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Obligatory inclusion of uncertainty avoids systematic underestimation of Danish pork water use and incentivizes provision of specific inventory data

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ABSTRACT

Livestock production is one of the most water-use-intensive economic sectors globally, and pork is the biggest of all meat sectors, necessitating continuous improvement of the sector's water use. Environmental product declarations are one way of incentivizing environmental performance, but with the majority of the water use occurring in primary pig and feed production, methods are required that quantify the water use over the entire value chain. Life Cycle Assessment applies such an approach, and the European Commission's Product Environmental Footprint framework uses this methodology. Product Environmental Footprint studies can use generic data for the pig production and feed cultivation stages and results communicated without uncertainty. Current study aimed to test if using database water footprint inventories could lead to a systematic underestimation of the water use in Danish pork production. A probabilistic surface- and groundwater footprint inventory assessment of the production of 100 g pork in Denmark was carried out. Danish average industry data was used to assess the possible range of water use for domestic Danish processes and FAOSTAT- and Water Footprint Network data for imported feed. Monte Carlo simulations were used to create water footprint inventory intervals, which were compared with intervals for three inventory databases: EcoInvent, Agribalyse, and Agri-footprint. The water footprint inventory intervals for Danish pork ranged from 3.8 to 9.2 L/100 g with a coefficient of variation of 21%. Database values were significantly ($p < 0.001$) left-shifted by 3.0–3.9 L/100 g and 4.4–6.6 L/100 g with significantly different ($p < 0.001$) coefficients of variation of 6.4% and 12.3% for Agri-footprint and Agribalyse respectively. This makes using generic data preferable to using primary data for producers with low water efficiency. Instead of demanding primary data, it is recommended that uncertainties in databases capture the observable variability, and that environmental product declaration results must be communicated with their associated uncertainty. This could incentivize provision of primary data and avoid deliberate underestimations of water use.

1. Introduction

Freshwater supplies are increasingly strained from consumptive use and pollution (WWAP, 2015). Food production is one of the largest contributors to the exploitation of this resource (Hansen et al., 2017), and meat production accounts for the majority of the sector's water use (Mekonnen and Hoekstra, 2012). In Denmark, agriculture is responsible for more than half the country's total direct water use, the majority of which is used for rearing livestock (DST, 2016). Globally, the pig industry is the largest of all the meat sectors, with an annual production exceeding 1.4 billion tons (FAO, 2016), and Denmark follows a similar trend. The Danish pig industry supports a standing population of more than 12 million pigs and piglets, twice the 5.78 million human residents (DST, 2016). This allows for the production of approximately 2 million tons of pork annually. The intensity and scale of pork production necessitates a continued focus on improving efficiency and sustainability

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of the sector's water use.

One way to stimulate more efficient and sustainable water use is by monitoring, reporting, and verifying it at the product level, as there is a positive correlation between environmental performance and willingness to pay. Additionally, consumers, civil society, and business can make more informed decisions when water use information is communicated through Environmental Production Declarations (EPDs) (EC, 2013). For pork production—and most other livestock products—the majority of water use occurs in the stages prior to the stables and slaughterhouses (de Miguel et al., 2015; González-García et al., 2015), predominantly for irrigating feed crops (Pfister and Bayer, 2014). Consequently, methods that assess the water use over the entire product lifecycle are required to accurately and justly account for the amount of water used to create the final product.

There are two standardized methods to assess water use applying a value chain perspective: the water footprint by The Water Footprint Network (Hoekstra et al., 2011) and the water footprint by ISO (ISO14046, 2014), which builds upon Life Cycle Assessment (LCA) (ISO14044, 2006). Results from studies on value chain water use (i.e. the Water Footprint Inventory (WFI)) vary significantly with values from 1.4 L/100 g of Danish pork following the ISO method (TS-RM, 2015) to 23.6 L/100 g of pork (Dutch) applying the Water Footprint Network method (Mekonnen and Hoekstra, 2012). While methodological differences exist, such as disagreement on whether a water footprint is volumetric or impact-oriented (Hoekstra, 2016; Pfister et al., 2017), both standards apply bottom-up approaches, and the way water use is accounted for is similar in the two methods (Hoekstra et al., 2011; ISO14044, 2006). Consequently, the difference in results stems from study-specific data-, parameter- and modelling choices. To minimize the effect of unavoidable normative choices, the EC-JRC developed the Product Environmental Footprint guideline (PEF) (EC-JRC, 2013) along with PEF Category Rules (EC, 2016) to harmonize EPDs, and enable a market value for sustainability (EC, 2013). The current paper focuses on issues arising from data use choices.

In LCA, a distinction is made between the specific system under analysis (i.e. the foreground system) and the systems providing inputs to the foreground system (i.e., the background systems). Foreground systems are generally modeled with specific data, while the background system can be modeled with generic data from databases (ISO14044, 2006). It has been shown that WFIs can vary up to an order of magnitude depending on which inventory database is used (Berger and Finkbeiner, 2010). Database values often come from studies at specific sites at specific times and, while inventory studies and databases sometimes provide country-specific values or compare one sector to another, they rarely focus on capturing the variability within the sector (Notarnicola et al., 2017). McAuliffe et al. (2018) found significant intra- and inter-farm variation; and generally, agricultural production systems have large spatiotemporal variability due to climatic-, soil-, and management heterogeneity (Notarnicola et al., 2017).

To understand how LCA or water footprint studies on pork production have addressed uncertainty, 49 scientific publications that contained the words “life cycle assessment” OR “water footprint” AND “pork” was identified (See Supplementary Information (SI-A) for details of the evaluation). While roughly two-thirds of the studies addressed uncertainty, 12 studies did not mention or discuss possible implications of uncertainty or variability at all. Roughly half the papers were case studies, of which one-third included water in the inventory ($n = 9$). Only one study reported the water footprint with quantitative uncertainty, with variance and uncertainty only attributed to the mass of the pigs. About a third of the cases applied Monte Carlo simulations, but less than half of these provided confidence- or prediction intervals, effect sizes, or any other statistical elaboration of their findings. No studies where the variability of the WFI of pork production is evaluated and results communicated with prediction intervals were identified, hence it is not possible to evaluate within which range to expect to find the value chain water use associated with Danish pig production.

Since the early nineties, LCA developers have argued that uncertainty should be quantified and reported in LCA studies (Huijbregts, 1998), yet standardized assessment and reporting of variance and uncertainty was not made a requirement in the PEF Guidelines (EC-JRC, 2013). Instead, the PEF guideline seeks to deal with the variability issue by defining which generic data sources to be used when product specific data is unavailable, and defining when primary data is required and when generic suffices, in product category specific data quality requirements (EC-JRC, 2013). The approved PEF product category rules for feed products require only the feed processing steps modelled with specific data (TS-FP, 2018), even though the majority of water use occur during cultivation of the feed ingredients.

