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Network Analysis as a tool for assessing environmental sustainability: Applying the ecosystem perspective to a Danish Water Management System

Massimo Pizzol [a,*], Marco Scotti [b], Marianne Thomsen [a,1]

[a] Department of Environmental Science, Aarhus University, Frederiksborgvej 399, 4000 Roskilde, Denmark
[b] The Microsoft Research – University of Trento Centre for Computational and Systems Biology, Piazza Manifattura 1, 38068 Rovereto, Italy

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A B S T R A C T

New insights into the sustainable use of natural resources in human systems can be gained through comparison with ecosystems via common indices. In both kinds of system, resources are processed by a number of users within a network, but we consider ecosystems as the only ones displaying sustainable patterns of growth and development. This study aims at using Network Analysis (NA) to move such "ecosystem perspective" from theory into practice. A Danish municipal Water Management System (WMS) is used as case study to test the NA methodology and to discuss its generic applicability. We identified water users within the WMS and represented their interactions as a network of water flows. We computed intensive and extensive indices of system-level performance for seven different network configurations illustrating past conditions (2004–2008) and future scenarios (2015 and 2020). We also computed the same indices for other 24 human systems and for 12 ecosystems, by using information from the existing scientific literature on NA. The comparison of these results reveals that the WMS is similar to the other human systems and that human systems generally differ from ecosystems. The WMS is highly efficient at processing the water resource, but the rigid and almost linear structure makes it vulnerable in situations of stress such as heavy rain events. The analysis of future scenarios showed a trend towards increased sustainability, but differences between past and expected future performance of the WMS are marginal. We argue that future interventions should create alternative pathways for reusing rainwater within the WMS, increasing its potential to withstand the occurrence of flooding. We discuss advantages, limitations, and general applicability of NA as a tool for assessing environmental sustainability in human systems.

1. Introduction

Human systems can be defined as series of interacting and interrelated man-made infrastructures and activities that provide a service to society. Examples of human systems are: systems for energy production and distribution (that include e.g. refineries, power plants, pipelines, and power lines), waste management systems (for the collection, recycling, and disposal of waste), transport systems (roads, railways, trucks, trains, and ships for moving people and goods). The persistence and future development of human systems are constrained by the challenge of making the transition towards sustainability possible (Pickett et al., 2008; Redclif, 1987). While the environmental performance of individual human activities may be quantified by measuring a single attribute (e.g. emission of CO_2 from fossil fuel-based energy production), human systems are complex entities where multiple users and technologies interact with each other. Therefore, information on multiple attributes must be combined to obtain a "systemic view" and a clear understanding of the nature of human systems. The use of indicators and indices (sets of weighted indicators derived by aggregating similar attributes measured in common units, e.g. Global Warming Potential GWP), can assist in this process (OECD, 2003). Nevertheless, even assuming that the environmental burden of human systems can be fully described by specific indicators, there is no apparent univocal metrics that absolutely quantifies sustainability *per se*. Thus, a common challenge in sustainability assessment of human systems is not only the choice of appropriate system indicators (Rowley et al., 2012), but also the identification of

* Corresponding author. Tel.: +45 871 58665.
E-mail addresses: mapi@dmu.dk, massimopizzol@gmail.com (M. Pizzol), marcoscot@gmail.com (M. Scotti), mth@dmu.dk (M. Thomsen).
[1] Tel.: +45 871 58665.

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sustainability-related attributes and of absolute metrics for sustainability.

