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Eco-efficiency of water and wastewater management in food production using value added and life-cycle assessment

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Keywords: Resource recovery, Wastewater treatment, Industrial water management, Holistic
decision-making, Food production, Value chain

ABSTRACT

Rising water demands and pressures on water resources call for water management in industrial production. An eco-efficiency framework may provide justification for water consumption and treatment, by combining environmental and economic dimensions in a life cycle perspective. We used principles of value added and value chain assessment in the life-cycle system boundaries for a consistent assessment of eco-efficiency. The method was demonstrated for a membrane bioreactor based decentralized wastewater treatment and reuse in a dairy. The LCA results showed that decentralized wastewater treatment and reuse in the dairy will improve the aquatic environment by 80% (freshwater) and 51% (marine water), but increase the negative impact on climate change by 27% compared to conventional wastewater treatment. The value chain assessment revealed that the decentralized alternatives would increase the value added by 1.3 to 1.4 € per m³ wastewater in the dairy, but also incur a value loss of 1.4 to 1.9€/m³ for associated stakeholders such as the centralized wastewater treatment plant. The case study revealed the critical nature of a well-defined system boundary that includes impacts along the entire value chain and exposed pros and cons for decision makers at both the system level and individual stakeholder level.

INTRODUCTION

Water scarcity is a growing concern for both developing and developed countries. Growing populations, changes in climate and inefficient use of our water resources are key drivers for developing water management of the future (Stakhiv and Stewart 2010). New technologies are introduced every day aiming to solve water issues; however it is often complex to assess whether they are suitable in life-cycle environmental and economic terms and for all involved actors. Eco-efficiency has been proposed as a holistic approach that considers a system, technology or a product from both an environmental and economic perspective, across the life-cycle and actors (Bidwell and Verfaillie 2000). The eco-efficiency assessment aims to support decisions that will create more economic value while reducing environmental impacts. The method was internationally standardized in 2012 by ISO 14045 (2012). The ISO standard requires use of the Life Cycle Assessment (LCA) framework for the environmental analysis. However, the ISO standard does not require a specific method for value assessment. Life Cycle Costing is a common measure of a systems monetary value (Lorenzo-Toja et al. 2016), but varying approaches are still explored in the literature (Martinez-Sanchez et al. 2015; Heijungs et al. 2013; Faragò et al. 2019).

As a closely related alternative to LCC, Value Added (VA) is a metric for economic performance that represents a core mission of businesses, i.e. value creation (Azapagic and Perdan 2000). It has been applied in earlier evaluations of the food industry (Laso et al. 2018; Rivera and Azapagic 2016). In the literature, VA is defined as “*the value from sales of all outputs minus the cost of inputs*”(Sikdar et al. 2004; Azapagic and Perdan 2000). However, the ambiguity of the VA concept is obvious because the definition of inputs and outputs used in the VA assessment varies across studies. Clift (2009) states that inputs in VA should only cover

ancillary materials and energy, while Sikdar et. al. (2004) advocates for the inclusion of investments too. On the contrary, Hunkeler et. al. (2008) wants inputs to also cover cost of services purchased (e.g labor), but do not want the sales of co-products (e.g. whey in the production of cheese) to be included in the outputs. VA has the strength of being a parameter businesses can relate to due to its resemblance of gross profit. VA prevents double counting (Moreau and Weidema 2015) and can be consistent with the boundaries of LCA and still include all upstream and downstream costs (e.g, End-of-Life costs) (Hunkeler et al. 2008). To fulfill the boundary of the LCA, VA has to include investments in new technology and anticipated future costs as explained by Martinez-Sanchez et al. (2015)