It remains unclear if the water use and uncertainties in generic databases in combination with EPDs being allowed communicated without descriptive statistics makes the use of generic data favorable to providing primary data in pork value chains with inefficient water use. To address this knowledge gap, the current study tested the hypothesis that using database derived water footprint inventory values for pork and feed production could lead to an underestimation of the water use in pork value chains.

2. Materials and methods

To test above hypothesis; first, a water footprint inventory assessment of Danish pork was carried out calculating the prediction interval for the water footprint inventory of Danish pork as well as the main feed components. Second, the prediction intervals were compared with values in databases.

2.1. Water footprint inventory methodology: Danish average pork production case

The case study was carried out with consideration of the ISO14046 Water Footprint Standard (ISO14046, 2014). Following subsections describe the water footprint inventory assessment study design, and are followed by a section describing the inventory analysis.

2.1.1. Goal and scope of assessment

The goal was to provide WFI prediction intervals that would capture any Danish pork value chain or production stage thereof and compare these with WFI from the Agri-footprint database (Blonk, 2015b), the Agribalyse database (Koch and Salou, 2016) and the EcoInvent database (Wernet et al., 2016). To enable comparison of the WFI in the current study with the WFI in databases, the methods and modelling approaches were aligned (ISO14044, 2006). Additionally, contribution analyses to estimate the percentage of water use occurring in processes required modelled with primary data in the PEF guidelines was carried out.

Uncertainties can be classified into data-, parameter-, and model uncertainty (Madsen et al., 2010). The water footprint inventory assessment in the current paper addressed data uncertainty only.

The study did not intend to shed light on any environmental impacts, but served solely to test the hypothesis that existing standards and databases in conjunction could lead to a systematic underestimation of water use values, and the strategic communication thereof. The intended audience was academics, politicians and professionals commissioning or carrying out EPDs or making standards governing these.

2.1.2. Reference flow

The reference flow '100 g of fresh pork at slaughterhouse exit gate sold for human consumption' was adopted. Moreover, "1 ton dry matter animal feed at the animal farm entry gate" was chosen as reference flow for feed production.

2.1.3. Boundary definitions

The downstream system boundary was set to the slaughterhouse exit gate (Fig. 1). The upstream boundary was set at the cultivation of feed ingredients, in Denmark and abroad. No agricultural processes and inputs apart from irrigation were included. Fig. 1 comprise the entire model system, and all processes were modelled as foreground processes with country average data collected accordingly. The WFI assessment focused solely on the volume of surface- and groundwater withdrawn and released in relation to production of the reference flows. Neither water quality parameters nor temporary depletion was included; hence, water withdrawn and released in the same watershed cancelled each other out.

2.1.4. Inventory approach

To create prediction intervals covering the least and the most water intensive production systems, the approach of a previous study on pork inventories by Basset-Mens and van der Werf (2005), was followed. Minimum values were made the function of favorable conditions, and maximum values the product of unfavorable conditions, while means were the function of all median or mean values (under the assumption of normality). To avoid subsequent citing of uncertain mean values, it was decided that all inventories in the current study should be communicated as 95% prediction intervals. The details of the inventory analysis can be found in sections 2.2.1–2.2.7.

2.1.5. Multi-functionality and allocation

Pork value chains are characterized by multi-functionality. Multi-functionality in LCA occur when one process have multiple outputs, and choices will have to be made about how much of each input should be attributed to each output (ISO14044, 2006). Allocation was done according to economic value, following attributional LCA methodology (EC-JRC-IES, 2010). Economic allocation values for plant processing was derived from Mogensen et al. (2018) and meat processing from the PEF Screening Study (TS-RM, 2015). Agribalyse provided only system processes and the documentation file did not provide the specific allocation keys. EcoInvent had no pork production process and was only used for crops which had no allocation between main and by-products. Agri-footprint included such allocation. The current study did not include multi-functionality of crops and allocated 100% of the water to the grain (see Table 1).

2.2. Inventory analysis

2.2.1. Feed use and feed components

The Danish national average feed composition for the pig rearing subsystems *Year-sows with suckling pigs, Weaned pigs, and Finishers* (DAFC and Vils, 2015) was combined with quartile data on pig feed intake, growth, and mortality obtained from 459 sow farms, 412 weaner farms, and 494 finisher farms in Denmark (Jessen, 2016) (see Table 2 and SI-B). For the feed ingredients potato protein concentrate, oil/fat, amino acids, and minerals were omitted. The total weight of the feed intake per life pig type was calculated by multiplying the feed unit per kilogram growth, the feed weight per feed unit, and the weight increase for each pig type, applying the above-described inventory approach to create intervals.

2.2.2. Feed origin

The origin of each feed ingredient was calculated based on data

Table 1
Allocation values used in the current study and Agri-footprint and the ratio between the two. No allocation was used for the included unit-processes in EcoInvent, and no information about allocation in Agribalyse could be found.

Product	Current study (%)	Agri-footprint (%)	Ratio
Crops:			
Barley grain	100	76.9	1.30
Oat grain	100	76.9	1.30
Rape seeds	100	100	1
Soy beans	100	100	1
Sugar beets	100	100	1
Sunflower seeds	100	100	1
Wheat grain	100	76.9	1.30
Processing:			
Soy-bean meal	56	58.5	0.95
Sugar molasses	5	4.3	1.16
Rapeseed meal	24	23.9	1.01
Sunflower meal	20	20.2	0.99
Pork for human consumption	97.5	97.5	1.00

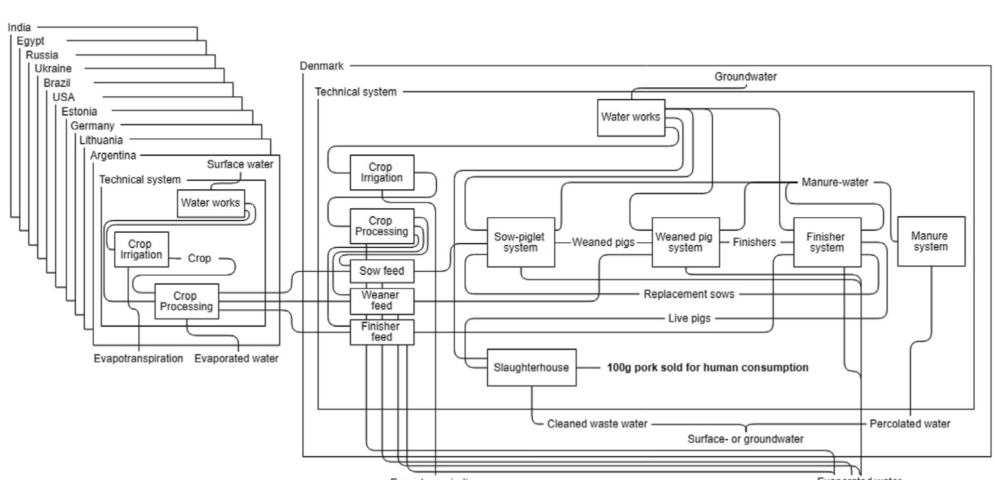

Fig. 1. Overview of compartments of the water use for pork considered (or modelled).

