of phosphorus MFA (modified from (Cordell et al., 2012))*

Author	Geographical scale					Temporal scale	Main scope		Sectors					
	Global	Multi-national	National	Regional	City		Pollution and wastewater management	Scarcity and food security	Mining & fertilizer production	Agriculture production	Food production	Household consumption	Waste/ wastewater treatment	Pollution & leakage
(Antikainen et al., 2005)			Finland			1995-1999	x	-	-	x	x	x	x	x
(Montangero, 2007)					Hanoi	2006	x	-	-	x	x	x	x	x
(Neset et al., 2008)					Linköping	1870-2000	x	-	-	x	x	x	x	x
(Matsubae-Yokoyama et al., 2009)			Japan			2002	x	x	x	x	-	x	x	x
(Cordell et al., 2009)	x					2005	-	x	x	x	x	x	x	x
(Smit et al., 2010)			The Netherlands			2005	x	-	-	x	x	x	x	x
(Suh & Yee, 2011)			US			2007	-	x	x	x	x	x	-	-
(Senthilkumar et al., 2012b)			France	4 regions		1990-2006	-			x	-	-	-	x
(Senthilkumar et al., 2012a)			France			2002-2006	-	x	-	x	x	x	x	x
(Cooper & Carliell-Marquet, 2013)			UK			2009	-	x	-	x	x	x	x	x
(Klinglmair et al., 2015)			Denmark	3 regions		2011				x		x	x	x
(Jedelhauser & Binder, 2015)			7 EU countries				-	-	-	x	x	x	x	x
(Li et al., 2016)			China	North China Plain	Quzhou county	2004-2009	-	x	x	x	x	x	x	x
(van Dijk et al., 2016)		EU-27				2005	-	-	-	x	x	x	x	x
(Yoshida et al., 2015)	Avedøre wastewater treatment plant (Copenhagen, Denmark)					2011	x	-	-	-	-	-	x	x

Methods and software

Three basic mathematical equations for calculating inputs and outputs in phosphorus MFA are summarized (**Error! Reference source not found.**, *Equation 2* and *Equation 3*; Montangero, 2007; Senthilkumar et al., 2012a; Li et al., 2016; van Dijk et al., 2016):

Phosphorus balance equation:

Equation 1

$$P_{\_stock} = \sum_{1}^{x} P\ inflows - \sum_{1}^{y} P\ outflows$$

where x and y represent the number of inflows and outflows in one process separately.

P mass in the specific goods:

Equation 2

$$P_{goods} = Mass_{goods} \times P_{concentration}$$

Phosphorus Utilization Efficiency equation (PUE):

Equation 3

$$PUE = \frac{Total\ useful\ output\ of\ P}{Total\ input\ of\ P} \times 100\%.$$

STAN is a substance flow analysis software, which is widely used to visualize phosphorus flows (Smit et al., 2010; Cooper & Carliell-Marquet, 2013; Klinglmair et al., 2015; Li et al., 2016; van Dijk et al., 2016). van Dijk (2016) also mentioned that the General Algebraic Modeling System (GAMS) was utilized for programming the mathematical model. Neset (2005) proposed a dynamic MFA model containing 81 equations (6 ordinary differential equations of first order, along with 75 algebraic equations) for incorporating time-dependency in MFA; equations were applied in a computer program SIMBOX developed by the Department System Analysis, Integrated Assessment and Modelling at EAWAG, Switzerland (https://www.eawag.ch/en/department/siam/software/). It has been shown that STAN is instrumental for MFA, and other software might also be needed for advanced modelling use.

Economic analysis

While technical possibilities have been exploited to recover phosphorus from wastewater, there is a lack of economic assessment of this process; it is important to understand the economic perspective of the phosphorus recovery process for effective planning of projects in order to apply phosphorus technologies at full and even commercial scales (Yetilmezsoy et al., 2017).

Economic calculation

With respect to economic analysis, researchers have focused on cost calculation with different calculation methods and indicators. It is generally acknowledged that CAPEX and OPEX are two components of the investment cost. CAPEX, refers to capital expenditure, which is one-time major purchase for companies to acquire or improve long-term assets (more than one accounting period), such as buildings, industrial plants and equipment. Once the asset is used, CAPEX is depreciated over time to divide the costs by the entire life of the asset. OPEX refers to operating expense, which is short-terms expenses and is spent by a company to meet the daily operation, and it only incurs during the current accounting period (Rumble, 2012).

When asset owners, operators and service providers consider purchasing an asset, it is not wise to compare alternative options only by analysing the capital expenditure. Life cycle costing (LCC) is a technique which describes entire costs of ownership over the whole estimated lifespan, including CAPEX and OPEX (Heralova, 2014). This calculus is useful for decision makers to assess different investment alternatives (Gluch & Baumann, 2004). Heralova (2014) stated that among all approaches, Net Present Value (NPV) and Equivalent Annual Cost (EAC) are the most popular and suitable methods for analysing LCC of construction projects. Besides NPV and EAC, van den Boomen et al. (2018) also mentioned a technique named capitalized equivalent worth (CW). Due to time value of money (TVM), a certain amount of money is worth more than the same amount of money in the future (Brigham & Houston, 2012). The discount rate, which can compensate time value of money via converting cash flows in the future into its equivalent value at present (Modigliani & Miller, 1958). The widespread method to make cash flows comparable and add up all discounted cash flows from different times is NPV (Gluch & Baumann, 2004). One disadvantage of NPV is that it cannot compare alternatives with variable replacement cycle units fairly while EAC converts overall costs of options spent over the whole lifespans into costs for one year (Heralova, 2014). CW converts the EAC of one replacement cycle into the present value of an endless number of life cycles, which however only used when repeating replacement cycles do not include initial investment (van den Boomen et al., 2018).

Equation 4

$$NPV = \sum_{t=0}^{T} \frac{c_t}{(1+r)^t}$$

The present value of the expected overall life cycle costs is calculated according to *Equation 4*, where C_t is the sum of all relevant costs generated during the period t, r is the discount rate, t is the analysed time (t=0...T), and T is the life cycle (Heralova, 2014).

Equation 5

$$EAC = NPV \times AF$$

Equation 6

$$AF = \frac{r \times (1 + r)^t}{(1 + r)^t - 1}$$

EAC is calculated as *Equation 5*, where AF refers to annuity factor (AF), where the functional notation AF expresses as follows (*Equation 6*): find the factor of annual sums, given the discount rate r and the expected life cycle t (Heralova, 2014).

Phosphorus economic analysis

There are controversial opinions if investment costs should be taken into account when comparing phosphorus recovery technologies. Balmér (2004) made cost estimations for 6 scenarios (source separation of urine, source separation of toilet wastes, recirculation of dewatered sludge, phosphorus recovery from sludge, phosphorus recovery from incinerated sludge ash and phosphorus recovery from a bio-P process), taking CAPEX and OPEX into account. However, Balmér (2004) did not mention the calculation method and extracted data from phosphorus recovery processing companies despite the possibly low data reliability.

Egle et al. (2016) performed cost calculation on an annual basis, in which annual costs comprised two components: annual capital costs and operating costs, based on *Equation 5* and *Equation 6* (Egle et al., 2016). Base on the cost analysis, Egle et al. (2016) pointed out that CAPEX are the main drivers of total costs, for example, installation of crystallization reactors; moreover, he discussed that one weakness of conducting economic assessment is for a low technology readiness level technologies, calculation of CAPEX is impossible at this stage of development due to the lack of data.

Yetilmezsoy et al. (2017) conducted a financial and economic feasibility analysis for struvite recovery from the fertilizer industry and extracted data from physicochemical treatability experiments; equipment, building and engineering service fees are taken into the calculation of investment costs, while material expenses, transportation and disposal, electricity, engineering, maintenance, labour, and analytical monitoring are components of operational costs.

Svanström (2014) proposed a roadmap for technical, economic and environmental assessment, particularly for advanced sludge processing (Figure 3). In this methodology of economic assessment, capital costs are ignored due to the main purpose of economic assessment is to compare different technologies and capital costs may bring a great difference between mature and new technologies and bias the result (Table 3).