Previous studies suggested the hypothesis that the most sustainable human systems are those mimicking ecosystems, and that environmental sustainability may thus be measured by comparing the two. This "ecosystem perspective" pertains to the concept of strong sustainability and has been proposed frequently and cross-disciplinarily in the scientific literature: in ecological economics (Azar et al., 1996; Ekins, 1993), ecology (Alberti et al., 2003; Bodini et al., 2012; Pickett et al., 2001), and industrial ecology (Andersson et al., 1998; Guinée and Heijungs, 2011; Odum and Odum, 2001). Ecosystems "*grow and decline in cycles that are repeatable and sustainable*" (Odum and Odum, 2001), and achieve an optimal resource management through the balanced interplay between growth (quantitative changes of the ecosystem's size measured in terms of total energy/matter processed within it) and development (qualitative changes in the ecosystem's structure, intended as levels of organization showed by energy/matter flows). Previous studies highlighted how e.g. cities can be seen as a "*particular example of ecosystem*" (Allesina and Bodini, 2008) where power and water supply networks resemble food webs, and where resource management represents a key factor in long-term urban planning (Agudelo-Vera et al., 2011). Since the ecosystem perspective – the idea that human systems and ecosystems are comparable (Pickett et al., 2011) – can be considered as "*more than a simple conceptual analogy*" (Allesina and Bodini, 2008), ecosystems may provide an absolute metrics for assessing the sustainability of human systems.

The subject of the present study is sustainable management of natural resources within human systems, which is a topic of increasing societal concern (e.g. EEA, 2011; UNESCO, 2009). In particular, we are interested in studying systemic aspects of sustainability, like human systems' efficiency in the use of resources, and the stability and resilience of human systems. These aspects can be investigated by using Network Analysis (NA) (Ulanowicz, 2004) as an environmental sustainability assessment tool. NA was originally conceived within the domain of the economic sciences and was first applied to ecosystem studies by Hannon (1973). There is an abundant literature concerning the use of the so-called "*Ecological Network Analysis*" in describing and understanding ecosystem dynamics (e.g., Baird and Ulanowicz, 1989; Bondavalli et al., 2006). Despite the wide application of NA in ecology, however, few studies relied on this tool to investigate sustainability in human systems (Bailey et al., 2004a,b; Bodini and Bondavalli, 2002; Bodini et al., 2012; Li et al., 2009; Li and Yang, 2011; Scotti and Vedres, 2012; Zhang et al., 2009, 2010, 2012).

Concerning sustainability assessment, the main features of NA are: (1) data on resource flows are organized in simple input–output matrices that can be obtained by applying well-established principles of material flow analysis; (2) intensive and extensive indices can be computed at system level to measure the performance of the system with respect to several sustainability-related aspects; (3) the method relies on a common element shared by human systems and ecosystems, i.e. the fact that both of them are constrained by limited resources; (4) NA allows for consistent comparisons between different systems and in particular between human systems and ecosystems.

We present the application of NA to assess the sustainability of the Water Management System (WMS) of Hillerød, a Danish municipality. Our objective is to use NA to provide realistic considerations on the potential for achieving the sustainable use of water within the WMS. Several tools have been applied for assessing the environmental performance and/or the sustainability of WMSs (e.g. Balkema et al., 2002; Muga and Mihelcic, 2008), including Water Footprint assessment (Hoekstra et al., 2011), Life-Cycle Assessment (Lundie et al., 2004; Lundin and Morrison, 2002), exergy assessment (Chen and Ji, 2007), and System Dynamics Simulation (Stave, 2003; Winz et al., 2009).

Recent research presented by Bodini and Bondavalli (2002), Bodini et al. (2012), Li et al. (2009), Li and Yang (2011), Zhang et al. (2010) shows that NA can be applied to study water as a resource and determine efficiency, resilience, and metabolic aspects in a WMS. In our study we extend the NA approach according to the hypothesis that new insights on the sustainability of human systems can be gained by comparing them with ecosystems. Our ambition is to use NA to move the ecosystem perspective from theory into practice. The Danish WMS is thus taken as a case study to present the potential of NA in this sense. Also, the case study is used to discuss the general applicability of NA as a tool for assessing sustainable management of natural resources in human systems.