The food industry is the third largest consumer of water worldwide, mainly due to cooling and heating processes and high hygienic standards (Rad and Lewis 2014). In Denmark, the food processing industry accounts for 43 % of the industrial water consumption, and the dairy industry comprises 9.2 % of food processing's water consumption (Naturstyrelsen 2013). Danish Nature Agency (Naturstyrelsen 2013) estimated a potential 20-30% reduction in the total water demand in dairies by reusing water from cleaning systems and excess milk water compared to a baseline with no on-site water reuse. Besides high drinking water consumptions, dairies are often accountable for a wastewater production exceeding the water consumption due to a 87 % water content in raw milk (Naturstyrelsen 2013). Wastewater from the dairy industry has a higher organic load than domestic wastewater, which, besides the high flow, puts additional pressure on the centralized wastewater treatment plants (WWTP). Industrial wastewater production can be restricted by pollution regulations, but in some areas also by the capacity of the local WWTP. Changes in dairy production may lead to changes in wastewater production volume and composition and therefore, some dairies experience two drivers for

efficient water use: reduced water import and reduced wastewater effluent. An on-site wastewater treatment plant can allow the dairy to increase the production capacity without increasing wastewater production. One example is the use of membrane bioreactors (MBR), achieving a high quality effluent due to the increased removal of organic matter, phosphorus, ammonium, particles, bacteria and viruses compared to conventional wastewater treatment (Deowan et al. 2015). MBR technology has the benefit of being compact and well suited to fit in existing production facilities as a decentralized WWTP. The small area requirement and the high treatment efficiency makes the MBR technology attractive for dairies. In combination with reverse osmosis (RO) MBR treatment of water can achieve drinking water quality and thereby facilitate unrestricted wastewater reuse. Water reclamation within a major food processing industry has broad relevance across the variety of food processing industries (Compton et al. 2018; Meneses et al. 2017).

Our study aims to develop and demonstrate a method for eco-efficiency assessment consistent with both concepts of value added and life-cycle assessment and use a uniform system boundary for both the economic and environmental assessments. Also, our study provides an example of the eco-efficiency potential of a decentralized wastewater reuse facility. The example is based on actual data from the dairy factory HOCO, located in Denmark and the anticipated implementation of a Grundfos Biobooster for local wastewater treatment and water recovery.

METHOD

Our eco-efficiency assessment was based on a five step methodology aligned with ISO (2012):

1) Goal and Scope definition, 2a) Life-cycle assessment, 2b) Value chain assessment, 3)

Quantification of eco-efficiency, and 4) Interpretation with sensitivity and uncertainty analysis. It

was the aim to have a direct link between the LCA and value chain assessment (VCA) that ensures consistent system boundaries for both assessments.

Goal and Scope definition. The goal was to evaluate the eco-efficiency of alternative ways of handling HOCO's wastewater and compare these to today's business-as-usual approach. In 2015, HOCO produced 17,000 tonnes of milk powder using 544,099 m³ of drinking water and 530,000 tonnes of raw milk (Nielsen 2016). The life cycle covered HOCO's water chain, defined as water related processes required for HOCO to produce milk powder and by-products. A consequential life cycle inventory (LCI) formed basis for both the environmental and economic assessment. Therefore, the system boundary only included processes affected by changes made to the original system. The system was simplified by only including processes that were expected to be affected more than 1 % per unit cost, mass or energy flow per year. Both foreground and background impacts (See S1) were included in the system. The functional unit (FU) was "treatment of 1000 m³ HOCO wastewater", and the analysis assumed a 20 years operation of the treatment plant, assuming constant wastewater composition based on 2015 (Table 1). Reclaimed water needed to fulfill the regulations of 2015 as stated by Ministry of Environment and Food of Denmark (Table 1) for both wastewater effluent (Miljø- og Fødevareministeriet 2016a) and drinking water (Miljø- og Fødevareministeriet 2015). 2015 was the reference year of the system (Bhupendra 2016).

Table 1. HOCO's wastewater characteristics in 2015 as well as a selection of relevant Danish water quality parameters for wastewater effluent and drinking water.

