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Publication date:
2019

Document Version
Publisher's PDF, also known as Version of record

Link back to DTU Orbit

Citation (APA):
Vezzaro, L., Arildsen, A. L., Johansen, N. B., Arnbjerg-Nielsen, K., & Mikkelsen, P. S. (2019). *Uncertainty analysis of model-based calculations of wet-weather discharges from point sources*. Technical University of Denmark.

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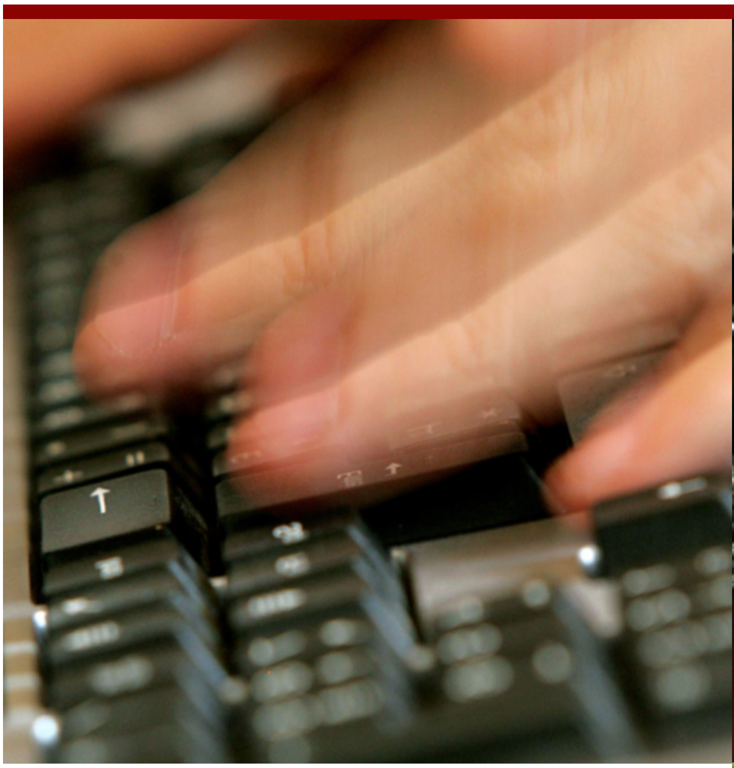
Usikkerhedsanalyse på modelberegninger af regn-betingede udledninger



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February 2019

DTU Environment
Department of Environmental Engineering

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February 2019

Uncertainty analysis of model-based calculations of wet-weather discharges
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February 2019

By
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Cover photo: Colourbox, DTU Environment
Published by: Department of Environmental Engineering, Bygningstorvet, Building 115, 2800 Kgs. Lyngby Denmark

Preface

The Ministry of Environment and Food of Denmark has requested the Technical University of Denmark to provide an assessment of the uncertainty in the modelling results that are used for reporting of yearly pollutant loads from wet-weather discharges (separate storm sewer systems and combined sewer overflows). The focus was on the pollutants that are reported in the *Punktkildedatabase* (*PULS* database): organic matter (expressed as 5-days Biological Oxygen Demand), Nitrogen, and Phosphorous. The analysis investigated the objectivity of the model calculations and the possible sources of uncertainty. Additionally, another objective was the identification of potential improvements in the current modelling methods which (i) can lead to a reduction in the results uncertainty, and (ii) can be applied across all the Danish discharge points.

The uncertainty analysis was carried out in the period from mid-November to December 2018 by using data extracted from the PULS database. The advisory group for the Ministry provided the data that were analyzed in the report, and it was formed by Jóannes Jørgen Gaard, Bo Skovmark, and Anne Gade Holm.

Kongens Lyngby, February 2019

Luca Vezzaro
Associate Professor

Content

Summary

Wet-weather pollutant discharges from point-sources in urban areas (discharges from separate storm sewers and Combined Sewer Overflows) are difficult to measure directly. Therefore, the yearly pollutant loads that are recorded in the national point-source database (PULS – *punktkildedatabase)* are mostly estimated by using mathematical (numerical) models.

All models are affected by sources of uncertainty, which can be characterized in terms of location, level, and nature of uncertainty. This report investigated the various sources of uncertainty in the models that are currently used to estimate the yearly loads of organic matter (expressed as BOD_5), total nitrogen (N) and total phosphorous (P) in Denmark. A great number of these uncertainty sources showed a high natural variability, i.e. they cannot be eliminated from the modelling procedure, but should be considered when analyzing the data in the PULS database. The magnitude of uncertainty could be quantified (in term of precision) based on results from previous studies for some sources. However, site specific conditions, combined with the inherent variability of wet-weather discharges require *ad-hoc* investigation to provide a more detailed evaluation of the model result uncertainty in terms of accuracy.

The PULS data from 2017 were analysed, highlighting the presence of additional sources of variability linked to the procedures for model application. Although the existing guidelines define the main characteristics of the models to be utilized, modellers have several degrees of freedom with regards to model structure, model parametrization, utilized inputs, etc. In order to compare the results of the model-based estimation of pollutant loads across catchments, it is therefore important to harmonize the modelling procedure across municipalities. This would eliminate important sources of variability, which are mostly linked to subjective choices, and it will allow for a general improvement of the data in the PULS database.

A list of different actions that can be taken to reduce the model result uncertainty was proposed. These options include different levels of complexity, depending on the number of available measurements. Model uncertainty is expected to decrease when an increasing amount of site-specific data is used. In ideal conditions, the expected uncertainty for a single event and a single discharge point may vary from above 150-200% (approach based on a simple water balance) to 30-35% (approach based on extensive monitoring of water quality). The expected decrease in uncertainty when looking at annual loads and at the catchment scale should be further investigated.

The current high level of uncertainty has a small importance when looking at accumulating pollutants (N and P) when compared to the overall load from other point and diffuse sources (as wet-weather discharges only represent only 2% (N) and 3% (P) of the total yearly load). However, when looking at short-term effects, the current uncertainty level hampers a detailed assessment of emissions from a single point of discharge. Also the level of uncertainty is higher compared to other discharges points of the integrated storm- and wastewater system (e.g. wastewater treatment plants). The adoption of specific guidelines on model application, aiming at harmonizing the calculation methods across all the Danish municipalities, would thus contribute to reduce such high levels of uncertainty and thereby create the conditions for a reduction of impacts from wet weather discharges.

1. Introduction

Pollution to water bodies originates from diffuse sources (runoff from agricultural areas, mass transport from contaminated sites, etc.) and from point sources. The latter include discharges from various localized human activities (e.g. aquaculture, industries, low density housing) and from urban areas. Cities and other high density human settlements discharge a constant flux of pollutants through outlets from wastewater treatment plants (WWTP – discharging treated wastewater), while additional pollutant fluxes are discharged during rain events through outlets from separated drainage systems (stormwater) and combined sewer overflows (CSO - mix of stormwater and wastewater) that both take place only during a limited number of events per year.

Point sources contribute to about 10% of the yearly discharges of nitrogen and phosphorous to the Danish water environment (Jensen et al., 2018; Thodsen et al., 2018). Wet-weather discharges represent a non-negligible fraction of this contribution (Figure 1). Therefore, Danish municipalities are required to report the loads of different pollutants (organic matter and nutrients) discharged from wet-weather discharges on an annual basis. This allows for a constant monitoring of the efforts aiming at reducing the pollutants discharged to the natural environment and for complying with the requirements of the existing environmental legislation. Data on annual discharges from point sources are recorded in the *Punktkildedatabase* (PULS database), which is managed by the Ministry of Environment and Food of Denmark (MFVM - Miljø- og Fødevareministeriet). The reported data provide the basis for the annual report on point sources (Miljøstyrelsen, 2017,2018).

Continuous pollution sources, such as WWTP outlets, are relatively easy to monitor since they are accessible, and representative samples can easily be collected from the continuous outlet flow. Wet-weather discharges are intermittent discharges, and this increases the logistical challenges associated with collecting representative samples. Furthermore, there are 19,773 wet- weather discharge points spread across the Danish municipalities; almost 5,000 CSO structures and about 14,500 outlets from separate systems, compared to 726 WWTPs with a capacity over 30 Person Equivalents (PE) (data for 2016, (Miljøstyrelsen, 2018)). Continuous monitoring of all the wet-weather outlets would thus require massive resources.

Mathematical models are widely used to estimate pollutant loads from wet-weather discharges, as they represent a valid and cost-effective alternative to continuous monitoring. These models utilize information on the physical characteristics of the urban catchment and data on measured rainfall, and they provide an estimation of discharged water volumes and pollutant loads. Different models (see Johansen and Petersen, 1990) have been applied during the last decades, and the wet-weather data currently reported in the PULS database are all calculated based on models. The guidelines (Miljøstyrelsen, 2012) specify two levels of calculations: Level 1 (areal unit numbers in lookup tables prepared based on standardised model simulations for synthetic catchments) and Level 3 (model simulations for each catchment).

All models are affected by sources of uncertainty, which can to some extent be identified (Warmink et al., 2010; Walker et al., 2003) and quantified. The sources of uncertainty can be located e.g. in the model inputs, the mathematical formulations and the chosen parameters.

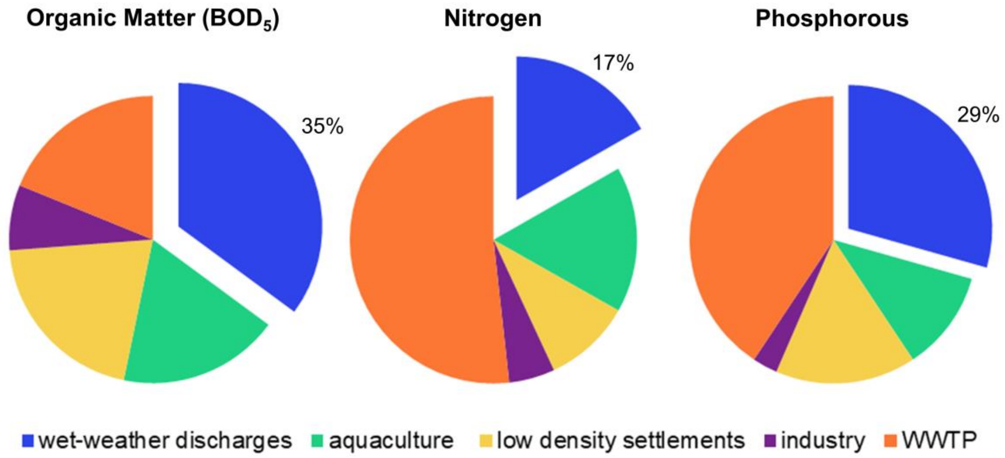


Figure 1. *Contribution of different sources to the overall loads from point sources in 2016 (Miljøstyrelsen, 2018). When looking at total nitrogen and phosphorus, it should be stressed that point sources only represent about 10% of the total yearly discharge (Jensen et al., 2018; Thodsen et al., 2018), i.e. wet-weather discharges represent 2-3% of the total emissions to the Danish water environment.*

Furthermore, subjective choices made by the modellers themselves can affect the final results, and some of these choices may involve elements of human error. Quantification of the uncertainty of model results increases the reliability of these results (Beven, 2009), and this is therefore recommended in any "good modelling practice" guideline or handbook.

This report investigates the uncertainty of the model calculations currently used in Denmark to estimate pollutant loads from wet-weather discharges, which are then reported in the PULS database. Sources of uncertainty are identified based on knowledge from the scientific literature as well as analysis of data recorded in the PULS database.

- *Chapter 2* describes the various indicators that are used to quantify discharges from point sources, and specifically from wet-weather discharges and WWTPs.
- *Chapter 3* focuses on the methodologies that are currently employed in Denmark to quantify the discharges from urban areas, i.e. monitoring and modelling techniques.
- *Chapter 4* introduces the framework that has been applied for identifying and classifying the different sources of uncertainty affecting the model results.
- *Chapter 5* explains the sources of uncertainty affecting the quantification of discharges from urban areas, which are classified based on the framework presented in the chapter 4. When possible, quantitative estimates of uncertainties have been given based on available literature and results.
- *Chapter 6* analyses a sample of the data recorded in the PULS database, investigating the reported water quantity and quality indicators, as well as differences in the different quantification methods
- *Chapter 7* provides an overview of possible actions that can be implemented to reduce the uncertainty of the model results reported in the PULS database.

2. System characterization

2.1 Discharges from the integrated urban storm- and wastewater systems

When describing wet-weather point discharges from urban areas, it is important to distinguish the different elements of the integrated urban storm- and wastewater systems that are considered in this report. These are schematized in Figure 2 below:

- *Combined Sewer Overflows (CSO)*: discharge structures that are activated only during medium-sized and large rain events, i.e. when the capacity of the combined sewer system is exceeded. Therefore, CSO discharges are intermittent discharges, with frequency that is significantly lower than that of rain events (e.g. typically around 5-10 times/yr, against the average 110-130 rainy days per year, (DMI, 2019a)). CSO discharges are a mixture of stormwater and wastewater, and their pollution levels are characterized by a high inter-event and intra-event variability. Further information on CSO pollution levels can be found in Vezzaro et al. (2018a,b).
- *Separate Storm Sewer Outlets*: discharge only stormwater collected by separate systems, i.e. there is a discharge every time a rainfall event occurs. Stormwater also carries a range of pollutants, whose levels depend on the pollutant sources in the drained catchment. Stormwater quality is thus also characterized by high inter-event and intra-event variability (see the overview in Vezzaro et al., 2018b). Whilst stormwater discharges tend to show lower BOD, N and P pollutant concentrations compared to CSO discharges, the concentrations and total pollutant loads for some other pollutants may be comparable to those from CSO or even larger.
- *Treatment Plant Outlets:* discharge the treated wastewater from WasteWater Treatment Plants (WWTP). WWTPs operate all year round, so these are continuous discharges, and their pollution levels are characterized by limited temporal variability. Variations in flow and quality can be caused by wet-weather events in case the WWTP treats discharges from a combined sewer system and by groundwater infiltration-inflow.

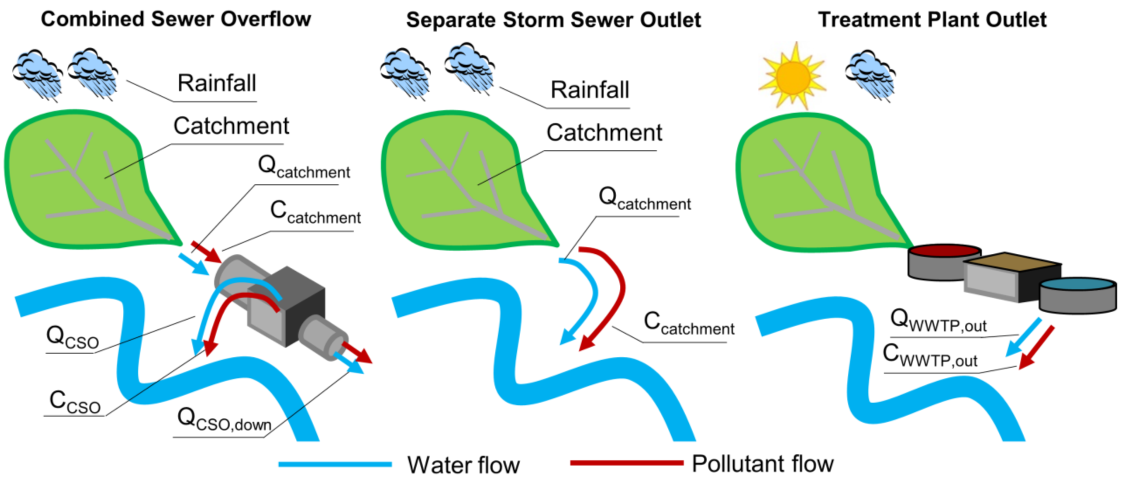


Figure 2. *Schematic representation of the factors that are involved in the estimation of pollutant discharges from point sources (abbreviations are explained in Table 1 below). The arrows refer to water flows and pollutant fluxes related to these water flows.*

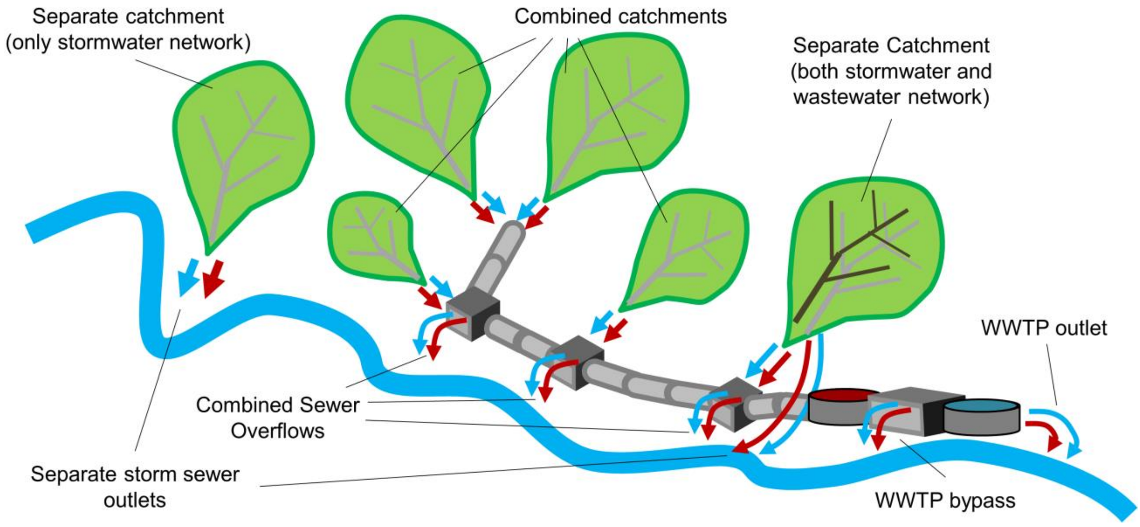


Figure 3. *Schematic representation of an integrated urban storm- and wastewater system, where the elements described in Figure 2 are interacting***.**

The three elements are often interconnected and their discharges can affect the same water body, as exemplified in Figure 3. For example, runoff from an upstream catchment can cause CSOs in downstream parts of the systems, or even bypass of the WWTP. A detailed description of the possible interactions between the various elements of an integrated system can be found in Vezzaro et al. (2018a,b).

Depending on the pollutant of interest, it is relevant to address the individual discharge points or look at the overall discharge from the integrated storm- and wastewater system. For example, for short-term acute effects (such as oxygen depletion), it is relevant to look at individual discharge points and their pollutant load during a rain event. For long term effects (such as eutrophication), it is relevant to look at the total mass of accumulating pollutants discharged to the water body over e.g. a year.

The quantification of the pollutant load *M* for each pollutant discharged from these discharge points is obtained by simply multiplying information on the water quantity (*Q* – typically expressed in m^3/s or m^3/d) and water quality (*C* – typically expressed in mg/l):

$$\mathrm{M} = \int Q(t) \cdot C(t) dt \tag{1}$$

Eq. 1 can be rewritten in a discrete form for intermittent discharges:

$$\mathrm{M} = \sum_{i=1}^{N_{events}} V_i \cdot EMC_i \tag{2}$$

where N_{events} [-] is the number of discharge events, and EMC_i [mg/l] is the Event Mean Concentration, which is calculated for each *i-th* event as:

$$EMC_i = \int \frac{Q(t) \cdot C(t)}{Q(t)} dt \tag{3}$$

While measuring the flow from WWTPs outlets is relatively straightforward, the direct measurement of Q_{CSO} and separate storm sewer is more complicated and it is often further hampered by the physical configurations of the CSO structures (i.e. measuring the flow after it has passed the overflow weir is a complex operation). Therefore, Q_{CSO} is usually estimated by utilizing the water level in the CSO structure (directly depending on $Q_{catchment}$) or the difference between the inlet to and the outlet from the CSO structure (i.e., $Q_{catchment}$- $Q_{CSO,down}$).

Table 1 provides an overview of the current status in the estimation of the variables that are needed to quantify pollutant discharges based on common practice at the international level. Flow measurements in both combined and separate systems ($Q_{catchment}$) are relatively well established, with a range of different measurement techniques available. However, only a limited number of water utilities has currently deployed an extensive network of flow measurements in their systems. Therefore, it is not possible to obtain flow measurements for all the regulated discharge points. Model calculations are then widely used to estimate discharge flows and volumes.

