Water Research 183 (2020) 115988

Contents lists available at ScienceDirect

Water Research

journal homepage: www.elsevier.com/locate/watres

Review

Nature-based solutions as enablers of circularity in water systems: A review on assessment methodologies, tools and indicators

C.E. Nika [a], L. Gusmaroli [b, c], M. Ghafourian [a], N. Atanasova [d], G. Buttiglieri [b, c], E. Katsou [a, *]

[a] Department of Civil & Environmental Engineering, Institute of Environment, Health and Societies, Brunel University London, Uxbridge Campus, Middlesex, UB8 3PH, Uxbridge, UK
[b] Catalan Institute for Water Research (ICRA), Emili Grahit 101, 17003 Girona, Spain
[c] Universitat de Girona, Girona, Spain
[d] Faculty of Civic and Geodetic Engineering, University of Ljubljana, Hajdrihova 28, Ljubljana, Slovenia

A R T I C L E I N F O

Article history:
Received 10 March 2020
Received in revised form
25 May 2020
Accepted 26 May 2020
Available online 18 June 2020

Keywords:
Nature-based solutions
Water systems
Circular economy
Assessment methods
Key performance indicators

A B S T R A C T

Water has been pushed into a linear model, which is increasingly acknowledged of causing cumulative emissions of pollutants, waste stocks, and impacting on the irreversible deterioration of water and other resources. Moving towards a circular model in the water sector, the configuration of future water infrastructure changes through the integration of grey and green infrastructure, forming Nature-based Solutions (NBS) as an integral component that connects human-managed to nature-managed water systems. In this study, a thorough appraisal of the latest literature is conducted, providing an overview of the existing tools, methodologies and indicators that have been used to assess NBS for water management, as well as complete water systems considering the need of assessing both anthropogenic and natural elements. Furthermore, facilitators and barriers with respect to existing policies and regulations on NBS and circularity have been identified. The study concludes that the co-benefits of NBS for water management are not adequately assessed. A holistic methodology assessing complete water systems from a circularity perspective is still needed integrating existing tools (i.e. hydro-biogeochemical models), methods (i.e. MFA-based and LCA) and incorporating existing and/or newly-developed indicators.

Contents

* Corresponding author.
E-mail address: evina.katsou@brunel.ac.uk (E. Katsou).

https://doi.org/10.1016/j.watres.2020.115988

Abbreviations

BCR	Benefit Cost Ratio
BOD	Biochemical Oxygen Demand
CBA	Cost-Benefit Analysis
CE	Circular Economy
CEA	Cost-Effectiveness Analysis
CEIP	Circular Economy Indicator Prototype
CET	Circular Economy Toolkit
DWTP	Drinking Water Treatment Plant
EbA	Ecosystem-based Adaptation
ESS	Ecosystem Services
GHG	Greenhouse Gas
GI	Green Infrastructure
GIS	Geographic Information System
H2HA	Harvest to Harvest Approach
IAS	Individual Appropriate Systems
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LCCA	Life Cycle Cost Analysis
MCA	Multi-criteria Analysis
MCI	Material Circularity Indicator
MFA	Material Flow Analysis
MFCA	Material Flow Cost Accounting
MSA	Multi-Sectoral Systems Analysis
NBS	Nature-based Solutions
NPV	Net Present Value
PP	Payback Period
RSA	Regionalized Sensitivity Analysis
SFA	Substance Flow Analysis
SS	Suspended Solids
SuDS	Sustainable Drainage Systems
SWB	Soil Water Balance
TM	Territorial Metabolism
UHA	Urban Harvest Approach
UWM	Urban Water Metabolism
UWWTD	Urban Wastewater Treatment Directive
WSUD	Water Sensitive Urban Design
WTP	Willingness to Pay
WWTP	Wastewater Treatment Plant

1. Introduction

Water plays a critical role in human well-being, socio-economic development, as well as in sustainable ecosystem services (UNEP, 2009). Water is itself the most valuable and universal resource and at the same time, water contains nutrients and water is viewed as a carrier of energy thus, water systems intersect with all sections of society, industry and the natural environment (IWA, 2016; Arup et al., 2018). Over the past years, the increased stress on limited water resources has reached a critical level, in terms of both reduced water availability and jeopardized water quality (Sgroi et al., 2018). The ecosystems could not function without sufficient water supplies of appropriate quality, making water scarcity a key stressor in many ecosystems (Voulvoulis, 2018). On top, global demand for water is expected to exceed viable resources by 40% by 2030, if we continue business as usual (Wintgens et al., 2016). Therefore, the challenge is to meet or manage the competing demand for water, to minimize the damage to the environment and to regenerate the natural ecosystems (WWAP, 2018).

Water in nature represents one big cycle maintained by natural processes (e.g. precipitation, infiltration, evapotranspiration, condensation, etc.), which is interrupted by urbanization and by man-made water systems. Therefore, water has been pushed into the linear model of "take-make-consume and dispose", which is economically unsustainable and causes a successive degradation of water quality as water travels through the system (Stuchtey, 2015). In response to the linear pattern of growth, the adoption of the circular economy (CE) model is proposed that decouples economic growth and development from the consumption of finite sources (Murray et al., 2017; Babbitt et al., 2018; Hofmann, 2019). In order to deploy and enhance circularity, a number of studies have focused on identifying priority areas of action (Hislop and Hill, 2011; EC, 2015a; EMF, 2017; EMF and WEF, 2017; EMF, 2018; EMF and Arup, 2019). Water was identified as one of the key priority resources by Hislop and Hill (2011). The rest of the working groups mainly focus on the consumer goods sector mentioning the preservation of water, but the importance of closing the water loops is not well addressed. Single indicators assessing the circularity of products − either qualitative or quantitative − have been developed in various academic studies. The portfolio includes the Circular Economy Toolkit (CET) (Evans and Bocken, 2013); the Material Circularity Indicator (MCI) (EMF and Granta Design, 2015); the Circular Economy Indicator Prototype (CEIP) (Griffiths and Cayzer, 2016); the Longevity Indicator (Franklin-Johnson et al., 2016); and an economic value-based circularity metric (Linder et al., 2017). These indicators focus on the technical cycles, while disregarding the biological cycles, which are of major importance in the water sector. They target at materials preservation with strategies, such as recycling, which is only one aspect of circularity and even misleading in the water sector. Water cannot be "manufactured" by recycled materials as water is a raw material itself. Thus, preservation of materials can only be applied in the drinking water treatment plants (DWTPs) and wastewater treatment plants (WWTPs) in order to recover nutrients, salts and metals that can be used in other interconnected systems, such as agriculture and industry. These strategies are good for preserving raw materials stocks but the preservation of water stocks is poorly addressed. On the other hand, Pauliuk (2018) proposed a dashboard of new and established indicators − based on material flow analysis (MFA), material flow cost accounting (MFCA) and life cycle assessment (LCA) - to assess five main characteristics, i.e. restore, regenerate, maintain utility, maintain financial value, and maintain nonfinancial value, as well as four complementary characteristics, i.e. resource efficiency, climate, energy, and sufficiency.

In the water sector, three reports have been published up to date (i.e. Stuchtey, 2015; IWA, 2016; Arup et al., 2018), conceptually describing what should be considered in order to create circular water systems. They highlight the importance of establishing different water functionalities (e.g. consumptive and non-consumptive water, water as a durable) that would enable the balance between water withdrawals and return flows; the consideration of water, materials and energy pathways in order to create synergies within and outside the water sector. The need for an integrated urban resource management is also highlighted, looking at the water cycle "from catchment to consumer and back to catchment", following a systems approach that reveals interconnections between the human-managed and nature-managed systems. At the core of circular water systems lies the realization of the three principles identified by Arup et al. (2018), i.e. “Design out waste externalities”, “Keep resources in use” and “Regenerate natural capital”. However, to the best of our knowledge this approach has not yet been applied into practice for developing an assessment methodology for circular water systems.

The transition to circular water systems requires the redesign of the water infrastructure, the utilization of recent developments in technology and the integration of nature-based ecosystems to the grey infrastructure (i.e. hybrid infrastructure) (O’Hogain and McCarton, 2018). Existing concepts and approaches using and enhancing nature, such as ecosystem-based adaptation (EbA), green infrastructure (GI), ecosystem services (ESS) and nature-based solutions (NBS), have gained momentum as they tackle challenges (e.g. climate mitigation and adaptation, water management, degradation and loss of natural capital, disaster risk reduction, etc.) in a more sustainable way, compared to the conventional hard engineering. While all four concepts share the common principle of multifunctionality, NBS can be considered as an umbrella to the other concepts with a strong solution-oriented focus and biodiversity lying at its core (Pauleit et al., 2017). Up to date, several definitions have been applied to describe NBS, e.g. the definitions provided by EC, 2015b; Cohen-Shacham et al. (2016); Raymond et al. (2017); O’Hogain and McCarton (2018); Langergraber et al. (2019a). According to them, NBS should be cost-effective, resource efficient and locally adapted. NBS are systemic interventions that bring more, and more diverse, nature and natural features and processes. They address either a specific problem (i.e. societal challenge) or multiple challenges and simultaneously provide environmental, social and economic benefits, such as biodiversity, climate change mitigation and adaptation, resilience, human well-being etc. While CE seeks to reduce environmental stress of socio-economic activities, NBS have the potential to enhance environmental and ecological status and to address human demand for natural resources. NBS can restore the crucial natural processes – by changing the fluxes of water, sediment, nutrients and pollutants – that drive the water cycle and thus, return the circularity to the water systems. NBS are also capable of resources’ recovery from water, like nutrients, which fits in the natural water and nutrients cycles facilitating the transition from open to closed-loops. Therefore, the synergies fostered between the two concepts bring NBS to the forefront of enabling the realization of circular water systems.

This study is conducted in order to shed light on the assessment of circular water systems and the integration of NBS as enablers to water circularity, by addressing two main questions. What are the main parameters that should be considered in circular water systems (i.e. Section 2)? And what methods could be deployed in order to holistically assess the circularity of water systems? Considering that NBS can be used as means to integrate nature to human-managed systems, a thorough investigation on the current state of NBS is carried out. This is followed by a literature review on the assessment methodologies that have been applied to evaluate their performance, including indicators that have been used/developed as metrics of performance (i.e. Section 3). Current regulations and policies that act as barriers or facilitators for implementing NBS are also reviewed (i.e. Section 4). The third part (i.e. Section 5), reviews existing tools, methods and indicators for assessing the performance of water systems and elaborates on their suitability to evaluate key circularity aspects. The last part (i.e. Section 6) concludes on what methods and indicators can be used and what is still missing in order to holistically assess the circularity of water systems.

2. Circularity in water systems – what needs to be measured?

In view of the lack of a water circularity definition, the white papers of Stuchtey (2015), IWA (2016) and Arup et al. (2018) are considered in this study, to understand what needs to be measured for assessing circularity in water systems. The three principles of “Regenerate natural capital”, “Keep resources in use” and “Design out waste externalities” should be assessed, following the water, energy and materials/nutrients pathways. The three principles indicate the requirement of a systems approach as well as the consideration of the interactions between natural and human-managed systems. Water-related human-managed systems encompass various socio-economic sectors, i.e. urban water sector, agricultural/food sector, energy sector, and industrial sector, with agriculture and industries accounting for the highest global water withdrawals (FAO, 2016). Therefore, water circularity needs to consider all water users and not being managed at a sectoral level. To this extent, water circularity can only be achieved in a multi-sectoral system, including both human-managed and nature-managed systems. Natural and anthropogenic water cycles should be closed and symbiotic management of resources be promoted, avoiding burden shifting both from one sector to the other, but also from the anthroposphere to the environment. To achieve and assess water circularity, the goals of the different CE principles are explained:

The goal of the “Regenerate natural capital” principle is to ensure functional environmental flows and stocks, by reducing the anthropogenic water uses, preserving and enhancing ecosystems, and ensuring minimum disruptions from human interactions and use. In order to assess this principle, selected assets of ecological integrity and of regulating, provisioning and cultural ecosystem services are proposed to be considered. Ecological integrity targets at reducing water and nutrient loss, and storage capacity of nutrients and water (i.e. soil nutrient retention, soil organic matter, soil water storage). Regulating ecosystem services are targeted at climate regulation (i.e. sources and sinks of GHGs), groundwater recharge and nutrient regulation. Provisioning ecosystem services target at crops, livestock and fodder, while cultural ecosystem services focus on recreation and aesthetic values.

The goal of “Keep resources in use” principle is actually to close the water and water-related materials and energy loops within the system. It can be achieved by optimizing resource yields, optimizing energy and resource extraction, and by maximizing recycling and reuse. Recirculation of resources to close the loops requires sufficient quantity of the reused resources and suitable quality to meet the internal demands, which would result in a reduction of the amount of resources that are abstracted from the nature and the amount of returning flows.

The “Design out waste externalities” principle focuses on both the reduction of waste and the economic efficiency of the system, i.e. the costs of reducing waste by one unit is equal to the economic and environmental benefits of having one less unit of waste. The reduction is achieved by taking actions to achieve the “keep resources in use” principle, while the remaining waste (i.e. gaseous, liquid and solid) has impacts on the environmental system, affecting the “regeneration of natural capital” principle. Therefore, the environmental impacts, the avoided negative environmental impacts and the economic benefits and costs need to be considered in the assessment.

All three CE principles can be achieved by modifying current systems and grey infrastructure. While classical grey infrastructure can be designed to enhance water reuse and resources recovery, it differs significantly from the concept and paradigm of NBS. NBS are using natural processes, i.e. they work with nature, while grey systems (infrastructure) use additional energy to achieve circularity. Introducing NBS to urban water management naturally enhances circularity of the urban water system, hence shifts the urban water management into CE. Water in the CE should be placed without unnecessary additional energy use, which for the modification of the existing grey water infrastructure would be inevitable. In particular, the implementation of NBS can lead not only to modifications, but rather to new (hybrid) systems, more sufficiently applying the three principles. For example, constructed wetlands treating road runoff will prevent pollution of water bodies, recharge groundwater and increase biodiversity. Wetland roofs will collect and treat rainwater to be used for non-potable domestic purposes, while simultaneously acting as a natural “air conditioning” that is cost-effective and resource efficient. Therefore, NBS by exploiting their multifunctionality can facilitate the transition to circular water systems. The implementation of a single NBS may simultaneously obtain multiple co-benefits related to circularity and achieve the same result with traditional grey infrastructure, where multiple mono-functional engineering solutions are required. Additionally, the fact that NBS interconnect nature-managed to human-managed systems leads to their consideration as a key element to identifying methods to holistically assess circularity of water systems.

Any water circularity assessment can be undertaken at three different scales, i.e. micro (single components), meso (interconnected components forming a system) and macro (catchment, city, region, or national). Assessment methodologies at any scale need to take into consideration the other scales of assessment as well, in order to add significant value and become applicable. Therefore, the interlinkages that the analyzed system has with other systems and scales needs to be considered as well, by providing the information required for the assessment at higher scales. Fig. 1, which illustrates how the information obtained from the assessment at a small scale provides feedback to the next assessment scale until the puzzle of water circularity of a basin is completed.