Firstly, we introduce the case study and describe the modeling approach adopted for network construction. In particular we focus on novel aspects as the setting of a definite boundary between techno-sphere and natural environment, and the use of Material Flow Analysis (MFA) to calculate specific flows. Second, we show the evolution and performance of the WMS by considering past configurations and future scenarios and by using several system-level indices. Third, we compare the scores of indices computed for the WMS of Hillerød with outcomes extracted from the analyses of other human systems and ecosystems. We evaluate the sustainability trajectory of the WMS with respect to several sustainability-related systemic properties (quantifying e.g. growth and development), and we provide recommendations for improvements. Finally, we discuss advantages and limitations of using NA as a tool for assessing the ability of human systems to manage natural resources in an environmentally sustainable manner.

2. Material and methods

2.1. The Hillerød municipality

Hillerød is a Danish municipality located north-west of the country's capital Copenhagen, on the island of Zealand; it covers an area of approximately 200 km^2 and has around 48,000 inhabitants (2011 estimate). We discuss two critical issues regarding the water management within Hillerød: the risk of flooding in critical areas during heavy rain events, and its adaptability with respect to future growth.

Although flooding events occur very rarely under Danish climatic conditions, their frequency has increased in the last ten years. This led to partial flooding of some areas within Hillerød municipality due to the inability of the sewer system to cope with the excess rainwater (Larsen et al., 2012). As for many ancient Danish municipalities, the sewer system of Hillerød was not originally designed to collect wastewater and rainwater separately; only 29% of the catchment area has a sewer system specific for rainwater collection. Due to the mixing of rainwater, household, and industrial wastewater in the sewer system, the municipality of Hillerød is allowed to discharge into the environment the excess rainwater, without wastewater treatment, in emergency situations only. The municipal water and wastewater plans currently take into account a future increase in the frequency of heavy rain events (Hillerød and Orbicon, 2009; Hillerød and Watertech, 2006).

The municipality plans to develop and grow significantly in the oncoming years. A new hospital covering an area of approximately 200,000 m^2 will be built between 2016 and 2020, in the southern part of the Hillerød municipality. A significant infrastructural development is subsequently expected, leading to an increase in the number of households and industrial facilities and resulting in changes in water consumption and wastewater production.

As stated in the municipal drinking water plan and in the wastewater plan, Hillerød aims at achieving a sustainable future water management. The plans suggest to: (1) decrease human pressure on natural reservoirs by cutting per-capita consumption and collecting rainwater for domestic uses; and (2) avoid the loss of drinking water by reducing leakages from the water distribution pipelines.

2.2. Quantification of water flows within the WMS

The Hillerød WMS has been modeled as consisting of different users exchanging the water resource between them. The physical boundaries of the WMS are the administrative borders of the municipality. The theoretical boundaries of the WMS are between the techno-sphere and the natural environment. Techno-sphere is here defined as the whole spectrum of man-made infrastructures and human activities (alternatively identifiable with the economy). All the water that is withdrawn from the natural reservoirs (aquifers, rivers, lakes, sea, and clouds) and used by humans in production and consumption activities is accounted within the WMS. Water exchanged between natural reservoirs as part of the natural water cycle is not accounted within the WMS. We structured the WMS according to the parsimony principle: "*as simple as possible, as complex as needed*". We defined the users according to two criteria: (1) difference in function and characteristics; (2) significance for the modeling of future scenarios. Users are numbered in ascending order (1, 2, 3, etc.). We adopted the notation proposed by Allesina and Bodini (2008), which is well grounded in the domain of NA (Ulanowicz, 1986, 2004). Each water flow t (measured in m^3/year) between users was denoted according to its direction: t_{12} is the flow from user 1 to user 2 and t_{21} is the reverse flow direction from user 2 to user 1. Water withdrawals from the natural reservoirs into the WMS were named "*Imports*" (I), and wastewater effluents returned to the environment outside the WMS were named "*Exports*" (E). Water flows that undergo transformation into a form that cannot be reused in the system (e.g. steam or the water content in sludge) were named "*Dissipative flows*" (D). As such, the water import, export, and dissipation associated to e.g. user 1 are named t_{I1}, t_{1E}, and t_{1D}, respectively. The model of the WMS is shown in Fig. 1.