Table 2
Estimated feed intake for Year-sows, Weaners and Finishers, average and highest and lowest quartiles.

	Year-sow	Weaned pig	Finisher
Feed in total, kg, Q25	1512	43.5	216
Feed in total, kg, mean	1513	49.6	229
Feed in total, kg, Q75	1520	56.8	245
Feed composition, %$_{(\text{wet weight})}$			
Wheat	36	45	46
Barley	40	25	25
Soy-bean meal	7	13	10
Sunflower meal	3	–	6
Rapeseed cakes	3	2	6
Wheat bran/oat	5	–	2
Fish meal	–	2	–
Potato concentrate	–	5	–
Oil/fat	1.5	3	1
Molasses from sugar beets	1.2	1	1
Amino acids	0.5	1	1
Minerals	2.8	3	2

from the Danish National Statistics Office (DST, 2016) and FAO (FAO, 2016) and values can be found in the report from Aarhus University (Mogensen et al., 2017) (see Table 3 for import percentages). The method applied identified the countries from which crops were imported, then the import-export mix for these countries were assessed. It is a matrix based approach with a cut-off at third tier importer. Overall, there was a balance between cereal and rapeseed production and their use in Denmark (DST, 2016). Thus, all cereals and rapeseed cakes were assumed to have been produced in Denmark. Soybean meal was estimated to be produced in and imported from Argentina (64%), Brazil (28%) and the USA (8%). Sunflower meals were estimated to have been produced in and imported from Russia (40%), Ukraine (35%), Estonia (14%), Germany (5%) and Lithuania (6%). Sugar beet molasses were estimated to originate from India (36%), Russia (34%), Ukraine (16%), Denmark (10%) and Egypt (4%) (FAO, 2016). Only mean values for the import mix were available and no parameterization was carried out to evaluate the variability induced by shifting import patterns.

2.2.3. Water use in feed production

As data availability and quality on irrigation differ between domestic and imported crops, different methods were applied to assess water use of domestic and imported feed constituents (see SI–C).

2.2.3.1. Water use in domestically cultivated feed. Two approaches were applied to determine the Danish irrigation coverage. The first approach analyzed data from the MarkID database covering all field in Denmark (Jørgensen et al., 2015). Irrigation permits are provided at farm level, and the irrigated area was extracted for cereals on sandy soils using ArcGIS and R-Studio 1.1.383. The assumption of limiting the search to the subset of sandy soils (i.e., with >65%$_{(w)}$ sand particles 0.002–2 mm and 0–10%$_{(w)}$ clay particles <0.002 mm (following the international soil classification)) was made because other soil types in Denmark are primarily irrigated for the cultivation of vegetables. The second approach was based on a dataset including data from 9000 Danish farms, collected between 2007 and 2011, containing the area under irrigation as well as farm type. Cattle farms were excluded as they were assumed to produce feed only for their own herds, resulting in a subset of pig- and plant cultivation farms. The irrigation coverage was estimated to be 11% and 12% for the first and second approaches, respectively, and a mean value of 11.5% was applied. The dataset provided farm level permits, and to translate this into water use, the utilization degree of irrigation permits was calculated. From the Jupiter database on water withdrawal in Denmark (GEUS, 2017), the actual withdrawn water was divided by the permitted withdrawal for each Danish irrigation well and averaged. From 2010 to 2016, 45% of the allowed irrigation water was utilized at the national level. The utilized fraction varies significantly between regions and lower and upper bounds were defined to 30% and 60% respectively. The irrigation coverage and degree of utilization was multiplied with Danish national recommendations for irrigation amounts per crop type obtained from Hvid (2011), where data is based on water deficit calculations by the Danish Meteorological Institute. A sample period from 2001 to 2010 was chosen, as these data are more representative for moisture deficits than earlier periods (Refsgaard et al., 2011). The lowest and highest moisture deficit was applied to define lower and upper bounds, respectively. All irrigated water was assumed to be evapotranspired, as farmers only irrigate when needed due to cost optimization.

The water footprint inventory for irrigation was calculated for two scenarios; 1) All Danish feed crops irrigated according to moisture deficit (FULL), and 2) Best estimate of mean irrigation, as described above. The full irrigation scenario was used to assess the potential upper limit for water withdrawal for crop production, and the mean irrigation scenario was the best estimate of the likely prediction interval for the functional unit for meat and the reference flow for feed.

Table 3
The origin and import mix along with mean, upper and lower bounds for estimated annual irrigation for each crop type used in pig feeds in Denmark.

Crop	Country of origin	Contribution	Irrigation [m^3/ha]			Moisture deficit [m^3/ha]		
		%	Min	Mean	Max	Min	Mean	max
Barley or oats [a)]	Denmark	100	72	549	1449	210	1061	2100
Sugar beets	Denmark	10	104	362	725	100	700	1050
	Egypt	4	8400	9200	10400			
	India	36	0	0	0			
	Russia	34	3	585	2812			
	Ukraine	16	4	140	575			
Rape-seeds	Denmark	100	128	655	1608	370	1270	2330
Soy beans	Argentina	64	0.4	33	322			
	Brazil	28	0.01	2	8			
	USA	8	0.01	299	2457			
Sunflower seeds	Estonia	14	0	0	0			
	Germany	5	0	0	0			
	Lithuania	4	0	0	0			
	Russia	42	0.1	13	133			
	Ukraine	35	0.1	35	148			
Wheat	Denmark	100	159	727	1697	460	1400	2460

[a)] There was no moisture deficit data for oats available, so values for barley were applied instead.