Measuring water quality ($C_{catchment}$, C_{CSO}) from intermittent discharges is a challenging task, which requires extensive resources for establishing and maintaining the monitoring equipment. Monitoring the quality of intermittent discharges is thus often limited to research/demonstration projects. Given the difficulties in collecting water quality data from intermittent discharges, developments of effective water quality models have been hampered by a lack of detailed information regarding some the pollution generation/transport processes across catchments (e.g. accumulation, resuspension and transport of particulate pollutants in complex drainage networks). Therefore the application of these models is characterized by high uncertainty, especially for particulate pollutants.

Table 1. Current status for estimation of the variables that are necessary for the quantification of pollutant discharges from point sources. The evaluation is based on current practices at the international level.

Type of point source	Abbreviation	Description	Measured	Modelled
Wet-weather – Combined Sewer Overflow (CSO)	$Q_{catchment}$	Flow at the outlet of the catchment	AP	SP
	$C_{catchment}$	Concentration of a pollutant at the outlet of the catchment	RE	RE
	Q_{CSO}	Flow discharge from the overflow weir	AP	SP
	C_{CSO}	Concentration of a pollutant in the overflow from weir	RE	RE
	$Q_{CSO,down}$	Flow downstream the CSO structure	AP	SP
Wet-weather – separate storm sewer Outlet	$Q_{catchment}$	Flow at the outlet of the catchment	AP	SP
	$C_{catchment}$	Concentration of a pollutant at the outlet of the catchment	RE	RE
Outlet from wastewater treatment plant (WWTP)	$Q_{WWTP,out}$	Flow at the outlet of the WWTP	SP	DE
	$C_{WWTP.out}$	Concentration of a pollutant at the outlet of the WWTP	SP	DE/RE

SP: Standard practice: widely applied in everyday operation; AP: Advanced practices: applied in everyday operation in few cases, where advanced methods are used (early adopters); DE: Practices applied for everyday operation in few cases (demonstrated by innovators); RE: Applied on at research/demonstration level

Table 2. Overview of the methodologies used to estimate the loads reported in the PULS database

Type of point source	Variable	Measured	Lookup Tables	Dynamic models
Wet-weather – Combined Sewer Overflow (CSO)	$Q_{catchment}$		(X)	X
	$C_{catchment}$		X	
	Q_{CSO}		(X)	X
	C_{CSO}		X	
Wet-weather – separate storm sewer Outlet	$Q_{catchment}$			X
	$C_{catchment}$		X	
Outlet from wastewater treatment plant (WWTP)	$Q_{WWTP,out}$	X	(X)	
	$C_{WWTP,out}$	X	(X)	

X: Standard Practice; (X) Used when no measurements or model results are available

Given their continuous nature, discharges from WWTP are easier to monitor, both in terms of water quantity ($Q_{WWTP,out}$) and quality ($C_{WWTP,out}$). While flow measurements are often available at high time resolutions (in the order of few minutes), water quality measurements are often collected as flow- or volume-proportional samples (see Section 3.3 and Figure 4 for a description of the different sampling methods), i.e. they are available as discrete values or aggregated as composite samples. WWTPs with a capacity bigger than 1000 PE are required to collect volume proportional samples. Water quality measurements with high-time resolution are available only for selected WWTPs and for selected pollutants (e.g. turbidity measurements, converted into Total Suspended Solids).

2.2 Current status for pollutant load reporting in Denmark

Annually, each municipality must report the total discharged volumes and pollutant loads from all wet-weather discharges in their municipality for selected water quality indicators (5-day Biological Oxygen Demand (BOD_5), Chemical Oxygen Demand (COD), total nitrogen (N), and total phosphorous (P)). These indicators are related to short-term acute effects, such as oxygen depletion (COD, BOD_5), or to long-term accumulative effects, such as eutrophication (N, P). The discharged pollutant loads are reported to the Danish Environmental Protection Agency.

Discharges from point sources can be measured, estimated by using lookup tables, or dynamically modelled. Lookup tables can be both based on data from past measurement campaigns and on results from long-term simulations. Therefore they are used to quantify the discharge from a specific discharge point when: (a) measurements are missing or (b) there are no information and/or resources to model the discharges. An overview of the different methodologies used to quantify the pollutant loads reported in the PULS database is provided in Table 2 above, while a more detailed description of each method is provided in Section 3.

2.2.1 Intermittent discharges

Discharges from combined sewer overflows and separate storm sewer outlets have been reported by using different calculation methods (described in Johansen and Petersen, 1990) since the 1990s. Currently, only two modelling approaches can be applied (Miljøministeriet, 2012): the so-called Level 1 or Level 3 models.

Table 3. List of main information that are recorded in the PULS database for each wet weather discharge point

Type of information	Variable	Unit/Note
Pollutant load indicators (updated on a yearly basis)	Overflow volume	m^3/yr
	BOD_5 load	kg/yr
	COD load	kg/yr
	Total-N load	kg/yr
	Total-P load	kg/yr
	Rainfall	mm/yr
	Rain scaling factor	Compared to the reference value of 650 mm
Catchment description (updated when physical changes of the catchment or of the modelling approach are occurring)	Location of discharge point	
	Type of overflow structure	e.g. overflow without basin or with detention basin
	Basin volume	if included at the overflow structure
	Relative outlet capacity of basin	see equation 8
	Infiltration flow	l/s
	Dry weather flow	l/s
	Total area	Ha
	Reduced area	Ha
	Calculation method	
	Type of lookup table	If a catchment specific table is used (see Section 3.1.1)

A total of 42 fields are available for each discharge point, mostly dealing with location and characterization of the discharge point.

The first approach (defined as Level 1) utilizes areal unit numbers in lookup tables prepared based on standardised model simulations for synthetic catchments, while the more complex approach (defined as Level 3) is based on dynamic model simulations for the actual catchment in question. Referring to Table 2, Level 1 is mostly based on lookup tables, while Level 3 is mostly based on dynamic models. Both Level 1 and Level 3 methods are valid for combined and separate systems. In 2017, 38% of the reported loads were estimated by using Level 1 calculations, while 62% were based on Level 3. Monitoring of wet-weather discharges is regulated by specific guidelines (Naturstyrelsen, 2012a), but only a limited number of discharge points have been monitored over long time periods (see Appendix A).

In 2016, quantities from a total of 4,880 CSO structures and 14,689 separate storm sewer outlets were reported across Danish municipalities (Miljøstyrelsen, 2018). Only discharge points with a linked impervious area of > 1,500 m^2 must be included in the reporting.

2.2.2 Discharges from Wastewater Treatment Plants

The legislation regulating the monitoring of discharges from WWTPs include the EU Wastewater directive (91/271/EEC) and its Danish implementation (BEK nr 1469 af 12/12/2017). While the EU regulation mainly aims at monitoring compliance with discharge limits (i.e. concentrations expressed as Emission Limit Values - ELV), the Danish legislation also enforces a monitoring of discharged loads. Furthermore, discharged pollutant loads (for N,P, BOD_5) are taxed (Skatteministeriet, 2016), creating an economic incentive for the reduction of pollutant outlet concentrations significantly below the allowed ELV.

Collection, preservation, transport and analysis of wastewater samples are described in specific technical guidelines (Miljø- og Fødevareministeriet, 2016). Daily samples are collected by using a volume-proportional approach (see Section 3.3 for further details on sampling terminology) for plants with a capacity above 1,000 Person Equivalent (PE). For smaller plants, time proportional samples are accepted (grab samples are also accepted for small plants with a capacity below 200 PE). The frequency of daily samples depends on the WWTP dimensions. Given that both inlet and outlet flows from WWTPs are relatively continuous throughout the day compared to wet-weather discharges, monitoring of WWTP discharges is relatively simpler than monitoring of wet-weather discharges. Therefore, complex dynamic mathematical models are mostly use for planning and optimizing WWTP operations, but they are currently not employed to quantify pollutant loads.

Depending on the available measurements, the total yearly load discharged from a WWTP is then calculated by using one of the approaches listed in Table 4 below. The WWTP should also report the amount of industrial wastewater discharged to the plant and the fraction of inlet volume due to groundwater infiltration. Both these two quantities are estimated based on empirical methods. For example, infiltration can be estimated based on a graphical analysis of flow data in dry weather, the measured drinking water consumption in the catchment, and the measured rainfall data in the catchment. The existing guidelines also provide a list of standard values and lookup tables (yearly inlet loads, removal efficiency of different treatment technologies) that should be used whenever measurements are not available

Table 4. Overview of the methods used to calculate yearly pollutant loads from WWTP , listed in order of preference (Miljø- og Fødevareministeriet, 2016).

Calculation method	Available data		Note
	Water quantity	Water quality	
Average concentration (weighted average of daily samples) * yearly discharged volume	Continuously measured Volume measured in the day when samples were taken	Values from daily samples	
Average daily load = (avg. volume * avg. conc.) * 365 days	Average daily volume	Values from daily samples	
Average daily load from calculated plant load (in PE)	Not available – standard value of 300 l/PE/d is used	Values from daily samples	The plant load [PE] is back-calculated from the water quality data by using a standard value of 60 $gBOD_5$/PE/day. Depending on the treatment technology, standard pollutant removal efficiency values are used.
Average daily load from declared plant load (in PE)	Not available– standard value of 300 l/PE/d is used	BOD_5 value not available	The plant declared capacity [PE] is used Depending on the treatment technology, standard pollutant removal efficiency values are used
Standard values	Not available – standard value of 110 m^3/yr/PE is used	Not available – standard values are used: 21.9 kg BOD_5/yr/PE 45 kg COD/yr/PE 4.4 kg tot N/yr/PE 1.0 kg tot P/yr/PE[a]	The plant declared capacity [PE] is used Depending on the treatment technology, standard pollutant removal efficiency values are used

[a] *According to* Arildsen and Vezzaro (2019)*, the total P load has decreased by about 34% since 2007, i.e. this value is expected to be updated.*

3. Current methods for quantification of urban point discharges in Denmark

The following section provides a detailed description of the quantification methods that are applied in Denmark according to the existing guidelines (Miljø- og Fødevareministeriet, 2016; Miljøministeriet, 2012).

3.1 Model-based estimation of wet-weather discharges

3.1.1 Level 1

The Level 1 calculations are based on simple water balances and lookup tables (Miljøministeriet, 2012; Johansen and Petersen, 1990). This approach is based on generalizations and assumptions aiming at providing a method that can be used across the country. The flow from the catchment is calculated as:

$$Q_{catchment} = A_{impervious} \cdot \; Rain_{effective} + Q_{DWF} \tag{4}$$

where $A_{impervious}$ [m^2] is the impervious (or reduced) area of the catchment, $Rain_{effective}$ [μm/s] is the rainfall that contributes to generate runoff, and Q_{DWF} [l/s] is the dry weather flow (which includes both wastewater production and infiltration). The effective rainfall $Rain_{effective}$ is calculated by account for an initial loss, set to 0.6 mm, and a hydrological reduction factor, set to 0.8.

The dry weather flow Q_{DWF} is calculated based on the expected population and its wastewater production:

$$Q_{DWF} = A_{total} \cdot PopDensity \cdot Q_{PE,day} \tag{5}$$

Where *PopDensity* [PE/ha] is the typical population density in an urbanized catchment (set to 40 PE/ha), and $Q_{PE,day}$.[l/PE/day] is the daily wastewater production, and A_{total} [ha] is the total area of the catchment, calculated as:

$$A_{impervious} = \phi \cdot A_{total} \tag{6}$$

Where ϕ [-] is the runoff coefficient of the catchment, expressing the fraction of the catchment area contributing to the runoff in the sewer system. The default value for $Q_{PE,day}$ is set to 250 l/PE/day, and it also includes groundwater infiltration.

For separate systems, equation 4 is simplified, since there is no contribution from dry weather flow. The volume calculation is therefore straightforward, since it is sufficient to multiply the effective annual rainfall ($Rain_{effective}$) by the impervious area ($A_{impervious}$).

The Level 1 lookup table consider the presence of basins in the system. Therefore, two relative area-specific indicators are used (Winther et al., 2011): the relative storage capacity ($h_{storage}$ – defined in the guidelines as basin volume - *bassinvolumen*) and the relative outlet capacity (*a* – *afløbstal* in Danish*):*

$$h_{storage} = \frac{V_{storage}}{A_{impervious}} \quad (7)$$

$$a = \frac{(Q_{CSO,out,max} - Q_{DWF})}{A_{impervious}} \quad (8)$$

where $V_{storage}$ [m^3] is the storage volume available at the CSO structure, and $Q_{CSO,down,max}$ [l/s] is the maximum flow that is discharged downstream the CSO structure (i.e. the threshold for which overflow would occur once $h_{storage}$ is esceeded). The lookup tables then provide areal unit numbers prepared based on standardised model simulations for synthetic catchments. Based on these results, it is possible to utilize the values of $h_{storage}$ and *a* to estimate the relative yearly CSO volume (expressed as $m^3/ha_{reduced}/yr$ - Table 5).

The pollutant loads are then calculated by multiplying the estimated volume by standard concentration values ($C_{catchment}$ - Table 6). These are defined both for stormwater runoff from separated systems, wastewater, CSO water (water flowing above the CSO weir), and "combined stormwater". The latter (*overvand* in Danish) represents the pollution in CSO water when the contribution from wastewater is subtracted, i.e. is a fictive value which represents the pollution fraction from runoff and resuspension of sediment in the drainage network. This concept was first mentioned in the literature from the 1980s, and only proposed again by Metadier and Bertrand-Krajewski (2011a), who focused on the estimation of the dry weather contribution to the overall CSO pollutant load. The standard concentrations have not significantly changed in the last decades, as the existing guidelines still refer to the values listed in Johansen and Petersen (1990). Nevertheless, these values can be updated when new information and measurements become available (see for example Appendix A or Vezzaro et al. (2018b,a)).

Table 5. Example of lookup table used for quantification of CSO volumes (expressed as [$m^3/ha_{reduced}/yr$]) from detention basins with different outlet capacity and relative storage capacity (Miljøministeriet, 2012).

Relative outlet capacity (*a*)	Relative storage capacity ($h_{storage}$)			
	0 mm	**2 mm**	**10 mm**	**25 mm**
0.1 µm/s	3730	2080	530	120
0.3 µm/s	2410	1050	220	50
1.0 µm/s	910	310	100	20
2.0 µm/s	480	210	70	10

Table 6. Example of average concentration values (expressed as mg/l) used for Level 1 calculations (Miljøministeriet, 2012).

	Combined stormwater (average pollution level)	**Wastewater**	**CSO water (average pollution level)**	**Runoff from separate systems**
BOD_5	25	160	30	6
COD	160	320	180	50
N	10	43	12	2
P	2.5	13	2.9	0.5

Table 7. Example of lookup table used for quantification of pollutant loads from separate systems (Miljøministeriet, 2012).

Parameter	Reference value
Yearly rainfall (total)	650 mm
Yearly rainfall (effective)	485 mm
Yearly runoff volume	4850 $m^3/ha_{reduced}/yr$
BOD_5	30.3 $kg/ha_{reduced}/yr$
COD	243 $kg/ha_{reduced}/yr$
N	9.7 $kg/ha_{reduced}/yr$
P	2.4 $kg/ha_{reduced}/yr$

Pollutant relative loads are calculated based on the average concentrations listed in Table 6.

By combining the lookup tables with the standard concentration values it is possible to calculate areal discharges for BOD_5, COD, total nitrogen and total phosphorous (expressed as $kg/ha_{reduced}/yr$ – e.g. Table 7). All these values are then multiplied by the impervious area ($A_{impervious}$) to obtain the total yearly discharges. All the values in the lookup tables are calculated by using a rain series with a total yearly volume of 650 mm. The yearly discharges are then calculated by scaling the reference values by the total rainfall for the specific year (e.g. for a year 900 mm, a scaling factor of 1.38 should be applied to the reference values).

Theoretically, Level 1 calculations can be used for single-catchments discharge points. However, the existing guidelines (Miljøministeriet, 2012) suggest its use only for calculating the overall discharge from several outlets discharging to the same water body, i.e. the guidelines discourage the use of Level 1 calculations for single CSO structures.

3.1.2 Level 3

Level 3 models are dynamic rainfall-runoff models, exemplified in eq. 9: rainfall time series (*Rain(t)*) are used as input to a dynamic numerical model *f(),* with catchment specific parameters $\theta_{catchment}$, giving as output the outflow at the discharge point.

$$Q_{catchment}(\mathrm{t}) = f(Rain(t), \theta_{catchment}) \quad (9)$$

The level of complexity of Level 3 models can vary (see Obropta and Kardos, 2007; Zoppou, 2001), and two main model categories can be identified:

- *Conceptual hydrological models*
- *Detailed hydrodynamic models*

The model structure f*()* can be subdivided into four main submodels:

- *Rainfall submodel*: the input to the model can be measured or extrapolated if no rain gauges are present in the catchment. Spatial variability of rainfall can be included in the model calculations by utilizing data from different rain gauges or radar rainfall measurements.

- *Rainfall-runoff submodel*: converts the rainfall fallen on the catchment into runoff. This can be affected by processes that can be constant throughout the rain event or that start/end during the rain (e.g. wetting losses, saturation of soil infiltration capacity, runoff contribution from pervious areas).
- *Runoff routing submodel:* simulates the transport of the runoff across the drainage network and structures (overflows, basins).
- *Pollution generation and transport submodel:* this can be further subdivided in other submodels considering the pollutants sources, the pollutant release processes, and their transport across the network.

Table 8 provides a schematic comparison of the two main model categories. The main difference are noticed for the *Runoff routing* submodel, while similar formulations can be adopted for both the rainfall and the rainfall-runoff submodules. Due to the uncertainties in modelling water quality, linked to the lack of measurements (needed to calibrate model parameters) and of mathematical formulations capable to fully represent the observed processes, the *pollution generation and transport* submodels often adopt simple formulations. Dynamic modelling of sewer water quality is in fact a currently active research area. Therefore, the standard practice relies on the concentration values from the lookup tables used for Level 1 (standard concentration values) or on simple approaches (e.g. simple pollutant advection, dilution calculations).

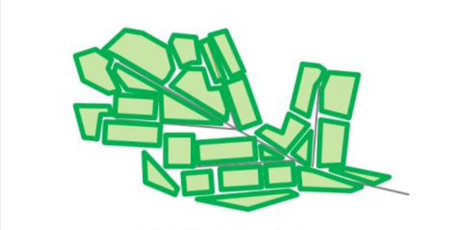
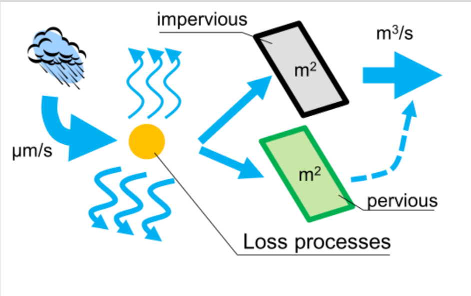
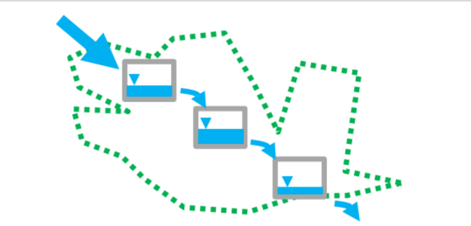
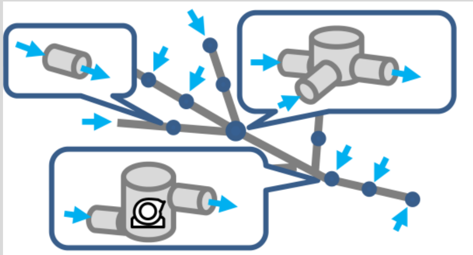
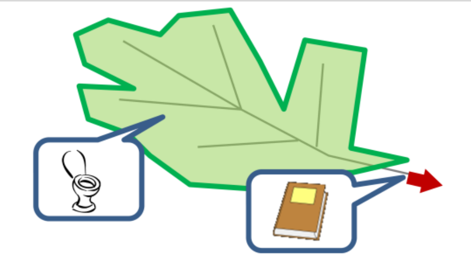
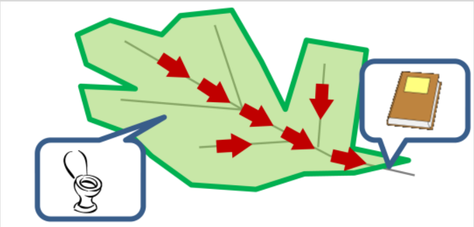
Conceptual hydrological models

These models utilize a simplified mathematical description of runoff routing across the drainage network. These approaches do not consider the spatial variability of the problem, i.e. the flow of runoff across the drainage network, but rather utilize a zero-dimensional approach which only estimates $Q_{catchment}$ at the discharge point. Typical hydrological models include linear and nonlinear reservoir approaches, time-area methods, and the Muskingum routing method.