We identified seven users. The public water supplier PBWS (1) together with the private suppliers PVWS (2) extract drinking

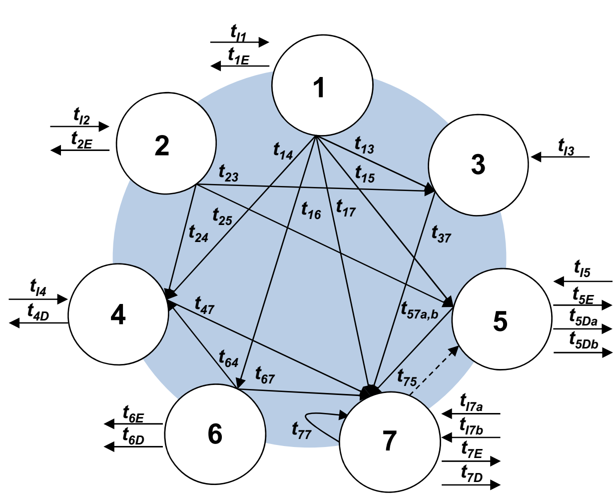

Fig. 1. Graphical representation of the WMS network. Intensities of water flows are summarized in Table 1. (1) PBWS, Water Supplier, Public; (2) PVWS, Water Supplier, Private; (3) PS, Public Service; (4) HH, Households; (5) ID, Industry; (6) EP, Energy Production; (7) WWT, Waste Water Treatment. I, Import; E, Export; D, Dissipation.

water from underground reservoirs and distribute it to the public sector PS (3), households HH (4), industry ID (5), energy production EP (6), and wastewater treatment WWT (7) (flows t_{13}, t_{14}, t_{15}, t_{16}, t_{17}, t_{23}, t_{24}, t_{25}, respectively). WWT receives the sewage produced by PS, HH, ID and EP (flows t_{37}, t_{47}, t_{57}, t_{67}, respectively) as well as the rainwater inflow to the sewer pipelines (t_{I7a}). *Exports* from the system are: leakages from drinking water pipelines (t_{1E}, t_{2E}), effluent wastewater discharged from EP into the freshwater bodies (t_{6E}), water loss via infiltration after irrigation (t_{5E}), and municipal wastewater effluent returned to freshwater bodies (t_{7E}). Apart from the groundwater extracted by PBWS and PVWS (t_{I1}, t_{I2}), rainwater inflow to the sewer pipelines (t_{I7a}) and private pumping wells (t_{I4}, t_{I5}, t_{I7b}) constitute *imports* to the system. Dissipation occurs in the form of steam from both ID (t_{5Da}) and EP (t_{6D}), and as water content in sludge from WWT (t_{7D}). The use of water for nutrition by humans (t_{4D}) and plants/animals (t_{5Db}) also represents a dissipative flow out of the WMS. Recycling options refer to district heating (t_{64}) and recirculation of cleaned wastewater by the WWT user (t_{77}).

Users PBWS and PVWS incorporate all facilities needed to extract and supply drinking water, i.e. both pumping wells and distribution pipelines. The same applies to user WWT, which incorporates all infrastructures needed to provide the service of wastewater treatment, i.e., both sewage pipelines and treatment plants (seven facilities of different size). User ID represents the industrial sector and includes the pharmaceutical industry – which has the highest water consumption – as well as other non-industrial activities like agriculture and horticulture. User EP includes a single facility for the combined production of heat and power from natural gas.

We were able to calculate five water balances for the WMS by using yearly data from 2004 to 2008. The value of each water flow between users was calculated based on publicly available data obtained from the official Danish National statistics website (Statistics Denmark, 2012), reports from DANVA – the Danish water and wastewater association (DANVA, 2010), the Hillerød water supply plan and wastewater plans (Hillerød and Orbicon, 2009; Hillerød and Watertech, 2006) and green accounts produced by different companies and by the wastewater treatment company (BioGen Idec, 2011; Hillerød, 2008; Novo Nordisk, 2011; Vattenfall, 2011). The amount of dissipated water by households is set equal to a human consumption of 1.2 l/(person·day) (Bodini and Bondavalli, 2002).