2.2.3.2. Water use in feed cultivated abroad. Surface and groundwater use values (blue water) were derived from Mekonnen and Hoekstra (2011) and calculated WFI_{min}, WFI_{mean} and WFI_{max} values by multiplying the WFI in the county with the lowest, average, and highest surface- and groundwater footprint, with the lowest, average, and best yield, for the period 2010–2014 obtained from FAOSTAT (2016). The WFI of palm oil was omitted, as the surface and groundwater use values in Mekonnen and Hoekstra (2011) were 0 and in Siebert and Döll (2010) <0.5 m^3/ton and the fraction of oil in the feed used for pork production is minimal.

2.2.4. Water use in the stables

Two approaches for assessing the water use in stables for drinking and cleaning were used to assess withdrawal and release factors.

The first approach utilized the water to feed ratio of 3 L of water per 1 kg of feed, with a minimum value of 2:1 and a maximum value of 5:1 (Schiavon and Emmans, 2000). The rates were multiplied by the feed intake calculated earlier, with identical fate modeling. The second approach was based on pig residency times and the average amount of water a pig drinks and excretes per day, according to pig category (Pig time) (see SI-D). The water use in the stables was calculated per year-sow and for the number of weaned piglets and finishers per year sow (see Table 5). The format was chosen for methodological transparency and ease of repeatability as well as to enable data inclusion in subsequent studies.

Data was compiled from eight studies reporting pig drinking water use in a comparable way (Andersen, 1985; Larsson, 1997; Li et al., 2005; Magowan et al., 2007; Phillips et al., 1990; Predicala et al., 2013; Tavares et al., 2014; Torrey et al., 2008), and used this to identify min, mean and max values. Data on pig rearing time and weight, and data on spatial requirements was obtained from Jessen (2016) and SEGES (2016a). Pig physiology data was obtained from Schiavon and Emmans (2000). Several studies of pork production have measured pig water use (e.g., Gutiérrez et al., 2016), but only one study reporting the water use specifically for cleaning of pens (Predicala et al., 2013) was identified and included in the current study. The pigs were assumed moved twice prior to transport to the slaughterhouse (when weaned and when moved into the finisher pens) and cleaning taking place after moving. Equation (1)–4 were used to estimate the water withdrawn and released for each pig type and the parameters used shown in Table 4.

$$Finisher_{water\ use,\ \ min} = a_{min} \times c_{min} \times e_{min} + h \times k_{min} \tag{1}$$

$$Finisher_{water\ intake,\ \ min} = Finisher_{water\ use,\ \ min} \times (1 - q) \tag{2}$$

$$Finisher_{water\ to\ air,\ \ min} = Finisher_{water\ intake,\ \ min} \times p + 0.05 \times h \times k_{min} \tag{3}$$

Table 4
List of parameters used to estimate the water use and the fate in the stable.

ID	Parameter [unit]	n(obs)	Min	Mean	Max	References
a	Finisher per year sow [No. of pigs]	459 [b)]	28.15	30.85	32.85	See SI–B
b	Weaned pigs per year sow [No. of pigs]	459 [b)]	29.20	31.80	33.80	Jessen (2016)
c	Finisher rearing time [days]	480 [b)]	75.50	90.75	106	Jessen (2016)
d	Weaned pigs rearing time [days]	404 [b)]	45.50	49	52.50	Jessen (2016)
e	Finisher water use [L/(pig*day)]	53992 [c)]	5.26	7.44	9.68	(See SI-D)
f	Weaned pigs water use [L/(pig*day)]	1440 [c)]	1.02	2.76	4.79	(See SI-D)
g	Year sow water use [L/(pig*day)]	144 [c)]	9.30	19.25	37.90	(See SI-D)
h	Area required for one finisher [m^2]	–	–	0.65	–	SEGES (2016b)
i	Area required for one weaned pig [m^2]	–	–	0.30	–	SEGES (2016b)
j	Area required for sow-year [m^2] [a)]	–	–	8.12	–	SEGES (2016b)
k	Average cleaning water use [L/m^2]	336 [c)]	22.90	28.90	34.90	Predicala et al. (2013)
l	Finisher weight [kg]	480 [b)]	–	110	–	Jessen (2016)
m	Weaned pigs weight [kg]	404 [b)]	–	30	–	Jessen (2016)
n	Water content in pigs [%]	–	–	70	–	Schiavon and Emmans (2000)
p	Water evaporated [%]	150 [c)]	–	29.53	–	Schiavon and Emmans (2000)
q	Finisher direct water waste [%]	–	–	31	–	(See SI-D)
r	Weaned pigs direct water waste [%]	–	–	31	–	(See SI-D)
s	Sow direct water waste [%]	–	–	40	–	(See SI-D)

[a] Sum of the legal area required for farrowing and gestating sows multiplied with litters per year.
[b] Number of farms.
[c] Number of animals.

Table 5
Estimated drinking and cleaning water used in the stable (mean, minimum and maximum) for each stage of pig rearing and the amount that ended up in manure and was evaporated, respectively, for each of two estimation methods. All values were related to the number of pigs produced per year sow, which are provided in the first column.

Category & number	Estimation method	Water used [L]			Water to manure [L]			Water to air [L]		
		min	mean	max	min	mean	max	min	mean	max
Year-sow	Pig time	3581	7259	14116	2610	5595	11194	613	1259	2471
1	Water:Feed	3025	4539	7601	1875	2859	4864	862	1339	2318
Weaned	Pig time	1467	4311	8510	547	2751	6045	300	883	1745
31.8	Water:Feed	2764	4733	9037	1714	2982	5783	788	1396	2756
29.2–33.8										
Finishers	Pig time	13452	20843	33733	8168	14205	24671	2740	4246	6872
30.8	Water:Feed	13346	21185	37725	8275	13347	24144	3804	6250	11506
28.1–32.9										

$$Finisher_{water\ to\ manure,\ \ min} = Finisher_{water\ use,\ \ min} - Finisher_{water\ to\ air,\ \ min} - a_{min} \times l \times n \quad (4)$$

To estimate the total water use, for each animal category drinking water use was multiplied with the number of pigs produced by one year-sow and with the rearing time (see equation (1), calculating the Finisher min. water use). The cleaning water use per pig was estimated by multiplying water use per square meter with the minimum Danish legal area requirement for each pig category. Evaporated water was estimated by subtracting the percentage of direct wastage of water from the total water use and multiplied with the fraction of water evaporated. Additionally, it was assumed that 5% of the water used for cleaning dissipated as mist through the ventilation system (see equations (2) and (3)) while the rest ended up in the manure. The water that wasn't embedded in the animals (70% of the live weight) or evaporated ended up in the manure (equation (4)). Manure-water was assumed returned to the same watershed through the field application, as irrigation or soil water cover evapotranspiration. Hence, manure-water was only temporally displaced which does not count as consumptive water use.