The puzzle diagram is followed in the present review of the currently available methodologies assessing NBS (at different scales) and water systems, in order to identify the “required information” that would shed light on how to assess water circularity.

3. NBS enabling circularity in water systems – systematic literature review

A systematic literature review was conducted to identify the current state of NBS to enhance circularity of water systems, targeting at publications from scientific journals identified from the databases of Scopus and Science Direct. The key words that were selected – i.e. “nature-based solution”, “water”, “water system”, “circular economy”, “circularity”, and “assessment” – were used in multiple combinations (as presented in Fig. 2) and the search (conducted throughout the whole text) was expanded to all article types. Among the identified papers, 323 studies were categorized based on the main societal challenge to be tackled and their numerical proportions are illustrated in the pie diagram of Fig. 2.

The connection of NBS to the concept of circularity, especially in water systems, is not well addressed by the researchers (bar chart results). Additionally, the pie diagram of Fig. 2 illustrates that only 7% and 3% of the research focuses on water security and water quality, respectively.

The studies dealing with water security and water quality were analyzed to understand the extent to which they address water-related issues. The water-related studies assessing different aspects of NBS performance are summarized based on their context in Table 1.

The study of Castonguay et al. (2018) is the only one identified dealing with social aspects. This study focuses on the evaluation of different strategies for technology adoption to help decision-making by simulating the interactions of regulatory bodies and households within an agent-based model, integrating economic and environmental factors. Their results indicate that using economic instruments alone may have been insufficient to promote

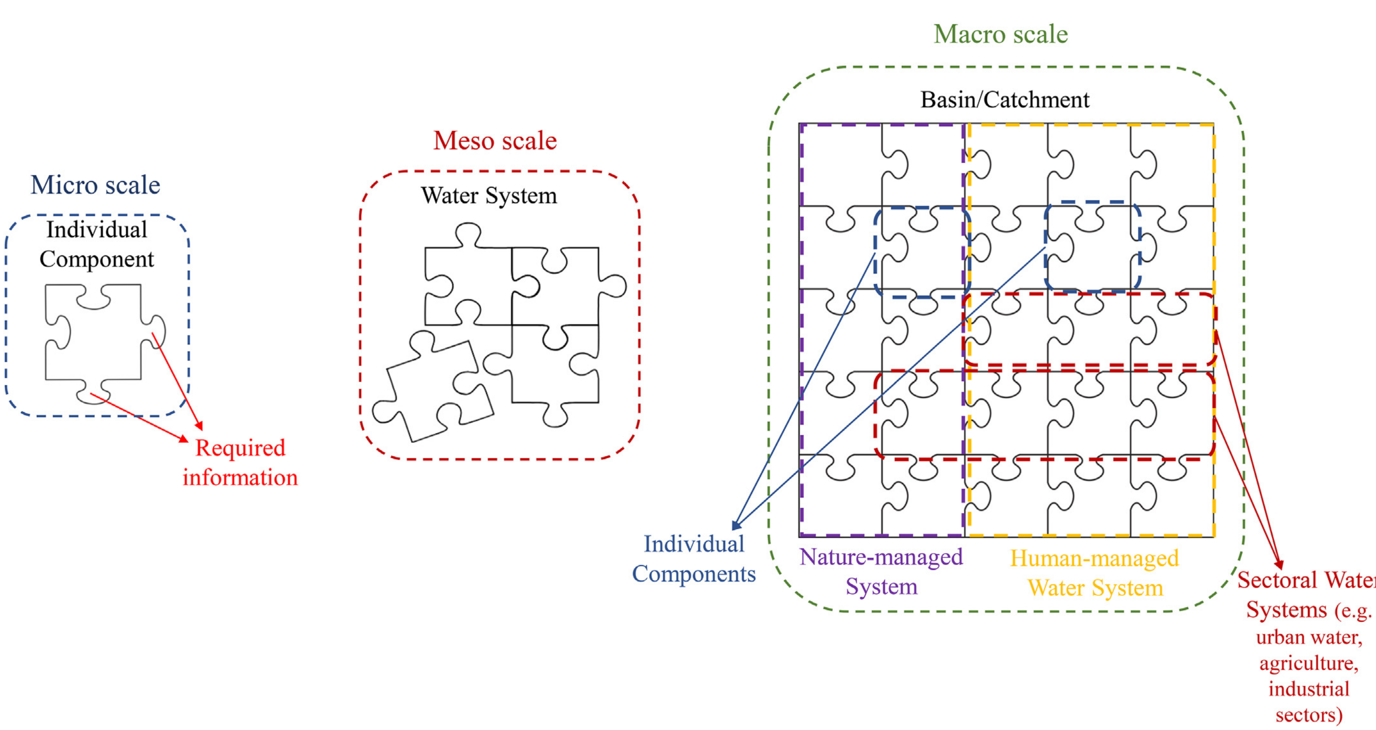

Fig. 1. Puzzle diagram of different assessment scales.

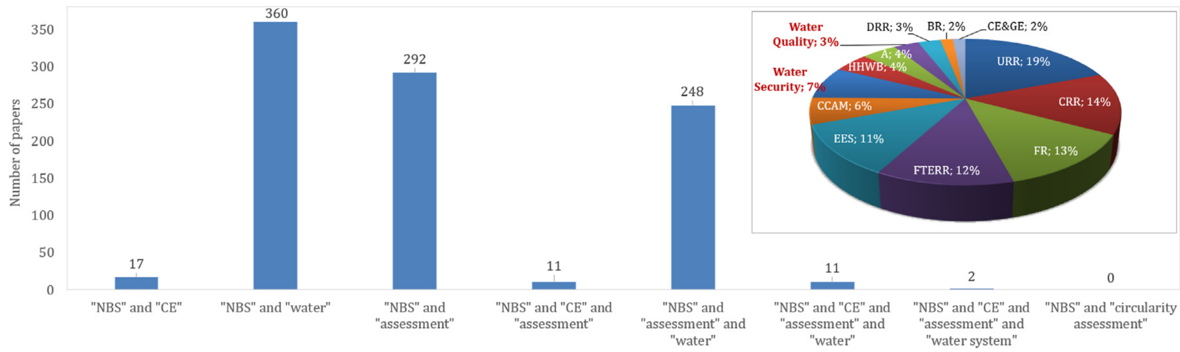

Fig. 2. The focus of the research in the area of CE, NBS, assessment and water; and distribution of the main societal challenges among the reviewed studies (pie diagram). Acronyms: URR – Urban Regeneration and Resilience, CRR – Coastal Resilience and Restoration, FR – Flood Risk, FTERR – Freshwater and Terrestrial Ecosystem restoration and Resilience, EES – Enhancement of Ecosystem Services, CCAM – Climate Change Adaptation and Mitigation, HHWB – Human Health and Well-being, A – Agriculture, DRR – Disaster Risk Reduction, BR – Bioremediation, CE&GE – Circular and Green Economy.

Table 1
Categorization of the identified water-related studies based on their context.

	Water Security	Water Quality/*Water Quality+	Water Quality & Flood Risk/*+	Water Quality & Reuse/*+	Water Quantity & Quality	Water Supply Regulation
Social aspects		Castonguay et al. (2018)				
Economic aspects			Reynaud et al. (2017)		Reddy et al. (2015)	Vogl et al. (2017)
Scenario analysis	Boelee et al. (2017)		Zhang et al. (2019)			
Environmental sustainability		Garfí et al. (2017)				
Effectiveness		*Hernández-Crespo et al. (2017); Geronimo et al. (2019); Haddis et al., 2020; *Krzeminski et al. (2019)	*Chow et al. (2014); *Liquete et al. (2016); *Li et al. (2017a); Jurczak et al. (2018); *Masseroni et al. (2018); *Radinja et al. (2019)	*Licciardello et al. (2018)		
Optimization		Andrés-Doménech et al. (2018); Cáceres et al. (2018)	Moezzibadi et al. (2019)			
Groundwater accounting					Bricker et al. (2017)	

*+ indicates the papers that use multiple criteria in their analysis.

the adoption of rainwater tanks, and that water restrictions have had a major impact on the uptake. Additionally, the study of Reynaud et al. (2017) used a contingent valuation approach to estimate the willingness to pay (WTP) of households to different multipurpose infrastructures (conventional or green) for managing flood and water pollution. They concluded that there is an excessive willingness to pay for green infrastructure in comparison to conventional systems. The WTP in this study was influenced by people's income and their visits from Gorla Maggiore Water Park.

The optimization of certain parameters of stormwater and wastewater treatment systems (including NBS) to improve their design has been performed in three studies. Andres-Domenech et al. (2018) considered runoff characterization and volume and rainfall depth to improve the design of source control systems. Cáceres et al. (2018) developed a statistical tool to select the most adequate withdrawal depth for optimizing the wastewater treatment processes. Moezzibadi et al. (2019) evaluated the filtering performance – related to suspended solids loads – of constructed wetlands to improve their design.

The technical evaluation of the NBS has been performed in the studies of Hernandes-Crespo et al. (2017), Geronimo et al. (2019), Haddis et al., 2020, Krzeminski et al. (2019), Jurczak et al. (2018), Masseroni et al. (2018), and Licciardello et al. (2018). These studies consider water quality parameters and/or water retention capacity as the main aspects of their evaluation, including in some cases costs (Licciardello et al., 2018; Masseroni et al., 2018; Krzeminski et al., 2019) and biodiversity (Hernández-Crespo et al., 2017, Hernandes-Crespo et al., 2017). However, the scope of these studies is narrow – focusing on the technical performance or the design optimization – and they do not assess holistically the proposed solutions (e.g. potential co-benefits are excluded from the evaluation).

The studies of Chow et al. (2014), Liquete et al. (2016), Boelee et al. (2017), Bricker et al. (2017), Garfí et al. (2017), Li et al. (2017), Radinja et al. (2019) and Zhang et al. (2019) were identified as more relevant for the purpose of this work. They either deploy a more holistic assessment or consider aspects relevant to assess water circularity (sub-section 3.1). The studies considering economic aspects of NBS are analyzed in sub-section 3.2.

3.1. Methodologies and indicators assessing NBS for water circularity

The eight studies identified in the previous section are categorized based on the methods that they deploy for assessment – i.e. water balance, LCA, modeling and combination of different tools. The indicators developed/used in the reviewed studies are presented in Table 2 and they are compared to the different water management aspects identified in EKLIPSE framework (Table 2) developed by Raymond et al. (2017). The sufficiency of the different

Table 2
List of indicators used in the reviewed studies and their comparison to the water management criteria identified in the EKLIPSE framework.

		Water management criteria identified in the EKLIPSE framework															
Study	**Applied indicators**	Runoff reduction	Flood peak reduction	Reduction of load from runoff into sewerage systems	Reduction of risk of flooding from flush-floods	Reduction of costs related to loads into sewerage systems	Reduction of risk of flooding from rivers	Infiltration increase	Water storage increase	Water retention capacity enhancement	Risk reduction of damages from drought	Evapotranspiration increase	Risk reduction from urban heat island effect	Human health & well-being improvement	Water quality improvement and pollutants reduction	Biodiversity increase	Carbon storage capacity increase
Bricker et al., 2017	–							✓	✓								
Garfi et al., 2017	Metal depletion																
	Climate change																
	Terrestrial acidification																
	Marine eutrophication																
	Fossil depletion																
	Ozone depletion																
	Freshwater eutrophication																
Boelee et al., 2017	Mean species abundance						✓		✓						✓	✓	
Zhang et al., 2019	Flow reduction		✓														
	TSS, TP & TN load reduction														✓		
Liquete et al., 2016	Peak flow reduction		✓														
	Reduction of flooding downstream						✓										
	No. of visitors/users													✓			
	Frequency of visits													✓			
	Load reduction of dissolved organic carbon														✓		
	Load reduction of nitrogen														✓		
	Biodiversity															✓	
	Landscape diversity (Shannon's diversity index)															✓	
	Value of wood																
	Total construction costs																
	Total maintenance costs																

aspects considered in the studies to holistically assess NBS for water management and their potential contribution to circularity of water systems are investigated.

A water balance method is used in the study of Bricker et al. (2017) that presents a vision exercise to study the impacts of different interventions on groundwater balance at city level. Water balance can be used to assess the increase in infiltration and water storage, which is one of the main considerations when assessing NBS for water management according to EKLIPSE framework (Table 2). However, this study does not apply a holistic assessment methodology as only one aspect (i.e. groundwater storage) is evaluated. The study of Bricker et al. (2017) is the only one identified – among the studies analyzed in this section – that deploys a semi-quantitative water balance approach to investigate how specific interventions impact water systems, considering both anthropogenic and natural water flows. The consideration of mass balances from a circularity perspective (i.e. to close the water loops) are of major importance and similar studies are reviewed in Section 5.

LCA is used by Garfí et al. (2017) to assess the environmental impacts of a conventional infrastructure and two NBS technologies as alternatives for wastewater treatment in small communities. The indicators used for the assessment are presented in Table 2. One of the strongest advantages of LCA is the fact that it is a standardized and well-established method for the evaluation of the environmental impacts for the entire life cycle of a product, process, or service (ISO, 1997; Hertwich, 2005) and it is widely used to compare the environmental sustainability of different water technologies (e.g. Pan et al., 2011; Manso et al., 2018; Guertin et al., 2019; Pan et al., 2019; Oquendo-Di Cosola et al., 2020). However, when it comes to the assessment of NBS following the EKLIPSE framework (Table 2), none of the highlighted considerations can be addressed using LCA. There are three main constraints. Firstly, the high level of abstraction associated with the LCA results leads to potential rather than actual impacts, thus there is a generalization of the LCA impacts that do not refer to specific cases (Pizzol et al., 2015; Bai et al., 2018). NBS on the other hand should be "locally

	Load reduction of dissolved organic carbon														✓		
	Load reduction of nitrogen														✓		
	Biodiversity															✓	
	Landscape diversity (Shannon's diversity index)															(✓)	
	Value of wood																
	Total construction costs																
	Total maintenance costs																
Chow et al., 2014	Increased water reuse							✓	✓								
	Reduced floodplain						✓										
	Reduced runoff volume	✓															
	Improved water quality														✓		
	Reduced energy use																
	Reduced carbon													(✓)			
	Improved air quality													(✓)			
	Creation of new habitats															✓	
	Capital expenditure																
	Operational expenditure																
	Land-take costs																
	Reduced water bills																
	Increased house price																
	Reduced treatment cost																
	Electricity savings																
	Natural gas savings																
	Avoided cost of CO_2																
	Avoided cost of					(✓)											
	Amenity value													(✓)			
	Runoff volume reduction	✓															
	Peak discharge reduction									✓							
Li et al., 2017a	Flood peak retardation		✓														
	Pollution reduction (SS, COD, TN, TP)														✓		
	Civil construction costs																
	Maintenance charge																
	Utilization of rainwater								(✓)								
	Landscape value													✓			
	Ecological function												✓			✓	
Radinja et al., 2019	Reduction of combined sewer overflow (CSO)			(✓)													
	Amenity													✓			
	Biodiversity															✓	
	Feasibility																

✓: directly addressed; (✓): indirectly addressed

adapted", mandating environmental evaluations to be tailored to local environments. Secondly, LCA focuses more on the environmental costs (e.g. biodiversity loss) rather than the environmental gains (e.g. carbon sequestration) (Rugani et al., 2019). Thirdly, the consideration of feedback loops between processes across the anthropogenic and the natural environment are generally neglected in LCA (Weidema et al., 2018). This means that the effects on ecosystems driven by changes occurred in the anthropogenic systems are not considered, which further neglects the consideration of the effects of those changes back to the anthropogenic systems, underestimating the actual load of the life cycle impacts. Therefore, LCA is very useful for comparing alternative solutions but not for assessing actual environmental impacts and the interconnections between natural and human-managed systems.