Parameter	HOCO	Wastewater effluent criteria	Drinking water criteria	Unit
Volume	624,735	-	-	m ³

COD	1500	< 75	-	mg/l
Total nitrogen (T-N)	118	< 8	-	mg/l
Total phosphorus (T-P)	30	< 1.5	< 0.15	mg/l
Temperature	30	-	< 12	°C
pH	9	-	7-8.5	-

Case description. A Baseline case represented the current wastewater management at HOCO. In the *Baseline*, Nibsbjerg waterworks in Holstebro provided drinking water to HOCO and the centralized Holstebro wastewater treatment plant (WWTP) treated wastewater before discharge to the river Storå. Wastewater sludge was sent to Maabjerg Energy Center (MEC), a biogas plant that produces electricity and heat. Stabilized sludge from MEC substituted fertilizer on the fields in local farm areas (Figure 1). Two decentralized alternatives were considered. The first alternative *A1 MBR-nature* would treat the wastewater at HOCO to the same effluent standards as in the Baseline and discharge it to the same river. Sludge would be sent by truck from HOCO to MEC. In the second alternative *A2 MBR-reuse*, the MBR was amended by a RO plant and chlorine disinfection to produce reclaimed potable water. The reclaimed water would have drinking water quality and substitute drinking water import from Nibsbjerg waterworks to the dairy. The RO concentrate would be pumped to Holstebro WWTP for treatment. Similar to the other alternatives, sludge would be treated at MEC and stabilized sludge used as fertilizer (Figure 1).

Figure 1. The conceptual water chain for the baseline and alternatives evaluated in the eco-efficiency assessment. Solid lines indicate flow of water, and dotted lines indicate flow of sludge. WWTP: Wastewater Treatment Plant, MEC; Maabjerg Energy Center, MBR: Membrane Bioreactor, RO: Reverse Osmosis.

Life-Cycle Assessment. The environmental impacts were modeled using EASETECH (Technical University of Denmark 2016) (Version 2), which is based on mass flow analysis and

keeps control of homogeneous and heterogeneous flows (Clavreul et al. 2014). The main stakeholders related to the water system were Nibsbjerg waterworks, Holstebro WWTP, MEC, and the farmers in the area who would receive the stabilized sludge from MEC.

Inventory. E-mail and telephone correspondences with contact persons from the different stakeholders provided the majority of the foreground data. Data were process data from 2015, whenever it was available. If not available, data were estimated based on literature and stakeholder employees. The design of the MBR was provided by Grundfos and based on HOCO's data from 2015. Background data, e.g for production of chemicals, were collected from Ecoinvent (Version 3.3) (Frischknecht et al. 2005) (See S1 for general background processes and S2 for data and assumptions for MBR and RO technology).

A mass balance for Holstebro WWTP formed basis for our substance flow analysis (Table 2). For example, the total amount of water, COD, T-N and T-P discharged to the river from Holstebro WWTP in 2015 was scaled to HOCO's relative contribution to the total inflow in 2015. The same was done for the distribution in the wastewater sludge, except that COD was estimated from the total solids (TS) in the sludge as in Henze et. al. (1999). The mass balances including carbon and nitrogen flows for the MBR were established by design engineers (Bhupendra 2016), and the assumptions and approach used to estimate COD from TS from Holstebro WWTP were repeated for the MBR alternatives. Methane release from wastewater treatment was estimated as suggested by the IPCC guideline (Doorn et al. 2006). Furthermore, it was assumed that T-N in stabilized sludge substitute 45 % of average T-N fertilizer, and that T-P in sludge substitute 100 % of T-P fertilizer, which reflects that the uptake of T-N is not as efficient as T-P (Miljø- og Fødevareministeriet 2016b).

Table 2. Water and mass flows to Holstebro WWTP and contributions from HOCO in 2015.

Parameter	Total inflow	From HOCO	HOCO Contribution [%]
Flow (1000 m ³ /yr)	5,882	625	11
COD (1000 kg/yr)	4,782	937	20
T-N (1000 kg/yr)	337	74	22
T-P (1000 kg/yr)	67	26	39