Due to data scarcity we were not able to retrieve direct information about all water flows in the WMS. Based on the principle of mass conservation, missing flows were estimated with the software STAN®, designed for material flow analysis (TUWIEN, 2011). The studied WMS is an open system, and we assumed no accumulation and no "stock" of water within the system during the 1-year time frame considered in the analysis. Therefore, the amount of water entering the system (I) equals the amount of water leaving the system (E + D). Water flow values used as input parameters for the NA, as well as details regarding their source and calculation method, are provided in Table 1.

2.3. Description of future scenarios for the WMS

Projections were calculated for two short-term future WMS scenarios at year 2015 and 2020. The scenarios take into account four characteristics that affect the WMS: (1) increased population and associated changes in water use; (2) improved water savings; (3) collection of rainwater by households; (4) re-use of rainwater collected in new sedimentation basins.

Based on a linear extrapolation of the observed population growth in Hillerød during the last ten years, we projected future populations of 49,700 inhabitants in year 2015 and 52,000 in year

Table 1
Water flows of the WMS (past and future scenarios), measured in m^3/year. Methods and data sources used to determine each flow are reported. Water users: (1) PBWS, Water Supplier, Public; (2) PVWS, Water Supplier, Private; (3) PS, Public Service; (4) HH, Households; (5) ID, Industry; (6) EP, Energy Production; (7) WWT, Waste Water Treatment. DW, Drinking Water; WW, Wastewater; GW, Groundwater; TC, Transfer Coefficient; HM, Hillerød Municipality.