Modeling projected scenarios are deployed in the studies of Boelee et al. (2017) and Zhang et al. (2019) to assess the performance of NBS under future climatic conditions. Boelee et al. (2017) developed a model to evaluate the performance of alternative scenarios to address simultaneously water management challenges (i.e. water shortages, pollution, deterioration of aquatic ecosystems) and biodiversity loss, under future projections in terms of urbanization, climate change and increasing demands for food production and hydropower. Zhang et al. (2019) used a model to examine the implications of climate change on future rainfall and evaluate the reliability of Water Sensitive Urban Design (WSUD) stormwater infrastructure in pollution reduction, flow frequency mitigation and reliability as an alternative water supply. Both studies are able to address some of the key points identified in the EKLIPSE framework (Table 2), under dynamic time- and spatial variations of the system. However, scenario analysis modeling is more relevant for investigating options for the future rather than the actual state of the system. Therefore, process understanding modeling is considered more appropriate for water circularity assessment and relevant studies are reviewed in Section 5.

More holistic approaches – resulted from combination of different tools and methods – were adopted in the studies of Liquete et al. (2016), Chow et al. (2014), Li et al. (2017a) and Radinja et al. (2019). Liquete et al. (2016) deployed a runoff model, water quality measurements, biological samplings, surveys, satellite images, ArcGIS and Fragstat software to assess multiple environmental, economic and social benefits of a set of CWs surrounded by a park to treat the excess flow of mixed sewage and rainwater during heavy rain events. Chow et al. (2014) developed a systemic multi-criteria decision support framework to evaluate the design of grey/green drainage infrastructure based on quantitative measures (i.e. indicators) covering energy, environment, water quantity and quality criteria, and monetary costs and benefits. Li et al. (2017) developed a benefit evaluation system for low-impact development of urban stormwater management measures, based on Analytic Hierarchy Process and urban stormwater model, including environmental (i.e. water quantity and quality), economic (i.e. civil construction and maintenance costs), and social (i.e. rainwater reuse, landscape value, and ecological function) benefits. Radinja et al. (2019) used a framework based on hydrology-hydraulic modeling and Multi-criteria Analysis (MCA) to evaluate co-benefits (i.e. combined sewer overflow reduction, CAPEX, OPEX, amenity, biodiversity, and ownership) of Sustainable Drainage Systems (SuDS), resulting in favorable scenarios for stormwater control measures. Although these studies consider several water management criteria for their evaluations (see Table 2), water quantity – which is important to investigate the extent of closing the water loops and the enhancement of the natural water cycle – is not well addressed.

Although there is a variety of developed/used indicators among the different studies, they are not able to cover all the different aspects of water management, i.e. the water management criteria identified in the EKLIPSE framework as shown in Table 2. Most of the studies focus on the main societal challenge to be addressed by NBS (i.e. water quality issues and flood risk), underestimating the enhancement of the natural environment (e.g. the natural water cycle is not considered) and many co-benefits (e.g. carbon storage capacity is not evaluated by any of the reviewed studies). Another issue is the scale of assessment. NBS are assessed as individual components (i.e. detached from the complete water system), neglecting the whole supply chain (i.e. upstream and downstream processes and flows), the systems thinking, the interconnections between these processes and feedbacks and therefore, the impacts to the natural and urban water system. A fragmented approach of understanding isolated parts of the water system increases the risk of implementing solutions that may be inefficient due to overlooking many dynamic and aggregated effects that emerge at larger scales (i.e. due to the complex interactions of the different components with their surroundings) (Thorslund et al., 2017). This may result in overestimation of the effectiveness of NBS (if assessed at small scales) but also insufficiency to assess circularity of water systems.

3.2. Methodologies and indicators economically assessing NBS

While NBS appear to be capable of addressing water management challenges and enhancing ecosystem services, the debate on how to valuate nature, its attributes and services in monetary terms is still open, as they are not goods directly traded in the markets (Bockarjova and Botzen, 2017). This section focuses on commonly used methods for economic evaluation of NBS – i.e. Life Cycle Cost Analysis (LCCA), Cost-Benefit Analysis (CBA), Cost Effectiveness Analysis (CEA) and Natural Capital Accounting.

LCCA is an economic assessment that accounts for all the relevant costs throughout the entire life cycle of a product, service or process, enabling the comparison between different alternatives (Bhoye et al., 2016). Life Cycle Costing (LCC) considers investment, implementation, operation, maintenance and end-of life (e.g. disposal and residual value) costs, benefits cash flows. According to the Directive/24/EU, 2014 Directive 2014/24/EU (2014) it can also include costs related to environmental externalities (e.g. greenhouse gas emissions). The results of LCCA are expressed using an economic measure, e.g. net present value (NPV), benefit cost ratio (BCR), payback period (PP) and (annual) LCC. Economic assessment based on LCCA has been performed by many researchers to evaluate the economic impacts, e.g. of rainwater harvesting systems (e.g. Roebuck et al., 2011; Amos et al., 2018a), green roofs (e.g. Ziogou et al., 2018 – including the monetization of the avoided environmental deterioration), waste management alternatives (Lee et al., 2020), urban vegetation (Sicard et al., 2018). One of the main limitations of LCCA is that environmental and economic benefits that are not easy to be monetized (e.g. reduction in urban flooding, biodiversity increase, increased property value due to aesthetic improvement, etc.) tend to get lost, potentially prejudicing the decisions (Amos et al., 2018b; Ziogou et al., 2018).

The same limitation stands for CBA, as it calculates the total costs related to a project, monetizes the obtained (environmental and social) benefits and compares the results to identify the most efficient alternative (Hansson, 2007). However, CBA is considered as one of “the most widely applied tools for economic analysis” (Balanay and Halog, 2019) and it is used to economically evaluate NBS. For example, Feng and Hewage (2018) used CBA to assess the payback period of green roofs in different markets, considering life cycle costs, public and individual benefits. CBA considering public and private costs and benefits was also deployed by Reddy et al. (2015) to assess water shortages risk of alternative scenarios (including NBS).

CEA is proposed by Boerema et al. (2018) as an alternative economic evaluation method for environmental management, in order to overcome the limitation of CBA of expressing benefits into explicit monetary terms. Therefore, CEA identifies the most cost-effective strategy by comparing the investment costs to achieve a specific goal (or measure of effectiveness, expressed in any unit), e.g. biodiversity conservation (Helm and Hepburn, 2012), greenhouse gas emission reduction (MacLeod et al., 2010), water provision improvement (Yang, 2011). However, in CEA each measure of effectiveness is treated separately (due to the different units), thus their integration for assessing whether the total benefits exceed the total costs is not always possible.

Natural Capital Accounting is defined by Philips (2017) as “a series of interconnected physical and monetary accounts that provide a structured set of information related to the stocks of natural capital and flows of services supplied by them”. Monetary accounts are referred to annual monetization of selected services, resulting in an “overall valuation of the natural asset’s ability to generate future flows of services”. For example, Vogl et al. (2017) discussed the importance of watersheds’ natural capital valuation increase (i.e. the cost of maintaining the natural capital in healthy watersheds that contributes to public good and private values) of the uptake and impact of investments in watershed services. Although non-market goods – such as many ecosystem services – do not have direct exchange values (i.e. the supply value equals the use value), Natural Capital Accounting is based on the concept of exchange value. Thus, the monetary valuation of non-market goods is based on estimations as “if a market existed” (Philips, 2017). Therefore, such estimated exchange values increase the uncertainty of this method.

Table 3 presents indicators that have been used in literature. While some established economic values (e.g. NPV, BCR, PP, CAPEX, OPEX) exist and are used in the studies, most indicators are case-specific. Even within the LCCA method, the LCC indicator is interpreted differently among the studies, e.g. Lee et al. (2020) consider the revenue from products, while Ziogou et al. (2018) consider the environmental cost of emissions. Especially for CBA and CAE, the costs and benefits/effectiveness measures strongly depend on the scope of each study and often the economic assessment is based on economic analysis (e.g. economic and environmental or ecologic balance) of the system rather than development of indicators.

4. NBS for water implementation barriers and potential risks

In recent years, the concepts of circular economy and NBS have been promoted and encouraged at EU level (EC, 2015a; EC, 2015b). However, in some cases current policies and regulations, designed for a linear economy paradigm, are likely to hinder, rather than encourage, their implementation. As evidenced in the study of Stewart et al. (2016), regulations can act as barriers that prevent the adoption of innovative sustainable approaches, e.g. giving unclear messages, being complex and rapidly changing, and lacking space of manoeuvre for innovation.

The European Commission put forward a proposal for a regulation setting EU standards for reclaimed water in May 2018. This proposal is based on a JRC report (Alcalde-Sanz and Gawlik, 2017) and is related to the 2015 circular economy Action Plan (EC, 2015a), the seventh environment action programme (Decision No. 1386/2013/EU, 2013), and, globally, to the UN's sustainable development goals. The aim is to reduce water stress by promoting the use of treated wastewater in agriculture and encourage the free circulation of products irrigated with reclaimed water (Alcalde-Sanz and Gawlik, 2017) proposing a regulation that sets minimum quality standards. At present, only few EU member states have set requirements for wastewater reuse. The proposed requirements include microbiological (E. coli, Legionella spp. and intestinal nematodes) and physico-chemical parameters (biochemical oxygen demand (BOD), suspended solids (SS) and turbidity), while micropollutants are not mentioned. The proposed legislation would consider irrigation as the only application, therefore disregarding possible utilizations in NBS. Quality standards would be set according to the fit-for purpose approach. Following the World Health Organization guidelines, a risk management plan based on the multiple-barrier approach and the hazard analysis and critical control points system, analogue to the Water Safety Plan, would also be established (Alcalde-Sanz and Gawlik, 2017). However, this proposal considers solely reclaimed wastewater, that is, water complying with the quality standards detailed in the Urban Waste Water Treatment Directive (UWWTD) (Council Directive 91/271/EEC, 1991) excluding other possible sources (e.g. harvested rainwater, greywater, etc). It should be noted that the forthcoming EU standards, focused only on reuse of treated wastewater, propose improved end-of-pipe solutions that represent only a mitigation, and not a solution, of the water stress problem. Advanced concepts that would tackle reuse at decentralized level and facilitate resource recovery by source separation and source control rather than treatment need to come into force. The UWWTD states that in case the investments to build a WWTP would not produce environmental benefit or would entail excessive costs, "individual systems or other appropriate systems (IAS) which achieve the same level of environmental protection shall be used". This creates a sort of a free zone and, as a result of this vagueness in legislation, a recent report highlights that IAS represent a significant source of pollution in EU, making the refit of the UWWTD an impellent need (EC, 2019). In this case, the concept of IAS could be an opportunity, at EU level, for the implementation of NBS in the form of small-scale, decentralized, water treatment, proven that the systems comply with the required quality standards. An example of NBS that could be a suitable option for small-scale treatment is treatment wetlands. Although treatment wetlands cover larger surfaces compared with other technologies, they do not need external energy inputs to be operated if the location allows the avoidance of pumps (Dotro et al., 2017; Langergraber et al., 2019b). This results in energy savings and lower operation and maintenance costs, which may be proven crucial in areas with limited financial resources (Langergraber et al., 2019b).

When reviewing scientific literature on NBS, little or no relevant data on the risk posed by micropollutants were found. Micropollutants have been targeted at the Priority Substances Directive (Directive/39/EU, 2013, Directive, 2013/39/EU, 2013), which encompasses the Watch list system, introduced with a Commission Implementing Decision in a 2015 (EC, 2015c), then updated through Decision 2018/480 (EC, 2018). The Watch list is a list of micropollutants that may pose a significant risk to or via the aquatic environment but for which data are still insufficient to support their prioritization and must therefore be monitored Europe-wide by Member States (EC, 2018). Since the monitoring is to be carried out in freshwater only, there is a missing link between the occurrence of such pollutants in wastewater and the risk associated to their presence when reclaimed wastewater is reused. On the one hand, this legislative gap will delay the adoption of a common regulatory framework in the EU for water reuse/greywater/harvested rainwater or hinder its potential; in particular, this could prevent from implementing new technologies, practises and solutions – above all NBS. On the other hand, the lack of regulations on micropollutants might bring to underestimate the risk posed by such compounds.

Decentralized systems could provide an answer to this problem as they enable better source control and consequently bigger potential and safer resource recovery and reuse. Moreover, the adoption of certain NBS may even improve the removal of micropollutants compared to conventional WWTPs, as in the case of the treatment wetlands. In treatment wetlands the removal efficiencies of many compounds are reported higher than 90% thanks to the coexistence of anaerobic, anoxic and aerobic zones within surface flow, as well as the concurrence of different mechanisms, such as biodegradation, sorption, plant uptake and, in certain cases, photodegradation (Langergraber et al., 2019b; Li et al., 2014; Verlicchi and Zambello, 2014). In parallel to setting environmental quality standards, the legislation could also focus on preventing micropollutants from entering the water systems, for instance improving source control measures.

The situation outside the EU is different. In Switzerland, the Water Protection Ordinance established that WWTPs following certain criteria should be upgraded with advanced treatment by 2035 in order to ensure the removal of at least 80% of micropollutants loads, tracking WWTPs efficiency by means of periodical monitoring campaigns (Swiss Confederation, 2016). Other countries outside Europe, such as California (State of California, 2013), Australia (Australian Government, 2018) and Singapore (PUB, 2018) have implemented cutting edge policies on water reuse, including specific regulations on greywater and direct or non-direct potable reuse. In some cases, micropollutants are also monitored. The legislation of these countries might pave the way for implementation at EU level.

5. Methodologies and indicators assessing water systems

A lack of a holistic assessment methodology evaluating NBS as part of water systems and adequately addressing issues of water

Table 3
Economic indicators used in the reviewed studies (Chen et al., 2019).