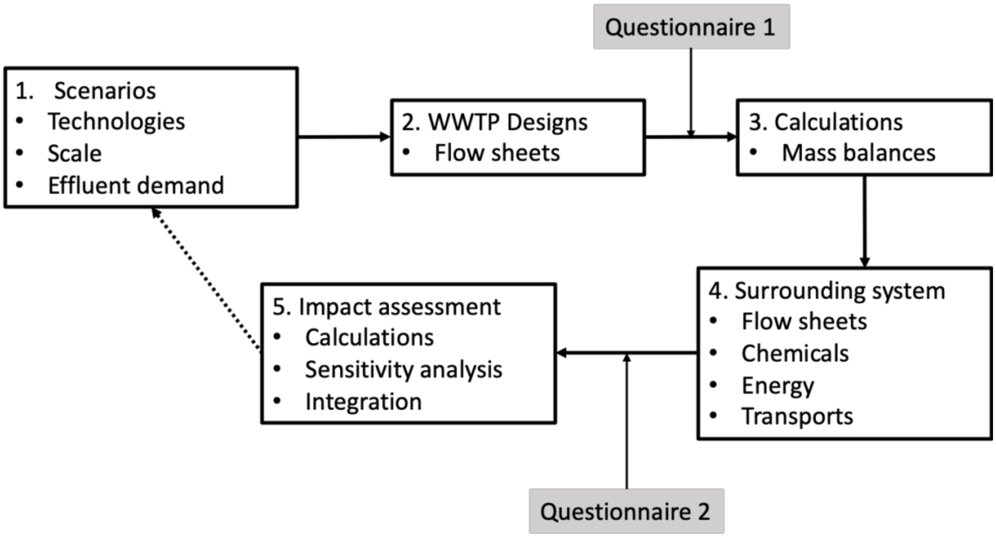

Figure 6. *Techno-economic-environmental assessment roadmap in the ROUTES project, adapted from (Svanström et al., 2014)*

Table 16. *Economic assessment components (Svanström et al., 2014)*

Cost	**Revenue**
Personnel	Recovered materials
Electric energy	Electric energy sale
Raw materials and reagents	Thermal energy sale
Reuse or disposal of solid/slurry residues	Treatment of organic fraction of municipal solid waste
Transportation	
Ordinary maintenance cost	

Regarding the cost-effectiveness, it has been argued whether the cost and price of recovered phosphorus should be compared with that of raw phosphate rock (Egle et al., 2016). A general consensus is that up to now, the recovered phosphorus is several times more expensive than mined phosphate rock.

Cornel and Schaum (2009) suggested not to directly compare recovered phosphorus and phosphate rock since they are of different qualities and phosphate rock needs to be processed before use. Generally, phosphorus recovery from sludge has higher costs than from sludge ash and from the liquid phase (Mayer et al., 2016). Balmér (2004) calculated costs of 33, 80 and 58 SEK/ kg P (1 SEK equals to € 0.09) for phosphorus recovery from wastewater, sludge and sludge ash separately and concluded that the cost of recovered phosphorus is higher than the market price of mineral fertilizer phosphorus.

Yetilmezsoy et al. (2017) believed that setting a price for struvite recovered through the precipitation process is the key step of economic analysis because the flexibility of the struvite sales price can highly influence the results. Operational cost equations were developed to find out the break-even point of the market price of struvite, which is

€482/ton in the study; the result means when the sales price of struvite exceeds €482/ton, expenses can turn into profits, while the struvite price ranges from €320 to $3,800/ton ($1 equals to €0.93) in spite of the great difference across countries (Yetilmezsoy et al., 2017).

Risk and Uncertainty

Stochastic modelling

As mentioned above, uncertainty occurs in TEA. As a result, stochastic modelling has been introduced in TEA, especially from a financial perspective to forecast the potential returns to investment. Nevertheless, stochastic TEA has not yet been employed in phosphorus and fertilizer production sector while some studies have been carried out to calculate breakeven prices and net present values (NPV) in renewable energy production (Zhao et al., 2015; Yao et al., 2017; Diniz et al., 2018). Compared to deterministic analysis, stochastic analysis shows more reliability since it inherently assesses risk and uncertainty (Diniz et al., 2018). Zhao et al. (2015) presented a stochastic comparison of eight biofuel cellulosic biofuel production pathways while considering uncertainties from capital cost, conversion technology yield, hydrogen cost, natural gas price and feedstock cost. Yao et al. (2017) applied time-series price projection, which captures the uniqueness of the movements of single product markets based on historical prices. In addition to economic viability prediction, Diniz et al. (2018) also performed policy analysis to see how incentives can increase the attractiveness of renewable energy options.

Source of uncertainty

Many studies have discussed uncertainty and sensitivity. Egle et al. (2016) summarized information and data for the work and categorized them into the following topics: (1) resource demand, (2) material flow data of phosphorus and heavy metals, (3) nutrient content, (4) heavy metal pollutant contents in the recovered products, (5) organic micropollutant contents in the recovered products, (6) solubility and plant availability, (8) investment costs, (9) operational costs, and (10) revenues and savings. It shows that in a comprehensive study, there are many parameters and they might be obtained from different sources.

A review of previous studies reveals that the major sources of uncertainty can be summarized as follows: (1) information and data are collected from different sources and of different qualities; (2) same data varies in different data sources; (3) regarding economic analysis, there is limited data at both laboratory and pilot-scales as investment costs are unknown and it is difficult to predict the actual values; (4) the market price of recovered materials is unknown because of the novelty of recovered phosphorus fertilizers (Senthilkumar et al., 2012a; Egle et al., 2016; Yetilmezsoy et al., 2017).

Jedelhauser and Binder (2015) discussed how previous research dealt with data uncertainty regarding MFA and concluded that many of them did not consider uncertainties in their quantitative analysis. One example is Senthilkumar et al. (2012a), only compared descriptive data between studies. Egle et al. (2016) also considered data uncertainty in a qualitative concept by dividing data quality into 3 levels: low (+), moderate (o), high (-) and very high (--). Hedbrant and Sörme (2001) developed an approach consisting of 5 uncertainty levels (intervals from */1.1 to /10) determined by the type and reliability of the data source, which roughly imitated traditional statistical methods. For example, if the number of cows is 1000 heads with the uncertainty interval */1.1, the uncertainty range can be calculated as 1000/1.1=909 and 1000*1.1=1100; as a result, the very likely (with 95% probability) number of cows is between 909 to 1100 heads (Antikainen et al., 2005). This method is further adapted by Antikainen et al. (2005) and Cooper, Carliell-Marquet (2013) and Klinglmair et al. (2015). Based on this method, Antikainen et al. (2005) suggested using an average confidence interval for each flow when a number of calculations for one flow.

To estimate the phosphorus loads, Mekonnen and Hoekstra (2018) conducted a Monte Carlo analysis with 10,000 simulation runs; a normal distribution is applied to change the main variables and parameters within a standard deviation of 20% of their central estimate; some examples of these parameters are P load from non-sewered, wastewater P removal with sewage treatment and fertilizer application rate. Due to data scarcity, Montangero et al. (2007) chose a few easily assessed or measured parameters in material flow model, which are expressed by probability distribution (normal, lognormal or uniform); each flow is expressed by a function of parameters and its uncertainty is assessed by Monte Carlo simulation.

In terms of economic analysis, Egle et al. (2016) developed a sensitivity assessment considering the impacts of plant size and expected revenues; additional costs or revenues are presented as percentage deviations to predict the value between the worst (no revenues for the recovered materials, no up-scaling, no consideration of other benefits) and best scenario (maximum revenues for the recovered material, up-scaling, full consideration of other benefits) in the cost calculation.

Conclusion

Phosphorus is one of the fundamental and irreplaceable nutrients for all forms of life on the planet. Today, phosphorus is primarily obtained from raw phosphate rock. However, the increasing demand and use of phosphorus, especially in agriculture production, has posed challenges depletion unequal distribution over the world. Phosphorus security has become a big concern for Europe and other regions without or with little reserves.

Many studies have addressed the importance of recycling phosphorus from waste streams. There is a high phosphorus recycling potential in dairy wastewater as phosphorus concentration is several times higher than in domestic wastewater. We believe that with a sustainable phosphorus recovery from dairy wastewater, a great number of chemical phosphorus fertilizers applied in agriculture can be substituted. Therefore, particularly for Europe, the phosphorus security challenge can be mitigated.

To implement phosphorus recovery technologies at industrial scales, technical and economic feasibility crucially needs to be addressed. This report gives a brief introduction of different types of phosphorus recovery technologies and reviewed previous techno-economic analysis related to dairy industry and dairy wastewater. Such research activities are scarce, however, assessment methods, indicators and frameworks used for other industrial sectors or recovery process of other materials, might be of help in this case.

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A review of studies analysing drivers and barriers for the adoption of sustainability initiatives in the dairy processing value chain (ESR13)

This document contains a literature review on drivers and barriers for the adoption of sustainable alternatives in the dairy industry with a value chain perspective by the EU Marie Skłodowska-Curie action funded Early Stage Researcher 13 (ESR13) active in Work Package 3 of the REFLOW European Training Network.

Several science-based documents on the context of the deployment and marketing of alternative agricultural products were reviewed with a strong focus on waste-based fertilizers such as the ones generated by REFLOW project. Upon recognition, available information for specifically phosphorus-recovered fertilizers is scare. The general knowledge on the field gave insights into the drivers and barriers to be considered when addressing the concept of sustainability improvements in value chains.

For this literature review, main actors from the dairy value chain are presented. It was encountered that the adoption of sustainability initiatives, strongly depends on commitments from each actor of the value chain. As well as limitations of technical expertise and governmental support.

From the dairy value chain perspective, scholars agreed that there is a positive trend for sustainability actions. Nevertheless, it is recognized that there is a lack of available information addressing the problem which opens a window of opportunities for exploration of driver and barriers and concept development of new business models. In doing so, activities for further steps of this research project are discussed.