2.2.5. Water use at slaughterhouse

Data on water use was obtained for six of Danish Crown's (the largest meat processor in Denmark) slaughterhouses in Denmark for the years 2014–15 and 2015–16 (see SI-E). The six slaughterhouses account for approximately 80% of the Danish pork production (DST, 2016). Min, mean and max water use values were calculated for each of the six slaughterhouses. The processing of pork requires with 95% confidence between 0.158 and 0.25 L/100 g surface- and groundwater, with 91%–99% being sent to treatment as wastewater and discharged to surface waters.

2.2.6. Comparison of calculated WFI with WFIs from LCI databases

To test the hypothesis stated in the goal formulation, the prediction intervals of WFI was compared with the WFI prediction intervals in the LCI databases EcoInvent- (Wernet et al., 2016), Agri-footprint- (Blonk, 2015a, b), and Agribalyse (Koch and Salou, 2015).

Agri-footprint contained uncertainty information on 1.1% of the processes related to the production of pork for human consumption (Blonk, 2015b). There was no variability of the slaughterhouse processes, modeled as a foreground system. The animal rearing stage had the weight of the pigs parameterized, while feed manufacturing, the consumption mix, drying, and processing contained no variability. For crop cultivation, the yield and the straw to grain ratio was parameterized. There was no uncertainty information on water-related elementary flows in any of the stages. Agribalyse provided variability for 63 percent of all processes related to the production of pork for human consumption and the coefficient of variations ranged from 2% to 20% for freshwater elementary flows, with all but one being above 10%. The variability and uncertainty in the Agribalyse and Ecoinvent was assessed through expert judgment using a pedigree-matrix approach (see Ciroth et al., 2016 for elaboration on the method), where qualitative parameters such as temporal and spatial representativeness determine the standard deviation of the mean, with the actual variability remaining unknown.

The geographical coverage of the three evaluated databases differed: while EcoInvent and Agribalyse contain no values specific for Denmark, Agri-footprint had WFIs for Danish cereals. Therefore, to compare database values with the WFIs obtained in the current study, unit processes for European countries between the 45th and 60th northern latitude, were merged into joined processes. For example, EcoInvent had no rape seed LCIs for Denmark, and hence rape seed production processes for Germany (2), France (1) and Switzerland (3) were merged into one process, and the prediction interval for the merged process was assessed. For imported crops the import mix was mimicked to the extent that the values were available, such as soy beans in Agri-footprint, but for EcoInvent only soy bean processes for Switzerland were available. Agribalyse provided only one process for wheat and one for sugar beets, and had no LCIs for soy, sunflower, or rape seed, and the barley LCI contained no irrigation water.

2.2.7. Data analysis and calculations

Data preprocessing was done in MS Excel 2016 and R Studio Version 1.1.383.

The calculated minimum, mean and maximum values were manually entered into the LCA modelling software SimaPro 8.5.0.0 and a triangular distribution was applied. Following the standard definition, of the triangular distribution the minimum value was the lower bound of the probability density function, the mean was the mode and the maximum was the upper bound of the probability density function. Monte Carlo simulations were carried out with 1000 iterations to enable bootstrapping of probability density functions and the individual simulation runs exported as a txt-file. The seed-value used in SimaPro's Monte Carlo random number generator was set to 1234, ensuring full repeatability. Statistical analysis of the model runs was done in R Studio (see SI–F for R scripts and SI-G for model runs). The distributions were visually evaluated for normality with histograms and qqplots. Fisher's test and Wilcoxon-Mann-Whitney rank-sum test were carried out to test if the distributions were significantly different from each other.

To enable statistical contribution analysis, the inventory model was designed so that the production stages: *feed cultivation*, *stable* and *slaughtering* were assigned with the respective elementary flows, *irrigation*, *well* and *lake* in SimaPro. The same approach was applied to the database processes from EcoInvent, Agri-footprint and Agribalyse databases where new test unit processes were defined and only surface- and groundwater flows included. An R function was designed that grouped and summed all water flows according to production stages for each model run. The contribution of the WFI for each stage to the overall WFI was calculated as a percentage for each model run. Prediction intervals for both the WFI and the contribution of each production stage were calculated under the assumption of normality using the student's t-distribution. To further evaluate the robustness of the contribution analysis, a pairwise evaluation of the WFI for each stage, was done for each Monte Carlo simulation iteration, resulting in frequencies for $WFI_{slaughter} > WFI_{stable} > WFI_{Feed}$.

The effect of using database WFI values instead of values specific to the Danish context was assessed. EcoInvent had no process for pork meat at slaughterhouse exit gate, and hence the effect size for the reference flow for meat was only estimated for Agri-footprint and Agribalyse. Effect sizes for the reference flows for feed were estimated for all databases. The effect size was calculated as the difference between each production stage for each study for each model run, following the same procedure for grouping as described above. The WFI was subtracted by the WFI of Agri-footprint and Agribalyse. The mean difference as well as the prediction interval of the differences were calculated and reported.

To evaluate the difference in variability induced by using secondary instead of primary data, coefficients of variations were calculated for both reference flows for each of the production stages in the current study and in the three databases.

3. Results

3.1. Water footprint inventory of feed

The WFI attributable to feed cultivation and processing was with

95% confidence between 3.1 and 8.3 L/100 g pork at the slaughterhouse exit gate. The difference between the findings and the WFI in Agri-footprint was 1.8–7.1 L/100 g. The difference from Agribalyse was −1.2–4.4 L/100 g, and hence included zero. The Wilcoxon rank sum test revealed a significant ($p < 0.001$) left shift of the WFI values for both databases. The effect of using database values was much greater when it was assumed that a sampled Danish farm irrigate all feed crops according to moisture deficit. In this scenario, the WFIs would with 95% confidence be underestimated with 33–94 L/100 g and 31–92 L/100 g if Agri-footprint and Agribalyse were used respectively.

The estimated WFIs were significantly higher in the current study than in any of the databases for all crops, with the exception of sunflower seeds, where the prediction intervals overlapped for Agri-footprint, Agribalyse and the current study (Table 6). However, the upper bound of the 95% prediction interval in the current study exceeded the upper bound of Agri-footprint, with 17 m^3/ton. The maximum WFI estimated for wheat cultivation, if farmers irrigated according to the moisture deficit (490m^3/ton), which is the case on many sandy soils in Denmark, exceeded the maximum value in Agri-footprint (5m^3/ton) by more than 90 times (see Table 6). The 95% prediction interval of the difference between the full irrigation scenario and Agri-footprint was 88–454 m^3/ton. This difference was even greater when the results were compared to EcoInvent and Agribalyse.