Indicator	Description	Study	Methodology	Categorization of monetized values		
				Direct economic	Environmental externalities	Natural capital
Net present value [NPV]	The sum of the annual net cash flows (i.e. the difference between cash outflow and inflow reduced by an appropriate discount rate) over the project's life time	Amos et al., 2018a	LCCA	✓		
Benefit-cost ratio [BCR]	The sum of discounted costs divided by the sum of discounted benefits (e.g. water savings) as they occur over the project's life time			✓		
Payback period [PP]	The time required to recover an investment or loan			✓		
Life Cycle Cost [LCC]	Changes in economic welfare due to the avoided environmental deterioration, i.e. consideration of construction and operational costs and environmental costs of the emissions	Ziogou et al., 2018		✓	✓	
Life Cycle Cost [LCC]	Use of the present value method, including infrastructure, operation and maintenance, collection and transportation costs, tipping fee, and revenues from beneficial products (i.e., electricity, heat, and digestate or compost)	Lee et al., 2020		✓		
Annual life cycle cost per amount of air pollutant's removal	Includes installation, planting, operation and maintenance costs, normalized per kg of O_3 removal	Sicard et al., 2018		✓	✓	
Capital expenditure [CAPEX]	Money spent to acquire, upgrade, and maintain physical assets	Radinja et al., 2019	–	✓		
Operating expenses [OPEX]	Ongoing costs for running a product, business, or system			✓		
Gross value added [GVA]	The value of goods minus the value of intermediate consumption required for the production; expressed as money per amount (e.g. euro/tonne)	Chen et al., 2019	Input/ Output	✓		
Environmental flows						✓
Farm operations savings	Monetary estimates in 2012 US$, present value (PV)	Reddy et al., 2015	CBA	✓		
Soil conservation						✓
Avoided costs from reduced pesticide use and nutrient runoff				✓		
Lower utility costs				✓		
Decreased operating costs for treatment plants				✓		
Lower water treatment costs				✓		
Direct payments				✓		
Cost	The cost per hectare of implementing a measure, per	MacLeod et al., 2010	CEA/ marginal abatement cost curve (MACC)	✓		
Cost-effectiveness	The cost of reducing GHG emissions			✓		
Abatement potential (AP)	The total amount of GHG emissions that are reduced per year				✓	
Abatement rate (AR)	The rate at which a measure can reduce the GHG				✓	

quantity, as well as the regeneration of natural capital was identified in Section 3. Therefore, in this section, the research is expanded on currently available methodologies that have been applied to assess the effects of complete water systems on the physical and environmental performance in order to investigate their potential of measuring water circularity.

This section is organized in three sub-sections according to the type of approaches deployed in the reviewed studies – i.e. studies assessing water systems and environmental compartments using MFA-based approaches (Chapter 5.1), Consumption-based approaches (Chapter 5.2), and Modelling approaches (Chapter 5.3). The links between these methods, NBS and circularity of water systems are identified.

5.1. MFA-based approaches

MFA is one of the most widely used methods to evaluate circularity (Elia et al., 2017; Pauliuk, 2018; Moraga et al., 2019). It enables a systemic quantification of materials flows and stocks, helping towards the management of resource use and the development of closed-loop systems. Based on the principle of mass conservation in a well-defined system, MFA focuses on loadings (instead of concentrations) and provides an overview of the total system – enabling the integration of NBS to the complete water system – by linking and examining the relationship between the human-managed system and the natural environment (Hendriks et al., 2000). Focusing on water systems, many MFA-based assessment methodologies have been developed to evaluate the metabolic and sustainability performance of a specific area or system, including the development of indicators used for the assessment (Table 4).

In 2011, Kenway and his colleagues developed the concept of Urban Water Metabolism (UWM) that provides a systematic mass-balance framework to quantify all anthropogenic and natural water flows into and out of the urban environment, resulting to quantitative performance indicators. Since then, UWM has been expanded to include energy and nutrient flows (Farooqui et al., 2016; Renouf et al., 2017, 2018). Therefore, UWM can be used to assess the transition from linear to more circular metabolism of complete water systems (including the integration of NBS) that create a self-regulating sustainable relationship with the biosphere.

Similarly, Verger et al. (2018) used Territorial Metabolism (TM) to analyse the metabolism of a peri-urban area through its nitrogen, phosphorus, carbon and water flows. The main contribution of this study is the inclusion of the occurring natural processes of nutrients as part of the analysis. From a circularity perspective, the integration of nutrients in the nature-managed system is important as it is one of the aspects indicating the achievement of regeneration of natural capital principle.

Agudelo-Vera et al. (2012) developed a methodology (i.e. Urban Harvest Approach – UHA) – based on the concept of urban metabolism – evaluating and quantifying the multiple potentials of different primary and secondary (already used) resources that can be utilized within a water system (from building to city scale) in order to become self-sufficient. In 2015, Leusbrock and his colleagues expand the application of UHA to consider energy flows. UHA is close to the concept of circularity as the three deployed strategies (i.e. minimizing demand, minimizing outputs, and multi-sourcing) are similar to the circularity concepts of reduce, reuse, recycle.

Wielemaker et al. (2018) – based on the UHA – developed the Harvest to Harvest Approach (H2HA) to assess the match between the supply by new sanitation systems and the demand from urban agriculture for nitrogen, phosphorus and organic matter, in terms of quantity and quality, to foster a circular metabolism and to optimize interconnecting systems.

Numerous indicators are resulted from MFA-based studies that are able to quantitative and qualitatively measure the metabolic performance and self-sufficiency of water systems. The developed water-, nutrient- and energy-related indicators cover both anthropogenic and natural flows, but extended natural nutrient flows are disregarded (i.e. the consideration of natural nutrient cycles). This way, the extent of closing the water-loops and the regeneration of natural capital in terms of water is achievable. However, emissions and natural nutrient processes and flows and other ecosystem services are disregarded in these indicators.

MFA-based assessment approaches can form the basis for a water circularity assessment methodology of water systems as they have the capability of including both the human-managed and the nature-managed system (i.e. particularly useful when NBS are part of the analyzed system), and simultaneously consider flows, stocks and loadings of the water, materials and energy pathways. However, other approaches need to be incorporated in order to consider additional environmental impacts, additional environmental benefits and ecosystem services, as well as economic aspects in order to fully assess the ability of regenerating the natural capital and designing out waste externalities.

5.2. Consumption-based approaches

Consumption-based approaches quantify the resources (including water) required to produce goods and services "consumed" by society and estimate the associated embodied life-cycle environmental impacts, whether those impacts occur inside or outside the defined boundary of the system (Baynes and Wiedmann, 2012). LCA is one of the most representative methods of this category and one of the most widely-applied to assess the environmental impacts of water systems (see Section 3), as well as to assess systems from a CE perspective (Elia et al., 2017; Pauliuk, 2018; Baleta et al., 2019; Moraga et al., 2019). The indicators used in LCA are well-known and they are not presented in this study.

Different practices or technologies to recover raw materials (e.g. phosphorus) and energy, as well as to reuse water from wastewaters have been investigated, by assessing or evaluating technical (e.g. Zhou et al., 2017), environmental (e.g. Pintilie et al., 2016; Amann et al., 2018; Dominguez et al., 2018; Hao et al., 2019; Pradel and Aissani, 2019; Sylwan et al., 2019) and economic (e.g. Laitinen et al., 2017) aspects. Additionally, Buonocore et al. (2018) used LCA to compare the environmental impacts of linear, partially circular and circular scenarios of energy recovery and water reuse in WWTP. Leong et al. (2019) compared the environmental (LCA) and economic (LCC) impacts of centralized and decentralized options for non-potable water uses at a domestic and commercial building. Similarly, Zanni et al. (2019) compared the environmental impacts and other technical aspects (e.g. system's complexity, tanks, pumping system etc.) of centralized and decentralized water systems at single dwelling and apartment buildings. Assessment of environmental impacts of the entire urban water system was investigated in the studies of Lemos et al. (2013) (considering environmental impacts), of Lane et al. (2015) (comparing conventional to diversified urban water infrastructure) and of Xue et al. (2019) (considering both environmental and monetary costs). However, the above-mentioned studies – trying to address circularity issues by focusing mainly on the environmental impacts – fall short of keeping water as the protagonist in terms of circularity (i.e. there is no proof of closing the water loops). Thus, they are not able to make an overall evaluation of the CE benefits (i.e. elimination of waste and regeneration of natural capital). It is also reported that the application of LCA in complex multifunctional circular water value chains with multiple outputs (i.e. water, energy, materials) and water uses is still challenging (Reap et al., 2008; Bobba et al., 2018). Therefore, although these studies have assessed aspects of circularity (i.e. environmental impacts, economic efficiency, etc.) of different components of water systems, they cannot holistically assess the circularity of water systems.

However, LCA if combined with other tools/methods can provide useful information in terms of environmental sustainability as part of a water circularity assessment. For example, the CE principle of design out waste externalities, can be assessed with LCA especially if combined with LCC or other economic assessments to incorporate the economic aspects of the system. Additionally, LCA has been integrated to metabolic approaches (in Goldstein et al., 2013; García-Guaita et al., 2018; Sohn et al., 2018) in order to assess the sustainability of cities' metabolisms, considering the anthropogenic flows of different materials. However, to the best of our knowledge, such methodological integration has not been performed to water-related metabolisms.

5.3. Modelling approaches

MFA-based and Consumption-based approaches can be used to evaluate the circular metabolism (i.e. in terms of water in both human- and nature-managed systems, of energy and of nutrients mainly in human-managed systems) in a stationary (snap-shot) mode and the potential environmental impacts of the entire life cycle to the environment, respectively. However, the actual environmental impacts, the environmental benefits and in general the degradation or regeneration of the natural environment (e.g.

Table 4
Developed indicators in the MFA-based studies.

Study	Indicator	Description/Equation	Components						
			Anthropogenic system				Natural system		
			Water	*Nutrients*	*Energy*	*Emissions*	*Water*	*Nutrients*	*Other ES*
Kenway et al., 2011	Intensity of water use	[Total water use]/[Area]	✓						
	Overall balance of inputs and outputs	[Total inputs]/[Total outputs]	✓				✓		
	Supply centralization	[Centralized supply]/[Total water use]	✓						
	Rainfall harvesting	[Decentralized sources]/[Rainfall]	✓				✓		
	Centralized supply replaceability	[Rainwater or wastewater or stormwater]/[Centralised water supply]	✓						
	Total use replaceability	[Rainwater or wastewater or stormwater]/[Total water use]	✓						
	Replaceability of total use with wastewater and stormwater	[Wastewater + Stormwater flows]/[Total water use]	✓						
	Anthropogenic turnover rate	[Anthropogenic system inputs]/[Stored water]	✓				✓		
	Rainfall turnover rate	[Natural system inputs]/[Stored water]					✓		
	Total turnover rate	[Total inputs]/[Stored water]	✓				✓		
Farooqui et al., 2016	Internal harvesting ratio	[Internally harvested freshwater volume]/[Total water volume supplied to meet demand]	✓						
	Internal recycling ratio	[Internally water recycled volume]/[Total water volume supplied to meet demand]	✓						
	Water extracted	[Extracted water volume from external sources]/[Population of the urban area]	✓				✓		
	Energy used	[Total water-related energy use]/[Population of the urban area]			✓				
	Stormwater runoff ratio	[Post-development stormwater runoff]/[Pre-development stormwater runoff]	✓				✓		
	Total stream discharge ratio	[Post-development discharge]/[Pre-development discharge]	✓				✓		
	Infiltration ratio	[Post-development groundwater infiltration]/[Pre-development groundwater infiltration]					✓		
	Evapotranspiration ratio	[Post-development evapotranspiration]/[Pre-development evapotranspiration]					✓		
Renouf et al., 2017	Urban water efficiency per person	Total use of 'environmental' water per person	✓				✓		
	Urban water efficiency per unit of functionality	Total use of 'environmental' water per unit of urban function	✓				✓		
	Water – related energy efficiency per person	Total energy use for the water system per person			✓				
	Water – related energy efficiency per unit of functionality	Total energy use for the water system per unit of functionality			✓				
	Nutrient recovery from urban water	Proportion of the nutrient load in wastewater that is beneficially utilized		✓					
	Water supply internalization	Proportion of total water demand met by internally harvested/recycled water	✓						
	Water use within safe operating space	Rate of surface and groundwater drawn from supplying catchments relative to the sustainable urban water allocation					✓		
	Water pollutant load within safe operating space	Point-source and diffuse nutrient loads discharged to surface and groundwater relative to sustainable discharge rates						✓	
	Hydrological performance	Post-urbanized hydrological flows/fluxes relative to pre-urbanized flows/fluxes	✓				✓		
	Supporting diverse functions	Water needed to maintain desired functions relative to water allocated for the functions	✓				✓		
Verger et al., 2018	Efficiency for nitrogen, phosphorus and carbon flows	[Local consumption]/[Total production] **and** [Local consumption]/[Total consumption]		✓				✓	
	Self-sufficiency capacity for nitrogen, phosphorus and carbon	[Production]/[Consumption]		✓				✓	
Agudelo-Vera et al., 2012	Demand minimization index (DMI)	[Baseline demand – Minimized demand]/[Baseline demand]	✓						
	Waste output index (WOI)	– [Exported waste]/[Minimized demand]	✓						
	Self-sustainable index (SSI)	[Harvested resources – Exported resources]/[Minimized demand]	✓						
	Resource export index (REI)	[Exported resources]/[Minimized demand]	✓						
Leusbrock et al., 2015	Energy recovery index (ERI)	– [Recovered and reused energy]/[Minimized demand]			✓				
	Self-sufficiency index (SSI) for thermal, electric and total energy	[Thermal/electric/total energy produced]/[Minimized thermal/electric/total energy demand]			✓				
	Resource export index (RXI)	[Exported energy]/[Minimized demand]			✓				
Wielemaker et al., 2018	Self-sufficiency index (SSI) for nutrients	[Resource reused]/[Minimized demand]		✓					

considering the nutrients natural cycles) is not well-addressed. Therefore, modelling approaches are reviewed in this sub-section in order to bridge this gap by studying different material cycles simultaneously, in addition to interpreting them individually, while gaining insight into the magnitude of the associated flows. The purpose is to present the state-of-the-art of existing models (as tools that can potentially be incorporated in assessment methodologies) and existing methodologies (deploying modelling approaches) that can be used towards a holistic water circularity assessment.

Process-based and conceptual hydrologic and biogeochemical models have been developed to simulate the water and nutrients (or solutes) transport, fate and cycling. Such models are based on a theoretical understanding of relevant ecological processes (i.e. using partial differential equations, kinetic laws, stoichiometry and balance equations) (Cuddington et al., 2013) that enables the consideration of multiple and complex interactions between climate, soil, geology, vegetation, hydrology and nutrient balances. In addition to this feature, computer models interpret dynamically the analyzed system due to the inclusion of temporal and spatial variation and resolution – in comparison to the stationary nature of the MFA- and Consumption-based approaches. However, the main drawback of such models is the required amount of data (field and experimental data and model results), which increases with increasing mechanization of the model, as well as the complexity of the model. Data-intensive complex models impede their wide implementation and use, thus more generic models are required that are simpler and easier to apply (Vadas et al., 2013).