Introduction

Food consumption patterns are rapidly changing due to a concern and increasing awareness regarding the environment, nutritional value and health issues, primarily as stated by Tsakiridou et. al, 2008. While agricultural productivity has demonstrated significant achievements in terms of productivity, several authors have continuously argued that significant environmental impacts from industrialized agricultural production have resulted in a significant threat to natural resources arising from intensive water use, soil degradation, climate change, natural resources depletion, pesticides use, among others. (Notarnicola, et al. 2017,Parris, 2011,Adger, 1991).

To a major extent, the success of the expansion of the agricultural industry globally has relied on mineral fertilizers. Certainly, fertilizers have played a key role in high crop yields, but the downstream impact of the same nutrients degrade environmental quality and human well-being (Vitousek et. al 2009). A potential threat to food security globally is due to the consequences of the use of mineral fertilizers, particularly via leakage of Phosphorus (P) and Nitrogen (N) into the natural environment in the form of nitrates, ammonia, and phosphates (NH_3, NH_4, and PO_4).

The consequences of the industrialization of agriculture impact upon an extremely complex system and any response needs to take account of the increasing demand for food with a strict attachment to sustainable practices (Rosin & Campbell, 2013). With the recognition of adequate environmental management, food production systems are constantly changing with the aim of reducing their environmental impact, but also trying to satisfy the needs of the population with better quality products.

Future improvements of agriculture will entail a shift in market dynamics in the traditional (ex. mineral) fertilizer sector. Alternative bio-based fertilizers and specifically waste management technologies for nutrient recovery is expanding. Nevertheless, there is a knowledge gap in terms of the attitude, knowledge, and role of stakeholders regarding the low market adoption (Tur-Cardona et. al, 2018, Loo et al., 2013). An understanding of the acceptance of bio-based fertilizers, will facilitate the development, production, and marketing of innovative and sustainable alternatives across the entire value chain (Yiridoe et al., 2005, Rich et al., 2011, Ho et al., 2018, Hung, 2016,).

Context and study case

In the European Union (EU), the agricultural sector accounted for 1.1% of the total GDP in 2018. In that context, the dairy industry ascends to the biggest output sector with 13.2% followed by vegetables and horticultural plants with 13% with an average cumulative growth of 0.8% p.a. (Eurostat,2019, FAO, 2017). Displaying a threat to food security in Europe, several sustainability initiatives and regulations have raised to reduce the environmental impact and ensure a sustainable future for agriculture and the integrity of natural resources. As a result, and based on the concept of a circular economy, a potential improvement over the use of chemical fertilizers is the recovery of phosphorus from P-rich effluents from the dairy industry.

In this context, REFLOW proposes an interdisciplinary cross-sectoral European Training Network merging world-leading scientists and key stakeholders in dairy processing, fertilizer production, and phosphorus recycling. The main REFLOW objective is to address important technical and socio-economic challenges associated with the recovery of phosphorus from dairy processing wastewater and its recycling into fertilizer products enabling sustainable expansion of the dairy industry in Europe

The overall research goals of REFLOW are to develop and demonstrate processes for the recovery and reuse of phosphorus products from dairy processing waste (DPW), to establish their fertilizer value and optimum application rates, field trials and to address the environmental, social, food safety and economical challenges, ultimately finding market-driven solutions for the new processes and fertilizer products (ELO,2020). This research work uses anticipated REFLOW fertilizer products as the case of study for the development of market models for vertical integration of value along with stakeholders in the dairy value chain with a deep exploration of drivers and barriers for their deployment. Outcomes are intended to reveal which REFLOW fertilizers are more acceptable to the market and how value can be sustainably distributed along the entire value chain.

Value chains and sustainability actions

According to Kaplinsky & Morris (2000),the value chain describes a full spectrum of activities that are required to bring a product/service from an idea and to different phases of production, transformation, distribution, retailing, delivery to consumers and disposal after use. Nevertheless, this is a simple definition of value chains and the process is much more complex. By instance each of the steps first mentioned can have one or several subcategories. Within value chains, some key participants handle responsibilities of labour and capacities to perform tasks, and the roles of coordination within them reflect an important element in the governance and innovation of value chains (Kaplinsky & Morris, 2000). Researching elemental innovation aspects from a value chain perspective can be helpful to identify and understand factors that drive and facilitate improvements on products and processes, such as sustainability.

This individual REFLOW project aims to develop models for the integration of the economic value of recycled fertilizer products across the entire value chain. The entire supply chain of the dairy industry is a complex process with remarkable interactions, thus their internal processes and positions for the improvements and integration of added value sub-products (i.e. fertilizers) will require an analysis of stakeholder groups.

Understanding their internal processes will give elements to allocate social, environmental and economic benefits of the REFLOW fertilizer products and the development of new financial models to distribute the value across all stakeholders (ELO, 2020). The first step for this research project is to define the most important stakeholders and determine their position towards sustainability improvements, specifically around waste-based fertilizers. In doing so, a simplified boundary setting for this literature review is described in the diagram below.

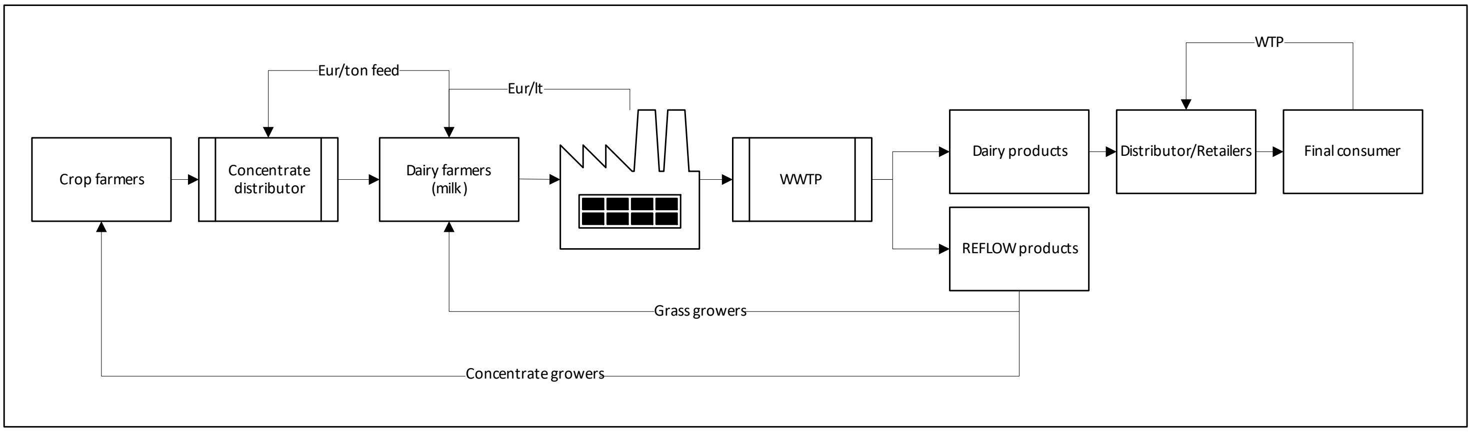

Figure 7. *Flow diagram for P-recovery from the dairy industry (REFLOW study case).*

Report outline

This literature is structured as follows; each chapter drives into the next chapter. The outline of the literature review falls into three main sections. Firstly, an initial exploration of different stakeholders towards sustainability actions towards Phosphorus-recovery processes in the dairy industry is done. This exploration also concerns about their positions in the value chain, as well as barriers and drivers to act on sustainability issues. Secondly, a brief description of fertilizer's legal context is presented and thirdly a section on economic principles, instruments for environmental regulations and financial models is included. Every section includes activities and questions that will drive activities of this individual research.

A key element for the success or failure of the concept of waste-based fertilizes, is the exploration of previous farmer's experiences and documentation of drivers and barriers involved in the adoption of sustainability measures, particularly fertilizers from non-mineral sources. The first section of this report starts exploring factors enabling the acceptance of fertilizers from biogenic wastes and secondary raw materials. As well, challenges and opportunities that should be addressed in the process.

Results from Literature Review

Farmers and sustainability actions (Fertilizers acceptance)

Large quantities of chemical fertilizers are imported in Europe Union to satisfy the demand for food production systems as stated by Tur-Cardona et al. 2017. Nevertheless, the consequences of the intensive use of mineral fertilizers attempting to achieve food security and human well-being are prominent. The concept of REFLOW fertilizers is a promising alternative to reduce the leakage of nutrients into the environment as well as a potential business case. However, an exploration of drivers and barriers to the deployment of the concept of REFLOW fertilizers must be analyzed. This section of the report integrates experiences from previous scholars regarding farmer's acceptance of more sustainable products based on their attributes in the context of the dairy industry.

The availability of alternatives to fertilizers from mineral is wide, but the adoption of them has been slow displaying a lack of studies on the attitude, roles and preferences of farmers which is crucial for technology development, the decision-making process as well for policymakers as reported by Tur-Cardona et al. 2017 and Case et al. 2016.