Impact assessment. The environmental impact assessment followed the guidelines of ILCD 2011 (JRC - European commission 2011). Fourteen impact categories were modelled in EASETECH (Table S3-S1 S3). Additionally, the impact on freshwater resources, Freshwater Withdrawal Impact (FWI) was estimated as suggested by Godskesen et al. (2013) and Hybel et al. (2015). Because all water comes from within the same small region, we chose FWI over the recommended AWARE method (Boulay et al. 2018). Our approach for calculating FWI was developed specifically for groundwater based water supply and based on best available data at the time of the assessment, and at a finer scale than available in the AWARE dataset. The FWI was calculated as:

$$FWI = Q_g \cdot \frac{WU}{WR - EWR} \quad (1)$$

where Q_g is the water use along the value chain, WU is the total water use, WR is the total water resource and EWR is the environmental water requirements, all given in [m³/yr]. WU, WR and EWR were found for the groundwater resource in Nisum Bay catchment, covering the case study area (Miljøministeriet 2011). Water use, Q_g , is calculated as groundwater abstracted for water supply minus the water returned to the catchment from the WWTP. All water used is sourced within Nisum Bay catchment.

Life-cycle impacts were normalized relative to the average European person with factors from Laurent et. al. (2013). The normalization factor for FWI was also estimated for an average European person using the method suggested by Godskesen et. al. (2013).

Value Chain Assessment. VA across the entire value chain was considered, and so the VA assessment is referred to as VCA (Macfadyen et al. 2012). The VCA consisted of the same steps as the LCA. The VA for one process k in the value chain was found as:

$$VA = \sum_k UP_k \cdot Q_k - \left[\sum_j (UP_j \cdot Q_j) + \sum_q FC_q \right] \quad (2)$$

where the revenues [€/yr] and costs [€/yr] were calculated as the unit price of product (k) (UP) × supply/demand of product (Q)(Kleinfeld 1993). Costs(j) covered variable costs i.e. costs that vary with production such as electricity, and transfers, e.g., emission taxes and fees on wastewater discharge permission. The costs did not include labor and equity and that left VA as a measure of profit to pay such expenses (Sikdar et al. 2004). In other words, salary counted positively towards the economic value of the system. The future cash flows (FC) [€/yr] included investments (q) that were also relevant for the LCA, which in our case was the investment in new technology. FC was calculated by (Kleinfeld 1993):

$$FC = PI \left[\frac{i(1+i)^n}{(1+i)^n - 1} \right] - SV \left[\frac{i}{(1+i)^n - 1} \right] \quad (3)$$

The present investment (PI) and the salvage value (SV) was given as one-time costs [€], with PI as the current investment price of the technology and SV as the value of the technology at the end of its lifetime. The annual costs are based on assumed lifetimes n [years] and annual interest i (%). The interest rate was assumed to be 3% (Bankrate 2016) and it was assumed that the salvage value would be 10% of the initial investment. All costs are reported for year 2016.

We defined the Total Value Added (TVA) as the sum of VAs for each process and summarized for each actor along the value chain.

VA was calculated for all processes included in the LCA based on the mass flows. Air emissions were not associated with any expenses and as such the only mass flows not associated with a cost to the stakeholders. Inherent to the consequential approach, only costs and revenues that change between the cases were included. It was assumed that costs and revenues were not affected by external policy choices e.g. a new tax system.

Inventory. MBR investment costs and chemical costs were based on observed costs from a similar Swedish dairy factory and provided by Grundfos (Bhupendra 2016). Silhorko (Dalsgaard 2016) provided the investment costs of the RO unit as well as a cleaning and maintenance schedule, however costs of cleaning and treatment chemicals were estimated by an expert (Bhupendra 2016). Costs of electricity, drinking water, wastewater treatment and heat were all provided by the utility Vestforsyningen A/S (2015). These unit prices were also used to calculate the respective revenues e.g. for the waterworks.

HOCO does not have an actual income related to their sales of products. Instead they receive a yearly amount of money from the mother company Arla corresponding to their expenditures. It was assumed that the dairy production will remain the same for Baseline and both alternative cases. Hence, VA was only reflecting the difference between costs for each case.