The reviewed tools (i.e. models) are categorized according to their focus. Pure hydrological models focus on the quantification of the hydrological partitioning, i.e. partitioning of precipitation into streamflow, evapotranspiration and storage change. Agro-hydrological models simulate hydrology, agricultural water management and in some cases nutrient loads of agricultural areas. Hydro-biochemical models simulate the transport fate and cycling of nutrients on soils and land use, including agricultural areas.

Starting with the reviewed hydrological models, Bellot and Chirino (2013) developed an eco-hydrological modelling approach (i.e. HYDROBAL) for assessing the water balance with a daily resolution. More precisely, HYDROBAL investigates the temporal variability in soil–water content determined by vegetation water uptake as a function of climatic conditions (i.e. daily rainfall and micrometeorological variables) and the model outputs include actual evapotranspiration, runoff, and aquifer recharge (deep percolation). Zhang et al. (2020) developed a conceptual catchment water balance model based on the proportionality hypothesis (denoted PWBM) to model the hydrological partitioning across spatial and temporal scales. The PWBM model inputs require precipitation, potential evapotranspiration and leaf area index and the model outputs are streamflow, evapotranspiration and storage change. Westenbroek et al. (2010) developed a modified Thornthwaite-Mather Soil-Water-Balance (SWB) code – combining geographic information system (GIS) data layers and tabular climatological data – to calculate spatial and temporal variations in groundwater recharge. Li et al. (2017b) developed a model – integrating conceptual models in the vadose zone (considering various landscape units, e.g. farmland, grassland, surface water, bare soil, etc.) and the groundwater flow model FEFLOW under GIS – to simulate the hydrological processes under various scenarios of water-saving activities. The model was applied in the Heihe River Basin in Gansu Province of China and validated by comparing the simulated evapotranspiration, groundwater levels and the total water balance with remote sensing results, previous studies and monitored data.

One of the most extensively used agro-hydrological model at the field scale is the FAO-56 dual crop coefficient (FAO-2Kc, Allen et al., 1998) model that estimates the crop water requirements by means of the simulated evapotranspiration and its two components, i.e. evaporation and transpiration. Another agro-hydrological model that can be used to investigate irrigation, nutrient and salt management strategies is the research version of the SWB model (i.e. SWB-Sci) that is a mechanistic, real-time, generic crop growth, soil water, nutrient and salt balance model (Annandale et al., 1999, 2000; van der Laan et al., 2010, 2014), consisting of different submodules (e.g. water balance sub-module and the nitrogen submodule). The Soil and Water Assessment Tool (SWAT) simulates water quality and quantity, the impact of land use, management practices and climate change (Arnold et al., 1998 & Arnold et al., 2012). Automatic irrigation algorithms in SWAT have been tested, improved and validated to correctly simulate the hydrological processes in agricultural catchments in response to climate change and water management scenarios (e.g. Dechmi et al., 2012; Githui et al., 2016; Wei et al., 2018; Uniyal and Dietrich, 2019).

Hydro-biogeochemical models simulate the transport, fate and cycling of nutrients on agricultural areas and soils, considering the interactions with the hydrological processes. DRAINMOD-P (Deal et al., 1986; Tian et al., 2012; Askar, 2019), HYPE (Lindström et al., 2010), INCA-P (Wade et al., 2002; Jackson-Blake et al., 2016), RZWQM2-P (Ma et al., 2012; Sadhukhan and Qi, 2018) and Simply-P (Jackson-Blake et al., 2017) are some of the existing models focusing on the phosphorus transport in soils, summarized in the review of Pferdmenges et al. (2020). The models presented here are the ones that are capable of simulating both surface (diffusion, desorption and erosion) and subsurface (infiltration and transport in soil) phosphorus processes, as well as phosphorus plant uptake. Additionally, all the presented models include all the hydrological compartments (i.e. surface water, infiltration, groundwater and streamflow), which are important if water cycle is included in the analysis. Their differences mainly lie on the different water flow and solute transport approaches that they use (e.g. storage routing representation or empirical or Darcy or Richards equations, and single/dual porosity and dual permeability etc.) and on the spatial scales (i.e. soil profile, plot/field and catchment). Additionally, the process-based CENTURY model (Parton et al., 1987) has been extensively used to dynamically estimate the soil organic carbon stocks. The CENTURY model is designed to simulate carbon, nitrogen, phosphorus and sulfur dynamics in natural or cultivated systems, using a monthly time step (Parton et al., 1988; Shaffer et al., 2001), allowing the simulation of the anthropogenic (land management) and natural (climate and soil) drivers, with the possibility to assess the effect of alternative scenarios. Other models developed to track and estimate the carbon sequestrated or emitted from both plantations and managed native forests are the CO_2Fix (Mohren and Klein-Goldewijk, 1990), GORCAM (Schlamadinger et al., 2000) and FullCAM (Richards and Evans, 2004; Waterworth et al., 2007).

The review of hydrological, agro-hydrological and hydro-biogeochemical models revealed that there is a variety of available models, capable of simulating the environmental processes of water and nutrients. Hydrological and agro-hydrological models can provide more accurate quantification of the hydrological processes on different temporal scales (daily, monthly, seasonal, annual) that can be used as inputs to water balance equations, considering both anthropogenic and natural water flows, in order to holistically assess the circularity of water systems. Biogeochemical models can shed light on the underestimated nutrients cycling in nature and carbon sequestration, that would further improve the environmental and physical assessment of the system, as well as the evaluation of the environmental benefits. Therefore, such models can effectively include NBS (as most of them use environmental processes to address societal challenges) and can be used within a water circularity assessment methodology integrating the ecosystem perspective, which is currently the missing element within the other reviewed approaches and methods. However, to the best of our knowledge, these models have not been incorporated to water assessment methodologies; thus, indicators targeting at the ecosystems cannot be identified in this case. The selection of the model to be incorporated in a water circularity assessment methodology is of major importance as models with high complexity would burden their wide use and adoption, while very simplistic models would not accurately describe the system, resulting in unrealistic conclusions. The model should also be able to sufficiently and simultaneously describe the environmental processes of water and all nutrients in order to avoid the need of incorporating different models for different purposes, which would result in high complexity and computational time. Attention should be given in case that such models are used in parallel with consumption-based approaches (such as LCA), as the actual environmental impacts resulting from the former and the potential impacts resulting from the latter may conflict with each other and be double counted.

The previously reviewed models have been used as standalone tools by the researchers. Villarroel Walker (2010) developed a Multi-sectoral Systems Analysis (MSA) methodology for understanding and managing the metabolism of complex systems, supported by a set of socio-ecological indicators. The environmental model was coded in MATLAB, incorporating Substance Flow Analysis (SFA) and Regionalized Sensitivity Analysis (RSA), in order to track and account for the movement of water, energy, nitrogen, phosphorus, and carbon into, around, and out of a regional-city system, considering multiple socio-economic sectors (i.e. water, energy, food, forestry, and waste management) and the interactions amongst them. However, the focus of this study is on the production system rather than the water system itself with the developed indicators (Table 5) covering aspects of circularity (i.e. the direct or indirect regeneration of natural capital and the design out waste externalities from its environmental dimension). The indicators are not able to evaluate the principle of keeping resources in use by measuring the extent of closing the water, energy and nutrients loops.

6. Compilation of findings

The current review paper addressed two main questions regarding circularity in water systems: What needs to be measured to assess water circularity? and How can water circularity be measured?

The answer to the first question was found in the white papers of Stuchtey (2015), IWA (2016) and Arup et al. (2018), and it is the realization of the three CE principles of natural capital regeneration, keeping resources in use and designing out waste externalities. The water, nutrients and other materials, as well as the energy pathways need to be followed within both human- and nature-managed systems (Arup et al., 2018). This analysis would enable the consideration of physical (natural and anthropogenic water, nutrients, materials and energy flows), environmental (actual impacts, e.g. increase or loss of biodiversity, and potential impacts, e.g. global warming potential) and economic (e.g. eco-efficiency) aspects. Most of these aspects can be measured using already developed indicators (Table 6).

Table 5
Developed indicators in Villarroel Walker and Beck, 2012).

Study	Indicator	Description	CE principles: Regeneration of natural capital – Direct	Regeneration of natural capital – Indirect	Keep resources in use – Water	Keep resources in use – Nutrients	Keep resources in use – Energy	Design out waste externalities – Environmental	Design out waste externalities – Economic
Villarroel Walker et al., 2012	Productivity indicator (PRI)	Measure of useful products generated within the system per unit of resources consumed		✔					
	Resources usage indicator (RWI)	Measure of resources consumed per unit of waste requiring disposal		✔				✔	
	Eco-efficiency indicator with respect to wastes (PWI)	Measure of the amount of products per unit of disposed waste						✔	
	Eco-efficiency indicator with respect to emissions (EEI)	Measure of the amount of products per unit of emission to the environment, either to the atmosphere or to water bodies	✔					✔	
	Health of air emissions (HAE)	Measure of the ratio between the actual amount of emissions to the atmosphere and a healthy emission level	✔						
	Health of water emissions (HWE)	Measure of the ratio between the actual amount of emissions to water bodies and a healthy emission level	✔						
	Waste equals food (WEF)	Compares the amount of products versus the quantity that the system would generate if no flows are classified as waste and all emissions correspond to healthy emissions		✓				✓	
	Eco-effective indicator (E2I)	Encloses together the concepts of waste equals food and healthy emissions, describing thus the overall eco-effectiveness of the system						✔	

✔: directly addressed; ✓ : indirectly addressed

Table 6
Methodologies, tools and indicative indicators measuring different aspects of circularity.

	Evaluation of:												
	Regeneration of natural capital					Keep resources in use			Design out waste externalities			Integrated system	Additional aspects
	Water cycle		Nutrients (N, C, P) cycle	Biodiversity	Demand minimization of resources	Water	Nutrients	Energy	Waste reduction	Environmental impacts	Economic impacts		
	Quantity	Quality											
Tools													
Hydrology-hydraulic models	✓												✓
Water quality measurements		✓											✓
Biological samplings				✓									
Surveys													✓
Satellite images													
ArcGIS													✓
Fragstat													
Hydrological models	✓									✓			
Agro-hydrological models	✓	✓	✓							✓			
Hydro-biogeochemical models	✓	✓	✓							✓			
Methods													
UWM	✓	✓				✓	✓	✓				✓	
TM						✓	✓						
UHA					✓	✓		✓	✓				
H2HA					✓		✓		✓				
LCA										✓			
MSA			✓		✓				✓	✓		✓	
LCC											✓		
Indicative indicators	e.g. Hydrological performance (Renouf et al., 2017)	e.g. Water pollutant load within safe operating space (Renouf et al., 2017)	–	e.g. Mean species abundance (Boelee et al., 2017)	e.g. DMI (Agudelo Vera et al., 2012)	e.g. Internal harvesting & recycling ratio (Farooqui et al., 2016)	e.g. SSI (Wielemaker et al., 2018)	e.g. ERI (Leusbrock et al., 2015)	e.g. WEF (Villarroel Walker et al., 2012)	**Actual:** e.g. HWE (Villarroel Walker et al., 2012) **Potential**: e.g. Global warming potential (GWP) (from LCA)	e.g. Avoided cost of pollutants (Chow et al., 2014)	e.g. PRI (Villarroel Walker et al., 2012)	e.g. Landscape value (Li et al., 2017)

✓: directly addressed; ✓: potentially addressed

Regarding the second question, a variety of existing methodologies and tools were identified that have been presently used to assess NBS for water management, water systems, and water-related environmental compartments. These methodologies are capable to measure the different aspects of water circularity, as illustrated in Table 6.

The information given in Table 6 can be used to get better insights on what is still missing to holistically assess the circularity of water systems. The lack of a holistic water circularity assessment methodology is apparent, as well as a water circularity assessment framework to rigidly frame the assessment methodologies. However, there is a variety of existing methods and tools that if incorporated are able to cover all the different aspects of circularity assessment in water systems. Additionally, although a wide set of indicators exists in the literature covering most of the aspects of circularity, indicators measuring the environmental performance of water systems in terms of natural nutrients cycles are still missing. The existence of numerous indicators does not necessarily mean that they are capable of adequately and holistically measure circularity in water systems. The development of a water circularity assessment methodology would prove their adequateness and a potential requirement for the use of new indicators.

7. Conclusions

The concept of water circularity is not well addressed within the research community and there is an apparent lack of a water circularity assessment framework. Currently available circularity methodologies – focusing mainly on the consumer goods sector – are not easily transferable to the water sector. A failure to adequately evaluate water circularity by considering both human-managed and nature-managed systems, would result in scattered and fragmented actions, insufficient to deal with the degradation of our most essential and valuable resource and of our natural capital in turn. Therefore, the present study identifies the positioning of water within the CE concept and how NBS can enable the transition to CE, by shedding light on what needs to be measured and how to measure water circularity.

NBS – as human interventions to address societal challenges by enhancing natural processes –can play a key role in re-connecting nature-to human-managed water systems. However, researchers mainly focus on the technical performance of NBS and the evaluation of some of their co-benefits. The connection of NBS to the wider water system and environment, water quantity issues, as well as a holistic assessment methodology still remain gaps.

Concerning the assessment methodologies of water systems, MFA-based approaches are capable of mapping all the natural and anthropogenic water and water-related flows in a well-defined system and provide a wide set of indicators. However, the natural cycles of nutrients are not holistically considered. LCA -based approaches include well-established indicators focusing on the potential environmental impacts of anthropogenic water systems, but fall short both of showing the actual impacts and of incorporating natural elements in the assessment. To bridge those gaps, hydro-biogeochemical models can be incorporated enabling the consideration of the natural environment in the assessment. The selection of the appropriate model should be based both on its capability to consider different aspects and on its simplicity that would enable its wider use and application.

A question however remains on how to incorporate different methods and models within a holistic water systems circularity assessment. Therefore, future research in the topic should address current challenges, requirements, limitations and barriers leading in the development of a water circularity assessment framework and enabling researchers and practitioners to work towards this direction.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

This research was supported by the Horizon 2020 research and innovation program HYDROUSA (grant agreement No 776643). Part of the work was performed within the COST Action CA17133 Circular City. Gianluigi Buttiglieri acknowledges the Ramon y Cajal research fellowship RYC-2014-16754 and the CLEaN-TOUR project (CTM2017-85385-C2-1-R) from the Spanish Ministry of Economy and Competitiveness. The authors thank Generalitat de Catalunya through Consolidated Research Group 2017-SGR-1318. ICRA researchers thank funding from CERCA program. Nataša Atanasova acknowledges the project (Closing material flows by wastewater treatment with green technologies, ID J2-8162), financially supported by the Slovenian Research Agency.

References

Agudelo-Vera, C.M., Mels, A., Keesman, K., Rijnaarts, H., 2012. The urban harvest approach as an aid for sustainable urban resource planning. J. Ind. Ecol. 16 (6), 839–850. https://doi.org/10.1111/j.1530-9290.2012.00561.x.