Tur-Cardona et al. 2017 identified the most important attributes for the acceptance of bi-based fertilizers as replacement for mineral based fertilizers and conducted a discrete choice experiment designed to reveal farmers' preferences and willingness to pay for these attributes. The authors included seven types of attributes: price reduction compared to mineral fertilizers, the form of the fertilizer (i.e. liquid, pellets, etc.), certainty in the content, fertilizer equivalence value, presence of organic carbon, hygiene, and speed of release of the nutrients. Among their findings, the authors stated that farmers found bio-based fertilizers inconvenient due to insufficiency for crop requirements, uncertainty in yields as well as lack of trust.

An alternative study conducted by Case et al. 2017, also pointed to undesired attributes of organic fertilizers such as odor, uncertainty in nutrients and difficult planning of use, whilst Lienert et al (2003) pointed out also the importance of micropollutant-free fertilizers as a potential motive for rejection. Altogether fertilizer attributes can have undesired effects on milk. Previous experiences reported in the literature, demonstrate the effects of milk from altered production systems on cheesemaking. These effects are commonly reflected in changes associated to milkfat, protein content, and minerals, such as calcium, phosphate, and citrate which are related to yield and quality of cheese as stated by Augustin et al. (2013)

These statements suggest that any bio-based alternative, such as waste-based fertilizers, laboratory testing and trials on the field are compulsory, as well as an increase in information regarding fertilizer equivalence value, yield information and composition are recommended since they will give certainty to farmers of the product's performance. Regarding the structure, it was found that farmers prefer granular solid fertilizer forms. Nevertheless, preferences among regions differed. Besides, it was found that farmers across Europe on average are willing to pay 76.6% of the current chemical fertilizer, as stated by Tur-Cardona et al. (2017).

Even though farmers positively display preferences for bio-based fertilizers, technological, legal and logistic issues should be addressed (Tur-Cardona et al. 2017), as well as the interaction of these factors in a specific study case (i.e. waste-based fertilizers) given that findings from several studies differ even if considering similar agricultural innovations (Case et al. 2017).

Case et al. (2017) conducted a study on current uses of organic fertilizers in Denmark. They specifically included three types of fertilizers in their study: 1) Unprocessed manure, 2) Processed manure and last 3) Urban wastes.

Among the characteristics that influence the adoption of agricultural innovation, aspects such as farmer capacity (age, size, access to capital), attitudes and awareness of environment-related issues. Also, external factors influence the adoption of agricultural innovations such as local regulations, policies, and market context. Similar statements were made earlier on in this report, with the relation between companies and farmers as mentioned by Case et al. (2017).

In the same study, the authors stated that almost three quarters of surveyed farmers had used bio-based fertilizers. Putting into perspective the study case of the REFLOW project, the category where waste-based fertilizers fit best, is in the third fertilizer category (Urban waste) as they evaluated mineral concentrates such as struvite

Despite the high share of farmers that have used bio-based fertilizers (almost 75% of participants), just 9% reported having used urban waste. Further, on future intentions of organic fertilizer use, they answered that unprocessed and processed manure is preferred before urban waste. Among types of urban waste, mineral concentrates were the least preferred alternative.

An opportunity is documented by Lienert (2010), as they conducted a study to determine the acceptance of the reuse of human urine as fertilizer in European countries. The authors found that 85% of respondents marked the alternative as a good idea (50% of farmers) and 70% would purchase food that is produced using urine-based fertilizer. In an alternative study, Lienert et al. (2003) determined that the price of such agricultural alternatives should be moderate as 4% of farmers would pay more to their current fertilizer use and 50% would pay less.

The importance of willingness to pay is also remarked by Dahlin et al. (2016) as their study on insights of consumer preferences for green fertilizers stated that price (40%) scored the highest importance, followed by brand name (19%). Thus, it shows the importance of conducting a study on the general perceptions, market segmentation and communication strategies to evaluate the potential success of waste-based fertilizers.

This section of the report highlighted farmers' preferences related to the use of bio-based agricultural fertilizer alternatives. It is observed that they do follow a similar pattern as for final food product consumers. The commercial success thereupon strongly relies on proving and communicating that new fertilizer developments match with their requirements at a competitive price. Within the barriers and drivers mentioned in this section, a set of activities are going to be delivered in a later stage of the research. Such activities will be focused on answering the questions mentioned below:

1. *Which are the most suitable crops and potential farmers for use of waste-based fertilizers in the context of dairy industry*
2. *What are the most preferred types of fertilizers and what physico-chemical attributes are required for farmers ?*
3. *What is the general attitude towards the concept of waste-based fertilizers for farmers? (concept experiment for P-recovered phosphorus from dairy waste)*
4. *What is the WTP for specific waste-based fertilizers?*
5. *What institutional, policy and financial arrangements can potentially foster the adoption of waste-based fertilizers in the dairy industry context?*

Dairy industry sector and sustainability actions

As regards associated environmental impacts, industries are observed to increase their awareness, leading to sustainability adoptions into their management systems in many sectors, including the dairy industry. Traditionally, safety and quality have been the subjects of innovation. Nowadays, sustainability initiatives are rapidly taking the position in the innovation agendas, as stated by Augustin et al (2012).

Several cases of initiatives to improve sustainability in the dairy industry sector have shown short and long-term effects on competitiveness by increasing revenues and decreasing outputs such as waste, cost savings, improved compliance, efficiency, and liability, as referred by Tailor, 2006. Hence, the future industry must center efforts on continuous improvement and the ability to produce milk while remaining economically competitive within a context of social changes and legislation, according to Augustin (2012).

Several studies have described that ccompanies with proactive Corporate Social Responsibility (CSR) initiatives can influence and engage economic, policy, and societal actors in initiating environmental transformations. Also, with a commercial perspective, a global trend of companies refusing to do business with non-certified (environmentally) companies can be seen. This derives into a general lose in competitiveness with domestic and international markets due to lack of action towards environmental issues. (Massoud et al. 2010, Chan, 2008, Thongplew et al. 2015)

Massoud et al. (2010), conducted a study to determine drivers, barriers and incentives for the implementation of environmental management systems in the food industry based in Lebanon. The authors stated that governmental regulations and stakeholders are widely recognized to put pressure for the adoption of sustainability measures. The authors emphasized that lack of government support, incentives, benefits of certifications, as well as no customer demand can easily be a barrier to environmental action. Hence it is important to conduct a review on barriers and drivers of the food industry to incorporate sustainability actions.

Farmer's commitments tend to be voluntary and dependent on the market and less likely to engage if there is no contract in between. But there is the scenario in which farmers and cooperatives do not pressure to adopt more sustainable practices as they sometimes face important issues such as productivity, milk quality, and management, thus resulting into a low priority aspect (Korthong, P., 2010 cited by Thonplew et al.,2015).

A similar statement is argued by, Augustin et. al (2012), which conducted a study determining the associated challenges from historical changes on-farm practices. The authors highlighted potential threats for dairy factories in the future. One of the statements provided is that an inevitable consequence of sustainability initiatives is altered raw milk. This alterations as side effect of a technological improvements or practice changes, such as fertilizing methods, processing, storage etc. Thereby any process adjustment should study the potential variations on milk quality. Negotiations between dairy processors and farmers are crucial for more sustainable practices.

Taylor (2005) examined concepts of the favorable implementation of cleaner production methods in the industry. The author argues that cleaner production measures in corporations are often displayed with a slow adoption rate. Awareness and technical expertise inside and outside the corporation is important. Alternative opportunities for cleaner production measures regulatory-driven initiatives. Nevertheless, the author stated that regulations must be rational and achievable for all sizes of corporations (i.e. small, medium), and interests of stakeholders should be adequately addressed.

Innovation opportunities in the dairy value chain

Environmental challenges and societal pressure will require that the dairy industry reformulates their processing approach. In doing so, the dairy industry should look at the entire supply chain to take cost-effective measures for milk conversion into final products. Value chain coordination can enable the introduction of incremental innovations with stable technological and organizational arrangements to address consumer trends. (Augustin et al. 2012 and Thonplew et al. 2015).

Thonplew et al. (2015), integrated a triad-network model based on corporate social responsibility principles and they distinguished three types of networks that influence dairy industries in the Netherlands and Thailand.

- Economic networks: interactions between the dairy industry and economic rules and resources.
- Policy networks: Interactions between the dairy industry and political-administrative rules and resources
- Societal networks: Interactions between the dairy industry and societal movements (consumers, NGOs) and their influence on environmental reform and/or economic actors.

The relation between dairy industry and retailer is strong, and they constantly negotiate on price, but they sustainability initiatives separately. Experiences in the Netherlands showed positive dairy industry-retailer cooperation with the push of organic milk The dairy processor conducted a campaign in cooperation with retailers to allow the company to do marketing in the shops. This marked a breakpoint to the era of organics in most of the stores as other retailers followed due to success, whilst the pressure in Thailand is inclined from the retailers to the industry, as stated by Thonplew et al. (2015).

Ultimately, Thonplew et al. (2015) recognize the influence of external organizations (i.e. consultancy, lobby production chain organizations, etc.) as a key element leading the market development and bringing actors together.