The VCA included costs of fertilizer. The content of T-N in the sludge was the limiting factor for fertilizer substitution and the changes in costs between cases were calculated based on the T-N content. Baseline had the highest content of T-N in the sludge and could cover the largest area with fertilizer. The optimal amount of T-N on a field depends on the soil and crop type. We used a value of 150 kg N/ha based on the assumption that the crop is wheat and that

sludge has the same fertilizer value as manure from animals (SEGES 2015). Costs and environmental impacts of producing and using conventional fertilizer negatively impacted the alternative systems per hectare not covered with fertilizer compared to the Baseline. The cost of conventional fertilizer was calculated as 80 €/ha (PlanteavlsKonsulenten, No date).

Quantification of Eco-efficiency. The eco-efficiency indicator (EEI) combines the environmental and economic dimensions into a single number:

$$EEI_{j,k} = TVA_j \cdot \text{Reduced Impact}_{j,k} \quad (4)$$

An EEI score was calculated for each impact category (k) for each scenario (j). Only processes that changed were included in the analysis, meaning that EEI should be kept a relative term (Saling et al. 2002; Huppel and Ishikawa 2007; ISO 2012) compared to the Baseline. An increased EEI score will indicate an improved eco-efficiency performance.

Sensitivity and uncertainty. A contribution analysis, a perturbation analysis and finally an uncertainty propagation analysis assessed the sensitivity of results to inventory parameters and assessed the overall uncertainty. The method consisted of the following steps (Clavreul et al., 2012):

1. Contribution analysis to find the parameters that contributed with more than 5 % to the total results (S7).
2. Perturbation analysis to identify sensitive parameters from the contribution analysis. Parameters were changed by 10 %.
3. Uncertainty propagation to quantify overall uncertainty of results based on 1/3 of the most sensitive and uncertain parameters (S7). Probability theory and Monte Carlo sampling were applied. Parameters were assumed to have a normal distribution with a standard deviation

estimated from the assessment of the data quality matrix defined by Weidema and Wesnaes (1996) (S4). 10,000 Monte Carlo iterations were performed.

RESULTS AND DISCUSSION

The largest difference between the Baseline and the alternatives were found for climate change, freshwater eutrophication, marine eutrophication and the freshwater withdrawal impact (S3, Figure S3-S2 and Figure S3-S3). These environmental impacts depend largely on changes in water abstraction for water supply and energy requirements for the MBR treatment of wastewater at the dairy (Table 3).

Table 3: A selection of the most influential parameters for the LCA and VCA inventories per 1000 m³ of HOCOs wastewater. The unit prices for electricity, drinking water and wastewater are 0.2 €/kWh, 0.7 €/m³ and 2.2 €/m³ respectively. For the full inventory see Table S5.

Parameters	Amount			Comments
	Baseline	A1	A2	
Nibsbjerg waterworks				
Groundwater abstraction [m³]	937	937	132	Groundwater extracted from nearby wellfields. The volume was calculated by adding 7.1 % water loss to HOCO's drinking water demand in 2015.
Electricity consumption [kWh]	249	249	35	Scaled to volume purchased by HOCO in 2015. Assumed linear relation between energy consumption and water flow. Cost of electricity is based on (Vestforsyningen A/S 2015).
HOCO dairy				
Drinking water consumption [m³]	871	871	123	Volume was based on HOCO's total drinking water demand in 2015 for Baseline and A1 MBR-nature. The amount in A2 MBR-reuse was calculated as total demand minus reclaimed wastewater.

Wastewater to Holstebro [m ³]	1,000	-	249	Volume was based on HOCO's total wastewater production in 2015 for Baseline. Volume for A2 MBR-reuse was RO concentrate. No wastewater production in A1 MBR-nature.
Electricity consumption [kWh]	-	1,705	2,277	For wastewater treatment only. Based on a MBR design specifically for HOCO.
Methane emission to air [kg]	-	83	83	For wastewater treatment only. Expected emissions according to IPCC 2006 guidelines.
<i>Holstebro WWTP</i>				
Electricity consumption [kWh]	1,064	-	72	The electricity consumption and associated costs were scaled to HOCO's contribution to the total COD and T-N to Holstebro WWTP in 2015, which was 20 %.
Methane emission to air [kg]	75	-	2	Expected emissions according to IPCC 2006 guidelines.