Alcalde-Sanz, L., Gawlik, B.M., 2017. Minimum Quality Requirements for Water Reuse in Agricultural Irrigation and Aquifer Recharge - towards a Water Reuse Regulatory Instrument at EU Level. EUR 28962 EN. Publications Office of the European Union, Luxembourg. https://doi.org/10.2760/887727, 978-92-79-77176-7, PUBSY No.109291.

Allen, R.G., Pereira, L.S., Raes, D., Smith, M., 1998. Crop Evapotranspiration: Guidelines for Computing Crop Water Requirements, FAO Irrigation and Drainage Paper 56. FAO, Rome, Italy, 92-5-104219-5.

Amann, A., Zoboli, O., Krampe, J., Rechberger, H., Zessner, M., Egle, L., 2018. Environmental impacts of phosphorus recovery from municipal wastewater. Resour. Conserv. Recycl. 130, 127–139. https://doi.org/10.1016/j.resconrec.2017.11.002.

Amos, C.C., Rahman, A., Gathenya, J.M., 2018a. Economic analysis of rainwater harvesting systems comparing developing and developed countries: a case study of Australia and Kenya. J. Clean. Prod. 172, 196–207. https://doi.org/10.1016/j.jclepro.2017.10.114.

Amos, C.C., Rahman, A., Karim, F., Gathenya, J.M., 2018b. A scoping review of roof harvested rainwater usage in urban agriculture: Australia and Kenya in focus. J. Clean. Prod. 202, 174–190. https://doi.org/10.1016/j.jclepro.2018.08.108.

Andrés-Doménech, I., Hernández-Crespo, C., Martín, M., Andrés-Valeri, V.C., 2018. Characterization of wash-off from urban impervious surfaces and SuDS design criteria for source control under semi-arid conditions. Sci. Total Environ. 612, 1320–1328. https://doi.org/10.1016/j.scitotenv.2017.09.011.

Annandale, J.G., Benade, N., Jovanovic, N.Z., Steyn, J.M., Du Sautoy, N., 1999. Facilitating Irrigation Scheduling by Means of the Soil Water Balance Model. Water Research Commission. Report No. 753/1/99, Pretoria, South Africa.

Annandale, J.G., Campbell, G.S., Olivier, F.C., Jovanovic, N.Z., 2000. Predicting crop water uptake under full and deficit irrigation. An example using pea (Pisum-sativum cv Puget). Irrigat. Sci. 19, 65–72. https://doi.org/10.1007/s002710050002.

Arnold, J.G., Srinivasan, R., Muttiah, R.S., Williams, J.R., 1998. Large area hydrologic modeling and assessment: Part I. Model development. J. Am. Water Resour. Assoc. 34, 73–89. https://doi.org/10.1111/j.1752-1688.1998.tb05961.x.

Arnold, J.G., Moriasi, D.N., Gassman, P.W., Abbaspour, K.C., White, M.J., Srinivasan, R., Santhi, C., Harmel, R.D., Van Griensven, A., Van Liew, M.W., Kannan, N., 2012. SWAT: model use, calibration, and validation. Transactions of the ASABE 55 (4), 1491–1508. https://doi.org/10.13031/2013.42256.

Arup, Antea Group, Ellen MacArthur Foundation, 2018. Water and Circular Economy: White Paper. https://nextgenwater.eu/wp-content/uploads/2018/10/Water_and_circular_economy-Co.Project_White_paper.pdf.

Askar, M., 2019. DRAINMOD-P: A Model for Simulating Phosphorus Dynamics and Transport in Artificially Drained Agricultural Lands. North Carolina State University, Raleigh, North Carolina, USA. http://www.lib.ncsu.edu/resolver/1840.20/36548.

Australian Government, 2018. Charter: National Water Quality Management Strategy. https://www.waterquality.gov.au/about/charter.

Babbitt, C.W., Gaustad, G., Fisher, A., Chen, W.Q., Liu, G., 2018. Closing the loop on circular economy research: from theory to practice and back again. Resour. Conserv. Recycl. 135, 1–2. https://doi.org/10.1016/j.resconrec.2018.04.012.

Bai, S., Wang, X., Zhao, X., Ren, N., 2018. Characterizing water pollution potential in life cycle impact assessment based on bacterial growth and water quality models. Water 10 (11), 1621. https://doi.org/10.3390/w10111621.

Balanay, R., Halog, A., 2019. Tools for circular economy: review and some potential applications for the Philippine textile industry. Circular Economy in Textiles and Apparel 49–75. https://doi.org/10.1016/B978-0-08-102630-4.00003-0.

Baleta, J., Mikulčić, H., Klemeš, J.J., Urbaniec, K., Duić, N., 2019. Integration of energy, water and environmental systems for a sustainable development. J. Clean. Prod. 215, 1424–1436. https://doi.org/10.1016/J.JCLEPRO.2019.01.035.

Baynes, T.M., Wiedmann, T., 2012. General approaches for assessing urban environmental sustainability. Current Opinion in Environmental Sustainability 4 (4), 458–464. https://doi.org/10.1016/j.cosust.2012.09.003.

Bellot, J., Chirino, E., 2013. Hydrobal: an eco-hydrological modelling approach for assessing water balances in different vegetation types in semi-arid areas. Ecol. Model. 266, 30–41. https://doi.org/10.1016/j.ecolmodel.2013.07.002.

Bhoye, M., Pandya, M.H., Valvi, S., Trivedi, I.N., Jangir, P., Parmar, S.A., 2016. An emission constraint economic load dispatch problem solution with microgrid using JAYA algorithm. International Conference on Energy Efficient Technologies for Sustainability (ICEETS) 497–502. https://doi.org/10.1109/ICEETS.2016.7583805.

Bobba, S., Mathieux, F., Ardente, F., Blengini, G.A., Cusenza, M.A., Podias, A., Pfrang, A., 2018. Life Cycle Assessment of repurposed electric vehicle batteries: an adapted method based on modelling energy flows. Journal of Energy Storage

19, 213–225. https://doi.org/10.1016/j.est.2018.07.008.
Bockarjova, M., Botzen, W.J.W., 2017. Review of Economic Valuation of Nature-Based Solutions in Urban Areas. Universiteit Utrecht. Technical Report, Naturvation Project.
Boelee, E., Janse, J., Le Gal, A., Kok, M., Alkemade, R., Ligtvoet, W., 2017. Overcoming water challenges through nature-based solutions. Water Pol. 19 (5), 820–836. https://doi.org/10.2166/wp.2017.105.
Boerema, A., Van Passel, S., Meire, P., 2018. Cost-effectiveness analysis of ecosystem management with ecosystem services: from theory to practice. Ecol. Econ. 152, 207–218. https://doi.org/10.1016/j.ecolecon.2018.06.005.
Bricker, S.H., Banks, V.J., Galik, G., Tapete, D., Jones, R., 2017. Accounting for groundwater in future city visions. Land Use Pol. 69, 618–630. https://doi.org/10.1016/j.landusepol.2017.09.018.
Buonocore, E., Mellino, S., De Angelis, G., Liu, G., Ulgiati, S., 2018. Life cycle assessment indicators of urban wastewater and sewage sludge treatment. Ecol. Indicat. 94, 13–23. https://doi.org/10.1016/j.ecolind.2016.04.047.
Cáceres, L., Méndez, D., Fernández, J., Marcé, R., 2018. From end-of-pipe to nature-based solutions: a simple statistical tool for maximizing the ecosystem services provided by reservoirs for drinking water treatment. Water Resour. Manag. 32, 1307–1323. https://doi.org/10.1007/s11269-017-1871-7.
Castonguay, A.C., Urich, C., Iftekhar, M.S., Deletic, A., 2018. Modelling urban water management transitions: a case of rainwater harvesting. Environ. Model. Software 105, 270–285. https://doi.org/10.1016/j.envsoft.2018.05.001.
Chen, W., Oldfield, T.L., Katsantonis, D., Kadoglidou, K., Wood, R., Holden, N.M., 2019. The socio-economic impacts of introducing circular economy into Mediterranean rice production. J. Clean. Prod. 218, 273–283. https://doi.org/10.1016/j.jclepro.2019.01.334.
Chow, J.F., Savić, D., Fortune, D., Kapelan, Z., Mebrate, N., 2014. Using a systematic, multi-criteria decision support framework to evaluate sustainable drainage designs. Procedia Engineering 70, 343–352. https://doi.org/10.1016/j.proeng.2014.02.039.
Cohen-Shacham, E., Walters, G., Janzen, C., Maginnis, S., 2016. Nature-based Solutions to Address Global Societal Challenges. IUCN, Gland, Switzerland. https://doi.org/10.2305/IUCN.CH.2016.13 (en).
Council Directive 91/271/EEC, 1991. Council Directive 91/271/EEC of 21 May 1991 concerning urban waste-water treatment. Official Journal of the European Communities No. L 135, 40–52.
Cuddington, K., Fortin, M.J., Gerber, L.R., Hastings, A., Liebhold, A., O'connor, M., Ray, C., 2013. Process-based models are required to manage ecological systems in a changing world. Ecosphere 4 (2), 1–12. https://doi.org/10.1890/ES12-00178.1.
Deal, S.C., Gilliam, J.W., Skaggs, R.W., Konyha, K.D., 1986. Prediction of nitrogen and phosphorus losses as related to agricultural drainage system design. Agric. Ecosyst. Environ. 18, 37–51. https://doi.org/10.1016/0167-8809(86)90173-8.
Dechmi, F., Burguete, J., Skhiri, A., 2012. SWAT application in intensive irrigation systems: model modification, calibration and validation. J. Hydrol. 470, 227–238. https://doi.org/10.1016/j.jhydrol.2012.08.055.
Decision No. 1386/2013/EU, 2013. Decision No. 1386/2013/EU of the European parliament and of the council of 20 november 2013 on a general union environment action programme to 2020 'living well, within the limits of our planet'. Official Journal of the European Communities No. L 354, 171–200.
Directive/24/EU, 2014. Directive 2014/24/EU of the European parliament and of the council of 26 february 2014 on public procurement and repealing directive 2004/18/EC. Official Journal of the European Communities No L94, 65–242.
Directive/39/EU, 2013. Directive 2013/39/EU of the European Parliament and of the Council of 12 August 2013, amending Directives 2000/60/EC and 2008/105/EC as regards priority substances in the field of water policy. Official Journal of the European Communities No L226, 1–17.
Dominguez, S., Laso, J., Margallo, M., Aldaco, R., Rivero, M.J., Irabien, Á., Ortiz, I., 2018. LCA of greywater management within a water circular economy restorative thinking framework. Sci. Total Environ. 621, 1047–1056. https://doi.org/10.1016/j.scitotenv.2017.10.122.
Dotro, G., Langergraber, G., Molle, P., Nivala, J., Puigagut, J., Stein, O., Von Sperling, M., 2017. Treatment Wetlands, vol. 7. IWA Publishing, London, United Kingdom. https://doi.org/10.2166/9781780408774.
EC (European Commission), 2015a. Closing the Loop – an EU Action Plan for the Circular Economy. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. https://www.eea.europa.eu/policy-documents/com-2015-0614-final.
EC (European Commission), 2015b. Towards an EU Research and Innovation Policy Agenda for Nature-Based Solutions & Re-naturing Cities. Final Report of the Horizon 2020 Expert Group on "Nature-Based Solutions and Re-naturing Cities." https://doi.org/10.2777/765301.
EC (European Commission), 2015c. Commission implementing decision (EU) 2015/495 of 20 March 2015 establishing a watch list of substances for Union-wide monitoring in the field of water policy pursuant to Directive 2008/105/EC of the European Parliament and of the Council. Official Journal of the European Communities No L78, 40–42.
EC (European Commission), 2018. Commission implementing decision (EU) 2018/840 of 5 June 2018 establishing a watch list of substances for Union-wide monitoring in the field of water policy pursuant to Directive 2008/105/EC of the European Parliament and of the Council and repealing Commission Implementing Decision (EU) 2015/495. Official Journal of the European Union No. L141, pp. 9–12.
EC (European Commission), 2019. Staff Working Document on the Evaluation of the Urban Waste Water Treatment Directive. https://ec.europa.eu/environment/water/water-urbanwaste/evaluation/index_en.htm.
Elia, V., Gnoni, M.G., Tornese, F., 2017. Measuring circular economy strategies through index methods: a critical analysis. J. Clean. Prod. 142, 2741–2751. https://doi.org/10.1016/j.jclepro.2016.10.196.
EMF (Ellen MacArthur Foundation), 2017. A New Textiles Economy: Redesigning Fashion's Future.
EMF (Ellen MacArthur Foundation), 2018. Circular Consumer Electronics: an Initial Exploration.
EMF (Ellen MacArthur Foundation), Arup, 2019. Circular Economy in Cities: Project Guide.
EMF (Ellen MacArthur Foundation), Granta Design, 2015. Circularity Indicators: an Approach to Measuring Circularity.
EMF (Ellen MacArthur Foundation) and WEF (World Economic Forum), 2017. The New Plastics Economy – Catalysing Action.
Evans, J., Bocken, N., 2013. Circular Economy Toolkit. Cambridge Institute for Manufacturing. http://circulareconomytoolkit.org.
FAO (Food and Agriculture Organization of the United Nations), 2016. AQUASTAT - FAO's global information system on water and agriculture. Website accessed on 12/05/2020.
Farooqui, T.A., Renouf, M.A., Kenway, S.J., 2016. A metabolism perspective on alternative urban water servicing options using water mass balance. Water Res. 106, 415–428. https://doi.org/10.1016/j.watres.2016.10.014.
Feng, H., Hewage, K.N., 2018. Economic benefits and costs of green roofs. Nature Based Strategies for Urban and Building Sustainability 307–316. https://doi.org/10.1016/B978-0-12-812150-4.00028-8.
Franklin-Johnson, E., Figge, F., Canning, L., 2016. Resource duration as a managerial indicator for Circular Economy performance. J. Clean. Prod. 133, 589–598. https://doi.org/10.1016/j.jclepro.2016.05.023.
García-Guaita, F., González-García, S., Villanueva-Rey, P., Moreira, M.T., Feijoo, G., 2018. Integrating urban metabolism, material flow analysis and life cycle assessment in the environmental evaluation of santiago de Compostela. Sustainable Cities and Society 40, 569–580. https://doi.org/10.1016/j.scs.2018.04.027.
Garfí, M., Flores, L., Ferrer, I., 2017. Life cycle assessment of wastewater treatment systems for small communities: activated sludge, constructed wetlands and high rate algal ponds. J. Clean. Prod. 161, 211–219. https://doi.org/10.1016/j.jclepro.2017.05.116.
Geronimo, F.K.F., Maniquiz-Redillas, M.C., Hong, J.S., Kim, L.H., 2019. Nutrient concentration in sediments accumulated in pre-treatment basins of urban LID technologies. Water Sci. Technol. 79 (5), 1000–1006. https://doi.org/10.2166/wst.2019.033.
Githui, F., Thayalakumaran, T., Selle, B., 2016. Estimating irrigation inputs for distributed hydrological modelling: a case study from an irrigated catchment in southeast Australia. Hydrol. Process. 30 (12), 1824–1835. https://doi.org/10.1002/hyp.10757.
Goldstein, B., Birkved, M., Quitzau, M.B., Hauschild, M., 2013. Quantification of urban metabolism through coupling with the life cycle assessment framework: concept development and case study. Environ. Res. Lett. 8 (3), 035024 https://doi.org/10.1088/1748-9326/8/3/035024.
Griffiths, P., Cayzer, S., 2016. Design of indicators for measuring product performance in the circular economy. Smart Innovation, Systems and Technologies 52, 307–321, 978-3-319-32096-0.
Guertin, F., Halsey, K., Polzin, T., Rogers, M., Witt, B., 2019. From ash pond to riverside wetlands: making the business case for engineered natural technologies. Sci. Total Environ. 651, 419–426. https://doi.org/10.1016/j.scitotenv.2018.09.035.
Haddis, A., van der Bruggen, B., Smets, I., 2020. Constructed wetlands as nature-based solutions in removing organic pollutants from wastewater under irregular flow conditions in a tropical climate. Ecohydrol. Hydrobiol. 20 (1), 38–47. https://doi.org/10.1016/j.ecohyd.2019.03.001.
Hansson, S.O., 2007. Philosophical problems in cost–benefit analysis. Econ. Philos. 23 (2), 163–183. https://doi.org/10.1017/S0266267107001356.
Hao, X., Wang, X., Liu, R., Li, S., van Loosdrecht, M.C., Jiang, H., 2019. Environmental impacts of resource recovery from wastewater treatment plants. Water Res. 160, 268–277. https://doi.org/10.1016/j.watres.2019.05.068.
Helm, D., Hepburn, C., 2012. The economic analysis of biodiversity: an assessment. Oxf. Rev. Econ. Pol. 28 (1), 1–21. https://doi.org/10.1093/oxrep/grs014.
Hendriks, C., Obernosterer, R., Müller, D., Kytzia, S., Baccini, P., Brunner, P.H., 2000. Material flow analysis: a tool to support environmental policy decision making. Case-studies on the city of Vienna and the Swiss lowlands. Local Environ. 5 (3), 311–328. https://doi.org/10.1080/13549830050134257.
Hernández-Crespo, C., Gargallo, S., Benedito-Durá, V., Nácher-Rodríguez, B., Rodrigo-Alacreu, M.A., Martín, M., 2017. Performance of surface and subsurface flow constructed wetlands treating eutrophic waters. Sci. Total Environ. 595, 584–593. https://doi.org/10.1016/j.scitotenv.2017.03.278.
Hertwich, E.G., 2005. Life cycle approaches to sustainable consumption: a critical review. Environ. Sci. Technol. 39 (13), 4673–4684. https://doi.org/10.1021/es0497375.
Hislop, H., Hill, J., 2011. Reinventing the Wheel: A Circular Economy for Resource Security. Green Alliance. https://www.green-alliance.org.uk/page_77.php.
Hofmann, F., 2019. Circular business models: business approach as driver or obstructer of sustainability transitions? J. Clean. Prod. 224, 361–374. https://doi.org/10.1016/j.jclepro.2019.03.115.