The entire catchment is lumped into a single parameter, the impervious area. Different formulations are utilized to simulate rainfall-runoff generation processes. The characteristics of the network are expressed through few lumped parameters. For example, the reservoir constant employed by the reservoir approaches affects the shape of the outlet hydrograph, simulating different networks.

Thanks to their simple structure, conceptual models have low computationally requirements. Conversely, their lumped descriptions require a calibration process, i.e. flow measurements at the outlet point are necessary to obtain realistic simulation results. Parameters for models based on the time-area approach can be estimated by an analysis of the upstream drainage (i.e. looking at the length of the system).

Table 8. Schematic comparison of conceptual hydrological modes and detailed hydrodynamic model

	Conceptual hydrological models	**Detailed hydrodynamic models**
Catchment description	 Lumped description	 Detailed description
Rainfall submodel	 Measurements from rain gauge	 Measurements from rain gauge(s) – distributed rain
Rainfall-runoff submodel		
Runoff routing submodel	 Simplified process description (phenomenological)	 Detailed hydraulic description
Pollution generation and transport submodel	 Standard concentration values	 Simple transport (advection) or standard concentrations

A widely successful software package, based on the simple conceptual model described by Johansen et al. (1984), was the SAMBA model. SAMBA calculations provided the basis for the lookup tables listed in e.g. Johansen and Petersen (1990) and Arnbjerg-Nielsen et al. (2000). The software was not updated and was commercially phased out after 2000. Currently, several comparable software solutions are available on the market (e.g. SIMBA# - www.inctrl.ca; WEST - www.mikepoweredbydhi.com), and they are often applied for modelling of integrated urban storm- and wastewater systems (Langeveld et al., 2013) and for optimization of real time control strategies (Schütze et al., 2018; Löwe et al., 2016).

Detailed hydrodynamic models

These models are based on the conservations laws driving the behaviour of water in the urban drainage network: the conservation of volume, momentum or energy. These equations are solved by using numerical schemes, and they allow for estimating the water flow across the whole drainage network. Therefore, it is possible to estimate $Q_{catchment}$ in any node of the network, i.e. not only at the outlet of the catchment. Thanks to these characteristics, detailed hydrodynamic models are also used for detailed design (sizing of the network infrastructure) and for the simulation of pluvial flooding (Hammond et al., 2015).

Detailed hydrodynamic models require a precise description of the physical characteristics of the network (for example, all pipes should be characterized by diameter, slope and roughness – with the latter depending on the pipe material and age). Therefore, these models have been coupled with databases of pipe network and GIS interfaces. The majority of model parameters is seldom calibrated, and standard values are used.

Detailed hydrodynamic models are characterized by high computational requirements, and this hampers their usage in combination with highly computationally demanding techniques. Therefore, a great number of examples of applications in combination with automatic calibration routines or uncertainty analysis methodologies can be found in the scientific literature (Wagner et al., 2019; Del Giudice and Padulano, 2016; Tscheikner-Gratl et al., 2016; Thorndahl et al., 2008), but their application in everyday practice is still limited. Depending on the purpose of the model, different level of simplification and complexity are utilized. For example, models used for long term simulations (e.g. to estimate annual CSO loads), evaluation of Real Time Control strategies, or flooding risk assessment in specific areas of the city, can employ a less detailed description of the catchment and of the network. Therefore, it is common that several hydrodynamic models, characterized by different level of complexity, are available for the same urban catchment area.

There are several software solutions that are available in the market: CANOE (www.canoe-hydro.com), Infoworks ICM (www.innovyze.com), Mike Urban (www.mikepoweredbydhi.com), SWMM (www.epa.gov), etc. Mike Urban is well known internationally and it has gained a dominant position on the Danish market, being *de facto* the standard software for urban drainage modelling in the country.

3.2 Measurements of wet-weather discharges

The procedure for monitoring wet weather discharges is described in specific guidelines (Naturstyrelsen, 2012b,a), which detail all the procedures for collection of samples, definition of dry weather contribution, definition of outliers, etc. Nevertheless, monitoring of wet-weather discharges is limited: Appendix A provides a list of long-term monitoring campaigns recently carried out in Denmark (6 sites). Compared to the total number of discharge points (about 19,000), it is clear that only a minor fraction of wet-weather discharge points are monitored.

The difficulties in monitoring wet-weather discharges and the uncertainty affecting the available data have been widely discussed in the scientific literature (Métadier and Bertrand-Krajewski, 2011b; Bertrand-Krajewski, 2007; Bertrand-Krajewski et al., 2002,2003). The main challenges for these monitoring activities can be summarized as:

- *Stochasticity of wet weather discharges*: sampling depends on rainfall events, which cannot be predicted with high precision and accuracy. Therefore, equipment and personnel should be ready to be deployed with short notice at any time. This issue is exacerbated for CSOs, since the discharge is dependent on the magnitude of the rainfall events and on other factors (storage volume, online controls of the drainage network, etc.), which are difficult to predict in advance.
- *Difficulties in sampling representativeness*: drainage systems are underground structures that are difficult to reach. This creates some challenges for the installation and maintenance of sensors and sampling equipment, which might affect the representativeness of the collected data (e.g. Sandoval and Bertrand-Krajewski, 2016).
- *Maintenance of equipment*: both online sensors and automatic samples require periodic maintenance (down to a weekly frequency), with a consequent high financial burden.
- *The number of discharge points,* which are spread across wide areas.

These difficulties are recognized by the current international legislation, as only in few cases monitoring is explicitly required. For example, in France, continuous monitoring is required for representative CSO structures, defined by the magnitude of their discharged yearly loads (JORF, 2015).

3.3 Discharges from Wastewater Treatment Plants

The monitoring of discharges from WWTP is described in specific guidelines (Naturstyrelsen, 2012c), which details the installation of sampling equipment, location of sampling point, procedure for flow measurements, registration of samples, etc. Compared to wet-weather discharges, monitoring of WWTP outlets is significantly less challenging. In fact, wastewater flow is continuous throughout the year; the effluent is easily accessible; sampling points are located within the perimeter of the plant, where permanent equipment can easily be installed; maintenance can be easily be organized within a plant routine.

Since logistical issues are limited, a specific focus has been made to define the most representative sampling approach in order to estimate pollutant loads. The most common sampling approach is based on discrete samples, where a series of sub-samples are collected and then mixed together, creating a composite sample. WWTP discharges are based on daily samples, which typically are collected over two calendar days. In fact, for logistical reason the sampling is started during working hours, and it is concluded 24 hours later.

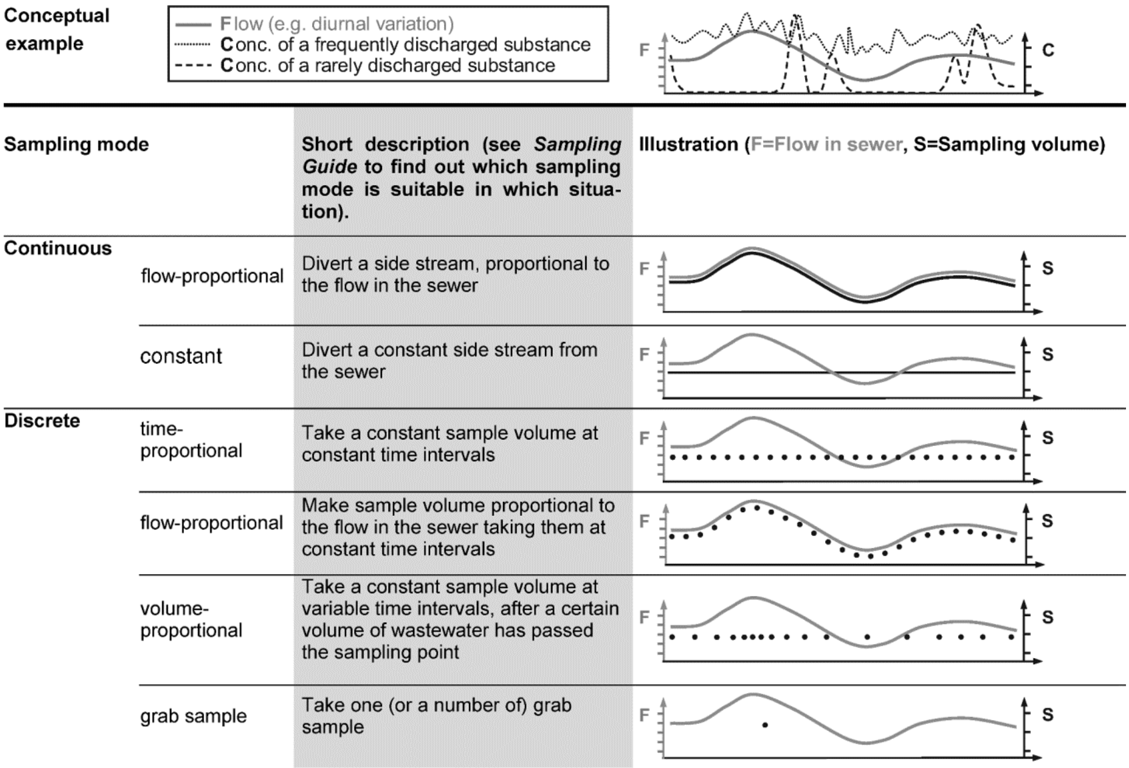


Figure 4. *Comparison of different sampling methods for wastewater monitoring (from Ort et al., 2010b).*

The collection of subsamples can be done in a time- flow- or volume-proportional manner (see Figure 4). Since wastewater generation is a dynamic process characterized daily variations, the collection of subsamples affects the representative of the composite sample. Ort et al. (2010b,a) presented an overview focusing on micropollutants (pharmaceuticals, personal care products, etc.), which are characterized by high temporal variability throughout the day. For the traditional pollutants (BOD_5, N, P), the temporal variability of the emission is limited. Nevertheless, a volume-proportional approach is recommended by the guidelines. It should be noted that in the Danish guidelines for sampling WWTP inlets and outlets (Miljø- og Fødevareministeriet, 2016), the term “flow proportional” is used as synonym for “volume proportional”.

4. Methodology

4.1 Classification of sources of uncertainty

Uncertainty can be defined as "*any departure from the unachievable ideal of complete determinism*" (Walker et al., 2003). There other definitions of uncertainty (see the overview in Warmink et al., 2010), but this definition is used in this report, since it is well suitable to models.

All mathematical environmental models are affected by different sources of uncertainty (Beven, 2009), which influence the different elements of a model (Figure 5). This uncertainty is augmented in the field of urban drainage, due to the logistical and epistemological issues in collecting sufficiently representative data, the inherent variability of the modelled processes, and the lack of detailed information regarding some processes taking place throughout the sewer network (Tscheikner-Gratl et al., 2019; Deletic et al., 2012; Bertrand-Krajewski, 2007). Therefore, the famous quote from Box (1976) "*all models are wrong, but some are useful"* should be the *leitmotiv* when selecting and using models to quantify wet weather discharges from point sources.

The different sources of uncertainty affecting environmental models can be classified according to the three-dimensional framework presented by Walker et al. (2003) and further expanded by Warmink et al. (2010). According to this classification framework, the three dimensions of uncertainty are:

- The *Location* where the uncertainty manifests itself within the various elements of the modelling procedure (Figure 5). In this report, the considered locations include context, input, model technical, model structure and model parameters.
- The *Level* of the uncertainty, expressing the modeller's ability to identify and potentially quantify the uncertainty. The four considered levels included statistical (see also Section 4.2), scenario, qualitative, and recognized ignorance.
- The *Nature* of the uncertainty, defining if the uncertainty is caused by a lack of data and/or understanding, or by the inherent nature of the modelled process. The three natures of uncertainty that were analysed in this report included: variability, ambiguity, and lack of knowledge.

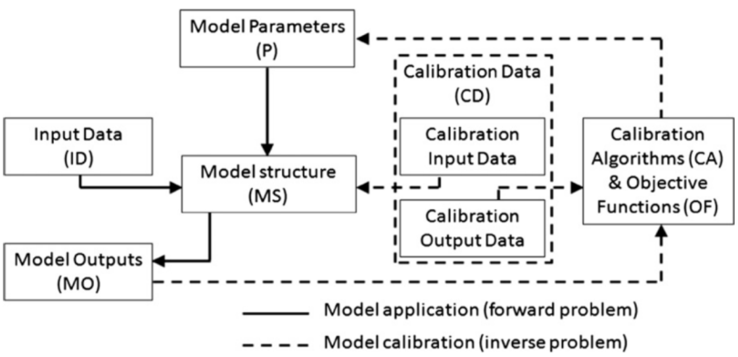


***Figure 5**. General scheme of model elements and potential sources of uncertainty (Deletic et al., 2012).*

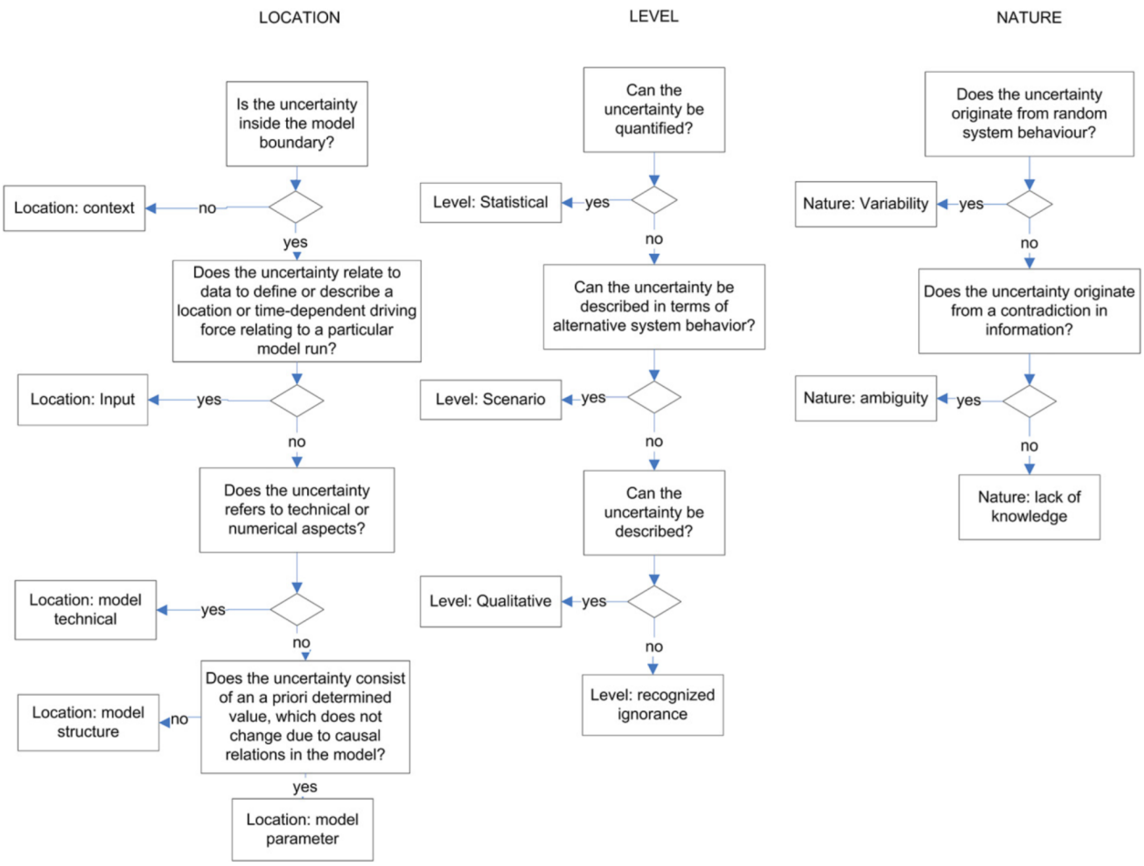


Figure 6. *Decision scheme for classification of uncertainties in environmental models (from Warmink et al., 2010).*

The decision scheme shown in Figure 6 was utilized to classify all the potential sources of uncertainty affecting the model-based quantification of pollutant loads from wet weather discharges (see Section 5).

The classification started from listing the different elements (inputs, parameters, equations) that are utilized for quantification of pollutant loads (see Section 3.1). The international scientific literature, along with results from Danish studies, was then reviewed for each single element, and the results provided the basis for the application of the decision scheme.

4.2 Quantification of statistical uncertainty (level)

According to the definition of level of uncertainty, this can only be quantified for the *statistical* level, i.e. for a level that can be expressed in probabilities or numbers. Uncertainty at the *scenario* level can be described, but not quantified in terms of probabilities.

Uncertainty in model results can be defined in terms of *accuracy* (also defined as *bias*), i.e. deviation of the model estimates from the true value, and *precision* i.e. variability of the model estimates around its mean value (Figure 7).

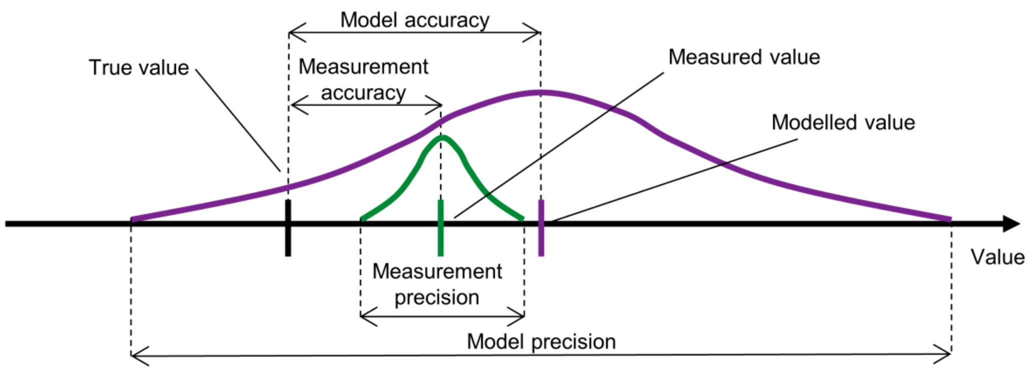


Figure 7. *Schematic representation of statistical uncertainty expressed in terms of accuracy and precision.*

In this report, the quantification of level of uncertainty was based on values reported in the scientific literature. The majority of the reported studies focused on the precision of the model results, while few studies provide results regarding the model accuracy. This is explained by the characteristics of the modelled process: measuring the different processes and parameters that are causing wet-weather discharges is very difficult. Measuring wet-weather discharges is also a cumbersome process (see Section 3.2), i.e. there is a general lack of available measurements for estimating model results accuracy. Therefore, the majority of available studies are based on the so-called *forward uncertainty analysis* (Beven, 2009), where input and parameters uncertainty is propagated through a model using a Monte-Carlo approach.

In studies where measurements were available, statistical uncertainty was conditioned to measurements, i.e. parameters ranges and probability distributions were estimated to match the available measurements. However, these measurements are often collected across the drainage network, in different location than the discharge point. Also they usually refer to hydraulic variables (flow, water levels), whereas there are seldom direct measurements of wet-weather discharges (specifically for CSO discharges). Therefore, most of the studies where uncertainty was estimated conditioned on observations report values in terms of model precision.

4.3 Analysis of existing data in the PULS database

A sample of the data recorded in the PULS (*Punktkildedatabase*) database was analysed in this report. The raw data were extracted from the database by the MFVM and they were subsequently imported and analysed by using the R software package (The R Foundation, 2018). The R software is an open source software package that is widely applied for statistical calculations and for data visualization.

The raw data extracted from the PULS database included

- Discharges reported for all the 98 Danish municipalities for 2017 (see Section 6.1). Data for separate systems were available for 96 municipalities, while data for combined systems were available for 89 municipalities.
- Discharges reported for three selected municipalities for 2016, where a different modelling approach was used as basis for the reporting in this year compared to 2017 (see Section 6.2).