Sustainable practices in the dairy industry are also linked with governmental regulations via subsidies for dairy farmers, such as the EU Common Agricultural Policy (CAP). These regulations helped to introduce organic certified milk into the market with acceptable price levels. Likely these policy interventions are also driven by external organizations, which their influential role should be considered when addressing corporate sustainability, as argued by Thonplew et al. (2015).

This section of the report is analyzed the dairy industry sector and remarked barriers and opportunities for sustainability innovations considering, technological, economical with a value chain focus. Within the barriers and opportunities mentioned in this section, a set of activities are going to be delivered in a later stage of the research. Such activities will be focused on answering the questions mentioned below:

1. *What are the best negotiations schemes between dairy companies and farmers to incorporate waste-based recovered fertilizers?*
2. *Which organizational arrangements will maximize business outputs between retailers and dairy companies?*
3. *What political and administrative regulations can trigger or foster the adoption of sustainable adoption measures for dairy processing companies specifically in the context of waste-based fertilizer?*
4. *What financial or economical instruments can come along with (3)?*

Up to now, general remarks of farmers and dairy companies are discussed. The next step downstream the value chain is retailer companies. Retailers represent one of the strongest links between production and final consumption. The following section describes what are the attitudes and drivers identified by scholars referring to food product retailers and the positions towards the adoption of sustainability practices.

Retailers and sustainability actions

As a key stakeholder between the production, distribution, and consumption of food products food product retailers (i.e. distributors, supermarkets, etc.) play an essential role. Retailers source products (i.e groceries, clothing, etc.) through various distribution channels to provide goods for consumers to generate profit.

An impotant opportunity to incorporate and promote sustainability campaigns is identified within the retailer sector. To that extent, efforts to address issues of agricultural concern in Europe have played a previous role before at retailer level. EUREPGAP, established in 1996 (an integration of a group of 13 large European retailers). EUREPGAP was meant to establish sector-oriented protocols of standards for good agricultural practices. Initially this included horticultural practices.

This a clear example of an alliance between retailer actors which has been active for at least 20 years in which fertilizer regulations can be embedded. From this point of view, several initiatives and several retailer companies have developed their sustainability strategies, however, along the way they have encountered several difficulties and opportunities that have been documented by scholars.

This section of this report analyses primarily two scientific publications. Firstly, Chkanikova and Mont(2015) which conducted a literature review of academic publications and reports as well as personal communications with a diverse range of people within retailer firms (i.e managers, etc.) to determine the factors that affect the willingness and ability of food retailers to launch sustainable supply chain initiatives. And secondly, Belz and Schimidt-Riediger (2010), who investigated characteristics and drivers of sustainability strategies in the retailer industry.

As cited by Chkanikova and Mont(2015), Hoffman (2000) identified four major categories that play a role in shaping the sustainability agendas in the retail industry: regulatory, resource market, and social forces. Based on that classification, Chkanikova and Mont(2015) documented the main drivers and barriers based on the retailer's experiences (i.e personal communications) and several other authors. A summary of the main findings on the barriers/drivers to adopting sustainability initiatives can be seen in **Error! Reference source not found.** below:

Table 17. *Market factors, drivers and barriers for sustainability agendas in-retailer firms*[2]*.*

Category	Drivers	Barriers
Market factors (customers, competitors, industrial association, service provider pressure)	-Consumer demand for greener and healthier food (opportunity though green product differentiation) -Industrial norms (voluntary industry agreements and certification schemes) -Sustainability awards from third party organizations	-Globalization and the search for cheap/junk food (difficulties to monitor sustainability improvements in supply chains) -Geographic dispersion of suppliers where insufficient government enforcement is achieved -Product quality attributes from more sustainable production do not justify higher supply chain costs -The proliferation of eco-labels leading to consumers confusion and inability to recognize quality goods

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[2] Complete references for drivers and barriers can be found on Chkanikova (2015).

Resource factors (shareholders, suppliers, and investors)	-Demands for an increase of financial returns by cutting operational costs from stakeholders -Increased investors appeal because of sustainability -Possibility to the strong brand name, thus competitive advantage	-Cost measures. Retailers tend to address sustainability aspects that require lower investments and economic savings -Lack of expertise for developing and implementing sustainability strategies upstream in the food supply chain (i.e no influence of the environmental performance of own-brand suppliers -Costly collaborative relationships with suppliers -Lack of power to influence suppliers
Regulatory factors	-Pressure from governments in the form of regulations -International regulations (UN declarations SDG, EU action programs, etc.) -Anticipated actions of future regulations	-Lack of leadership and support of governments Lack of harmonization of regulations between countries -Too costly and/or strict legislation
Social factors (society, NGO, media, academia, etc.)	-The emergence of consumers who view shopping choices as an exercise of ethics and moral responsibility -Critical consumer pressure to address sustainability issues -Sustainability issues drawing the attention of environmental advocacy groups (i.e Greenpeace, WWF.) -Negative publicity in media via documentaries and films addressing sustainability -Fear to face court due to socially irresponsible behavior (i.e unfair, exploitive, abusive labor)	-Low interest and awareness of sustainability issues from consumers -Concern among the public on whether sustainability is a constructed phenomenon rather than a reality -Lack of scientific evidence and an agreed methodology to balance various environmental and social aspects making difficult for retailers to engage

Generally speaking Chkanikova and Mont(2015) observed a high degree of homogeneity on retail organizations, their business models and how they target their marketing strategies. The authors from this study concluded that there is a window of opportunity to investigate the relative importance of barriers and drivers for sustainable supply chains initiatives.

Belz and Schimidt-Riediger (2010), stated that certain target groups and the positioning of products are strategic decisions for sustainability marketing. The authors identified three major consumer segments within which companies target their marketing strategies. A classification for retailers was performed as a function of their target group. The classification of consumer segments can be seen in ***Table 18*** below.

Table 18*. Consumer type and characteristics based on socio-environmental positions. Adapted from Belz and Schimidt-Riediger (2010).*

Consumer type	Characteristics
Socio-ecological actives [1]	Small group and represent the innovator consumers of sustainable products
Socio-ecological approachable [2]	WTP more, but reluctant to compromise when it comes to the quality of the product.
Socio-ecological passives [3]	Not particularly conscious about social and ecological issues. This group does not perceive socio-environmental features as value-added.

Retailer companies provide goods to specific markets. **Error! Reference source not found.**, summarizes the classification retailer segments types and their characteristics according to their consumer target based on their position regarding socio-environmental aspects.

Table 19.*Retailer type and consumer target Adapted from Belz and Schimidt-Riediger (2010)*

Retailer type	**Characteristics**	**Consumer target**
Performers	Offer products of a very high social and ecological quality addressing the whole product lifecycle. They charge premium prices, commonly through small distribution channels.	[1]
Followers	Offer products of a very high social and ecological quality but less than performers.	[1][2]
Indecisive	Low social product quality and medium product quality. and not seem to pursue distinct strategies.	[2]
Passives	Process food products with a medium to low social-ecological quality. Lower prices and distribution via conventional food retail chains (mostly larger companies are on this segment. i.e wholesale.)	[3]

Based on the consumers and retailer types described above, it makes sense that companies target specific market segments, but as Chkanikova (2015) mentioned, several factors for barriers and drivers can be identified within the retailer groups. Belz and Schimidt-Riediger (2010) investigated the characteristics to determine “who influences whom to adopt sustainability practices?” This research encompassed the formulation of multiple hypotheses (7) to examine the relative importance of each factor via binary logistic regression analysis. The theoretical framework was aimed at determining which of 7 different factors (consumers, retailers, competitors, legislators, top management/owners, public exposure and industry membership) had the higher pursuit of sustainability marketing strategies in the German food market according to their socio-ecological position (performers, followers, etc.).

Main findings indicated that the inclusion of social-environmental marketing depends not only on the industry sector but also in the market segment within which company is competing. In addition, depending on the market segments, the stakeholders influence differently. The correlations performed by Belz and Schimidt-Riediger (2010) showed that the performer's segment significantly perceives pressure from consumers, while indecisive and passives displayed the opposite by perceiving less influence in their commitment to sustainability marketing. Some other effects were discussed such as no perceived influence by retailers and competitors. Regarding legislation, the performer's group correlated positively while the indecisive do not feel pressure from legislators to enhance sustainability marketing. This suggests a good approach is the definition of “real” actors to determine their positions, views, etc. and then determine what is likely to promote their action towards sustainability based on their business strategies with a strong focus on the dairy industry sector.

This section of the report is analyzed the retailer sector and remarked barriers and opportunities for sustainability actions considering market, resource and social factors. Within the barriers and opportunities mentioned in this section, a set of activities are going to be delivered in a later stage of the research. Such activities will be focused on answering the questions mentioned below:

1. *Which retailer companies will be part of the study?*
2. *What is their position towards sustainability (i.e. performers, followers, etc.)?*
3. *What do they need to achieve their sustainability targets?*
4. *Which are their specific main barriers and opportunities for the dairy industry sector and the inclusion of more sustainable dairy products? As well,*
5. *What institutional/governance arrangements can work best to foster the adoption of sustainability measures?*

Consumers acceptance of sustainable food products

According to Grunert et. al (2011), when dealing with new product development, the use of consumer insight techniques has benefits such as the identification of market opportunities, technology acceptability, optimization of product concepts, related communication and prototype test before launch. Hence, the first important step in the investigation of consumer attitudes.