ISO, 1997. ISO 14040: Environmental Management – Life Cycle Assessment: Principles and Framework. International Organization for Standardization, Geneva, Switzerland.
IWA (International Water Association), 2016. Water utility pathways in a circular economy: charting a Course for Sustainability. https://iwa-network.org/water-utility-pathways-circular-economy-charting-course-sustainability/.
Jackson-Blake, L.A., Wade, A.J., Futter, M.N., Butterfield, D., Couture, R.M., Cox, B.A., Crossman, J., Ekholm, P., Halliday, S.J., Jin, L., Lawrence, D., 2016. The INtegrated CAtchment model of phosphorus dynamics (INCA-P): description and demonstration of new model structure and equations. Environ. Model. Software 83, 356–386. https://doi.org/10.1016/j.envsoft.2016.05.022.
Jackson-Blake, L.A., Sample, J.E., Wade, A.J., Helliwell, R.C., Skeffington, R.A., 2017. Are our dynamic water quality models too complex? A comparison of a new parsimonious phosphorus model, SimplyP, and INCA-P. Water Resour. Res. 53, 5382–5399. https://doi.org/10.1002/2016WR020132.
Jurczak, T., Wagner, I., Kaczkowski, Z., Szklarek, S., Zalewski, M., 2018. Hybrid system for the purification of street stormwater runoff supplying urban recreation reservoirs. Ecol. Eng. 110, 67–77. https://doi.org/10.1016/j.ecoleng.2017.09.019.
Kenway, S., Gregory, A., McMahon, J., 2011. Urban water mass balance analysis. J. Ind. Ecol. 15 (5), 693–706. https://doi.org/10.1111/j.1530-9290.2011.00357.x.
Krzeminski, P., Tomei, M.C., Karaolia, P., Langenhoff, A., Almeida, C.M.R., Felis, E., Gritten, F., Andersen, H.R., Fernandes, T., Manaia, C.M., Rizzo, L., 2019. Performance of secondary wastewater treatment methods for the removal of contaminants of emerging concern implicated in crop uptake and antibiotic resistance spread: a review. Sci. Total Environ. 648, 1052–1081. https://doi.org/10.1016/j.scitotenv.2018.08.130.
Laitinen, J., Moliis, K., Surakka, M., 2017. Resource efficient wastewater treatment in a developing area—climate change impacts and economic feasibility. Ecol. Eng. 103, 217–225. https://doi.org/10.1016/j.ecoleng.2017.04.017.
Lane, J.L., de Haas, D.W., Lant, P.A., 2015. The diverse environmental burden of city-scale urban water systems. Water Res. 81, 398–415. https://doi.org/10.1016/j.watres.2015.03.005.
Langergraber, G., Pucher, B., Simperler, L., Kisser, J., Katsou, E., Buehler, D., Garcia Mateo, M.C., Atanasova, N., 2019a. Implementing Nature-Based Solutions for Creating a Resourceful Circular City. Blue-Green Systems.
Langergraber, G., Dotro, G., Nivala, J., Stein, O.R., 2019b. Wetland Technology: Practical Information on the Design and Application of Treatment Wetlands. IWA Publishing, London, United Kingdom. https://doi.org/10.2166/9781789060171_0011.
Lee, E., Oliveira, D.S.B.L., Oliveira, L.S.B.L., Jimenez, E., Kim, Y., Wang, M., Ergas, S.J., Zhang, Q., 2020. Comparative environmental and economic life cycle assessment of high solids anaerobic co-digestion for biosolids and organic waste management. Water Res. 171, 115443. https://doi.org/10.1016/j.watres.2019.115443.
Lemos, D., Dias, A.C., Gabarrell, X., Arroja, L., 2013. Environmental assessment of an urban water system. J. Clean. Prod. 54, 157–165. https://doi.org/10.1016/j.jclepro.2013.04.029.
Leong, J.Y.C., Balan, P., Chong, M.N., Poh, P.E., 2019. Life-cycle assessment and life-cycle cost analysis of decentralised rainwater harvesting, greywater recycling and hybrid rainwater-greywater systems. J. Clean. Prod. 229, 1211–1224. https://doi.org/10.1016/j.jclepro.2019.05.046.
Leusbrock, I., Nanninga, T.A., Lieberg, K., Agudelo-Vera, C.M., Keesman, K.J., Zeeman, G., Rijnaarts, H.H.M., 2015. The urban harvest approach as framework and planning tool for improved water and resource cycles. Water Sci. Technol. 72 (6), 998–1006. https://doi.org/10.2166/wst.2015.299.
Li, Y., Zhu, G., Ng, W.J., Tan, S.K., 2014. A review on removing pharmaceutical contaminants from wastewater by constructed wetlands: design, performance and mechanism. Sci. Total Environ. 468, 908–932. https://doi.org/10.1016/j.scitotenv.2013.09.018.
Li, J., Deng, C., Li, Y., Li, Y., Song, J., 2017a. Comprehensive benefit evaluation system for low-impact development of urban stormwater management measures. Water Resour. Manag. 31 (15), 4745–4758. https://doi.org/10.1007/s11269-017-1776-5.
Li, J., Mao, X., Li, M., 2017b. Modeling hydrological processes in oasis of Heihe River Basin by landscape unit-based conceptual models integrated with FEFLOW and GIS. Agric. Water Manag. 179, 338–351. https://doi.org/10.1016/j.agwat.2016.09.007.
Licciardello, F., Milani, M., Consoli, S., Pappalardo, N., Barbagallo, S., Cirelli, G., 2018. Wastewater tertiary treatment options to match reuse standards in agriculture. Agric. Water Manag. 210, 232–242. https://doi.org/10.1016/j.agwat.2018.08.001.
Linder, M., Sarasini, S., van Loon, P., 2017. A metric for quantifying product-level circularity. J. Ind. Ecol. 21 (3), 545–558. https://doi.org/10.1111/jiec.12552.
Lindström, G., Pers, C., Rosberg, J., Strömqvist, J., Arheimer, B., 2010. Development and testing of the HYPE (Hydrological Predictions for the Environment) water quality model for different spatial scales. Nord. Hydrol 41 (3–4), 295–319. https://doi.org/10.2166/nh.2010.007.
Liquete, C., Udias, A., Conte, G., Grizzetti, B., Masi, F., 2016. Integrated valuation of a nature-based solution for water pollution control. Highlighting hidden benefits. Ecosystem Services 22, 392–401. https://doi.org/10.1016/j.ecoser.2016.09.011.
Ma, L., Ahuja, L.R., Nolan, B.T., Malone, R.W., Trout, T.J., Qi, Z., 2012. Root zone water quality model (RZWQM2): model use, calibration, and validation. Transactions of the ASABE 55 (4), 1425–1446.
MacLeod, M., Moran, D., Eory, V., Rees, R.M., Barnes, A., Topp, C.F., Ball, B., Hoad, S., Wall, E., McVittie, A., Pajot, G., 2010. Developing greenhouse gas marginal abatement cost curves for agricultural emissions from crops and soils in the UK. Agric. Syst. 103 (4), 198–209. https://doi.org/10.1016/j.agsy.2010.01.002.
Manso, M., Castro-Gomes, J., Paulo, B., Bentes, I., Teixeira, C.A., 2018. Life cycle analysis of a new modular greening system. Sci. Total Environ. 627, 1146–1153. https://doi.org/10.1016/j.scitotenv.2018.01.198.
Masseroni, D., Ercolani, G., Chiaradia, E.A., Maglionico, M., Toscano, A., Gandolfi, C., Bischetti, G.B., 2018. Exploring the performances of a new integrated approach of grey, green and blue infrastructures for combined sewer overflows remediation in high-density urban areas. Journal of Agricultural Engineering 49 (4), 233–241. https://doi.org/10.4081/jae.2018.873.
Moezzibadi, M., Charpentier, I., Wanko, A., Mosé, R., 2019. Temporal estimation of hydrodynamic parameter variability in stormwater constructed wetlands – the hysteresis effect during multi-rainfall events. Ecol. Eng. 127, 1–10. https://doi.org/10.1016/j.ecoleng.2018.11.002.
Mohren, G.M.J., Klein-Goldewijk, C.G.M., 1990. CO2FIX: a dynamic model of the CO2-fixation in forest stands. De dorschkamp, research institute for forestry and urban ecology. Report no 624(35).
Moraga, G., Huysveld, S., Mathieux, F., Blengini, G.A., Alaerts, L., Van Acker, K., De Meester, S., Dewulf, J., 2019. Circular economy indicators: what do they measure? Resour. Conserv. Recycl. 146, 452–461. https://doi.org/10.1016/j.resconrec.2019.03.045.
Murray, A., Skene, K., Haynes, K., 2017. The circular economy: an interdisciplinary exploration of the concept and application in a global context. J. Bus. Ethics 140 (3), 369–380. https://doi.org/10.1007/s10551-015-2693-2.
O'Hogain, S., McCarton, L., 2018. A Technology Portfolio of Nature Based Solutions: Innovations in Water Management. Springer, Cham. https://doi.org/10.1007/978-3-319-73281-7.
Oquendo-Di Cosola, V., Olivieri, F., Ruiz-García, L., Bacenetti, J., 2020. An environmental life cycle assessment of living wall systems. J. Environ. Manag. 254, 109743. https://doi.org/10.1016/j.jenvman.2019.109743.
Pan, T., Zhu, X.D., Ye, Y.P., 2011. Estimate of life-cycle greenhouse gas emissions from a vertical subsurface flow constructed wetland and conventional wastewater treatment plants: a case study in China. Ecol. Eng. 37 (2), 248–254. https://doi.org/10.1016/j.ecoleng.2010.11.014.
Pan, Y.R., Wang, X., Ren, Z.J., Hu, C., Liu, J., Butler, D., 2019. Characterization of implementation limits and identification of optimization strategies for sustainable water resource recovery through life cycle impact analysis. Environ. Int. 133, 105266. https://doi.org/10.1016/j.envint.2019.105266.
Parton, W.J., Schimel, D.S., Cole, C.V., Ojima, D.S., 1987. Analysis of factors controlling soil organic levels of grasslands in the Great Plains. Soil Sci. Soc. Am. J. 51 (5), 1173–1179. https://doi.org/10.2136/sssaj1987.03615995005100050015x.
Parton, W.J., Stewart, J.W.B., Cole, C.V., 1988. Dynamics of C, N, P and S in grassland soils: a model. Biogeochemistry 5, 109–131. https://doi.org/10.1007/BF02180320.
Pauleit, S., Zölch, T., Hansen, R., Randrup, T.B., Konijnendijk van den Bosch, C., 2017. Nature-based solutions and climate change – four shades of green. In: Kabisch, N., Korn, H., Stadler, J., Bonn, A. (Eds.), Nature-Based Solutions to Climate Change Adaptation in Urban Areas. Theory and Practice of Urban Sustainability Transitions. Springer, Cham. https://doi.org/10.1007/978-3-319-56091-5_3.
Pauliuk, S., 2018. Critical appraisal of the circular economy standard BS 8001: 2017 and a dashboard of quantitative system indicators for its implementation in organizations. Resour. Conserv. Recycl. 129, 81–92. https://doi.org/10.1016/j.resconrec.2017.10.019.
Pferdmenges, J., Breuer, L., Julich, S., Kraft, P., 2020. Review of soil phosphorus routines in ecosystem models. Environ. Model. Software 126, 104639. https://doi.org/10.1016/j.envsoft.2020.104639.
Philips, J., 2017. Principles of Natural Capital Accounting: A Background Paper for Those Wanting to Understand the Concepts and Methodology Underlying the UK Natural Capital Accounts Being Developed by ONS and Defra. Department for Environment. Food and Rural Affairs, United Kingdom. https://www.ons.gov.uk/economy/environmentalaccounts/methodologies/principlesofnaturalcapitalaccounting.
Pintilie, L., Torres, C.M., Teodosiu, C., Castells, F., 2016. Urban wastewater reclamation for industrial reuse: an LCA case study. J. Clean. Prod. 139, 1–14. https://doi.org/10.1016/j.jclepro.2016.07.209.
Pizzol, M., Weidema, B., Brandão, M., Osset, P., 2015. Monetary valuation in life cycle assessment: a review. J. Clean. Prod. 86, 170–179. https://doi.org/10.1016/j.jclepro.2014.08.007.
Pradel, M., Aissani, L., 2019. Environmental impacts of phosphorus recovery from a "product" Life Cycle Assessment perspective: allocating burdens of wastewater treatment in the production of sludge-based phosphate fertilizers. Sci. Total Environ. 656, 55–69. https://doi.org/10.1016/j.scitotenv.2018.11.356.
PUB (Public Utilities Board), 2018. Innovation in Water Singapore, vol. 8. PUB, Singapore's National Water Agency.
Radinja, M., Comas, J., Corominas, L., Atanasova, N., 2019. Assessing stormwater control measures using modelling and a multi-criteria approach. J. Environ. Manag. 243, 257–268. https://doi.org/10.1016/j.jenvman.2019.04.102.
Raymond, C.M., Berry, P., Breil, M., Nita, M.R., Kabisch, N., de Bel, M., Enzi, V., Frantzeskaki, N., Geneletti, D., Cardinaletti, M., Lovinger, L., Basnou, C., Monteiro, A., Robrecht, H., Sgrigna, G., Muhari, L., Calfapietra, C., 2017. An Impact Evaluation Framework to Support Planning and Evaluation of Nature-Based Solutions Projects. Report Prepared by the EKLIPSE Expert Working Group on Nature-Based Solutions to Promote Climate Resilience in Urban Areas. Centre for Ecology & Hydrology, Wallington, United Kingdom. https://www.preventionweb.net/publications/view/54143.