Figure 2. Normalized LCA results with estimated change in environmental impacts for the two considered alternatives compared to Baseline. A positive change indicates a reduction in environmental impacts, and thus an improvement compared to Baseline. Alternative 1, where treated wastewater is pumped to the recipient (A1 MBR-Nature) generally performs worse than Alternative 2, where wastewater is recycled at

The wastewater treatment efficiency of the MBR technology resulted in a better aquatic environment when considering freshwater and marine eutrophication (Figure 2). In A1 MBR-nature and A2 MBR-reuse, freshwater eutrophication impacts were reduced by 1.8 PE and 2.3 PE per 1000 m³. A2 MBR-reuse has the largest environmental savings compared to Baseline

because of improved treatment of wastewater from HOCO. In A2 MBR-reuse water passes the MBR and RO unit and RO concentrate is pumped to the Holsterbro WWTP. A2 MBR-reuse released approximately 97 % less phosphorous to the aquatic environment than A1 MBR-nature and approximately 99 % less than Baseline. Due to the extra treatment in A2 MBR-reuse, marine eutrophication would also decrease with less nitrogen ending up in the ocean. HOCO contributes indirectly to marine eutrophication through the release of N_2O to air caused by the 27% increase in electricity use for the MBR and RO unit (Figure 10 S6). The RO unit in A2 MBR-reuse would produce 467,443 m³ of drinking water from HOCO's wastewater. This amount can substitute 86 % of HOCO's yearly drinking water consumption and reduce the freshwater withdrawal impact by 9.3 PE per 1000 m³ wastewater. As such, freshwater withdrawal represented the highest saving of all considered impact categories. The reclaimed wastewater means less abstracted groundwater at Nibsbjerg waterworks resulting in a total saving of 443 PE compared to Baseline and A1 MBR-nature.

These savings for the aquatic environment came at the cost of increased climate change impacts (Figure 2). The increased impact on climate change stemmed primarily from a 60 to 114 % higher electricity consumption by the MBR and MBR plus RO alternatives compared to conventional wastewater treatment (Table 3). Furthermore, there was estimated a 12 % higher methane release from the MBR (Table 3) due to an improved conversion of organic matter. Considering the generally hard water groundwater in Denmark, water softening is necessary to improve overall life expectancy of production equipment. A2 MBR-reuse was found to avoid some electricity use for water softening compared to Baseline. It also reduced the need for gas heating by 29% due to demineralization by RO and higher temperatures of the reclaimed wastewater (Table S5). Still, such savings were not enough to offset the impacts from increased

energy use in the A2 MBR-reuse compared to Baseline and A1 MBR-nature. The electricity and chemical consumption of Holstebro WWTP dropped for the alternatives A1 MBR-nature and A2 MBR-reuse, because less water would be treated at the plant when HOCO treats their own wastewater. Since less drinking water would be produced at Nibsbjerg waterworks in A2 MBR-reuse, Nibsbjerg waterworks would also reduce its electricity consumption compared to Baseline. Again, the savings in impacts from Holstebro WWTP and Nibsbjerg waterworks were not found to off-set the performance of the MBR alternatives.

Table 4: The change in VA compared to Baseline given in €/functional unit i.e. 1000 m³ of HOCO's wastewater.

	Nibsbjerg waterworks	HOCO dairy	Holstebro WWTP	Biogas plant MEC	Agriculture	TVA
A1 MBR-nature	0	1,386	-1,918	2	-6	-536
A2 MBR-reuse	-454	1,307	-1,386	0	-4	-538

In three out of the four mentioned impact categories, HOCO's environmental performance decreases with a MBR implemented. The VCA, however, showed that HOCO's VA increases with approximately 40 % with the MBR and MBR plus RO implemented (Table 4). Costs associated with discharging wastewater to Holstebro WWTP was found to be around 4 times higher than costs related to the MBR. Contrary to the positive impact on HOCO's VA, VA for all other stakeholders was found negative. In A1 MBR-nature, Holsterbro WWTP VA dropped by 1,918 €/1000 m³. This loss corresponded to 11 % (Table 2) of the WWTP's yearly revenue, assuming the only revenue is wastewater treatment. The loss in revenue was less in A2 MBR-reuse because the RO concentrate contributes additionally 155,814 m³ wastewater to

Holstebro WWTP. In A2 MBR-reuse, the water utility experienced a 20% reduction in VA compared to Baseline due to an 86% reduction in water consumption at HOCO. In a broader systems perspective, the overall TVA would not favor the implementation of MBR at HOCO. Compared to the Baseline we estimated a total loss of 536 and 538 € VA per 1000 m³ for the two MBR alternatives.