The WFIs of the crops in the current study were characterized by high variability with coefficients of variation for all crops but sugar beets exceeding 40%. For comparison, the coefficient of variation for WFI for cereals in Agri-footprint was around 5%, in EcoInvent between 7% and 11%. The coefficients of variation in Agribalyse ranged from 0% to 46% (Table 7). The distributions were generally slightly left-skewed, albeit not approaching log-normal. The skewness in the current study was an effect of the choice of triangular distribution, where the mean values was closer to the minimum than the maximum value, creating a tailing effect (see description of the distribution above).

3.2. Stable

During the animal rearing stage, between 0.08 l and 1.06 L of surface- and groundwater was removed from the watershed to produce 100 g of pork for human consumption. Substantially more was temporally removed (~0.6 l–2.3 l) but returned in the form of manure. These prediction intervals were comparable to those of Agri-footprint 1.6–2.1 L/100 g and Agribalyse 0.7–1.6 L/100 g, albeit with a higher spread of values. 0.01–0.02 L/100 g of pork was used for cleaning, of which 95% percent was returned to the same watershed, making the contribution negligible.

3.3. The total water footprint inventory

The best estimate of the prediction interval of the WFI for pork produced in Denmark was, with a 95% confidence, 3.8–9.2 L/100 g (Table 8). The contribution analysis revealed that the feed stage contributed 78–96% or 3.0–8.3 L/100 g to the total water use, the animal rearing stage accounted for 2–16% or 0.1–1.0 L/100 g, and the slaughtering processes contributed 2–6% or 0.25–0.27 L/100 g. One-third of the feed related WFI could be attributed to imported crops and two-thirds to crop cultivation in Denmark. This contribution pattern was similar to the pattern observed in Agribalyse, while Agri-footprint had equal contributions from feed and stable (Table 8). As the probability density functions were not fully normally distributed, the robustness of the findings were evaluated with a pairwise comparison. Despite the large variation of each stage and the apparently large overlaps (Fig. 2), the pairwise analysis showed that only in 2.3% of the model runs did the stable WFI exceed the WFI of imported feed, which in turn only exceeded the WFI of the domestic feed in 6.1% of the model runs. The WFI of the slaughterhouse, while situated inside the probability density functions for the stable only exceeded the water use in stables in 8% of the simulations, underpinning that it was the least significant of the production stages in regards to quantitative water use.

From visual analysis of histograms and qqplots, the feed stages were generally slightly right-skewed, the stables approximated a normal distribution, and the slaughterhouse was left skewed with a wide hump. The slaughterhouse WFI for Agri-footprint was almost perfectly normally distributed. The total water use in all four studies were approximately normally distributed (Fig. 3). The F-test of equal variance further rejected the hypothesis of equal variance with p-values of less than 0.001. Thus, the comparison of the distributions was carried out with the non-parametric Wilcoxon-Mann-Whitney rank-sum test that yielded p-values of less than 0.001 for all comparisons. The pair-wise tests ratios further supported the findings. For feed, the current study exceeded that of Agri-footprint and Agribalyse in 100% and 88.4% of the simulations, respectively. For stables, the results were 0% and 6.1%, and for slaughterhouse they were 0% and 2.7% respectively. The total WFI of the Danish case 3.8–9.2 L/100 g exceeded the WFI in Agri-footprint 3.0–3.9 L/100 g in 99.1% of the cases and Agribalyse 4.4–6.6 L/100 g in 72.6% of the simulations.

It was found that, with 95% confidence, WFIs for Danish pork using Agri-footprint data underestimated the water use in ~99% of the studies with an effect size interval of 0.3–5.7 L/100 g. WFI estimated using Agribalyse underestimated the values in ~73% of studies with an effect size interval of −1.9–3.9 L/100 g. From Fig. 3, the lower variance of the probability density functions in the databases can be visually confirmed. The coefficient of variations increased from 6.4% to 12.3% for Agribalyse and Agri-footprint to

Table 6
Water used for irrigation of crops from LCI databases and the current study expressed as 95% prediction intervals. The number of countries included is shown in parentheses.

	Inventory databases [m^3/ton][a]			Current study [m^3/ton][a]	
	Agri-footprint	EcoInvent 3	Agribalyse	Best estimate	Full irrigation
Barley	0.21–0.23 (1) [b]	0.97–1.4 (1) [c]	0 (2) [c]	4.0–26 (1)	76-380 (1)
Oat	0.13–0.14 (1) [b]	–	–	5.0–31 (1)	93-460 (1)
Rapeseed	0.16–0.18 (1) [b]	2.4–3.4 (6) [c]	2.7–5.4 (2) [c]	9.5–58 (1)	200-900 (1)
Soy-beans	13-29 (3) [b]	1.2–1.7 (2) [c]	–	17-120 (3) [b]	–
Sugar beets	1.7–2.1 (2) [c]	0.07–0.09 (1) [c]	1.4–1.6 (1) [c]	11-32 (5) [b]	12-33 (5)
Sunflower seeds	8.2–34 (5) [b]	1.6–2.3 (1) [c]	17-35 (1) [c]	9.4–51 (5) [b]	–
Wheat[b]	4.6–5.4 (1) [b]	1.1–1.6 (4) [c]	3.0–3.6 (3) [c]	3.7–31 (1)	120-490 (1)
Finisher feed	4.1–4.1 (1) [c]	–	11.5 ([c]1)	10-35 (1)	120-270 (1)

[a] The reference flow for feed.
[b] Danish crops or exact import mix.
[c] Based on available non-Danish database processes for North European countries.

Table 7
Coefficients of variation for the modeled crops. Missing values was due to the crop not being present in the database.

Crop	Agri-footprint [%]	EcoInvent [%]	Agribalyse [%]	Current study Best Est. [%]
Barley	0	8	0	43
Oat	0	–	–	42
Rapeseed	6	11	45	43
Soy-beans	35	7	–	47
Sugar beets	6	0	0	29
Sunflower seeds	7	11	17	41
Wheat	4	8	6	42
Compound feed	0	–	0	21

Table 8
Water Footprint Inventory results for 100 g pork at slaughterhouse and contribution analysis of Agri-footprint, Agribalyse and the current study provided as 95% predictions. The relative contributions of the different life cycle stages are shown in parentheses.