“An attitude is defined as the evaluation of an object” (Scholderer, 2010 cited by Van Wezemael, 2011). Within a vast number of products, consumers are required to make purchase decisions daily. The reason for choosing a product merely depends on the attitude (positive or negative) towards the product; and attitude relies on a large number of product characteristics, as stated by Van Wezemael (2011). This combination of different product attributes and consequences inform the general evaluation of the product which leads to product choice, according to Shafie & Rennie (2012).

Nguyen Hoang Diem (2018) performed research on organic labelled food products in Vietnam. The author stated that when referring to consumer's attitudes and behaviours, three main components can be identified: i) cognitive factors, ii) affective factors and iii) conative behavioural factors. The first classification is related to the consumer's awareness of the product (ex. knowledge, thoughts, beliefs and perceptions). The second classification is referred to the emotional attachment (positive or negative) towards a product and thirdly, conative factors refer to the tendency to perform a behaviour (i.e. intention purchase, willingness to pay, purchase behaviour, etc.). Cognitive factors determine affective and conative factors in a decision-making context, as stated by Lavidge and Steined (1961), Grunert (2011) and cited by Nguyen Hoang Diem (2018). An adaptation to the conceptual diagram described by Nguyen Hoang Diem (2018) can be seen in Figure 8presented below:

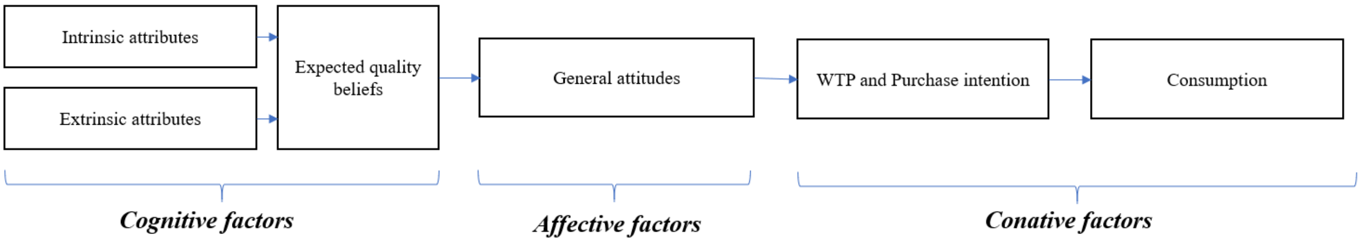

Figure 8. *A conceptual framework for consumer attitude and behavior towards food quality labels. Adapted from Nguyen Hoang Diem (2018)*

The construction of a general attitude (willingness to pay for final products/consumption), has its origins in intrinsic and extrinsic quality ques. The experiments conducted by Nguyen Hoang Diem (2018) on rice and vegetables exposed a positive relation of both intrinsic and extrinsic indicators of purchase intentions for safe products. This implies that consumers who have higher perceived importance of these attributes have higher purchasing intentions. However, the purchase intention coefficients between intrinsic and extrinsic indicators have a remarkable difference between them. This means that the influence of intrinsic indicators over final consumption is higher than the extrinsic attributes.

Nguyen Hoang Diem (2018) observed that food safety and health value (intrinsic) are one of the most important attributes for the evaluation of food products. Food safety encompasses food handling behaviours to all consumers and food handlers (cleanness, raw-cooked separation, safe water, and raw materials, etc.) to prevent foodborne diseases (WHO,2019). Healthy food refers to the dietary intake that protects to malnutrition and diseases that lead to global health risks (WHO,2018).

Both definitions are mutual and match with the findings from other scholars (Kutnohorska,2013, Hung,2015). Additionally, trustworthiness and quality are part of the intrinsic category. Quality attributes refer to appearance, texture and food flavor, whereas trustworthiness refers to anonymous production, publicity levels, transparency, and traceability. (Nguyen Hoang Diem, 2018, Meijboom et. al, 2006). These four factors are the most significant intrinsic attributes as mentioned by several authors. A categorisation of intrinsic attributes and their subcategories can be seen in Figure 9 presented below:

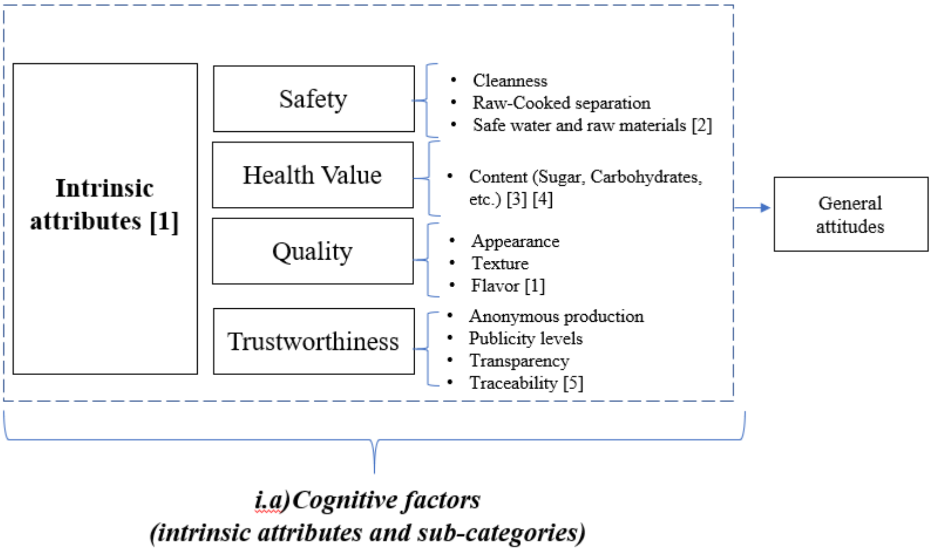

Figure 9. *Intrinsic attributes and subcategory disclosure.*

The allocation of environmental attributes to final consumer decisions might also play a role. Until now, none of the scholars cited identifies the category “Environment” as criteria for food product evaluation. The next section explains how the environment relates to intrinsic attributes and the degree of importance regarding final food product consumption.

Environment and health perceptions on final product acceptance

Consumers commonly associate environmental friendliness with healthiness. To prove that, Lazzarini, et. al (2016) conducted a study on consumer's perception healthiness and environmental friendliness of 30 different protein products on Swiss consumers. They found that these 2 variables are significantly correlated with life cycle assessments (LCA) and nutrient profiles, respectively. This is to say, that there is a direct relation between, environmental impact and perceived healthiness of food products merely based on common consumers.

Accordingly, both health and environmental food perceptions can be used in synergy. Environmental performance of food products can be included as a subcategory of intrinsic indication of purchasing behaviour documented by Nguyen Hoang Diem (2018) for either safety or healthy food. Int he same research (Lazzarini et. al (2016)) described the most mentioned criteria as predictors for both perceived healthy (Origin, organic, production, digestibility, nutrient content, etc.) and environmental friendliness (transportation, environmental impact, processing, labor input, etc.). In total, 9 factors make up environmental impact according to consumer's perceptions. Fertilizer use, and particularly phosphorus, can be included as a subcategory of environmental impact for eutrophication, according to Life Cycle Assessment methodologies**.**

The final identification of intrinsic attributes and perceptions can be seen in Figure 10 presented below.

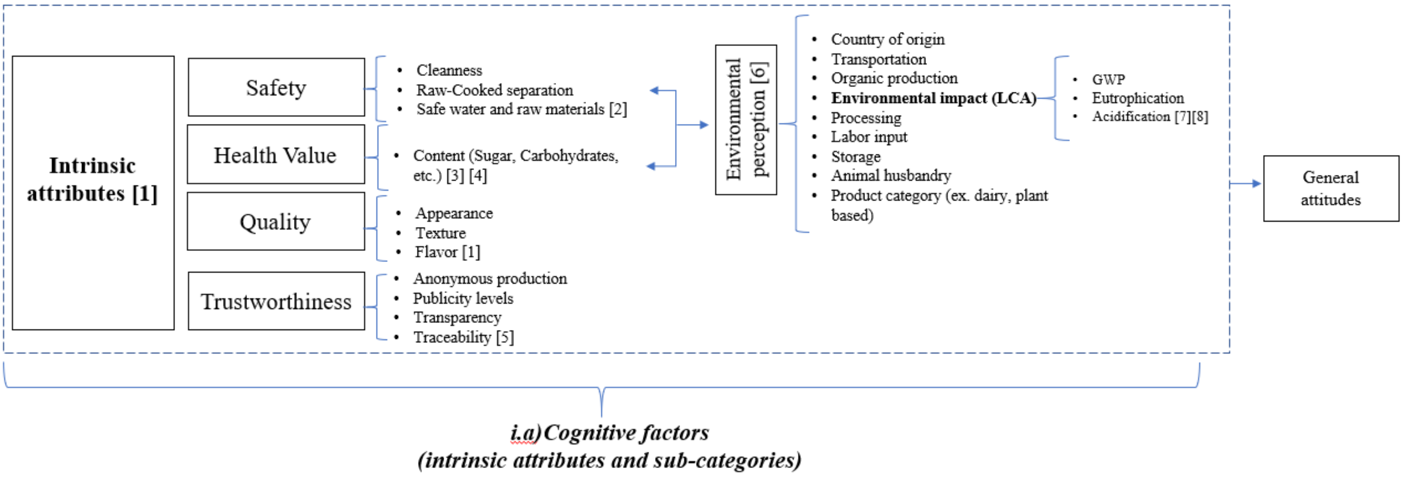

Figure 10. *Intrinsic attributes and subcategories.*

Diagrams references

[1] Nguyen Hoang Diem (2018) & Grunert et. al (2000).
[2] WHO (2019)
[3] WHO (2018)
[4] Bucher et al. (2015)
[5] Meijboom et al. (2006)
[6] Lazzarini et al. (2016)
[7] Thomassen et al. (2008)
[8]Djekic et al. (2014)