5. Quantification of model uncertainties

5.1 Uncertainties affecting Level 1 model

Table 9 below shows the classification of the sources of uncertainty affecting the results of Level 1 model. Each individual source of uncertainty is address in detail below:

Rainfall

According to the typology outlined in Section 4.1, the location, level, and nature of rainfall uncertainty are *input*, *statistical*, and *variability as well as lack of knowledge*, respectively. There are three factors that influence the input rainfall amounts and the uncertainty of these amounts. Each of them is discussed below, first by discussing each of the processes and how they would influence the results, secondly by discussing if a model of this forcing function could reduce the overall uncertainty of the calculations.

Variation in space: The mean annual rainfall varies in Denmark between 550 and 950 mm. Extreme properties of rainfall also exhibit similar important variations across the country and there is a relatively weak correlation between these two properties of rainfall. Together they give a good description of rainfall dynamics of Denmark. Both variables are used to predict extreme rainfall for design of pipes in sewer systems (Madsen et al., 2017; Spildevandskomitéen, 2014).

Table 9. Uncertainty analysis for Level 1 model calculations

Source of uncertainty	Location	Level	Nature
Rainfall data	Input	Statistical	Variability/ Lack of knowledge
Initial loss	Parameter	Statistical	Variability
Initial loss (used to account for implementation of LAR)	Parameter	Scenario	Lack of knowledge
Hydrological reduction factor	Parameter	Statistical	Variability
Hydrological reduction factor (used to account for implementation of LAR)	Parameter	Scenario	Lack of knowledge
Fraction of impervious areas (ϕ)	Parameter	Statistical	Lack of knowledge
Wastewater production ($Q_{PE,day}$)	Parameter	Statistical	Variability
Groundwater infiltration flow (included in $Q_{PE,day}$)	Parameter	Statistical	Lack of knowledge
Average pollutant concentrations in overflow water (C_{CSO})	Parameter	Statistical	Variability / Ambiguity
Calculations based on hydrological water balance	Model structure	Qualitative	Lack of knowledge
Lookup tables	Model structure	Statistical	Variability

Table 10. Yearly rainfall [mm] for the whole Denmark recorded over the last decade (DMI, 2019b)												
2006	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**
823	866	779	732	726	779	819	669	818	904	701	849	593

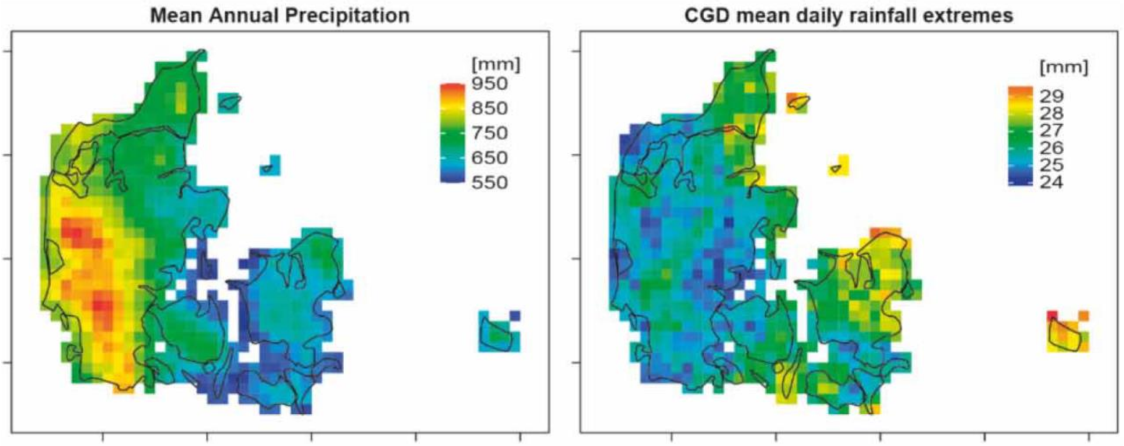


Figure 8*. Spatial variation of rainfall across Denmark: mean annual precipitation (left) and mean daily extremes (right). From Madsen et al. (2017). CDG: Climate Grid Denmark.*

Variation in time (Between years): It is well known that there is a substantial variation of the annual precipitation between years (Table 10): some years are very dry (as 2018) and some years are very wet (as 2017 or 2015). This will influence the discharged volumes significantly, especially when using models based on lookup values. WMO recommends that at least 20 years are used to calculate average statistics of annual properties of e.g. rainfall to account for inter-decadal variation.

Variation in time (observation period): For calculation of discharges from stormwater systems all the generated runoff ends up in the natural surface waters except for the hydrological losses discussed later in this chapter. Calculation of discharged CSO volumes is quite different and from a process based perspective these calculations resemble those performed for design of pipes. This is particularly true for emissions from modern CSO structures, which are typically designed to discharge to the surface waters with a low frequency (few annual discharges). By definition this statistic, which is based on one observation per event where there (almost) is an overflow, is much more uncertain than a value average over a whole year, such as the annual precipitation. Arnbjerg-Nielsen et al. (2002) discourages the use of rainfall series where the observation period is less than four times the length of the statistic to be estimated. However, other studies show that the requirement should be stricter than this. In particular, when comparing the difference in statistics between the large analyses of extreme statistics it is clear that this sampling uncertainty dominates the overall uncertainty of the rainfall and in fact dominates the overall uncertainty of the calculated discharged volumes.

It is crucial to select a representative rainfall series (as e.g. data from Egå or Kolding) and proper measurement period (e.g. 1979 – 2000 or 2000-2015) for calculating CSO volumes. This choice should be based on criteria that are close to the actual property of discharges from the CSO, e.g. concentration time of the catchment, storage volume, etc (Mikkelsen et al., 2005).

In reality, what is needed is an artificial rainfall series that can represent the properties at a given location, just like we have an artificial rain event that can represent single storms for pipe design.

We have the tools to create such a series by combining a number of analyses. What is needed as a minimum is a model that can produce a standardized artificial rainfall time series to be used as input and use it for calculations at Level 3 or to simulate standard tables for use at Level 1. For calculations of storm water emissions the mean annual precipitation would be the most important covariate, while for CSO both variables shown in Figure 8 would be important. These co-variates should then be supplemented with variables from the other variables discussed in this chapter if a new Level 1 is to be constructed.

Initial loss

This parameter reduces the total rainfall volume and thereby the volume of runoff causing the CSO discharges. The initial loss process accounts for losses at the beginning of a rain event due to e.g. wetting of surfaces, storage in local depressions. Therefore, it depends on the catchment conditions, and it might be affected by antecedent conditions. For example, in coupled rain events, the initial loss might be smaller than the default value of 0.6 mm. The value of the initial loss can be estimated by analysis of long time series of rain events. Arnbjerg-Nielsen and Harremöes (1996), for example, estimated an initial loss of 0.5 mm for yearly discharges from catchments with a surface between 2 and 20 ha. However, a larger value (2.0 mm) is suggested for larger events (i.e. above 10 mm, with return period above 0.1-0.2 years). The overview presented in Thorndahl et al. (2006) list literature values ranging from 0.48 to 1 mm.

Initial loss (used to account for effect of the implementation of LAR)

An increasing number of stormwater control measures, often defined in Denmark as LAR (Lokal Afledning af Regnvand), is implemented across urban areas (Fletcher et al., 2015). These measures are built to reduce the overloading of the existing drainage systems (and thereby flooding and CSO risk – see e.g. DHI (2017)).

The exact quantification of the effect of LAR on the flow discharged at CSO structures is still an active research topic (Jefferson et al., 2017), but the overall effects can be schematized as in Figure 9. Depending on the main process taking place in the implemented LAR, these can be accounted in Level 1 calculations by acting on the initial loss, the fraction of impervious area, or the hydrological reduction factor (as shown by Bell et al., 2016). It is therefore possible to account for LAR solutions when looking at the annual hydrological balance in an urban catchment (as in the example from Sørup et al., 2016).

However, some of the LAR elements are characterized by dynamic processes (e.g. saturation of green roof and infiltration trenches) and/or different LAR elements are combined within the same catchment. An exact quantification of their effect thus requires the adoption of Level 3 models. These tools are widely applied to assess the impacts of LAR solutions on the performance of the urban drainage systems (Eaton, 2018), even though their ability to realistically simulate LAR elements is not well documented.

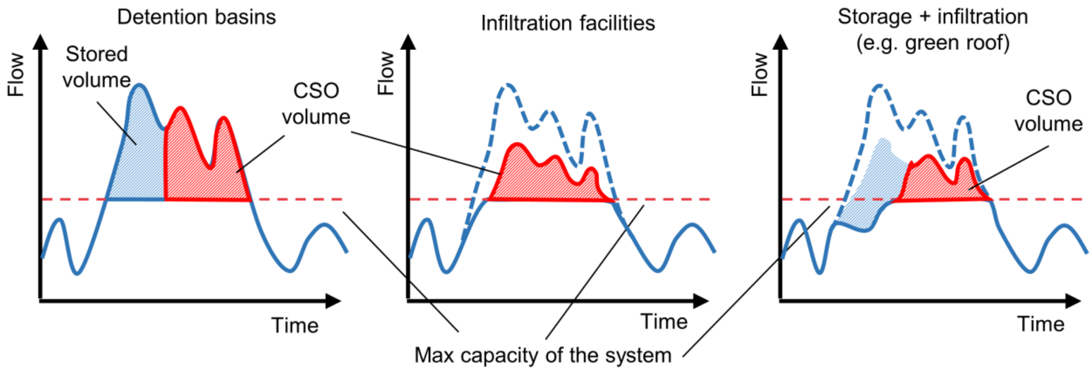


Figure 9. *Conceptual description of the effects of some LAR structures on the hydrograph at a CSO structure.*

Hydrological reduction factor

The hydrological factor is used to account for the uncertaintyof the characterization of the contribution impervious area and of the rainfall input, whose value is measured at a point station and then used to extrapolate at a whole catchment size. The overview presented in Thorndahl et al. (2006) list literature values ranging from 0.7-0.9, while their analysis listed values in the interval 0.42-0.60. Furthermore, the conclusion in Thorndahl et al. (2006) stressed the importance of catchment-specific estimation of the hydrological reduction factors, as this is affected by several local conditions (e.g. size of the catchment, degree of connection of impervious areas, groundwater infiltration flow, etc.).

Fraction of impervious areas

The uncertainty analysis performed by Sriwastava et al. (2018) showed how the fraction of impervious areas (also defined as runoff coefficient ϕ in eq. 6) is one of the major contributors to the uncertainty of CSO volume calculations. This is also confirmed by the overview provided in Fletcher et al. (2013). Redfern et al. (2016) highlighted how the land classification into “impervious” and “pervious” might be misleading when looking at the overall balance, since the different surfaces in urban catchments can react differently (e.g. different infiltration rates might be expected from pervious areas).

The estimation of the fraction of impervious areas is often performed by analysis of GIS information, aerial photography and remote sensing (Jain et al., 2016; Ravagnani et al., 2009), while new automatic methods using other measurement approaches are being developed (Ahm et al., 2013).

Wastewater production

The *per capita* production of wastewater is directly proportional to the consumption of drinking water. According to DANVA (2018), the water consumption in Denmark has fallen by over 40% since 1987, from a value of about 173 l/d/PE down to about 100 l/d/PE. The values used in eq. 5 should therefore account for such reduction.

Groundwater infiltration inflow

The analysis of the contribution of groundwater infiltration to the total dry weather flow (Q_{DWF} in eq. 4) presented in Andersen and Getreuer (2018) shows a high variability across Denmark. For different municipalities, the groundwater contribution to the total volumes treated by WWTP ranges from 10-20% to more than 50%. This underlines how the Q_{DWF} value used by Level 1 calculations needs to be adapted to the local catchment conditions. Furthermore, groundwater infiltration follows the seasonal variations of the groundwater table (e.g. Thorndahl et al., 2016), i.e. it also shows dynamic variations that are not taken into account by the simplified approach employed by Level 1 calculations.

Average pollution concentrations

Measurements from a long term monitoring campaign carried out in Lyon, France, with high resolution monitoring devices (Métadier and Bertrand-Krajewski, 2012) showed a high variability for both Event Mean Concentrations (EMC) and pollutant loads from separate and combined systems (Figure 10). A similar conclusion was obtained by the review of international values for CSO concentrations in Arnbjerg-Nielsen et al. (2000), where measured EMCs ranged from 62 to 1005 mg/l for COD, 1,5-22 mg/l for total N, and from 0,3 to 8,3 mg/l for total P. For separate systems, other values found in literature range from 47-163 mg/l for COD (Göbel et al., 2007), from 0.2 to 14 for total N, and from 0.001 to 4.4 for total P (Brudler et al., 2019). Average values from Danish monitoring campaigns are also within those ranges (see the overview in Appendix A and Vezzaro et al. (2018b)).

Despite several efforts, researchers have failed to build correlation methods that can explain all these wide variations. For separate systems, correlation have been found between total event loads and total event volume (Métadier and Bertrand-Krajewski, 2012), stressing the importance of using a modelling approach capable of simulating the single events (e.g. a Level 3 model). Also, these correlations are highly site specific and strongly depend on the number of available measurements, i.e. they cannot be generalized to other catchments.

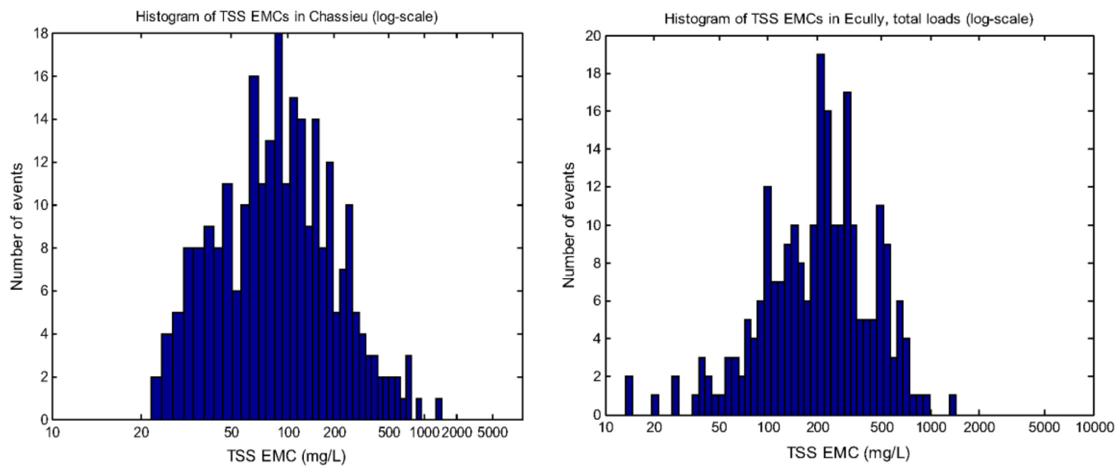


Figure 10. *Distribution of TSS EMCs in discharges from a separate (left) and a combined system (right) in Lyon, France (Métadier and Bertrand-Krajewski, 2012).*

The analysis of available datasets have shown that both EMCs (Figure 10) and discharged loads tend to follow a log-normal distribution (Métadier and Bertrand-Krajewski, 2012; Maestre et al., 2005; Van Buren et al., 1997). However, estimating the distribution parameters with a satisfactory accuracy requires a minimum number of measurements for each catchment/discharge point. Therefore, utilization of average concentrations implicitly involves a high level of uncertainty, that is summed to the inherent uncertainties linked to the high variability of wet-weather discharges (see the discussion in Bertrand-Krajewski, 2007).

Bertrand-Krajewski et al. (2002) presented a theoretical example on how the relative uncertainty in the estimation of the mean site EMC (also defined as Site Mean Concentration - SMC) is dependent on the number of measured samples (Figure 11). For example, assuming a coefficient of variation (COV) of 0.7 (as in Métadier and Bertrand-Krajewski, 2012) and an intensive monitoring campaign of a CSO over 2-3 years (i.e. 20 events), the relative uncertainty (accuracy) is about 30%. For a separate system (assuming a COV >1 based on the results from Metadier and Bertrand-Krajewski (2012)), similar level of uncertainties requires a bigger number of events. Furthermore, quality of stormwater system is strongly affected by the heterogeneous sources in the upstream catchment, i.e. average concentrations are highly site-specific. This is confirmed by the conclusions in Mourad et al. (2005), who showed that it is not possible to define a minimum number of measurements for defining a site-mean concentration for separate systems, due to the high inter-site variability.

As example of the uncertainty in currently available concentration data from selected Danish case studies (see Appendix A) were compared to the chart estimated by Bertrand-Krajewski et al. (2002). Figure 12 shows that the accuracy on the estimated SMC is in the range 30%-40% depending on the analysed pollutant, with important variability between case studies (in some cases the expected uncertainty is below 30%, in others above 60%). This uncertainty (linked to the number of available samples and the inter-event variability) is then evident when looking at the typical concentration values resulting from the different campaigns (Figure 13).

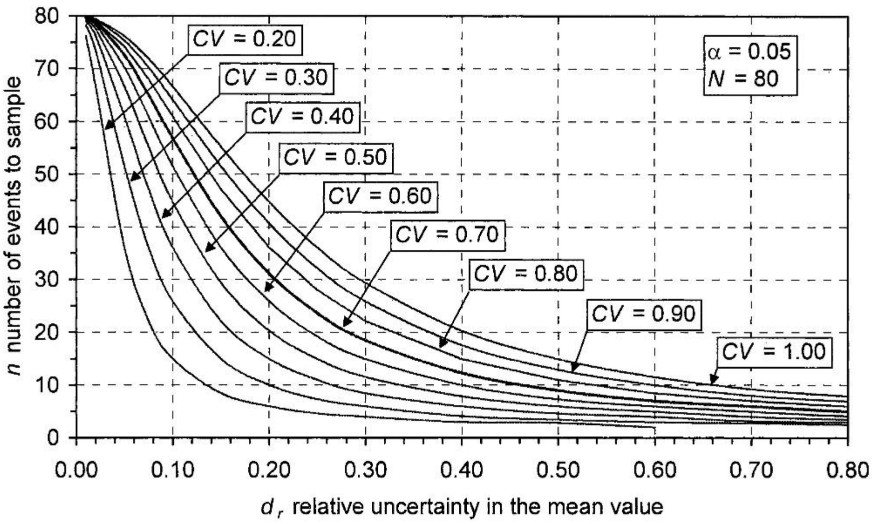


Figure 11. *Number of events n that need to be sampled in order to obtain a realtive uncertainty in the main value d_r (SMC) for different coefficient of variations (CV), as estimated by Bertrand-Krajewski et al. (2002). The reader is redirected to the original publication for further details on the assumptions made to estimate those curves.*