Reap, J., Roman, F., Duncan, S., Bras, B., 2008. A survey of unresolved problems in life cycle assessment. Int. J. Life Cycle Assess. 13, 290–300. https://doi.org/10.1007/s11367-008-0009-9.

Reddy, S.M., McDonald, R.I., Maas, A.S., Rogers, A., Girvetz, E.H., North, J., Molnar, J., Finley, T., Leathers, G., DiMuro, J.L., 2015. Finding solutions to water scarcity: incorporating ecosystem service values into business planning at the Dow Chemical Company's Freeport, TX facility. Ecosystem Services 12, 94–107. https://doi.org/10.1016/j.ecoser.2014.12.001.

Renouf, M.A., Serrao-Neumann, S., Kenway, S.J., Morgan, E.A., Choy, D.L., 2017. Urban water metabolism indicators derived from a water mass balance–Bridging the gap between visions and performance assessment of urban water resource management. Water Res. 122, 669–677. https://doi.org/10.1016/j.watres.2017.05.060.

Renouf, M.A., Kenway, S.J., Lam, K.L., Weber, T., Roux, E., Serrao-Neumann, S., Choy, D.L., Morgan, E.A., 2018. Understanding urban water performance at the city-region scale using an urban water metabolism evaluation framework. Water Res. 137, 395–406. https://doi.org/10.1016/j.watres.2018.01.070.

Reynaud, A., Lanzanova, D., Liquete, C., Grizzetti, B., 2017. Going green? Ex-post valuation of a multipurpose water infrastructure in Northern Italy. Ecosystem Services 27, 70–81. https://doi.org/10.1016/j.ecoser.2017.07.015.

Richards, G., Evans, D., 2004. Development of a carbon accounting model (FullCAM Ver. 1.0) for the Australian continent, 67. Australian Forestry, pp. 277–283, 4.

Roebuck, R.M., Oltean-Dumbrava, C., Tait, S., 2011. Whole life cost performance of domestic rainwater harvesting systems in the United Kingdom. Water Environ. J. 25 (3), 355–365. https://doi.org/10.1111/j.1747-6593.2010.00230.x.

Rugani, B., de Souza, D.M., Weidema, B.P., Bare, J., Bakshi, B., Grann, B., Johnston, J.M., Pavan, A.L.R., Liu, X., Laurent, A., Verones, F., 2019. Towards integrating the ecosystem services cascade framework within the Life Cycle Assessment (LCA) cause-effect methodology. Sci. Total Environ. 690, 1284–1298. https://doi.org/10.1016/j.scitotenv.2019.07.023.

Sadhukhan, D., Qi, Z., 2018. RZWQM2 Phosphorus Model (Technical Report). Department of Bioresource Engineering, McGill University, Quebec, Canada.

Schlamadinger, B., Marland, G., Canella, L., 2000. The model GORCAM (Graz/Oak ridge carbon accounting model). http://www.joanneum.ac.at/gorcam.htm.

Sgroi, M., Vagliasindi, F.G., Roccaro, P., 2018. Feasibility, sustainability and circular economy concepts in water reuse. Current Opinion in Environmental Science & Health 2, 20–25. https://doi.org/10.1016/j.coesh.2018.01.004.

Shaffer, M.J., Ma, L., Hansen, S., 2001. Modeling Carbon and Nitrogen Dynamics for Soil Management. CRC Press LLC.

Sicard, P., Agathokleous, E., Araminiene, V., Carrari, E., Hoshika, Y., De Marco, A., Paoletti, E., 2018. Should we see urban trees as effective solutions to reduce increasing ozone levels in cities? Environ. Pollut. 243, 163–176. https://doi.org/10.1016/j.envpol.2018.08.049.

Sohn, J., Vega, G.C., Birkved, M., 2018. A methodology concept for territorial metabolism-life cycle assessment: challenges and opportunities in scaling from urban to territorial assessment. Procedia CIRP 69, 89–93. https://doi.org/10.1016/j.procir.2017.10.005.

State of California, 2013. Policy for Water Quality Control for Recycled Water (Recycled Water Policy). Division of Water Quality, State Water Resources Control Board, California Environmental Protection Agency. https://www.waterboards.ca.gov/water_issues/programs/water_recycling_policy/.

Stewart, R., Bey, N., Boks, C., 2016. Exploration of the barriers to implementing different types of sustainability approaches. Procedia CIRP 48, 22–27. https://doi.org/10.1016/j.procir.2016.04.063.

Stuchtey, M., 2015. Rethinking the Water Cycle. McKinsey & Company Insights & Publications. https://www.mckinsey.com/business-functions/sustainability/our-insights/rethinking-the-water-cycle.

Swiss Confederation, 2016. Ordinance of the Federal Law of Introduction on Water Protection. Cantonal Ordinance on the Water Protection, OCPAc).

Sylwan, I., Zambrano, J., Thorin, E., 2019. Energy demand for phosphorus recovery from municipal wastewater. Energy Procedia 158, 4338–4343. https://doi.org/10.1016/j.egypro.2019.01.787.

Thorslund, J., Jarsjo, J., Jaramillo, F., Jawitz, J.W., Manzoni, S., Basu, N.B., Chalov, S.R., Cohen, M.J., Creed, I.F., Goldenberg, R., Hylin, A., 2017. Wetlands as large-scale nature-based solutions: status and challenges for research, engineering and management. Ecol. Eng. 108, 489–497. https://doi.org/10.1016/j.ecoleng.2017.07.012.

Tian, S., Youssef, M.A., Skaggs, R.W., Amatya, D.M., Chescheir, G.M., 2012. DRAIN-MODFOREST: integrated modeling of hydrology, soil carbon and nitrogen dynamics, and plant growth for drained forests. J. Environ. Qual. 41, 764–782. https://doi.org/10.2134/jeq2011.0388.

UNEP (United Nations Environment Program), 2009. Water Security and Ecosystem Services: the Critical Connection. A Contribution to the United Nations World Water Assessment Programme. WWAP, Nairobi, Kenya.

Uniyal, B., Dietrich, J., 2019. Modifying automatic irrigation in SWAT for plant water stress scheduling. Agric. Water Manag. 223, 105714. https://doi.org/10.1016/j.agwat.2019.105714.

Vadas, P.A., Bolster, C.H., Good, L.W., 2013. Critical evaluation of models used to study agricultural phosphorus and water quality. Soil Use Manag. 29, 36–44. https://doi.org/10.1111/j.1475-2743.2012.00431.x.

van der Laan, M., Stirzaker, R.J., Annandale, J.G., Bristow, K.L., Du preez, C.C., 2010. Monitoring and modelling draining and resident soil water nitrate concentrations to estimate leaching losses. Agric. Water Manag. 97, 1779–1786. https://doi.org/10.1016/j.agwat.2010.06.012.

van der Laan, M., Annandale, J.G., Bristow, K.L., Stirzaker, R.J., Du preez, C.C., Thorburn, P.J., 2014. Modelling nitrogen leaching: are we getting the right answer for the right reason. Agric. Water Manag. 133, 74–80. https://doi.org/10.1016/j.agwat.2013.10.017.

Verger, Y., Petit, C., Barles, S., Billen, G., Garnier, J., Esculier, F., Maugis, P., 2018. AN, P, C, and water flows metabolism study in a peri-urban territory in France: the case-study of the Saclay plateau. Resour. Conserv. Recycl. 137, 200–213. https://doi.org/10.1016/j.resconrec.2018.06.007.

Verlicchi, P., Zambello, E., 2014. How efficient are constructed wetlands in removing pharmaceuticals from untreated and treated urban wastewaters? A review. Sci. Total Environ. 470, 1281–1306. https://doi.org/10.1016/j.scitotenv.2013.10.085.

Villarroel Walker, R., 2010. Sustainability beyond Eco-Efficiency: a Multi-Sectoral Systems Analysis for Water, Nutrients, and Energy. Doctoral dissertation, University of Georgia, Athens, United States of America.

Villarroel Walker, R., Beck, M., 2012. Understanding the metabolism of urban-rural ecosystems. Urban Ecosyst. 15 (4), 809–848. https://doi.org/10.1007/s11252-012-0241-8.

Vogl, A.L., Goldstein, J.H., Daily, G.C., Vira, B., Bremer, L., McDonald, R.I., Shemie, D., Tellman, B., Cassin, J., 2017. Mainstreaming investments in watershed services to enhance water security: barriers and opportunities. Environ. Sci. Pol. 75, 19–27. https://doi.org/10.1016/j.envsci.2017.05.007.

Voulvoulis, N., 2018. Water reuse from a circular economy perspective and potential risks from an unregulated approach. Current Opinion in Environmental Science & Health 2, 32–45. https://doi.org/10.1016/j.coesh.2018.01.005.

Wade, A.J., Whitehead, P.G., Butterfield, D., 2002. The Integrated Catchments model of Phosphorus dynamics (INCA-P), a new approach for multiple source assessment in heterogeneous river systems: model structure and equations. Hydrology and Earth System Sciences Discussions, European Geosciences Union 6 (3), 583–606 ffhal-00304711.

Waterworth, R.M., Richards, G.P., Brack, C.L., Evans, D.M.W., 2007. A generalised hybrid process-empirical model for predicting plantation forest growth. For. Ecol. Manag. 238 (1–3), 231–243. https://doi.org/10.1016/j.foreco.2006.10.014.

Wei, X., Bailey, R.T., Tasdighi, A., 2018. Using the SWAT model in intensively managed irrigated watersheds: model modification and application. J. Hydrol. Eng. 23 (10), 1–17. https://doi.org/10.1061/(ASCE)HE.1943-5584.0001696.

Weidema, B.P., Schmidt, J., Fantke, P., Pauliuk, S., 2018. On the boundary between economy and environment in life cycle assessment. Int. J. Life Cycle Assess. 23 (9), 1839–1846. https://doi.org/10.1007/s11367-017-1398-4.

Westenbroek, S.M., Kelson, V.A., Dripps, W.R., Hunt, R.J., Bradbury, K.R., 2010. SWB – a modified Thornthwaite-Mather Soil-Water-Balance code for estimating groundwater recharge. U.S. Geological Survey Techniques and Methods 6–A31, 60.

Wielemaker, R.C., Weijma, J., Zeeman, G., 2018. Harvest to harvest: recovering nutrients with new sanitation systems for reuse in urban agriculture. Resour. Conserv. Recycl. 128, 426–437. https://doi.org/10.1016/j.resconrec.2016.09.015.

Wintgens, T., Nattorp, A., Elango, L., Asolekar, S.R., 2016. Natural Water Treatment Systems for Safe and Sustainable Water Supply in the Indian Context: Saph Pani. IWA Publishing. https://doi.org/10.2166/9781780408392.

WWAP (United Nations World Water Assessment Programme), 2018. The United Nations World Water Development Report 2018: Nature-Based Solutions for Water. UNESCO, Paris. https://unesdoc.unesco.org/ark:/48223/pf0000261424.

Xue, X., Cashman, S., Gaglione, A., Mosley, J., Weiss, L., Ma, X.C., Cashdollar, J., Garland, J., 2019. Holistic analysis of urban water systems in the Greater Cincinnati region:(1) life cycle assessment and cost implications. Water Res. X 2, 100015. https://doi.org/10.1016/j.wroa.2018.100015.

Yang, W., 2011. A multi-objective optimization approach to allocate environmental flows to the artificially restored wetlands of China's Yellow River Delta. Ecol. Model. 222 (2), 261–267. https://doi.org/10.1016/j.ecolmodel.2010.08.024.

Zanni, S., Cipolla, S.S., di Fusco, E., Lenci, A., Altobelli, M., Currado, A., Maglionico, M., Bonoli, A., 2019. Modeling for sustainability: life cycle assessment application to evaluate environmental performance of water recycling solutions at the dwelling level. Sustainable Production and Consumption 17, 47–61. https://doi.org/10.1016/j.spc.2018.09.002.

Zhang, K., Manuelpillai, D., Raut, B., Deletic, A., Bach, P.M., 2019. Evaluating the reliability of stormwater treatment systems under various future climate conditions. J. Hydrol. 568, 57–66. https://doi.org/10.1016/j.jhydrol.2018.10.056.

Zhang, S., Yang, Y., McVicar, T.R., Zhang, L., Yang, D., Li, X., 2020. A proportionality-based multi-scale catchment water balance model and its global verification. J. Hydrol. 582, 124446. https://doi.org/10.1016/j.jhydrol.2019.124446.

Zhou, K., Barjenbruch, M., Kabbe, C., Inial, G., Remy, C., 2017. Phosphorus recovery from municipal and fertilizer wastewater: China's potential and perspective. J. Environ. Sci. 52, 151–159. https://doi.org/10.1016/j.jes.2016.04.010.

Ziogou, I., Michopoulos, A., Voulgari, V., Zachariadis, T., 2018. Implementation of green roof technology in residential buildings and neighborhoods of Cyprus. Sustainable Cities and Society 40, 233–243. https://doi.org/10.1016/j.scs.2018.04.007.