Life cycle stage:	Inventory databases [L/100 g][a)]		Current study [L/100 g][a)]	
	Agri-footprint [b)]	Agribalyse [c)]	Best Estimate	FULL IRRI
Feed	1.0–1.5 (37–42%)	3.1–5.0 (66–81%)	3.0–8.3 (78–96%)	36-97 (96–99%)
Stable	1.6–2.1 (47–51%)	0.7–1.6 (14–28%)	0.1–1.0 (1–17%)	0.1–1.0 (1–3%)
Processing	0.38–0.42 (10–12%)	0.26–0.29 (4–6%)	0.25–0.27 (2–6%)	0.25–0.27 (0–1%)
Total	**3.0–3.9**	**4.4–6.6**	**3.8–9.2**	**36.8–97.8**

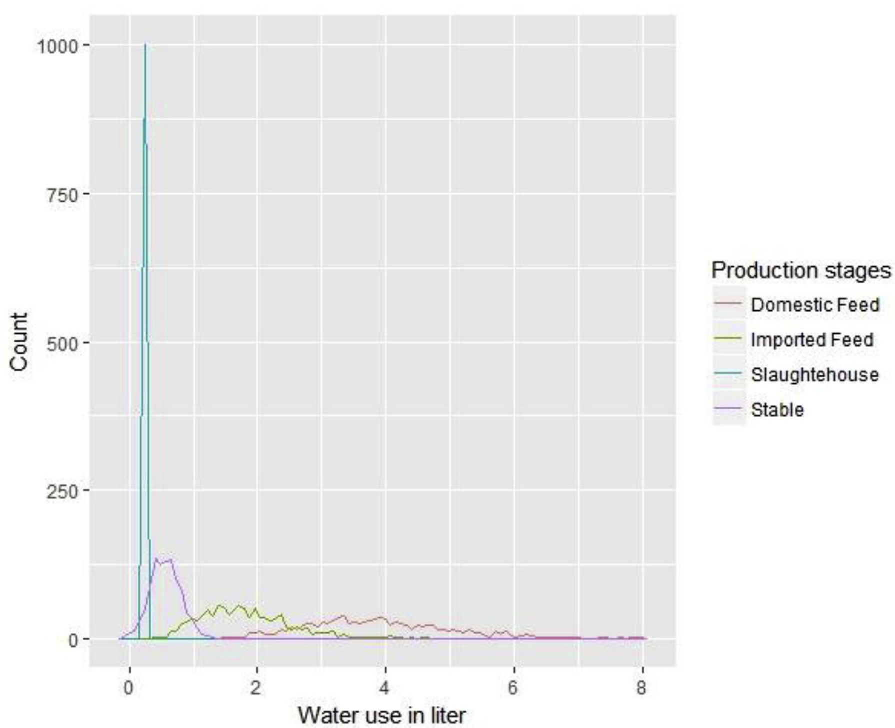

Fig. 2. Probability density function of the Water Footprint Inventory (WFI) in liters per 100 g of pork at the slaughterhouse exit gate.

21% and 23% the best estimate- and the full irrigation scenario respectively.

4. Discussion

The blue water footprint (inventory) of Dutch pig meat reported by The Water Footprint Network (23.6 L/100 g) (Mekonnen and Hoekstra, 2012) was more than two times higher than the 95th percentile in the current study, while the WFI reported in the PEF red meat screening study (1.4 L/100 g) (TS-RM, 2015), is less than half the 5th percentile. There is hardly any overlap of the 95% prediction interval of the Agri-footprint 3.0–3.9 L/100 g and the WFI in the current study 3.8–9.2 L/100 g. The predicted water use in Agribalyse is fully captured within the prediction interval in the current study, but is still significantly left shifted. The difference between the data sources became more pronounced when evaluating the feed constituents, where prediction intervals for five out of eight crops do not overlap (Table 6). The difference in allocation of 30% for the cereals between Agri-footprint and in the current study could not explain the witnessed discrepancies; and hence, the explanation should be found in the use of irrigation data.

The variability found in the current study was significantly

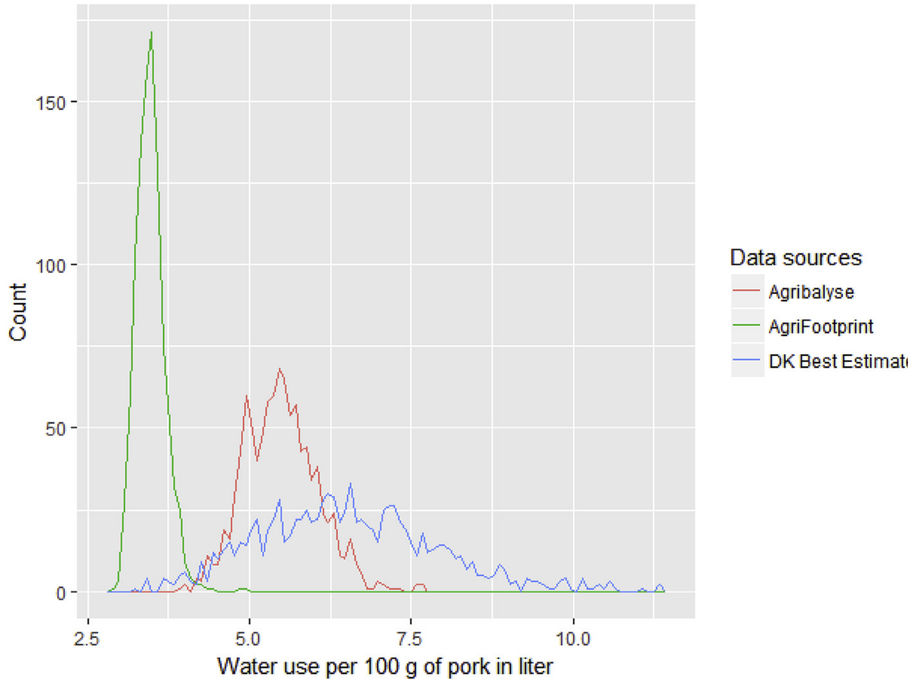

Fig. 3. Probability density functions of total WFI estimates based on the best estimate of Danish pork (current study), Agri-footprint and Agribalyse data, respectively.

higher than in all three LCI databases (Table 7). Since the first modeling in the current study was carried out, the Agri-footprint database has been updated. As a result, the variability and uncertainty of the LCIs decreased, with coefficient of variations consistently below 10%. This was inconsistent with the findings in the current study and appeared counter-intuitive for agricultural production characterized by large variability. Wheat constituted the most used feed ingredient with 144 g of wheat per 100 g of pork. With wheat sourced from Denmark, it could only be said with 95% confidence that somewhere between 0 and ~490 m^3 of water was used for irrigation per ton of wheat (Table 6). The current study clearly illustrated the significance of the choice of data when the WFI for wheat differed between Agri-footprint and the full irrigation scenario with 88–454 m^3/ton at the 95% confidence level.