Figure 3. The uncertainty propagation for the total results of the LCA and VCA estimated from 10,000 Monte Carlo iterations (used parameters and estimated SD in Table 12 S6). FU = Functional unit i.e. 1000 m³ of HOCO's wastewater.

The uncertainty propagation did not support a significant difference between Baseline and the alternatives for the climate change impact. The difference was significant for the freshwater eutrophication impacts between all cases (Figure 3). Baseline and A1 MBR-nature had less significant difference for marine eutrophication than Baseline and A2 MBR-reuse. For the economic evaluation, the difference between the two MBR alternatives was not significant. Also,

the uncertainty overlap between Baseline and A1 MBR-nature indicated that Baseline's TVA could potentially be as low as the MBR alternatives.

Overall, there are both benefits and drawbacks to the implementation of the MBR alternatives at HOCO. The LCA showed that the savings in quality and quantity of the aquatic environment were on behalf of increased climate change impacts. Similarly, the VCA suggests that the MBR implementation will be economically beneficial for HOCO, while a loss is expected for the entire value chain.

Eco-efficiency quantification. Assuming equal weights between environmental impact categories and value added, an eco-efficiency indicator shows that the economic losses were outweighed by the environmental benefits for freshwater withdrawal (A2 MBR-reuse), freshwater and marine eutrophication (Figure 4). This can be perceived an eco-efficiency improvement of the system compared to Baseline, for these impact categories only. On the other hand, if considering climate change and freshwater eutrophication (A1 MBR-nature) alone, the MBR worsens the eco-efficiency of the system compared to Baseline. A decision will therefore depend markedly on weights assigned by the decision maker to the environmental and economic impacts. Such a weighting scheme is considered outside the focus of our study, but suggestions are found in the literature, e.g. by letting a selection of stakeholders representing different preferences assign weights to the impacts (Søndergaard et al. 2018; Godskesen et al. 2018).

Figure 4. Eco-efficiency quantification of the four considered environmental impact categories and the total value added. Results are relative to Baseline placed in Origo.

Further studies should address the following six shortcomings of our case study: 1) By decentralizing the treatment of HOCO's wastewater, HOCO will be given an opportunity to increase their production, which adds value to HOCO. The potential realization of an increased production is unknown and therefore not included here, but it would benefit the choice of a decentralized MBR treatment plant. 2) With a decentralized wastewater treatment at HOCO, Holstebro WWTP may experience a loss in efficiency, because HOCO's warm wastewater increases the metabolism in the biological tanks (Agerbo 2016). The actual response of the WWTP to a changed inflow composition has not been modelled here, but it would probably punish the environmental performance of the decentralized treatment at HOCO. 3) The MBR technology is better at retaining microorganisms, such as antibiotics, pathogens and micro contaminants compared to conventional treatment. This is an environmental benefit not yet quantified by the

conventional LCA framework (JRC - European commission 2011) but with potential to be included in the future (Harder et al. 2017). 4) Vestforsyningen A/S are planning to build a new WWTP in 2022 as a consequence of the current WWTP reaching its end-of-life (Holstebro municipality 2008). 73 % of the incoming wastewater is from dairies and slaughterhouses (Agerbo 2016). If the trend for decentralized wastewater treatment increases, it needs to be included in the dimensioning of the new WWTP and this could lead to considerable savings in the re-investment for Vestforsyningen A/S. 5) It was not the goal and scope of this study to assess how and when VA is spent by stakeholders. The VA assessment provides a one-year snapshot of value generated for each actor. The change in water system operation was assumed to not directly affect prices of e.g. chemicals, change in interest rates, or affect dairy production and disposals. A verification of this assumption would require a second iteration of the VA based on a survey among stakeholders evaluating their responses to the proposed new water system. Their response could lead to new scenarios to be analyzed. 6) The method applied here was designed to quantify relative eco-efficiency. If the implementation of a water efficient solution leads to an overall increase in production, this might lead to an increase in drinking water demand, wastewater production and energy consumption. This would counteract the benefits and could lead to even higher emissions, the so called rebound effect (Hauschild 2015).