A general overview of consumer's motivations and priorities

By providing literature insights into consumer acceptance and food purchasing behavior for several food products, a global picture of the process was presented. It was also found that the process is composed of several subcategories and that environment indeed, can be specifically associated with food quality and health aspects.

Grunert et. al (2000) performed a study regarding quality perception and acceptance of dairy products by evaluating factors such as hedonic, health-related, convenience and quality aspects. The authors of this study stated that even while consumers may be motivated to process information (to lead acceptance/rejection of a food product), many do not possess the ability to process the information of the benefits of a given product. An example might be the production of a certain product with "waste-based fertilizers with a circular economy approach" because they do not have necessarily specific environmental knowledge.

This research suggests that if there are no specific psychological or health claims, there is little interest in final purchase intention for the enrichment/process modification and is unlikely to be translated into purchasing behaviour. Thus, environmental attributes should be addressed in terms of food health and safety rather than solely environmental benefits.

Nevertheless, a study conducted by De Graaf et al. (2016) analysed market opportunities for better sustainability practices of milk production and consumption, focusing specifically on animal-friendly milk in different consumer segments. The authors emphasized that consumers have an increasing, but highly variable interest in sustainability attributes. This was also identified by Van Loo et. Al (2013) who found that values differ significantly among countries, products and socio-demographic groups, as well on publicity (i.e. EU organic logo). According to Kutnohorska (2013), one of the main challenges for the adoption of organic food throughout the market is still it’s a very high price, limited availability, lack of trust and perceived value.

Despite the challenges mentioned by Kutnohorska (2013), Shafie (2012) highlighted the opportunity of sustainable food products (i.e. Organic label) since the results from this study found that participants are willing to pay approximately a 10% premium for organic food with an average gender segmentation of 9.5% by women and 11.4% by men. Van Loo et. al (2013) conducted a study of consumer attitudes, knowledge and consumption of organic yogurt. Among their findings is that the WTP for organic yogurt ranged from 15% for non-buyers to 40% extra for habitual buyers. Also, (VLAM,2012) cited by Van Loo et al. (2013) observed a willingness to pay from 55 to 64% extra for organic compared with conventional milk.

Up to now, general studies of final consumers of food products are described. A second step for this report is the analysis of the perspective and trends of food retailers as a key element between production and final consumption. The following section describes what are the attitudes and drivers identified by scholars referring to food product retailers and their attitude regarding the adoption of sustainability practices.

The legal framework around waste-based fertilizer innovations

REFLOW seeks to address important technical and socio-economic challenges associated with the recovery of phosphorus from dairy processing wastewater. Outputs from REFLOW include not only technical guidance and dissemination on novelty processes for phosphorus recovery but direct influence over the decisions taken in the policy, institutional and governance as well. In previous sections of this report is mentioned the importance of the regulatory framework for the success of sustainability initiatives implementation. Although Hukari et al. 2015 recognize that the “role of law harmonization, the inclusion of recycled phosphorus in existing fertilizer regulations and support of new operators would speed up market penetration of novel technologies”.

The performance, validation, and compliance of REFLOW products with regulatory frameworks of the EU will be provided to influence, elicit responses, and inform audiences with supporting evidence. Hukari et al. 2015 reviewed European Union laws and directives for the production, trade, and use of recycled phosphorus. One of the most important statements from this report is the recognition that the phosphorus recovery sector from biogenic sources faces fragmented and contradictory policy.

A general description of applicable legislation for phosphorus recovery processes included in Hukari et al. 2015 as well included by Huygens et al. (2018), is mentioned below.

Recovery and recycling

- Environmental Impact Assessment Directive (EIA): Directive from the European Commission to protect the environment and aim for the preparation of projects, plans, and programs with a view of environmental impact reduction. (EC, 2020)
- Directive on Industrial Emissions (IED): Directive from the European Commission with the objective of environmental and health by reducing harmful emissions from the industry sector across the EU member states through better application of Best Available Techniques (BAT)". (EC, 2020)

Both EIA and IED processes oblige operators to submit information about their processes and plants to the authorities such as wastewater treatment plants (WWTP), which in turn may grant an operation permit with or without certain conditions. Although this can differ in function of local legislation, as well as the registering type of the plant and input waste origins (waste imports for processing) as stated by Hukari et al. 2015.

Market placement

Waste framework directive: Directive from the European Commission who lays down waste management principles as well as basic concepts and definitions for waste recycling and recovery. As well states guidelines for wastes and secondary raw materials, also the difference by waste and by-products. This Directive states that waste ceases to be treated as waste when the object is used for specific purposes, a market exists, meets the standards applicable to products and the object does not lead to overall environmental or health problems. (EC, 2020, Hukari et al. 2015)

- European Chemicals Regulation (REACH): Directive from the European Commission aimed to identify the intrinsic properties of chemical substances with the protection of the environment and human health. The material operator needs to provide information to confirm these regulations in a central database in the European Chemicals Agency (ECHA) as well as the following:
- Classification and Labeling Regulation (CLP): Regulation specifying the guidelines to identify hazardous, chemicals and symbology for chemical products. This set of rules is part of the main European Chemicals Regulation. Fertilizer products are part of Regulations (EC) No 1069/2009 and (EC) No 1107/2009 and repealing Regulation (EC) No 2003/2003. OJ L 170/1(2019).

It is stated that some STRUBIAS materials already possess a registration under REACH regulations. On that extend, a verification to determine whether REFLOW products lay under this classification is important. Huygens et al. (2018)

Market segments for recycled materials

- Fertilizer regulation (Reg. EC 2003/2003). The regulation governing the placement of fertilizer products on the EU market. Recycled phosphorus can be applied as conventional fertilizer or for organic farming upon complying with fundamental principles of quality, safety, environmental, type of nutrients and must match with REACH and GLP regulations. As well, with the Sewage Sludge Directive for sludge application on agricultural lands. (Hukari et al. 2015)

Important to note that waste-based fertilizers have the opportunity to be commercialized as “organic” fertilizer, nevertheless, Hukari et al. 2015 reported that it is not well known whether sludge derived fertilizers has a real chance to be accepted by organic farming organizations. The same authors stated that current European regulation enables but does not support recycling efforts as no evidence of support systems such as recycling quotas, taxation of fossil phosphorus sources or similar is found.

Overall, this section of the report consisted of a superficial exploration of the legal background around waste-based fertilizers in the EU context. This section is limited to recovery and recycling, market placement and market segments for recycled materials. Further steps in this research should analyze legislation applicable to farmers. It is important to note that the available information, as well as the documentation of practical studies, is scarce. Further tasks for this research project must gather specific legal compliance from interested actors and local Directives, as they can vary among state members.

Conclusion

This report focused on reviewing mostly the position of stakeholders in the dairy value chain. It was aimed to document experiences towards the drivers and barriers for the adoption of sustainability initiatives. This review exposed that the environmental friendliness of food products is positively correlated with healthy products. Scholars have shown that there is a predisposition to pay more for more sustainable food products. Nevertheless, an important remark is that the opposite effect is displayed with the adoption of fertilizers from farmers.

These references are a clear indication of the potential market for the adoption of sustainability practices directly on food products. Nevertheless, special attention should be taken analysing preferences from the entire value chain. Further activities of this individual research will focus primarily on farmers, due to their tricky position towards the acceptance of waste-based products. With an understanding of their position towards the concept of REFLOW products, strategies (economic, technical, policy, etc.) will be referred.

Reviewed scholars continuously emphasized that lack of government support, incentives, benefits of certifications, as well no customer demand can be easily diminishing environmental action of dairy industry. This remarks the importance to understand their needs and requirements to effectively take sustainability measures. As well, how dairy industries can coordinate across stakeholder levels in the value chain. In addition, an exploration of legislation and economic instruments is recommended also.