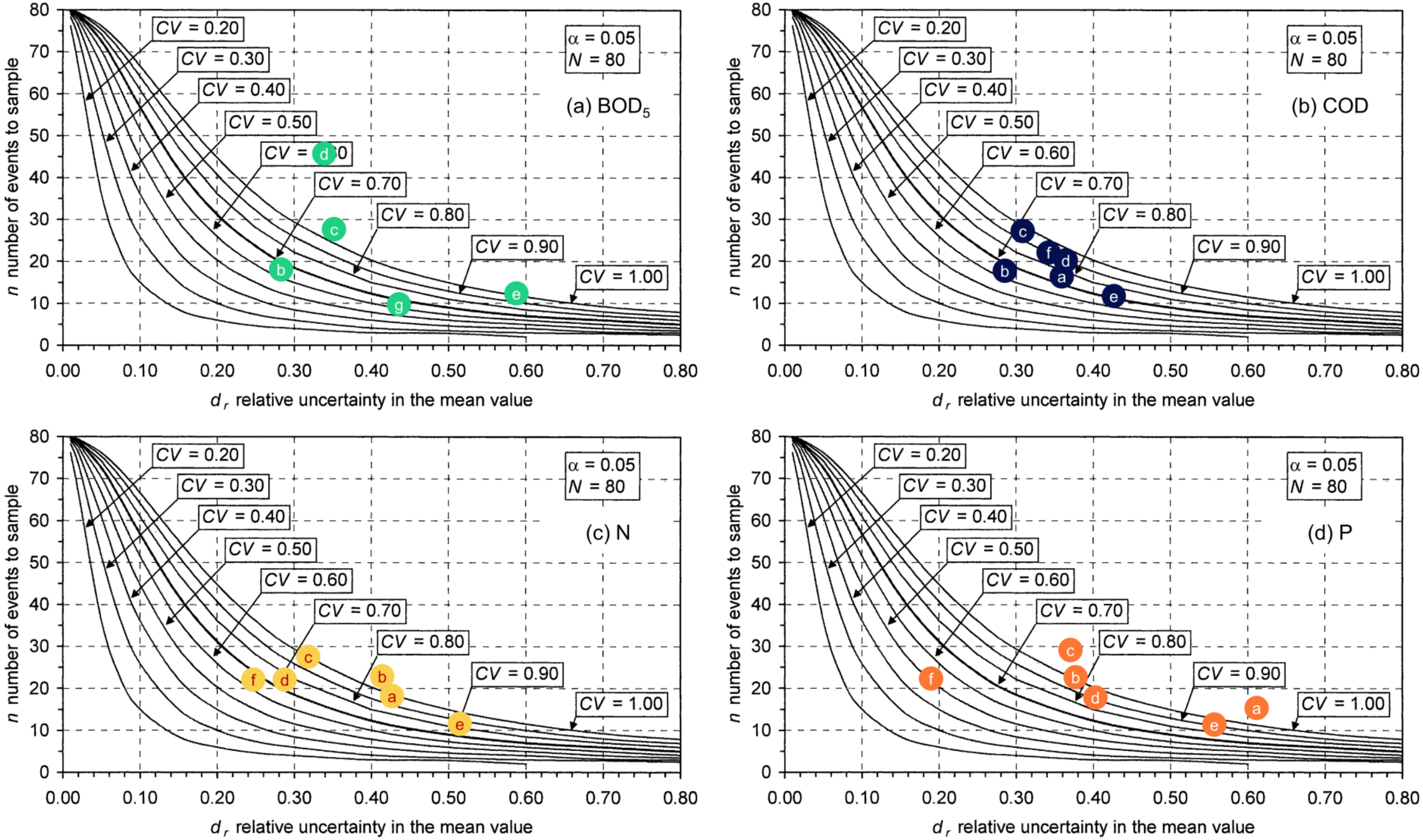


Figure 12. *Exemplification of expected uncertainty for selected Danish case studies (see Appendix A for further details), based on the analysis presented in Bertrand-Krajewski et al. (2002). The reader is redirected to the original publication for further details on the assumptions made to estimate those theoretical curves.*

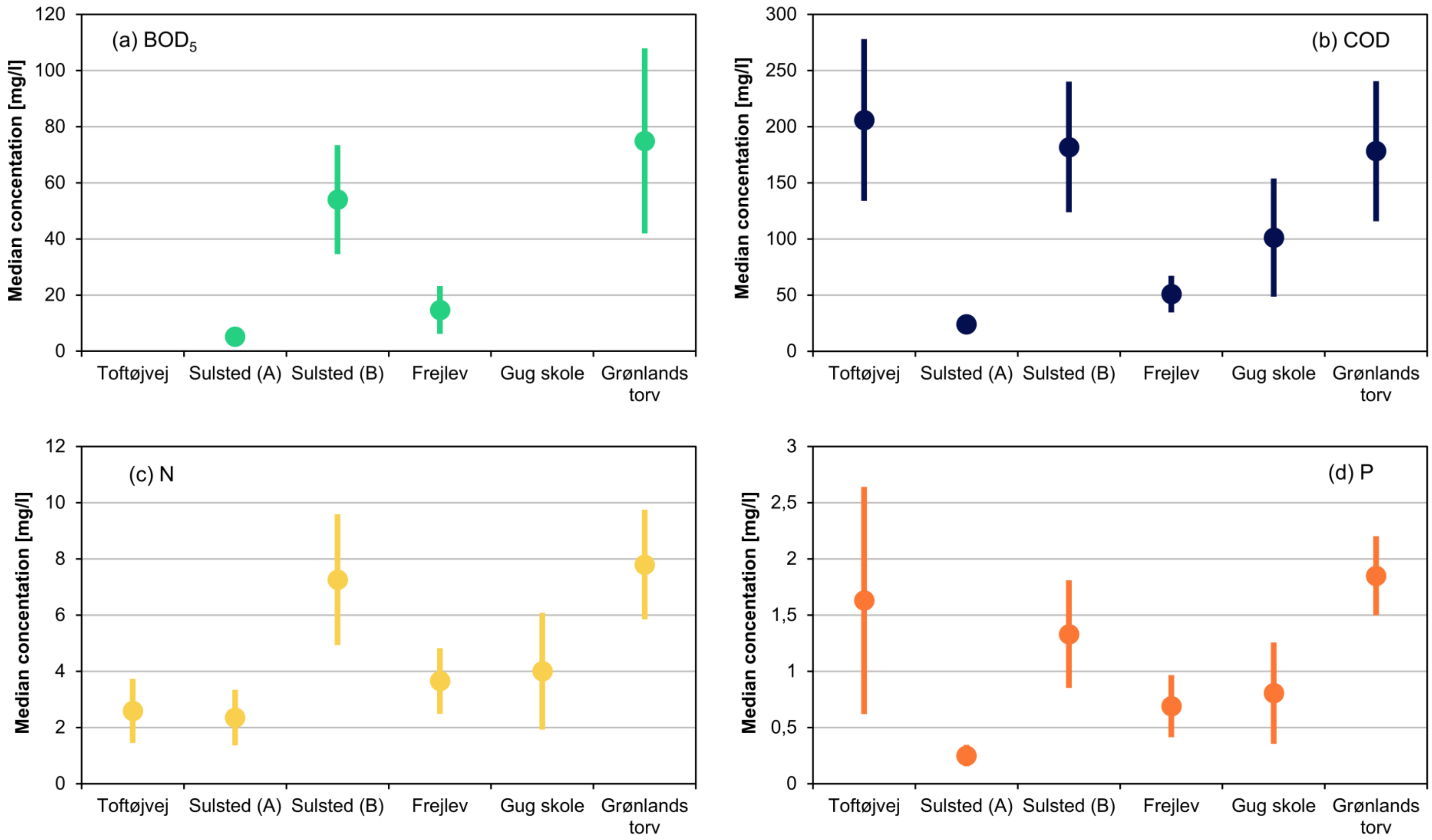


Figure 13. *Example of median concentrations estimated for data from selected Danish case studies (see Appendix A), where the bounds are estimated based on uncertainty levels (accuracy) extrapolated from Figure 12.*

Calculations based on hydrological water balance

Level 1 calculations utilize simple water balance calculations, assuming that the loads from intermittent discharges is linearly proportional to the total rainfall volume fallen on the catchment during one year. For separate systems, this also implies that the initial loss is not affected by seasonal variations or by the magnitude of the single event. These assumption are now questioned by several studies (e.g. Davidsen et al., 2018; Redfern et al., 2016), but it is still not clear how these uncertainties affect the overall volume discharged from separate catchments.

The assumption of linear correlation between total rainfall volume alone and discharges from CSOs is even more questionable. For example, Thorndahl (2009) obtained a correlation between the rainfall volume and the duration of the rain event. These results were obtained for a relatively simple, gravity driven, catchment. CSO volumes in more complex systems would depend on additional factors. For example, basins can be affected by discharges from upstream basins, by pumping stations or by the presence of Real Time Control (RTC). Also, CSO discharges are affected by the initial conditions in the system, i.e., a small rain event might result in a CSO volume in case of coupled events.

Lookup tables

The values defined in the lookup tables used for Level 1 calculations report general values, based on standardised model simulations for synthetic catchments. Therefore, they cannot provide the exact value for a specific discharge point. For example, the discharges listed in Table 5 do not consider dynamic behaviour (e.g. coupled events, contribution of permeable areas in medium-big events) of a specific year, but rather provide an average value. Furthermore, specific characteristics of the catchment, such as the presence of RTC, the presence of upstream basins, pumping stations, etc., cannot be represented by those general values. Also, the lookup tables have a low resolution, and they thus imply an interpolation for all the intermediate values that are not explicitly listed. For example, by assuming a relative discharge capacity of 0.3 µm/s and a relative storage of 5 mm, the CSO discharges can be extrapolated to 750 m^3/red ha/yr or 600 m^3/red (Figure 14) depending on if the interpolation is performed on the actual values (left) or on their log-transformed values (right), respectively.

Lookup tables should be calculated for each catchment, and by using rainfall data collected in the same area. When this information is missing, the lookup tables presented in Miljøministeriet (2012) can be used as backup. However, these are general reference values that add an additional source of uncertainty to the model results. For example, Arnbjerg-Nielsen et al. (2000) present lookup tables obtained by using data from four different data series recorded in Denmark. As shown in Figure 15, the relative CSO volumes differ for the different catchments. More interestingly, these results show how the use of total rainfall as scaling factor for defining CSO volumes across different catchments can lead to over-estimation of the discharged volumes. For example, the relative CSO volumes estimated with a station with a total rainfall of 850 mm (23% bigger than the standard 650 mm) are consistently lower (5-13%) than those listed by the guidelines. Similarly, the relative CSO discharges for a station with a total rainfall volume of 650 mm (i.e. equal to the one defined in the standard) differ in a range from -17% to +27% compared to the default values. Although the values shown in Figure 15 refer to different catchments, it can be assumed that similar behaviour can be observed by looking at different temporal intervals for the same catchment.

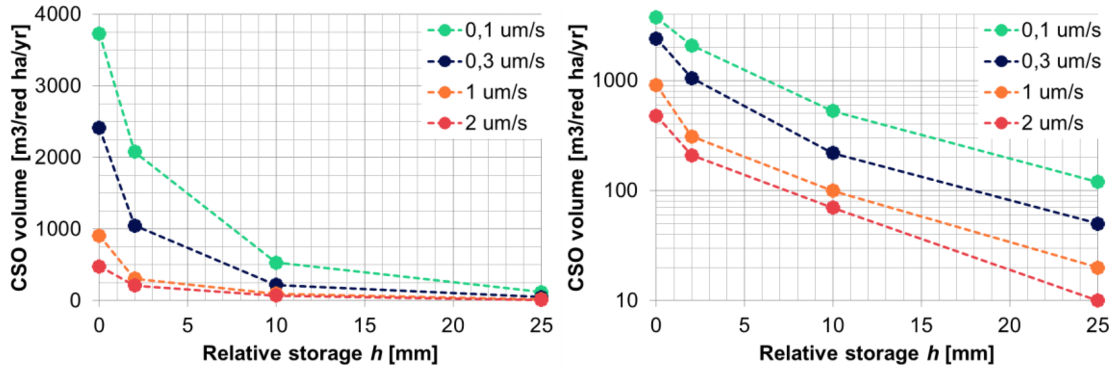


Figure 14. *Left: Relative CSO volumes listed in Table 5 for annual relative CSO discharges (volume as function of reduced area) for different relative storage capacities and relative discharge capacities. Right: same as left, with y-axis in logarithmic scale.*

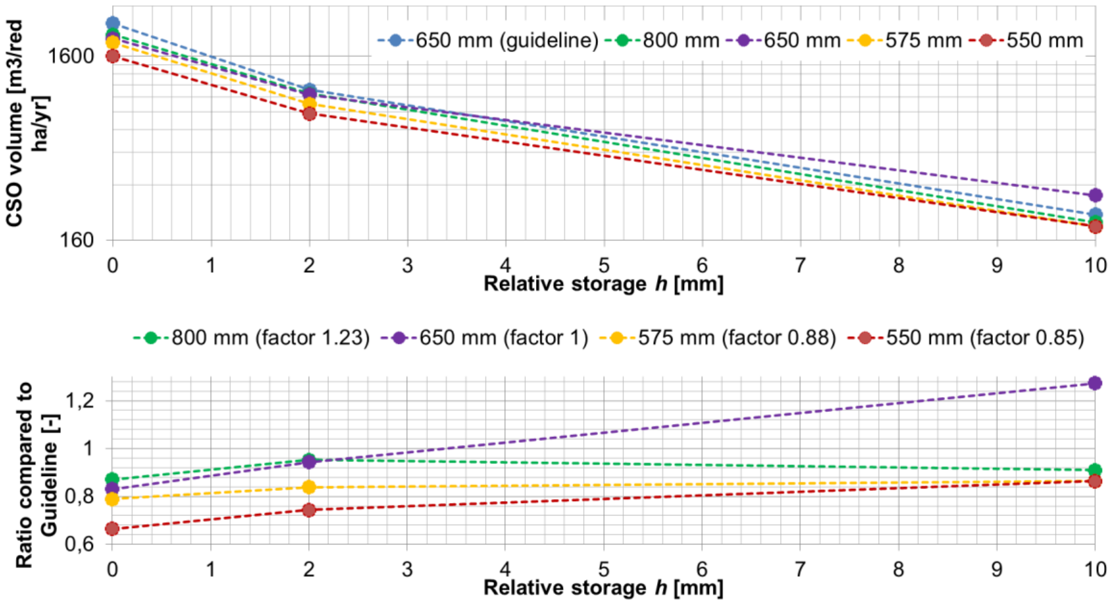


Figure 15. *Comparison between the relative CSO volumes suggested for Level 1 calculations and those estimated by Arnbjerg-Nielsen et al. (2000) using different rain gauges of a specific discharge capacity of 0.3 µm/s. Top: absolute values. Bottom: ratio between values from Arnbjerg-Nielsen et al. (2000) and the guideline values for different total rainfall volumes.*

While using a scaling factor is an acceptable approach for discharges from separate systems, this adds an additional uncertainty when looking at CSOs. In fact, an increase in the total annual rainfall does not necessarily result in a linear increase of CSO volumes. Theoretically, the same amount of yearly rainfall can be generated by several small events (not causing CSO) or few large events (causing CSO). Nevertheless, assuming that the distribution of rainfall events in the time series used to estimate the standard values is still representative, the annual rainfall volume can still be regarded an acceptable proxy when no other rainfall measurements are available.

5.2 Uncertainties affecting Level 3 model

Table 11 below shows the classification of the sources of uncertainty affecting the results of Level 3 model. Each individual source of uncertainty is address in detail below:

Rainfall

As outlined in Arnbjerg-Nielsen and Harremoës (1996b), rainfall input is one of the major sources of uncertainty when looking at model-based estimation of CSO volumes. For example, Schaarup-Jensen et al. (2009) used different rainfall series from different SVK rain gauges (Jørgensen et al., 1998) close to the same urban catchment in Aalborg, and obtained up to 150% differences in CSO volumes for extreme events depending on the different inputs. In a similar study, Müller and Haberlandt (2018) used different rainfall input combinations and obtained variation between 25% and 43% in CSO volumes for a 0.9 and a 4.4 years event respectively.

Initial loss

Thorndahl et al. (2008) performed an uncertainty based calibration of a MOUSE model for a catchment in Aalborg. The results of this calibration, performed by using a likelihood measure that considered several model output (runoff volume, CSO duration, etc.) suggested that the initial loss parameter was an insensitive parameter, i.e. its contribution to the total model output variance was negligible.

Table 11. Uncertainty analysis for Level 3 model calculations

Source of uncertainty	Location	Level	Nature
Rainfall data	Input	Statistical	Variability
Initial loss	Parameter	Statistical	Variability
Initial loss (used to account for implementation of LAR)	Parameter	Scenario	Lack of knowledge
Hydrological reduction factor	Input/parameter	Statistical	Variability
Hydrological reduction factor (used to account for implementation of LAR)	Parameter	Scenario	Lack of knowledge
Fraction of impervious areas (ϕ)	Parameter	Statistical	Lack of knowledge
Wastewater production ($Q_{PE,day}$)	Parameter	Statistical	Variability
Groundwater infiltration flow (included in $Q_{PE,day}$)	Parameter	Statistical	Lack of knowledge
Average pollutant concentrations in overflow water (C_{CSO})	Parameter	Statistical	Variability
Manning coefficient	Parameter	Statistical	Lack of knowledge
Pollution removal rate in storage basins	Parameter	Statistical	Lack of knowledge
Coefficient of overflow weir	Model structure	Scenario	Lack of knowledge
Simplification of drainage network	Model structure	Scenario	Lack of knowledge
Routing equations (conceptual vs-hydrodynamic model)	Model structure	Qualitative	Ambiguity
Numerical dispersion	Model technical	Statistical	Variability
Calibration	Model technical	Scenario	Ambiguity

Hydrological reduction factor

The results presented by Thorndahl et al. (2008) for a detailed hydrodynamic model highlighted the hydrological reduction factor as the most influential parameter. This result is confirmed by the uncertainty analysis presented by Freni et al. (2009) for a conceptual hydrological model. After calibration, Thorndahl et al. (2008) obtained a distribution of hydrological factors ranging between 0.4 and 0.8, while the parameter distribution obtained Freni et al. (2009) ranged between 0.3 and 0.5. This highlights how this factor is strongly related to the catchment characteristics and it needs to be estimated after a model calibration procedure.

Fraction of impervious areas

See the considerations made for Level 1 models

Wastewater production

See the considerations made for Level 1 models

Coefficient of overflow weir

Discharges from overflow structures are typically estimated by using standard Q-h relation curves. Ahm et al. (2016) compared the accuracy of CSO volumes estimated by using a detailed hydrodynamic model (Mike Urban) for a complex CSO structure located in Viby, Aarhus. The estimated error when using the default Q-h curve typically adopted by modellers was around 30%. The accuracy of the estimations based on a specific Q-h curve, estimated by using computational-fluid-dynamics model, was around 5%. This study stressed the importance of ensuring that the Q-h curves are adapted for each specific CSO structure.

Runoff routing

As outlined in Section 3.1, there are several options for rainfall-runoff modelling (Obropta and Kardos, 2007; Zoppou, 2001), and the current guidelines for Level 3 calculations leave the freedom to the modellers to select the most appropriate approach. Conceptual models (such as SAMBA) have been preferred in the past due to their simplicity and reduced computational requirements. The development in computational power, combined with wide application of detailed hydrodynamic models for planning and design of the urban drainage infrastructure, as well as urban flood assessment, has led to a wide application of hydrodynamic models (such as Mike Urban). In some cases, conceptual models are first calibrated against a detailed hydrodynamic model, and then applied to estimate CSO discharges. An example of this approach is presented Langeveld et al. (2013), where conceptual models have been implemented for simulating an integrated system, i.e. to simulate the impacts of the discharged pollutant loads on the receiving water body.

Detailed hydrodynamic models can also differ in the level of complexity: parts of the urban drainage network can be simplified in order to speed up the simulation time. The simplification procedure adds an additional uncertainty to the model simulations. Nevertheless, the results presented by Tscheikner-Gratl et al. (2016) showed that level of model detail did not have important influence on the estimation of the CSO volume, i.e. simple model structures are sufficient for the purpose of quantifying wet weather discharges.

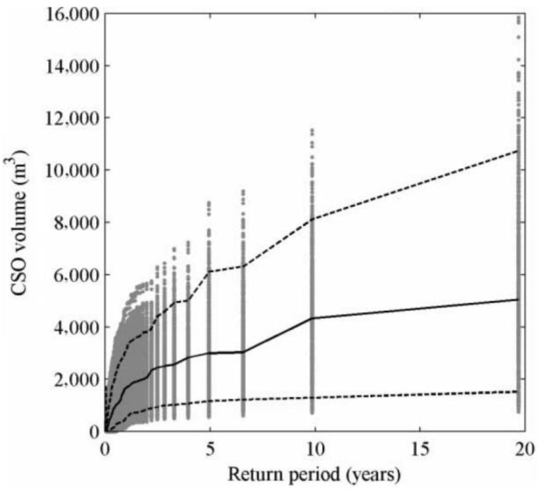


Figure 16. *Example of CSO volumes estimated with a MU model for the Frejlev catchment as function of return period (Thorndahl et al., 2008).*

Sriwastava et al. (2018) performed an uncertainty analysis on a detailed hydrodynamic model (SWMM), and estimated a Coefficient of Variation (COV) for CSO volumes between 0.113 and 0.431. Assuming a normal distribution, this correspond to a range for CSO volumes of ±80-86%. Thorndahl (2009) and Thorndahl (2008) also performed an uncertainty analysis of a detailed hydrodynamic model (Mike Urban) for discharges from a catchment in Frejlev. Based on the results presented in those studies, the uncertainty range for CSO volumes varied around ±50-80% of compared to the median value (Figure 16)

Calibration

As exemplified in Figure 5, the procedure of calibration has an important impact on the uncertainty of model results, irrespective of the model structure. Several studies investigated the effect of single factors on the calibration results, such as the choice of the objective function (Freni et al., 2008), or the influence on rainfall inputs (Kleidorfer et al., 2009). Tscheikner-Gratl et al. (2016) investigated the influence of different calibration scenarios (e.g. number of rainfall inputs, number of calibration events, model structure, etc.) on modelled CSO volumes, resulting in differences up to 152% in CSO volume. In a follow-up study, Vonach et al. (2018) obtained deviations up to 250% in CSO volumes when using different calibration scenarios (number and location of calibration points).