These six issues are likely to support the results we found, i.e. that the MBR has positive effects on HOCO's economy and the aquatic environment, but negative effects on the economy of dependent upstream and downstream stakeholders, except for the potential savings at the re-investment in central wastewater treatment. One important limitation to our study is the lack of measurements of direct GHG emissions from both conventional and MBR wastewater treatment. Future studies should investigate the changes in CH₄ and N₂O emissions related to the biogas

production and nitrogen removal processes. This eco-efficiency assessment documented the economical connection between water supply chain stakeholders, such as Holstebro WWTP and Nibsbjerg waterworks, with their large consumers (e.g. dairies), and how changes can affect them negatively. Considering the high wastewater costs, HOCO would either choose A1 MBR-nature or A2 MBR-reuse. Installing the MBR at HOCO would result in a net increase in their electricity consumption by 2.6 % for A1 MBR-nature and 3.6 % for A2 MBR-reuse. The increase in electricity for both alternatives does not have a major impact on the costs and environmental impacts already associated with HOCO's electricity consumption. The systematic value chain analysis embedded in our eco-efficiency method has revealed pros and cons important for the decision makers at both the system (authority) level and for the dairy company. On this background, it appears as a valuable framework for any future holistic assessments of water saving initiatives in the food processing industry.

Method evaluation. The method showed how a change for one stakeholder affected other stakeholders within the defined system boundary. This was observed for both environmental impacts and value added and is an important asset of the methodology with more focus on sustainability planning in a more holistic manner. Our method avoided appraisal of environmental impacts in economic terms, e.g. T-P discharge to nature, like in Cost–benefit analysis (Pearce et al. 2006). Monetizing such externalities would introduce more uncertainty to the analysis. Still, VA is a good indicator of the immediate effect on the economic activity. Different from similar studies (Laso et al. 2018; Rivera and Azapagic 2016), we chose to include the value of salaries in the value added as an acknowledgement of the increased economic activity for the stakeholders. These considerations, as well as the fact that the method can be based on the same mass flow

399 analysis and system boundaries set by an internationally accepted LCA method, makes the applied
400 eco-efficiency method useful and transparent for decision making.

401 As demonstrated above, the eco-efficiency framework developed is useful for
402 accommodating an increasing requirement from society for industries to report on topics like
403 environment, climate and societal costs. The framework further helps identifying links and
404 possible feedback responses of proposed changes to the water management, which will benefit
405 decision makers.

406

407 ASSOCIATED CONTENT

408 **Supporting Information**

409 List of background processes used in the LCA

410 Construction materials for MBR, RO and recycle potentials,

411 Impact categories and overall results of the LCA performed in EASETECH

412 Data Quality Matrix for LCA and VCA parameters

413 Full inventory for the LCA and VCA

414 Mass balances for COD, T-N and T-P for relevant processes and MC parameters

415

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419 **Notes**

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429 **ABBREVIATIONS**

430 LCA, life-cycle assessment; LCC, Life-cycle costing; VA, value added; VCA, value chain
431 analysis; WWTP, wastewater treatment plant; MBR, Membrane bioreactor; RO, reverse osmosis;
432 FU, functional unit; MEC, Maabjerg energy center; COD, chemical oxygen demand; T-N, total
433 nitrogen; T-P, total phosphorus; ILCD, International Reference Life Cycle Data; FWI, freshwater
434 withdrawal impact; TVA, total value added; EEI, eco-efficiency indicator

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