Regarding retailer firms, scholars recognized that the positions towards sustainability can potentially influence the acceptance of environmentally friendly products. Retailer firms can therefore guide consumers towards better-informed decisions. Nevertheless, their position towards sustainability is remarkably divided into segments. Certainly, the perception of new food products among retailer firms, will be biased accordingly to their positions in sustainability.

As a concluding remark, it is important to note that most of the studies here described applying the sustainability concept as well as drivers and barriers in a general term. Next steps from this individual research will focus specifically in the dairy value chain and deployment of REFLOW products.

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ANNEX I

Environmental impact due to the WWTP

According Finnegan et al., (2017)'s study was found that the global warming potential (GWP) associated to the dairy WWT was on average 4.43 kg CO_2 eq/m^3 wastewater treated, energy contributes on average 84% of the total GWP (on average 9% due to WWT); while the total fresh water aquatic ecotoxicity (FEW) and marine aquatic ecotoxicity (ME), due to the emissions associated with the WWT, contribute approximately 10% and 40%. In the second study, Djekic et al., (2014) does not show the relative contributions of each subsystem (three subsystems: dairy farm, dairy plant and waste and wastewater management) and each impact category calculated. Subsystem dairy plant and waste & wastewater management contribute to GWP, acidification potential (AP) and eutrophication potential (EP), as the major environmental impacts. In the Mondello et al., (2018)'s research is shown that the highest potential environmental impacts related to the WWT are climate change (CC) (8.6% - 1.87 kg CO_2 eq/FU), water depletion (WD) (17.3%) and fossil depletion (FD) (15.5%). In the Palmieri et al., (2017)'s study, three dairy chains were assessed: in two chains the whole amount of whey is mixed in the wastewater, while in the third chain the whey is used in the animal feed production. So, in this case, the impacts due to the WWT will depend also from whey and its transport from the dairy to the municipal treatment plant. On the base of that, WWT is mainly responsible for ME due to beryllium in the WWT for the disposal treatment of the cheese whey. In the case of the Broekema & Kramer, (2014) research, the results have shown the quantification of the contribution from general waste treatment, instead of the WWT, to the environmental impacts of semi-cured Gouda cheese and semi-skimmed pasteurized milk. The waste treatment in semi-skimmed pasteurized milk production is responsible for marine eutrophication (MEP) ($7.39*10^{-7}$kg N eq/FU) and waste treatment in semi-cured Gouda cheese production is responsible for CC (0.005 kg CO_2 eq/FU). González-García, Hospido, et al., (2013) presents the environmental results associated to the FU due to the WWT process: abiotic depletion potential (AD) 0.001 kg Sbeq, AP 0.002 kg SO_2eq, EP 0.006 kg PO_4^{-3}eq, GWP 0.30 kg CO_2eq, ozone depletion (OD) $2.07*10^{-8}$ kg CFC-11eq, photo-oxidation formation potential (FOFP) $2.66*10^{-5}$ kg C_2H_4eq, consume energy demand (CED) 2.81 MJeq. In this LCA is also considered the transport of the wastewater from the dairy to the WWTP. According (González-García, Castanheira, et al., 2013) the emissions of P and COD from the WWTP are responsible for the EP but are not shown the relative contributions of the WWT process. According Kim et al., (2013), the on-site WWTP is responsible for eutrophication impacts. On the base of the LCA results, the eutrophication (MEP 1.26 kg N eq/tonnes of cheese consumed, and FEP 7.86 kg P eq/tonnes of cheese consumed) depends on the manufacturing process, which includes the WWT. The research of Dalla Riva et al., (2018) explains that the plant does not have the equipment to treat the wastewater derived from the plant process, therefore the wastewater is directly discharged into the municipal WWTP. On the base of this research, the main impact due to WWT is FEP (around 60%). Canellada et al., (2018) notices an effect of the subsystem waste on water depletion (WD) (31%) due to the recycling plastic contains and to the wastewater management. According A. Flysjö, (2011) the energy associated with the treating wastewater from the dairy plant to the municipal WWTP contribute to the carbon footprint (CFP). G. Doublet, (2013)’s study shows that the water and wastewater are responsible for WD (20%) during the pasteurized milk production. In the Vergé et al., (2013)’s LCA study is showed that the water use and wastewater of the milk and yogurt production are responsible for 1.6% and 0.7% of the total green-house gases (GHG) emission. According to the research of Yan & Holden, (2019) WWTP is the largest contributor (32%) for the EP due to the nutrients emitted from the WWT during the butter production. According Dalla Riva et al., (2017) the wastewater treatment is the main contributor for WD.

ANNEX II

LCI of the 18 P-recovery technologies

Technology		Resource demand [$\text{kg/kgP}_{\text{rec}}$]		Energy [$\text{kWh/kgP}_{\text{rec}}$]		Product [$\text{kg/kgP}_{\text{rec}}$]		Waste [$\text{kg/kgP}_{\text{rec}}$]	
Recovery from liquid phase	REM-NUT®	Ion exchange resin	0,2	electricity	6,4-7,2	Struvite	10-12	waste resin	0,07
		Na2CO3	0,7						
		NaCl	11,0-11,5						
		MgCl2*6H2O	10,9-12,7						
		NaOH	4,2						
	Ostara Pearl ®	MgCl2*6H2O	7,7-8,5	electricity	4,9-6,6	Crystal Green ®	12,6		
		NaOH	0,20-0,22						
	PRISA	MgO	1,9-2,2	electricity	2,3-2,9	Struvite	9,3		
		NaOH	0,43-0,50						
	P-RoC	CSH	9-12	electricity	4,2	Loaded CSH	10-13		
	AirPrex®	MgCl2*6H2O	10,9-12,7	electricity	7,5	Berliner Pflanze ®	10		
		Air	60-70m^3						
	DHV Crystallactor ®	H2SO4	1,3	electricity	5,1	Struvite/Cap	8,8		
		acetic acid	0,9-1,1						
		Ca(OH)2	6,8						
		NaOH	0,9						
		sand	0,2						
Recovery from sewage sludge	Gifhorn	H2SO4	17,4	electricity	7,1	Struvite	8,3	acidified sludge	
		Na2S	2,5						
		Mg(OH)2	2,0						
		NaOH	8,9						
	Stuttgart	H2SO4	15,5	electricity	6,2	Struvite	8,3	acidified sludge	
		MgO	1,6						
		acetic acid	8,8						
		NaOH	6,7						
	MEPHREC®	iron	3,3	electricity	8,3	P rich slag	40-42	iron slug	5,6
		O2	6,7	coke	2,4			dust	3,2
		activated coke	0,2						
		Ca(OH)2	1,8						
		water	5,9						
	Aqua Reci®	O2	32,5	electricity	11,5	CaP	5,0	ash	25,30
		CaCO3	7,4	gas	119				
		NaOH	0,5						
	PHOXNAN	H2SO4	24,5	electricity	9,5	Struvite	8,0-8,5	ash	34
		O2	34,1					heavy metal slag	2,4
		catalyst							
		Mg(OH)2	4,9						
		NaOH	4,6						
		CaO	2,2						

This tables were adapted from Amann et al., 2018

Technology		Resource demand [kg/kgP_{rec}]		Energy [kWh/kgP_{rec}]		Product [kg/kgP_{rec}]		Waste [kg/kgP_{rec}]	
Recovery from sewage sludge ash	AshDec®	NaCl	0,50/0,45	electricity	1,28/0,96	depolluted P-ash	12,43	filter cake	0,79/0,78
		MgCO3	0,91/0,097	gas	6,93/3,71				
		NaHCO3	0,14/0,10						
		O2	0,57/1,02						
	LEACHPHOS®	H2SO4	5,6	electricity	2,3	CaP	7,6	waste ash	24
		CaO	2,1					gypsum	2,3
		NaOH	0,23					wastewater	92
		CaCO3	0,46						
		water	4,5						
	PASCH	HCl	6,8	electricity	0,75	CaP	6,8	waste ash	9,1
		solvent	0,04					heay metal sludge	0,3
		NH4 solution	0,94					wastewater	136
		NH4HCO3	0,26						
		Fe(III)Cl2	0,35						
		CaO	3,7						
		water	136						
	RecoPhos®	H3PO4	7,8	electricity	0,59	RecoPhosP16	20,1		
		water	3,1	oil	0,70				
	Fertilizer Industry	H2SO4	4,9	electricity	2,0	SSP	16,6	wastewater	5,5
		water	7,5	oil	2,2				
	EcoPhos®	HCl (100%)	5,4	electricity	0,42	Phosphoric acid	3,2	waste ash	6,0
		steam	8,4					Ca/Mg chloride	3,8
		resin	0,003					Fe/Al cloride	1,5
		water	43					heay metal sludge	27
								wastewater	16
	Thermphos	clay	1,3	electricity	13,2	P4	0,3	CaSiO3 slag	14,2
		coke	1,0	gas	1,2			wastewater	3,0
		SiO2	3,1					dust	2,6
		water	3,1						

This tables were adapted from Amann et al., 2018