5.3 General considerations

Level 1 is a simple calculation method which uses default values that needs to be adapted for the specific case area. Several factors show annual trends (e.g. wastewater consumption) or seasonal trends (e.g. groundwater infiltration) that, if not correctly taken into account, can lead to an over/underestimation of the CSO loads. Also, Level 1 seems to have challenges in accounting for new elements in the upstream catchment, characterized by a nonlinear effect on the runoff discharges (such as LAR).

Level 3 models ensure a better representation of the dynamic processes behind runoff generation and thereby CSO estimations. Also, they allow for simulation of new elements in the upstream system (such as LAR elements, although with additional uncertainty).

The quality of the used model is paramount to ensure a reduction in the estimation of discharges from separate and combined systems. Uncalibrated or poorly calibrated model can result in variations in the order of 80-100% of estimated CSO volumes. This uncertainty can be reduced by proper calibration procedures (e.g. Tscheikner-Gratl et al., 2016). An important source of uncertainty is represented by the rainfall input, which can add an uncertainty with a similar magnitude. Therefore, it can be assumed that uncertainties in the discharged water volumes can easily exceed 100%.

The majority of the studies found in literature focus on the water volume. However, the uncertainty on the estimation of the water volumes should then be combined with the high variations observed in pollutant concentrations, which can easily exceed a variation of 100% from the average value. Measured event pollutant loads (Métadier and Bertrand-Krajewski, 2012) showed variations around 180% from the mean load. It can be assumed that model uncertainties can easily exceed these values.

6. Analysis of existing data from the PULS database

6.1 Overview of recorded pollutant loads

6.1.1 Rainfall data

As highlighted in Section 5, rainfall input is one of the major sources of uncertainty for quantification of wet-weather discharges. The rainfall values reported in the PULS database were therefore compared against the rainfall data recorded across Denmark. Figure 17 shows the annual rainfall recorded by the Danish Meteorological Institute (DMI) across the country for 2017. Figure 18 shows the corresponding annual rainfall values that were utilized for the calculations of results recorded in the PULS database for combined systems. Figure 17 shows the rainfall pattern that is characteristics for Denmark, with a decrease in annual rainfall from the western part of Jutland moving eastward. This pattern is recognizable in the data reported in the PULS database.

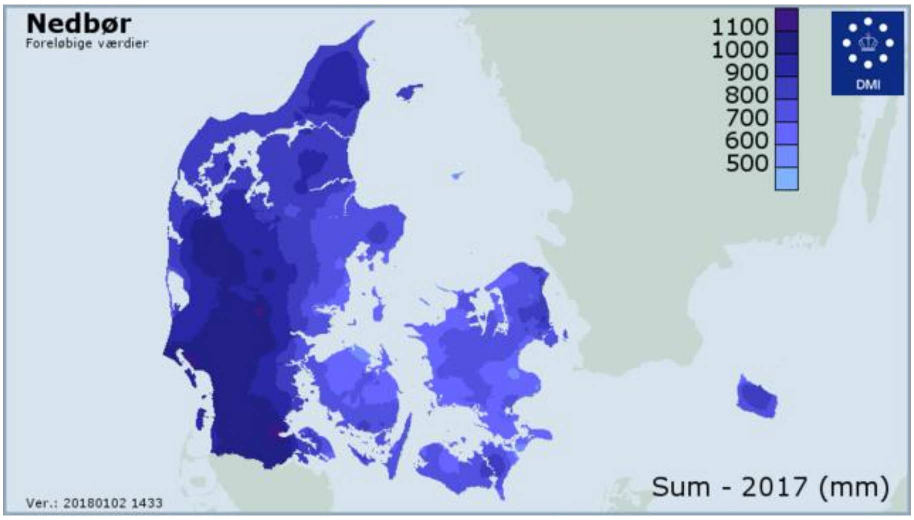


Figure 17. *Total yearly rainfall for 2017 (www.dmi.dk).*

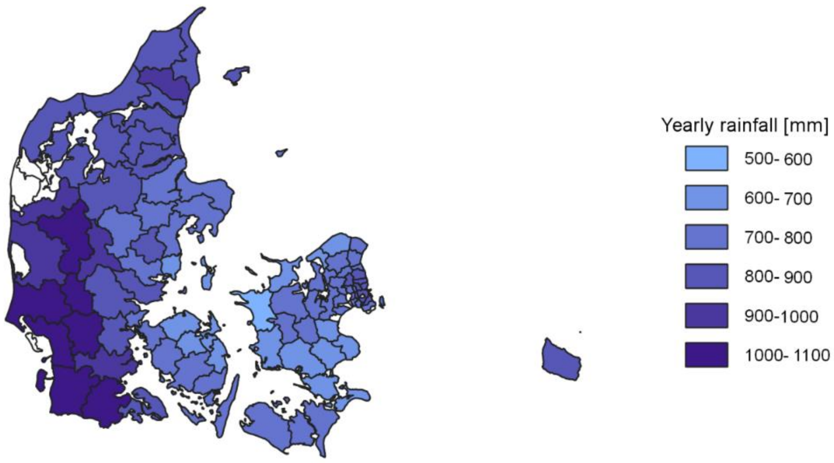


Figure 18*. Yearly rainfall reported in the PULS database for combined systems in 2017 (median of values reported for each municipality).*

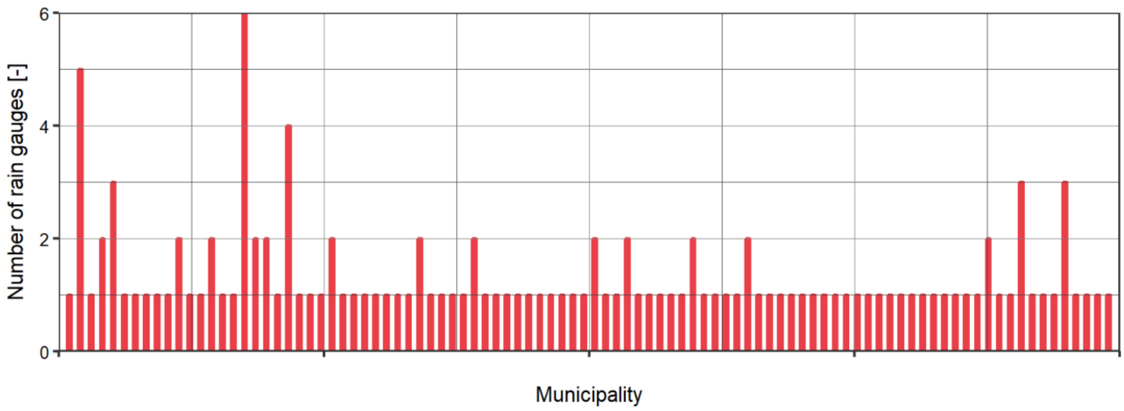


Figure 19. *Number of different rain inputs used by each municipality for separate systems (96 municipalities).*

The median of rainfall values reported for separate systems match those for combined systems for the majority of the municipalities: only 4 municipalities reported different values for separate and combined systems. This difference suggests the use of different rainfall values to represent the spatial variability across the different catchments of the same municipality. Figure 19 shows that only 19 municipalities (about 20% of the total) reported more than one rainfall value. It should be reminded that whenever rainfall values are not reported by the municipality, those are estimated by MFVM based on the data collected by DMI (Figure 17).

6.1.2 Discharged volumes

The relative discharge volume from each catchment illustrates how much volume is discharged for each unit of upstream impervious area. Since this variable is used for Level 1 calculations, it is a good indicator to explore the data recorded in the PULS database. Figure 20 show the relative discharges for separate and combined systems, respectively, grouped for each municipality. The comparison highlights how the median values reported for separate systems are mostly within the range 2500-7500 m^3/ha/yr, with a maximum median value below 9000 m^3/ha/yr for 2017. These values are in the same order of magnitude of those listed in the lookup tables used in Level 1 calculations. Few municipalities showed important variations in their relative discharges: the values for 20 municipalities (21%) had a difference above 2500 m^3/ha/yr between the 5% and the 95% percentile, while half of the municipalities (48) had differences below 50 m^3/ha/yr. These results suggest a relative homogeneity in the methods and parametrization used to calculate the discharges from separate systems.

The values reported for combined systems show a higher variability compared to separate systems. While the median relative discharges was below 1500 m^3/ha/yr for more than 80% of the municipalities, about 85% of the municipalities had more than 1500 m^3/ha/yr differences between the 5% and 95% percentiles of their catchments. Such variability for CSO relative discharges can be explained by several factors:

- *Errors in data reporting:* variability in the values due to errors in the process of recording data in the PULS database

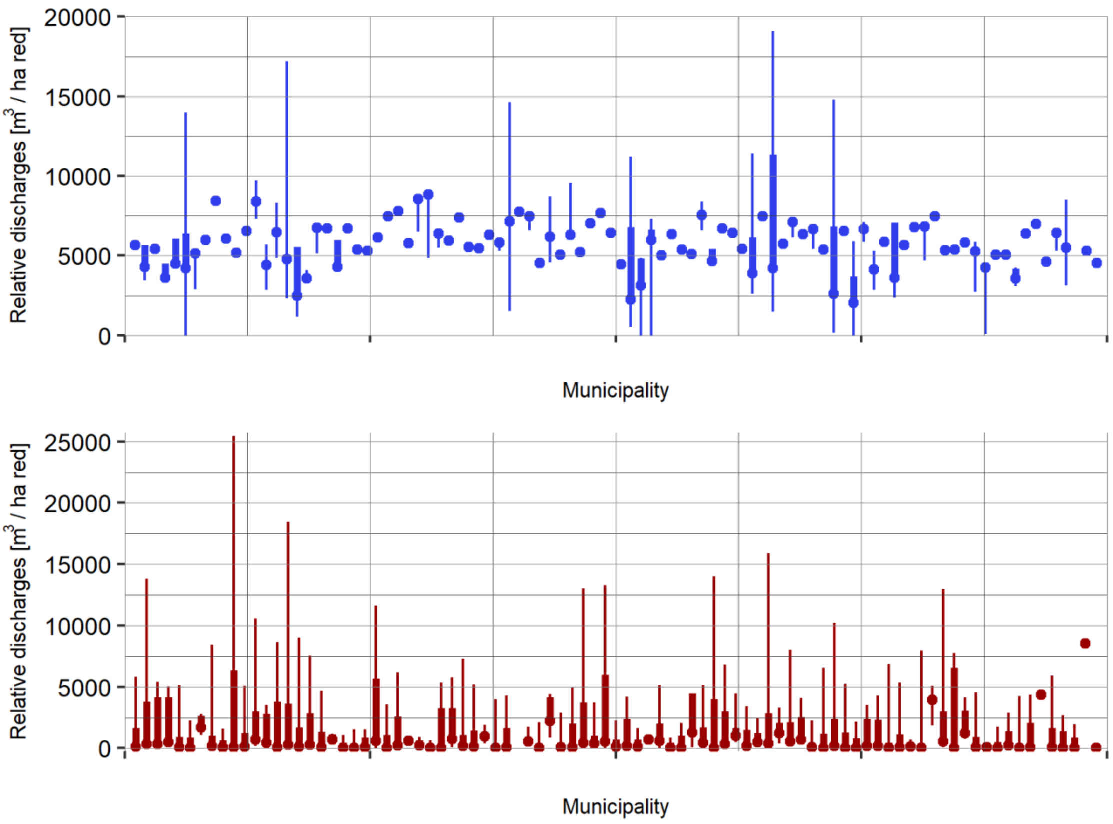


Figure 20. *Boxplot of relative yearly discharges from separate (top) and combined (bottom) systems for the available municipalities. Thin line 5-95 percentiles, thick line: 25-75 percentile, dot: median value.*

- *Classification of point discharges*: more than 2/3 of the total CSO volume reported in Denmark in 2017 was caused by 9 overflow structures. Specifically, one municipality has reported two overflow structures that contribute with more than 50% of the total accumulated overflow in Denmark. Although these data might appear as outliers at a first glance, these discharges refer to bypass of WWTPs. These are wet-weather discharges taking place when the WWTP treatment capacity is exceeded, i.e. they are affected by other processes than just exceedance of a hydraulic capacity. For example, different control operation of WWTP can increase the treatment capacity of a plant, while problems with the sludge settling properties in secondary clarifies might decrease the WWTP capacity. Also, the physical placement of the bypass channel can affect the level of pollution of the discharged water (see Vezzaro et al. (2018b) for further details. Therefore, although WWTP bypass are classified together with CSO, their characteristics are different, i.e. they should be excluded from the analysis when looking at relative discharges.
- *Inaccuracies in the connected areas*: 41 CSO structures had a relative discharge above 20,000 m^3/ha/yr. The upstream area was below 1 ha (or missing) for 28 (68%) of those CSO structures, suggesting that the reported upstream area is not representative of the actual upstream area. For example, for CSO located along interceptors and/or receiving runoff from other municipalities, the reported connected area is not directly linked to the actual CSO volume. Overall, this seems to be an issue to be further investigated, since 1,043 CSO structures have an upstream reduced area below 1 ha.

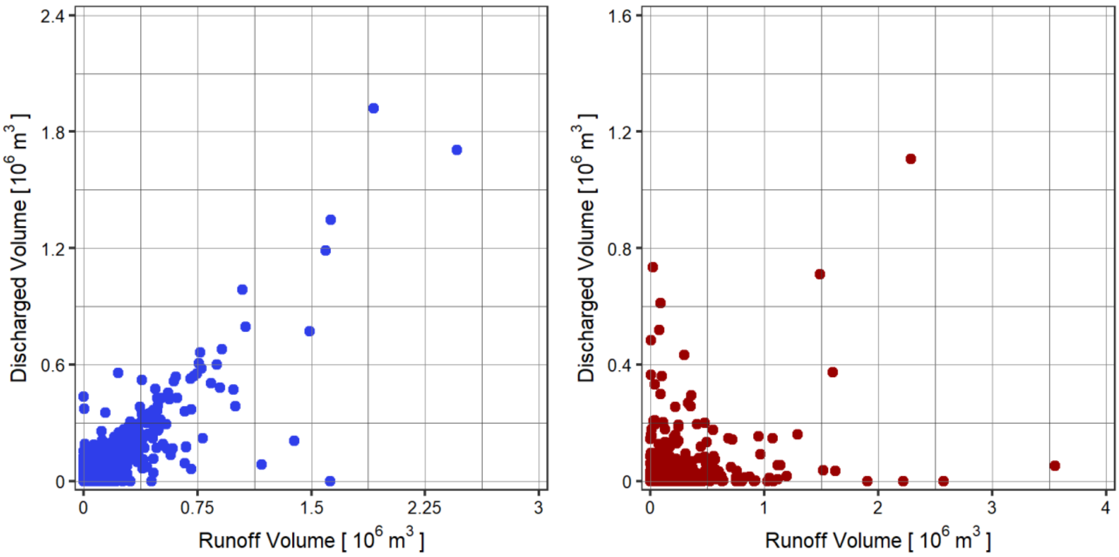


Figure 21. *Comparison against total runoff volume (x-axis) and discharged volumes (y-axis) recorded in the PULS database for 2017. Left: separate systems. Right: combined systems*

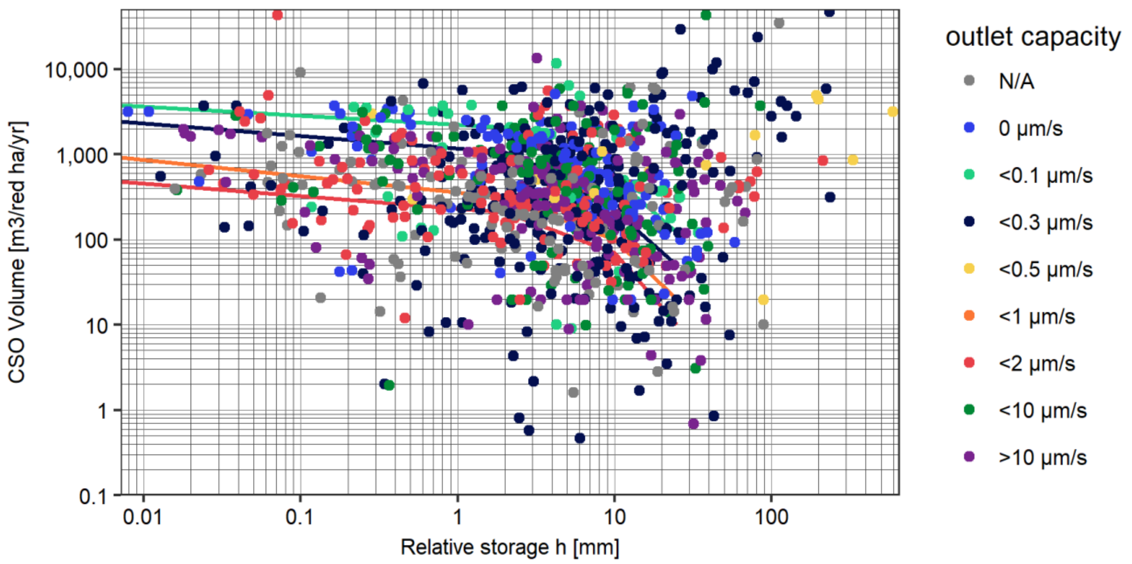


***Figure 22**. Comparison between relative storage volume and relative CSO discharges for different relative outlet capacities. Curves in background refer to the standard discharges values from the guidelines (also shown in Figure 14).*

- *The non-linear nature of CSO processes*: while discharges from separate systems are directly proportional to the generated runoff volume (obtained by multiplying the rainfall fallen on the catchment by the impervious area), CSO discharges depend on several factors, such as outlet capacity, storage volume, previous rainfall events (thus reducing the storage capacity of the drainage network), etc. This difference is illustrated in Figure 21, where the correlation between generated runoff and discharged volumes is evident for separate systems (left), while no clear correlations can be seen for CSO (right).

Another evidence of the non-linear behaviour of CSO discharges can be seen in Figure 22, which shows the relationship between relative storage capacity, relative CSO discharges, and relative outlet capacity. The patterns outlined in the Level 1 lookup tables (Figure 14) are not identifiable (this is also observed when CSO discharges are normalized to the reference rainfall value of 650 mm/yr – not shown here). Furthermore, the presence of CSO with reported storage volume but with relative outlet capacity set to zero and/or missing raises further concerns about the quality of the reported data.

6.1.3 Discharged pollutant loads

An overview of the volumes and pollutant loads reported in the PULS database is shown in Figure 23. By combining the available data, it is possible to calculate the average concentration for each discharge point. Figure 24 and Figure 25 show the distributions of the calculated average concentration for separate and combined systems, respectively. Figure 24 shows that, with few outliers, the greatest fraction of the reported discharges for separate systems used the reference values listed in the existing guidelines (Table 6), based on the data from Johansen and Petersen (1990)). Figure 25 show greater variability for CSO: the mode of the concentration values shows that the majority of the loads were calculated by using the lowest ranges that is available in the guidelines (combined stormwater with average pollution level). COD shows an exception, since the highest value from the guidelines ranges (CSO water with average pollution) was used. The distributions show also a second minor peak, corresponding to the second value listed in the guideline ranges. This shows a degree of subjectivity due to the ranges listed in the guidelines. The remaining values can be due to:

- *Errors in data reporting:*
- *Use of reference values other than those listed in the guidelines:* while the ranges listed in the guidelines refer to "average pollution levels", other values, reflecting the actual pollution level of the specific discharge points, can be used.