The downstream system boundary in the PEF category rules for red meat included retail packaging and transportation to consumers (TS-RM, 2016), whereas the stages after the slaughterhouse were omitted in the current study. However, as the values in the current study were both higher and more variable than the studies and databases under comparison, there would be no need to include additional production stages to confirm the hypothesis.

The WFIs provided by Agri-footprint and Agribalyse for drinking and cleaning purposes in stables were consistent with the values in the current study. Water was modelled as released back into the same watershed, hence water use was estimated to be 0.6 ± 0.21 L/100 g of pig. Had the fate modeling been identical to Agri-footprint and Agribalyse, the mean WFI for stables would have been more or less equal, and consequently further supported the hypothesis of the study. The issue of fate modeling was equally relevant for the slaughterhouses, where the current study modeled that 91%–99% of the water was returned to surface water. If the waste-water were discharged to sea or another catchment, the water use in processing would be almost 20 times higher, and the contribution pattern more equal to that of Agri-footprint. This would also have made the difference in assessed water use in the current study and compared the databases greater, which would have further supported the hypothesis. Assessing the effects of parameter and model choice uncertainty, such as fate modeling, would require development of fully parameterized LCI models and model-ensembling of different epistemological model representations respectively, which was beyond the scope of the project. Identifying and including more data on the water used for cleaning of pens would have enriched the picture, but would be unlikely to influence the results.

The contribution analysis in current study revealed that more than 95% of the water use occurred in the crop cultivation and animal rearing stages. However, these stages were considered outside the influence of the slaughterhouses and feed manufacturers in the PEF category rules for red meat (TS-RM, 2016) as well as category rules for feed products (TS-FP, 2018) and can thus be modeled with generic data and results reported as mean values, according to this guideline. While substituting primary for secondary data may not bias comparisons between slaughterhouses, it does mean that pork value chains, performing below average, can model and communicate PEF-compliant EPDs using generic data and achieve lower water footprints than if they compiled and reported specific data.

De-incentivizing primary data provision is an undesirable situation, and a first logical step to counter this could be to create data quality requirements that require modeling the life cycle stages contributing to the majority of the elementary flows with primary data. However, obtaining primary data broadly for all processes might be prohibitively costly. Another and easily implementable approach would be to make assessment and reporting of uncertainty obligatory when carrying out EPDs. This would have multiple derived benefits. When results are communicated as prediction intervals instead of descriptive values, EPDs could be tagged to any product, even when the only available information known is the product category. Products attributed with verified PEF declarations can be compared to non-declared products by saying, ‘The WFI of this piece of Danish pork, with 95% confidence lies within 3.9–9.2 L/100 g. In comparison, *our* product has a verified WFI of 4.1–4.7 L/100 g’. From a legal perspective, this approach would be immune to lawsuits for injuries, as the claims are indisputable, albeit imprecise. From a modeling and statistical perspective, it is crucial to assure that prediction intervals of elementary flows include the true value. From a sustainability perspective, it is of paramount importance to avoid overshooting ecological boundaries. If the variability of the LCI databases were set to reflect the intra-product category variability, it can be argued that, from an economic perspective, attributing products with prediction intervals would incentivize companies and value chains to provide specific data. The reason would be that each additional data point narrows the prediction interval, which improves the trust in the EPD, and thereby increase the consumer's willingness to pay. Ultimately, this could create an intrinsic value of primary data.

The limitations of the study include assessment of how normative choices (such as the choice of perspective in regards to allocation, data assumptions and boundaries and cut-offs at the micro- (processes) and macro level (product systems)) affected the prediction interval. The choice of only using economic allocation, and not modeling the effect of using mass- and biophysical allocation or substitution, limited the understanding of these sources of variability. The use of artificially low minima and high maxima served to capture the full width of the prediction interval of the WFI. Together with modeling all variables as independent, this could have potentially led to too-wide prediction intervals. The universal choice of triangular distribution reduced this effect compared to applying a uniform distribution, which is often the default when the distribution is unknown. Additionally, it could be argued that the true intervals could have been even greater than the prediction interval estimated because the use of pre-averaged data introduced a systematic bias with Monte Carlo simulations, and potentially omitted the tails of the distributions (McAuliffe et al., 2018). Choosing not to parameterize the feed import mix further reduced the variability and could have caused a shift in the contribution analysis. For instance, shifting import of soybean from Argentina to Brazil would have had a significant effect on the surface and groundwater use, as irrigation data in Table 3 illustrated. To fully understand the effect of each input parameter on the result, a parametrized model should be complemented with a sensitivity analysis. In the future, such types of models could inform producers on the key parameters to report in order to narrow the prediction intervals of their EPD.

5. Conclusions

The results of the current study were clear. The assessed WFI for pork and feedstuff reported in the Life Cycle Inventory (LCI) databases, applicable to Danish production, were significantly lower than the modeled WFI in the current study. Furthermore, the coefficient of variations were significantly lower in the LCI databases than in the current study. If the variance and uncertainty of the elementary flows for a product category are not properly accounted for in LCI databases, the use of generic data could provide a false sense of the robustness of the studies. Beyond the scientific shortcoming, this could compromise the validity of the assessment of the environmental sustainability of pork production. If the prediction intervals do not capture the true values and are systematically biased towards lower WFI values, there is a risk of underestimating the value chain water use.

The cultivation of feed ingredients is responsible for 78–99% of the surface- and groundwater use. However, the European Commission's Product Environmental Footprint (PEF) category rules for feed do not require primary data for these production stages. In conjunction with the systematically lower water use in the recommended databases, the water use for some feed crops could be underestimated by as much as 90 times their actual values, and when these results are communicated to consumers it qualifies as greenwashing. If the PEF guidelines were to demand specific data for the feed cultivation stages, the risk of underestimating water could be reduced. Additionally, such regulatory pull for data could increase the value of environmental data.

However, obtaining primary data broadly is prohibitively expensive. As an alternative approach, it is recommended that any Environmental Product Declaration (EPD) initiative, like PEF, include quantification and reporting of uncertainty as an obligatory requirement. Together with revising the LCI databases so that the variability of the inventories reflects the variability in real life, businesses are incentivized to provide data, as each data point would narrow the prediction interval resulting in more accurate EPDs. Ultimately, this is known to enhance consumer trust and willingness to pay.

Conflicts of interest

No conflict of interests has been identified. The main author is on a full and independent scholarship.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2019.06.057.

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