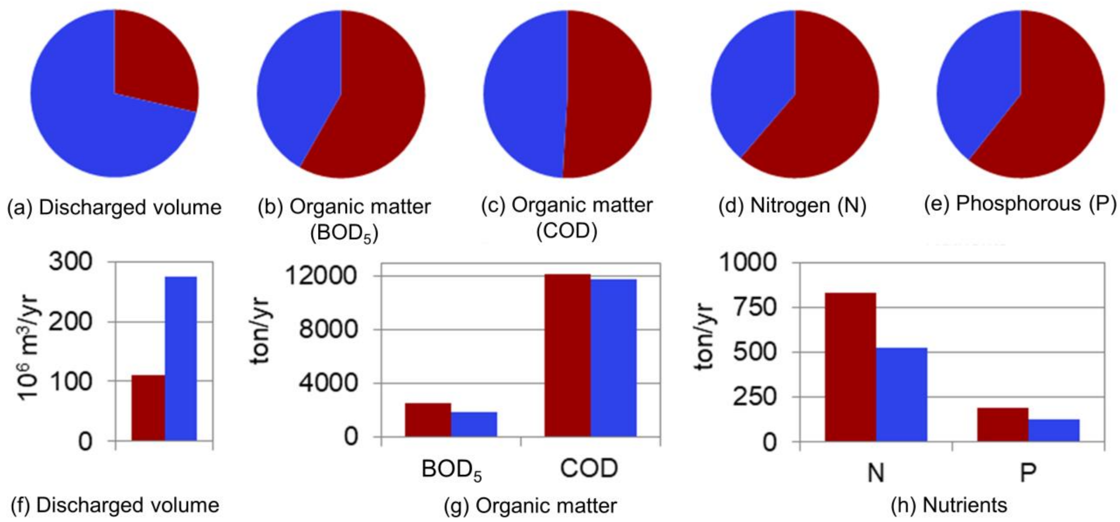


Figure 23. *Comparison between the total discharges for 2017 reported in the PULS database for separate (blue) and combined (brown) systems. Relative contribution (a-e) and absolute values (f-h).*

- *Different calculation methods:* data for separate systems and the majority of data for CSO discharges show that the simple multiplication of discharged volumes to average pollutant concentration (eq. 2) is the mostly used approach. However, other approaches, considering dilution, i.e. calculating the EMC in the discharges based on the contribution from wastewater and stormwater, can be used. This approach can easily be implemented when using Level 3 calculations which allow distinguishing between the two water flow contributions.

Overall, the data clearly show that the simple load calculations based on the use of a reference EMC are very popular, i.e. that the major modelling effort focuses on water quantity, while a dynamic modelling of water quality seems rarely applied.

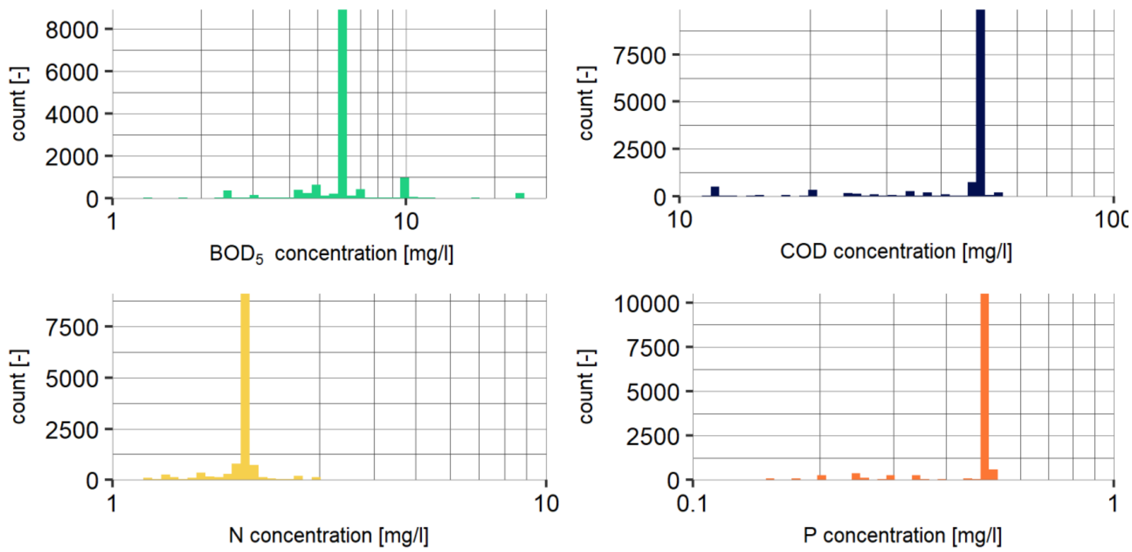


Figure 24. *Distribution of average concentrations calculated based on volumes and loads reported in PULS for separate systems (n=13168).*

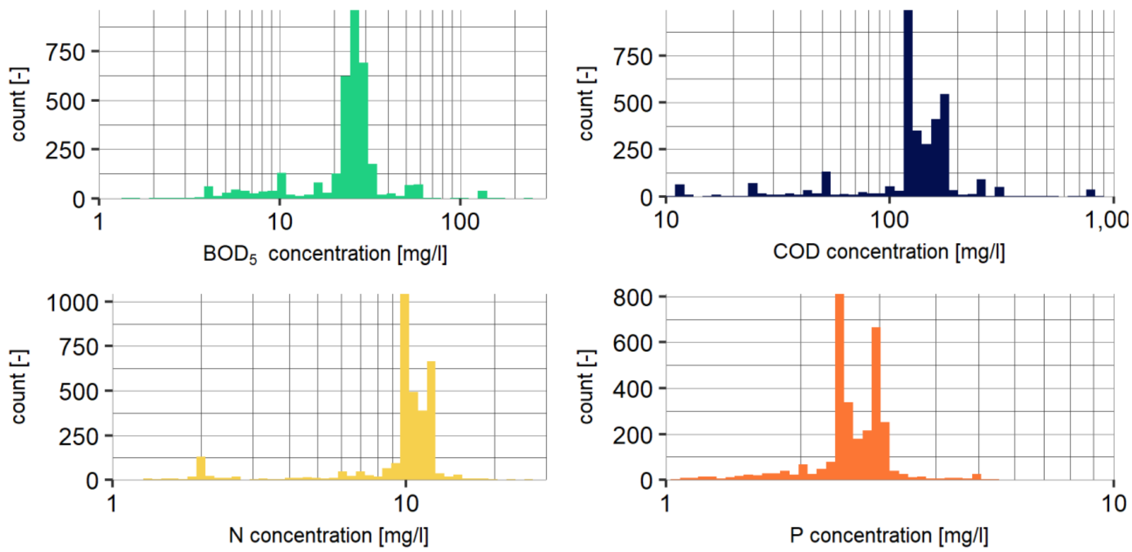


Figure 25. *Distribution of average concentrations calculated based on volumes and loads reported in PULS for combined systems (n=3435).*

Table 12. Statistics of the calculated average concentrations (expressed as [mg/l])

Water Quality Indicator	Separate systems			Combined systems		
	Median	Mode	Reference	Median	Mode	Reference
BOD_5	6.25	6.25	6	25	25	25-30
COD	50	50	50	130	180	160-180
N	2	2	2	10.2	10	10-12
P	0.5	0.49	0.5	2.54	2.5	2.5-2.9

The reference value is based on the one listed in the guidelines for calculation of wet weather discharges (Miljøministeriet, 2012). For combined systems, the listed values refer to "combined stormwater with average pollution level" (overvand *in Danish) and "CSO water with average pollution level", respectively (see Table 6).*

6.2 Differences in model-based estimations methods

The analysis of the PULS database shows that all the reported wet-weather discharges are based on estimates from Level 1 or Level 3 models (as described in section 3.1). In 2017, 38% of reported data were calculated by using the Level 1 method while 62% used Level 3. The proportion of PULS data calculated by using level 3 is expected to further increase as more detailed models of the existing sewer system are developed in the municipalities and utilities.

In an attempt to explore the differences and uncertainties in the results obtained by using the two quantification methods, data from three municipalities (referred to as municipality A, B and C) were selected and analysed. Over the last years, these three municipalities have switched from Level 1 to Level 3. This offered the possibility for a direct comparison of the results of the two methods.

For each municipality, the latest reported pollutant loads for each method have been compared. The comparison focused on the following reported data: reduced area, precipitation, overflow volumes and pollutant loads. The results of the analysis are found in Table 13 where the yearly sums of all reported discharge points are shown.

This partial analysis shows how discharged pollutant decreased after the introduction of Level 3. This suggests that the simple calculations and assumptions used in Level 1 lead to a consistent overestimation of the discharged loads. This is further emphasized by the fact that even if the total rainfall in 2017 was between 7% and 30% higher than in 2016, the reported volumes show a decrease between 13% and 57%.

The data for Municipality C cannot be used to directly compare the two methods, since significant modifications of the drainage system have occurred between 2016 and 2017. The total reduced area decreased by 24% (as results of area disconnections – see Section 6.3), while the available storage capacity shows a tenfold increase in 2017 compared to 2018. These important structural modifications of the drainage network mask the effect of the changes in the method for quantification of pollutant loads.

Table 13. Yearly sum of reported overflow quantities reported by the three analysed municipalities.

	Reduced area [ha]	**Storage volume [m3]**	**Precipitation [mm]**	**Discharged Vol. [10^6 m^3/year]**	**BOD_5 [kg/year]**	**COD [kg/year]**	**N [kg/year]**	**P [kg/year]**
Municipality A								
2016 (Level 1)	1104	51776	650	2.01	55888	279502	23240	6074
2017 (Level 3)	1104	51776	696	0.86	21527	137775	8611	2153
Change	*0%*	*0%*	*7%*	*-57%*	*-61%*	*-51%*	*-63%*	*-65%*
Municipality B								
2016 (Level 1)	1150	34349	690	1.49	39656	198356	16566	4252
2017 (Level 3)	1166	26377	894	1.27	38090	228540	15236	3682
Change [%]	*1%*	*-23%*	*30%*	*-15%*	*-4%*	*15%*	*-8%*	*-13%*
Municipality C								
2015 (Level 1)	1359	7209	762	3.25	87975	440012	36686	9518
2017 (Level 3)	646	80509	859	1.63	49030	294177	19612	4740
Change [%]	*-24%*	*1017%*	*13%*	*-50%*	*-44%*	*-33%*	*-47%*	*-50%*

More interesting are the cases for Municipalities A and B, where the reduced area is unchanged, and rainfall increased. Surprisingly, Municipality B reported a 23% smaller storage volume in 2017 compared to 2016. Despite all these factors would result in an increase of the total discharge volume (according to the equations listed in Section 3.1.1), the reported discharges show important decreases. These reductions can be explained by several factors such as the use of different hydrological reduction factors, the more realistic dynamic behaviour of stormwater flows (and thereby a more realistic representation of overflows processes), and differences in the rainfall inputs. All these factors underline how the Level 1 approach is not suitable for obtaining accurate estimation of wet-weather discharges when it is applied outside its boundary (i.e. if the rainfall and the catchment characteristics used to estimate the lookup tables do not match the catchment in question).

The results presented in Table 13 highlights the great structural uncertainty linked to the use of different quantification methods. Although only 3 out of 98 Danish municipalities have been analysed, it is expected that switching from the simple assumption of Level 1 models to the more realistic representation of Level 3 models would results in lower error in the estimation of discharged volumes and pollutant loads.

6.3 Differences in quantification procedure

The analysis of the differences between quantification methods cannot neglect the high influence that modellers can have on the model results. Different modellers can in fact affect the sources of uncertainty listed in Chapter 5 in different ways. The Level 1 estimation method is provided with a series of default/standard parameters (Miljøministeriet, 2012), where site-specific information (e.g. rainfall input, lookup tables, etc.) can be used if such information is available. On the other hand, the parametrization of Level 3 models is mostly defined by the modeller. Therefore, an analysis of the uncertainties in the data reported in the PULS database should also include a comparison of the procedures that are used by each municipality to quantify pollutant loads.

The three municipalities compared in Section 6.2 were contacted and the respective water utilities were interviewed regarding their simulation model used for reporting in 2017 (Level 3). Questions included their use of model inputs, hydrological reduction factors, as well as the validation and detail of their simulation model. The main outcomes and notable differences from the questionnaires are summarized in Table 14.

This overview highlights the great variability in the modelling approaches used by each municipality. For example, a variation bigger than 20% in the generated runoff can be obtained by simply changing from the hydrological reduction factor used by Municipality A to the one used by Municipality B. Other different aspects involve the inclusion of stormwater control measures (LAR) or the estimation of Q_{DWF}. Specifically, Municipality C included LAR in 2017 by modifying the total impervious area.

Common to all the three municipalities, the detailed models have not been calibrated, but their results are routinely compared against available measurements (flows and levels). In case of important discrepancies between measurements and models, the reasons for such deviations are further investigated. Furthermore, utilities often rely on external consultants, i.e. some aspects of the model parametrization might be overlooked or unknown by the utilities.

Overall, it can be stressed that the data reported in the PULS database are not consistent, in the sense that they have been estimated by using a variety of different estimation methods. Therefore it is not possible to perform a detailed uncertainty analysis across all the data and the municipalities included in PULS. Even for data reported by using the same method (Level 3), there is a wide range of different modelling procedures that would interfere with a statistical analysis. Most of the major sources of uncertainty listed in Chapter 5 are not reported in the database, i.e. it is not possible to investigate the reason for such variations.

None of the models were calibrated in the traditional sense, as it is usually intended in scientific literature (Jakeman et al., 2006; Jørgensen and Bendoricchio, 2001; Carstensen et al., 1997). However, the model simulation outputs are routinely compared against the available flow measurements, leading to a continuous improvement of the models. Therefore, the uncertainty of these Level 3 model simulations is expected to be lower compared to an uncalibrated model.

Without standardization and harmonization of the modelling procedure, variations in the modelling procedure are expected to be in the same order of magnitude than those due to natural variations and differences in the physical systems.

Table 14. Overview of the main differences in procedures and parametrization of Level 3 models.

	Municipality	Applied procedure/parameter value
Level 3 model(s) used for quantification of wet weather discharges	A	Detailed hydrodynamic model in combination with simple conceptual hydrological model
	B	Detailed hydrodynamic model
	C	Detailed hydrodynamic model
Subject running the model(s)	A	Water utility and consultant
	B	Consultant
	C	Consultant (with water utility in the process of establishing their own model team)
Calibration of model(s)	A B C	Models are not calibrated, but they are routinely compared against existing measurements from SRO system. In case of significant deviations, the model is further investigated
Validation of overflow structures models	A	Validation of overflow structures has been performed.
	B	Only some structures have been adjusted in the model to follow the correct Q-h curve.
	C	Validation of overflow structures has been performed.
Rain inputs	A	The model is run with a time series from one nearby rain gauge for a previous year. Corrections are then done to counter for differences in annual precipitation.
	B	Time series from 11 nearby rain gauges are used as input to the model for estimating CSOs
	C	Distributed rain from the rain gauges placed across the municipality
Inclusion of stormwater control measures in the model (LAR)	A	LAR solutions are not included in the drainage model.
	B	Few LAR solutions are incorporated in the model.
	C	LAR solutions have been included in the model
Estimation of groundwater infiltration	A	Groundwater infiltration is included in the estimated daily wastewater flow.
	B	Time series of groundwater infiltration is included in the simulation model. Numbers are obtained from correlation to discharges at the WWTP.
	C	Groundwater infiltration is not included
Daily wastewater production	A	180 l/person/day
	B	130 l/person/day
	C	Based on actual water consumption (adjusted every year)
Hydrological reduction factor	A	0.7
	B	0.9 for overflows structures with more than 5 observed overflows a year and 1 for the remaining with less overflows.
	C	0.8

7. Possible options for more certain quantifications of pollution loads from wet-weather discharges

7.1 Quantification approaches

The uncertainty analysis presented in the previous chapters underline three main results:

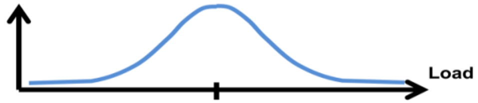
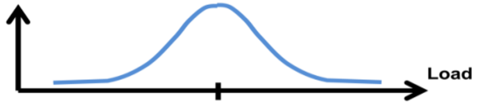
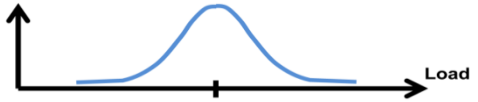
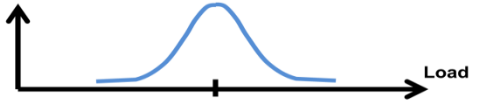
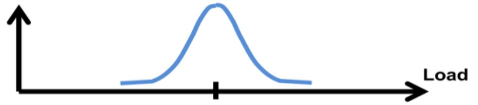
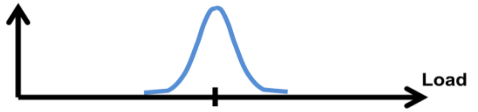
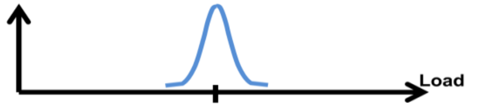
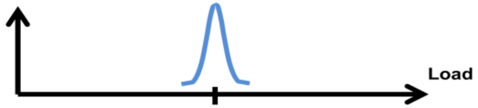
- Model-based estimation of pollutant discharges is affected by several sources of uncertainty that have an important effect on the accuracy and precision of the results. The magnitude of these uncertainties can be quantified, but the stochastic nature of wet-weather discharges implies that these uncertainties cannot be completely eliminated.
- The majority of the available information on the uncertainty of model-based load estimation has been carried out on point discharges, i.e. there are no results focusing on the analysis at the catchment/municipal scale. Similarly, most of results refer to single event estimations, i.e. the effect of aggregation over a yearly scale is often not investigated.
- The current pollutant loads recorded in the PULS database are characterized by high heterogeneity in terms of applied modelling tools, model parametrization and structure. This hampers the possibility of a general evaluation of the data reported in the PULS database, since the uncertainty values are highly site specific. This heterogeneity does not depend on technical aspects (i.e. the nature of the modelled phenomena), but it rather originates in the human involvement in the process of estimation and reporting of pollutant loads (i.e. utilities, municipalities, consultants).

Therefore, although uncertainty cannot be totally eliminated, specific actions can be taken to reduce and to better quantify it. Recognizing that a variety of modelling tools can be applied, different level of uncertainties can be associated to different quantification methods (as exemplified in Table 15). Furthermore, harmonizing and better describing the modelling procedure will enable the comparison of data estimated with different methodologies based on their estimated level of uncertainty.

Starting from the target of keeping uncertainty as low as possible, the most accurate quantification methodology is direct measurement of flows and pollution levels at every outlet point and for every rain event. This approach is currently adopted for quantification of loads discharged from WWTP outlets. Although this can be seen as the ultimate or reference method (equivalent to the Best Available Technology for quantification of pollutant loads), the accuracy of the measurements is highly dependent on the adopted equipment, sampling methodology, and maintenance. Also, the operational costs of this monitoring approach, applied at all discharge points, would be prohibitive.

More feasible and cost-effective quantification methods can be based on the combination of models and measurements. Clearly, simpler and cheaper methodologies will results in higher uncertainty, whose magnitude can though be estimated and compared. Based on these considerations, five different quantification levels are proposed (Table 15), with an increasing level of complexity and consequently decreasing uncertainty:

Table 15. Possible levels for quantification of pollutant loads from wet-weather discharges

Level	Input data	Method for quantification of discharged flows ($Q_{catchment}$, Q_{CSO})	Method for quantification of discharged pollutants (C_{CSO})	Estimated magnitude of uncertainty
A	Yearly rainfall	Simple hydrological balance based on measured data (rainfall, and WWTP inlet for QCSO)	Typical average concentration	
B1	Rainfall data (1 rain gauge or gauge outside/far from catchment)	Dynamic detailed hydrodynamic model		
B2	Rainfall data (several gauges within/close to catchment)			
C1	Rainfall data (1 rain gauge or gauge outside/far from catchment)	Calibrated dynamic model (detailed hydrodynamic or conceptual hydrological)		
C2	Rainfall data (several gauges within/close to catchment)			
D	Rainfall data (several gauges within/close to catchment)	Measured + Calibrated detailed hydrodynamic model used to estimated flow in ungauged locations		
E1	-	Measured	Measured (with autosampler – and sufficient data to estimate EMC distribution)	
E2	-		Measured (with online sensors)	

- Level A: Simple hydrological water balance for the total catchment for a treatment plant combined with standard values for the polluting substances.
- Level B: Dynamic modelling of the hydrological and hydraulic processes combined with standard values for the polluting substances.
- Level C: Calibrated dynamic modelling of the hydrological and hydraulic processes combined with standard values for the polluting substances
- Level D: Combination of measured hydraulic variables (flows, water levels), standard values for the polluting substances, and calibrated dynamic modelling of the hydrological and hydraulic processes (for ungauged locations).
- Level E: Measurement based quantification of pollutant loads (reference method).

These methods would require a modification of existing guidelines for quantification and reporting. For example, Level A would require additional fields in the PULS database with information of the WWTP connected to each CSO structure.

7.1.1 Level A - Simple hydrological water balance

In Level A, the basic assumption is that in an integrated urban catchment (as schematized in Figure 3), all the generated runoff entering the drainage system will end at a WWTP, a stormwater outlet or a CSO.

Discharge from separate stormwater outlets

In separated stormwater systems, a simple calculation based on the known catchment area, measured rain and the typical hydrological parameters will give the annual stormwater volumes discharged at the system outlet. Pollutant loads can then be calculated by using standard (or measured) concentrations. Basically, this method is equivalent to the current Level 1 calculations (see section 3.1.1).

Discharge from Combined System Overflow (CSO)

In case of combined systems, the water balance is slightly more complicated since the generated runoff is split in two flows: (i) the flows discharged over the overflow weir(s), and (ii) the main flows conveyed to the WWTP, treated and then discharged to the natural water bodies. However, the basic water balance for the integrated combined catchment can be calculated as:

$$\text{Inflow} - \text{Outflow} = \text{Overflow} \tag{10}$$

Where the *Inflow* of stormwater can be estimated as explained in eq. 4, by using the known catchment area, measured rain and the typical hydrological parameters. The values of the dry weather flow Q_{DWF} (including both wastewater and groundwater infiltration) can be defined by using the WWTP inflow measurements that are collected during dry days. Subtracting the Q_{DWF} from the WWTP measured inflow will therefore provide the stormwater contribution (i.e. the *Outflow* term).

Pollutant loads are then calculated by using standard concentration values. Conceptually, this method is very similar to the current Level 1 calculations, but it exploits the existing WWTP inflow measurements and it is applied at the WWTP catchment scale rather than at the municipal scale. It also reminds of the methods applied in France (see e.g. Métadier and Bertrand-Krajewski, 2011b,a). This method relies on existing data sources, and does not require new investments in new monitoring stations and/or better models.

The methodology implies an assumption, that the fraction of DWF that leaves through the overflow is very small compared to the rainwater flow. Also, the method is valid for discharges from entire combined catchments, i.e. it is not applicable to discharges to single CSO structure. In case of catchments subdivided across different municipalities, the CSO discharges should be subdivided based on the estimated contribution of reduced area.

Estimated uncertainty: the main source of uncertainty is represented by the estimation of the inflow, namely in the calculation of the generated runoff based on the measured rainfall. Estimation of inflow to a WWTP is judged to be the less uncertain variable to estimate (in the range of 10-20%), the impact of rainfall can have a slightly higher uncertainty (if rainfall input is well representative), while average concentration values have uncertainty greater than 30-40% (if inter-site variability is assumed to be small). Overall, the uncertainty on the estimated wet-weather loads is thus expected to easily exceed 150-200% in ideal conditions, i.e. in the same range as the current Level 1 simple hydrological calculations. However, thanks to the use of measured data rather than lookup tables, a better accuracy is expected, with the major fraction of the uncertainty linked to the natural variability of the process. In practice, this high level of uncertainty means that the uncertainty varies from having no discharge at all, to having a three times bigger discharge than modelled. This expected uncertainty value is based on conservative estimation: a more accurate quantification of the uncertainty for this method can be obtained if an analysis of the existing rainfall and WWTP inlet data, compared against the estimated CSO volumes will be carried out. Since the results from this method refer to a single discharge points, further analyses should focus on the uncertainty at the catchment/municipality scale. This method can also be used as a screening tool for validating the data reported in the PULS database, as well as for identifying possible errors and anomalies in the records.

7.1.2 Level B - Dynamic hydrodynamic models

The majority of the Danish water utilities have developed detailed hydrodynamic models to describe a great part of their drainage network. These detailed hydrodynamic models are commonly used for design, application for discharge permits, reporting, etc. These model are widely applied for quantifying discharges from wet-weather systems (the current Level 3 – see Section 3.1.2). Despite being often judged as the most accurate models due to their detailed hydrodynamic description, the uncertainty of their results can easily exceed 100-150% in the volume estimation, depending on the used rainfall data. Therefore, it is paramount to define a standard rain series that can be used to reduce the uncertainty in the rainfall input (see section 7.2). Furthermore, the definition of other hydrological parameters (such as initial loss, hydrological reduction factor, etc.) should be standardized and included in the reporting information. Also, the model detail level is often defined for other modelling objectives (e.g. planning, flood risk), i.e. their structural might not be optimized for simulation of CSO discharges.

Standard concentrations can be used to calculate the discharged pollutant loads.

Conversely to combined systems, where a correct simulation of flow dynamics is relevant to estimate Q_{CSO}, the use of detailed hydrodynamic models for separate systems would not have significant impacts on the quantification of the discharged volumes. Therefore, for separated systems Level A approaches can still be used without significant effects on the magnitude of the uncertainty.

Estimated uncertainty: is expected to be lower than for Level A, but still around or above 100% when both uncertainty on discharged CSO volumes and average concentrations are combined. Since these results refer to a single discharge points, further analyses should focus on the (likely lower) uncertainty at the catchment/municipality scale.

7.1.3 Level C – Calibrated dynamic models

The uncertainty in Level B models can be reduced by using available measurements for their calibration. Several water utilities have installed online sensors for recording water levels and flows across their infrastructure for operating purposes (SRO systems). Carefully selected and validated, these data, in combination with proper rain data, can provide the basis for calibration. Both detailed hydrodynamic models and simple conceptual hydrological models can be applied for the quantification of the discharged pollutant loads, given that they are properly calibrated.

Advanced parameter estimation techniques can be applied to reduce the uncertainty of several hydrological parameters (e.g. the Manning coefficient, the reduced area, the initial loss, etc.), leading to an overall reduction in the model result uncertainty. However, the calibration procedure should be standardized and have minimum requirements in terms of e.g. simulated rain events, number of measuring sites used for calibration. Also, uncertainty-based calibration procedures (Vezzaro et al., 2013; Thorndahl et al., 2008; McIntyre et al., 2002) are strongly encouraged in order to explicitly account for parametric uncertainty. For CSO, a correct calibration of the Q-h discharge curve is paramount for a correct estimation of the discharged volumes. This can be obtained by applying existing CFD techniques (Ahm et al., 2016).

Given the high uncertainty affecting existing dynamic water quality models (especially for particulate pollutants - e.g. Bonhomme and Petrucci, 2017; Bertrand-Krajewski, 2007) and logistical difficulties in the collection of a sufficient amount of water quality data, their calibration is judged as very difficult. Therefore, the use of standard concentration values is still an acceptable practice. Nevertheless, local monitoring campaigns, aiming at better estimating the average discharge concentrations, should be encouraged.

Estimated uncertainty: in an ideal condition, with a well-calibrated model (30-35% uncertainty in CSO volumes), a well-representative rainfall input (about 10% error), and a sufficient amount of site-specific EMC measurements (30-40% uncertainty), the overall uncertainty in the estimated pollutant annual loads can be expected to be around 50%. Since these requirements are very difficult to fulfil, uncertainty in practice will be higher. Also, it should be stressed that an incomplete or inadequate model calibration can easily lead to higher uncertainties that can equal (or even exceed) those of Level A and B calculations. Since these results refer to events at a single discharge points, further analyses should focus on the uncertainty at the catchment/municipality, which is expected to be lower.

7.1.4 Level D – Combination of measurements and dynamic models

This level reduces the uncertainty in the pollutant load quantification by combining the available measurements (flows, water levels) with the results of calibrated dynamic models (as for Level C) for ungauged locations. Further advantages of this approach consist in providing a backup in case of sensor malfunctioning (i.e. the model results can be used to integrate missing data) and in further reducing the uncertainties in the model results (i.e. providing a better representation of dynamic processes such as changes in rainfall losses, groundwater infiltration, etc.). If implemented in an online context, or with short time-lag between measurement collection and model verification, this approach can be seen as a sort of "continuous model calibration", which ensure that the model of the system is constantly updated based on the latest available information.

Estimated uncertainty: is expected to lie between those estimated for Level C and Level E, i.e. in the range above 35-50%. These values are an estimation based on ideal conditions in terms of model calibration, data availability, and input representativeness. Since these requirements are very difficult to fulfil, uncertainty in practice will be higher. Also, these results refer to a single discharge points, further analyses should focus on the uncertainty at the catchment/municipality scale.

7.1.5 Level E - Measurements

As stated earlier, direct measurement of water quantity and quality is regarded as the reference method that ensures the lowest level of uncertainty. Existing commercial flow sensors in sewer pipes can achieve quite accurate measurements, and Q-h curves can be accurately estimated by using existing modelling techniques.

Water quality measurements can be carried out by using automatic samplers (resembling the approaches used for monitoring WWTP outlets) or online sensors. The latter provide high-time resolution data, reducing the uncertainty linked to the sampling setup (Sandoval and Bertrand-Krajewski, 2016) and sampling techniques (e.g. Ort et al., 2010a,b), but are limited to few water quality indicators (TSS, ammonia, COD). Also, the performance of online sensor tends to quickly deteriorate if maintenance and data validation are not correctly performed. Therefore, operational expenses represent an important factor to take into account.

Estimated uncertainty: based on the experience from state-of-the-art monitoring campaigns, the uncertainty in the estimated annual pollutant loads is expected to be in the order of 30-35% (using an automatic sampler). Uncertainties are expected to be smaller if online water quality sensors are employed. However, this assumption requires a constant effort for sensor maintenance and calibration, as well as for data quality control routines.

7.2 Characterisation of rainfall input

Since the rainfall input represent one of the major sources of uncertainty for any model-based quantification of pollutant loads, it is necessary to clearly define how rainfall data should be used for such purpose.

Analyses of Danish extreme rainfall have established that the period 1979 – 2012 is representative for the current Danish climate. The substantial variability due to the relatively short observation periods at local sites should then be filtered out by constructing artificial series based e.g. on the approach by Sørup et al (2017). The method would have to be adapted from creating series that resemble future climates to resemble current climates and also the performance indicators would have to be reassessed. In principle this should be straight forward. Correction factors could then be employed to account for the variation in space that has been observed, giving more accurate assessments of the discharged volumes.

Other methods could be employed for generating artificial rainfall series that resemble current climate and indeed several methods have been developed and tested in a Danish context as discussed in e.g. Sørup et al. (2018).

If such methods are not employed, the contribution of the variability of rainfall to the overall uncertainty is likely to supersede the reduction expected in other sources of uncertainty when moving from Level A to higher levels. At least decades of (volumetric) measurements should be available for a catchment that has remained rather constant throughout the measurement period (to avoid ambiguity between the interpretation of different types of catchment responses).

7.3 Protocols for reducing uncertainty

Existing tools enable a quantification of the uncertainty affecting the model-based estimation of wet-weather pollutant discharges. However, the analysis of the existing data in the PULS database highlights the need for harmonizing the modelling procedure, thus enabling comparison and reproducibility of the recorded values. This can be achieved by:

1. Developing specific guidelines and training for each for the listed quantification approaches. A common modelling framework would allow for the reproducibility of the model results, avoiding the subjectivity seen in the current applications of Level 3 models. This framework should include a clear definition of the rainfall inputs, the procedure for the definition of hydrological parameters, protocols for data collections, calibration methods, etc. These guidelines can be developed based on analysis of existing datasets (e.g. the inflow data from WWTP could be used to evaluate Level A) and existing detailed models (currently used by water utilities for planning their drainage infrastructure).
2. Creating incentives towards a wider application of low-uncertainty methods (levels from C to E), with special focus on collection of measurements. This might include a better quantification of an “estimated magnitude of uncertainty” (as exemplified in Table 15), since the values listed in Section 7.1 are based on results from the international literature and more site-specific values are needed. Thanks to these uncertainty quantifications, it will be possible to estimate “typical confidence intervals”, which can then be used for defining “safety factors” (as currently done for design of urban drainage infrastructure). For example, load quantified by using Level A (assuming a 200% uncertainty) should be multiplied by a factor 3, load quantified by using Level C (assuming a 50% uncertainty) should be multiplied by a factor 1.5, etc. These “safety factors” would be based on conservative, (“worst case”) considerations, and they might be used in case of potential regulation and/or taxation of wet weather discharges.
3. Evaluating the results from the different quantification methods on annual basis in order to update the existing guidelines. This analysis will enable the comparison and the validation of the reported data, enabling the prompt identification of anomalies and highlighting the need for improvements in the existing quantification procedure. Furthermore, reference

values should be frequently updated. For example, recent analyses (Arildsen and Vezzaro, 2019) showed a decrease in the emission of phosphorous in domestic wastewater by one third over the last decade. Also, average pollutant concentrations should be re-estimated by including results from recent monitoring campaigns. For example, the values listed in the current guidelines (Miljøministeriet, 2012) were estimated before the monitoring campaigns listed in Appendix A.

4. Establishing a user group for exchange of experiences among the different subjects involved in the process (utilities, consultants, municipalities, environmental authorities, knowledge institutions), which can discuss the performance of the reporting system

5. Investigating the application of novel, cost-effective, monitoring approaches in order to improve the amount of available measurements of water quality control. Currently available water quality data are limited to a handful of discharge locations and events. Increasing the amount of measurements from wet-weather discharges will allow reducing the uncertainty linked to the estimation of average concentrations. Longer monitoring periods will allow a better evaluation of the inter-event variability, while a greater number of monitoring locations will enable a better understanding of inter-site variability. Given the logistical challenges and the financial resources required by traditional monitoring approaches, new techniques should be investigated. Overall, this effort should results in monitoring guidelines specifically targeting wet-weather discharges and their impacts.

8. Conclusions

This report assessed the uncertainty of the model-based estimation of the yearly pollutant loads (BOD_5, N, P) discharged from wet-weather discharges (separate systems and Combined Sewer Overflows) and currently reported by each Danish municipality in the PULS database.

All the modelling approaches used to quantify pollutant loads are affected by several sources of uncertainty which affect the accuracy and precision of their results. The uncertainty analysis carried out in this report identified several of these sources of uncertainty. These can be classified in terms of location, level, and nature of uncertainty. A great number of these sources showed a high natural variability, i.e. they cannot be eliminated from the modelling procedure. The magnitude of some uncertainty sources could be quantified (in terms of precision) based on results from previous studies. However, site specific conditions, combined with the inherent variability of wet-weather discharges require *ad-hoc* investigation to reduce the model uncertainty. The quantification of the model uncertainty in term of result accuracy (bias) can only be obtained by performing specific monitoring campaigns targeting both water quantity (e.g. measurements of CSO volumes, water levels, flows) and quantity (e.g. measurement of pollutant concentrations in discharged water).

The analysis of the current data in the PULS database showed that an additional source of variability is linked to the procedures for model application. Currently, modellers have several degrees of freedom with regards with model structure, model parametrization, utilized inputs, etc. which can affect the final result. For example, an initial analysis suggests that the use of simple models (Level 1) might result in higher estimation of pollutant loads compared to detailed models (Level 3). In order to compare the results of the model-based estimation of pollutant loads across catchments, it is therefore important to harmonize the modelling procedure across the municipalities. This would eliminate an important source of variability, which is mostly linked to subjective choices, and it will allow for an improvement of the data in the PULS database.

This report proposed a list of different actions that can be taken to reduce the uncertainty of model-based pollutant load estimations. These options improve the model accuracy by assimilating an increasing amount of information from measured data (e.g. from more accurate rainfall inputs, flow measurements, water quality measurements). Depending on the applied modelling approach, results on pollutant loads can be obtained at the catchment scale and/or for each single discharge point. The approach linking an increasing effort in data collection to a reduction in model uncertainty can provide an incentive to investments in more extensive monitoring across urban drainage systems.

In ideal conditions, the uncertainty in the estimated annual pollutant loads estimated by using the proposed methods might vary from above 150-200% (approach based on a simple water balance) to 30-35% (approach based on extensive monitoring of water quality). Since these ideal conditions are often difficult to fulfil in practice, uncertainty will in practice be higher. Since the majority of the available information in the scientific literature refers to single events and/or discharge points, the uncertainty on modelled yearly pollutant loads at the catchment scale could not be quantified. However, the uncertainty level is expected to be diminished due to the aggregation over several discharge points and over time.

The uncertainty levels estimated in this report mostly refer to the model precision, since difficulties in monitoring wet weather discharges limits the availability of the measurements necessary to assess the model accuracy. The high relative uncertainty (expressed as percentage) on quantification of wet-weather discharges should be put into perspective when compared to the absolute discharges (expressed as pollutant mass/year) from other types of point and diffuse sources. When looking at long-term effects (accumulating pollutants), wet weather discharges from combined and separate systems represent about 2% (N) and 3% (P) of the total yearly load. When considering short-term acute negative effects (driven by e.g. discharge of organic matter), the uncertainty analysis suggests that the current quantification methods are too imprecise to be used for detailed impact assessments. Also, the level of uncertainty is significantly higher compared to the one of other point sources of an integrated urban storm- and wastewater system, such as WWTP outlets.

Specific modelling guidelines can provide an important contribution to the reduction of model uncertainty, and they will reduce the possibility of subjective choices affecting the model results. Establishing new guidelines would require the involvement of all the stakeholders involved in the process of quantification of wet-weather discharges (municipalities, water utilities, consultant engineers), in order to harmonize the reporting across all the Danish municipalities.

The actions proposed in this report will likely result in a reduction of the model results uncertainty, thereby creating a more reliable data background in PULS. This can be further used for e.g. regulative purpose, evaluation of system performance, and identification of specific actions to reduce discharges from point sources.

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Appendix A Concentration data from monitoring of Danish CSO discharges

The Danish Environmental Protection Agency has carried out several intensive monitoring campaigns focusing on wet weather discharges from both separate and combined systems. The results of these monitoring campaigns often provide the basis for the estimation of the typical concentration values that are used in the estimation of pollutant loads.

In this report, data from selected monitored campaigns (Table A1) were analysed. The main characteristics of these datasets are listed in Table A. The reader is redirected to the original source for further details on the monitoring campaigns and on the single measurement values.

Table A1. List of monitoring campaigns included in the analysis

Catchment		Type of system	Source
a	Toftøjvej	Combined	Miljøstyrelsen (2006)
b	Sulsted (A)	Separate	Miljøstyrelsen (2006)
c	Sulsted (B)	Combined	Nordjyllands Amt (2006)
d	Frejlev	Combined	Nordjyllands Amt (2001)
e	Gug skole	Combined	Miljøstyrelsen (in preparation a)
f	Grønlandstorv	Combined	Miljøstyrelsen (in preparation b)

Table A2. Concentration data from Danish measurement campaigns (expressed as [mg/l])																	
Site		**BOD_5**				**COD**				**N**				**P**			
		μ	**m**	**σ**	**N**	**μ**	**m**	**σ**	**N**	**μ**	**m**	**σ**	**N**	**μ**	**m**	**σ**	**N**
a	Toftøjvej					187	206	121	15	4.27	2.59	4.16	17	59.59	1.63	172	16
b	Sulsted (A)	5.61	5.2	3.66	18	25.6	24	19.4	23	5.65	2.35	6.49	23	0.46	0.25	0.50	23
c	Sulsted (B)	114	54	146	27	355	182	338	27	10.2	7.26	10.4	27	3.23	1.33	4.43	27
d	Frejlev	33.3	14.7	36.2	13	87.8	50.9	79.9	20	3.85	3.66	2.86	22	1.03	0.69	0.95	18
e	Gug skole					135	101	96.6	12	5.03	4	4.24	12	1.20	0.81	1.11	12
f	Grønlandstorv	91.3	74.6	59.1	10	264	178	240	22	8.13	7.8	5.32	22	2.07	1.85	1.10	22

μ: mean value; m: median; σ: standard deviation; N: number of available observations

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