Flow	2004	2005	2006	2007	2008	2015	2020	Flow description and calculation method
t_{13}	259,140	255,554	268,596	258,273	270,541	248,696	236,344	DW to PS, municipal supply (PBWS) – Reported data, 14% of total municipal supply; HM (Hillerød and Orbicon, 2009; Hillerød and Watertech, 2006)
t_{14}	1,573,348	1,551,575	1,630,760	1,568,088	1,642,568	1,451,806	1,343,786	DW to HH, municipal supply (PBWS) – Reported data, 85% of total municipal supply; HM (Hillerød and Orbicon, 2009; Hillerød and Watertech, 2006)
t_{15}	101,510	106,254	104,885	100,848	110,624	110,624	110,624	DW to ID, municipal supply (PBWS) – Reported data, 1% of total municipal supply; HM (Hillerød and Orbicon, 2009; Hillerød and Watertech, 2006)[a]
t_{16}	2934	2934	3257	4326	3335	3335	3335	DW to EP, municipal supply (PBWS) – Reported data; Green account (Vattenfall, 2011)[a]
t_{17}	2896	3400	4385	5741	4660	4660	4660	DW to WWT, municipal supply (PBWS) – Calculated via STAN®[a]
t_{1E}	146,009	144,495	151,432	145,816	152,926	116,114	89,408	Leakage, PBWS – Calculated via STAN®, TC = 7% of total input to PBWS
t_{23}	19,934	19,658	20,661	19,867	20,811	19,131	18,180	DW to PS, private supply (PVWS) – Reported data, 2% of total private supply; HM (Hillerød and Orbicon, 2009; Hillerød and Watertech, 2006)
t_{24}	936,889	923,924	971,077	933,757	978,108	867,829	805,389	DW to HH, private supply (PVWS) – Reported data, 94% of total private supply; HM (Hillerød and Orbicon, 2009; Hillerød and Watertech, 2006)
t_{25}	39,868	39,316	41,322	39,734	41,622	38,261	36,361	DW to ID, PVWS – Reported data, 4% of total private supply; HM (Hillerød and Orbicon, 2009; Hillerød and Watertech, 2006)
t_{2E}	86,669	85,469	89,831	86,379	90,482	59,057	45,259	Leakage, private supplier – Calculated via STAN®, TC = 8% of total input to PVWS
t_{37}	321,074	317,212	331,257	320,140	333,352	309,827	296,525	WW production, PS – Calculated via STAN®
t_{47}	2,513,846	2,478,989	2,604,180	2,504,389	2,622,358	2,409,391	2,288,751	WW production, HH – Calculated via STAN®
t_{4D}	18,863	18,982	19,998	20,303	20,397	21,764	22,751	Dissipation, HH biological consumption – Estimate, 1.2 l/person/d; (Bodini and Bondavalli, 2002)
t_{57a}	72,000	68,000	72,400	64,400	77,600	77,600	77,600	WW production, pharmaceutical industry – Reported data; Green account (Hillerød, 2008)[a]
t_{57b}	28,737	28,682	28,882	28,723	28,912	31,776	31,586	WW production, other ID – Calculated via STAN®, TC = 10% of total input to ID; TC Assumed
t_{5Da}	29,510	38,254	32,485	36,448	33,024	33,024	33,024	Dissipation, steam from pharm. ID – Calculated via STAN®[a]
t_{5Db}	172,421	172,090	173,293	172,340	173,473	190,657	189,516	Dissipation, biological (plants and animals) – Calculated via STAN®, TC = 60% of total input to ID; TC Assumed
t_{5E}	86,210	86,045	86,647	86,170	86,737	95,328	94,758	Export due to infiltration in soil (agriculture) – Calculated via STAN®, TC = 30% of total input to ID; TC Assumed
t_{64}	1472	1472	1341	1847	1079	1079	1079	District heating – Reported data; Green account (Vattenfall, 2011)[a]
t_{67}	227	227	191	230	200	200	200	WW production, EP – Reported data; Green account (Vattenfall, 2011)[a]
t_{6D}	462	462	1007	1414	1449	1449	1449	Dissipation, steam from EP – Reported data; Green account (Vattenfall, 2011)[a]
t_{6E}	773	773	718	835	607	607	607	WW to environment, EP – Calculated via STAN®[a]
t_{75}	0	0	0	0	0	32,000	32,000	Reuse of treated water in industry – Estimate (planned new basin volume)
t_{77}	342,280	284,961	229,578	231,666	238,272	238,272	238,272	Recycling of treated WW (clean) – Reported data; Green account (Hillerød, 2008)[a]
t_{7D}	4010	3874	3532	4306	4046	4046	4046	Dissipation, sludge – Reported data; Green account (Hillerød, 2008)[a]
t_{7E}	6,496,928	5,825,579	6,444,524	8,121,441	6,771,460	6,505,832	6,371,699	Treated WW (clean) – Estimate (95% of wet weight, 80% of dry weight); Green account (Hillerød, 2008)
t_{I1}	2,085,837	2,064,212	2,163,315	2,083,092	2,184,654	1,935,236	1,788,158	GW extraction (wells), PBWS – Calculated via STAN®
t_{I2}	1,083,360	1,068,367	1,122,891	1,079,737	1,131,023	984,277	905,189	GW extraction (wells), PVWS – Calculated via STAN®
t_{I3}	42,000	42,000	42,000	42,000	42,000	42,000	42,000	GW extraction, PS – Reported data; HM (Hillerød and Orbicon, 2009; Hillerød and Watertech, 2006)[a]
t_{I4}	21,000	21,000	21,000	21,000	21,000	110,441	161,248	GW extraction, rainwater collection HH – Reported data; HM (Hillerød and Orbicon, 2009; Hillerød and Watertech, 2006)
t_{I5}	247,500	247,500	247,500	247,500	247,500	247,500	247,500	GW extraction, ID – Reported data; HM (Hillerød and Orbicon, 2009; Hillerød and Watertech, 2006)[a]
t_{I7a}	3,562,083	2,932,562	3,406,752	5,202,029	3,705,651	3,705,651	3,705,651	Rainwater to WWT – Calculated via STAN®[a]
t_{I7b}	75	381	9	95	2773	2773	2773	GW extraction, WWT – Reported data; Green account (Hillerød, 2008)[a]

[a] 2015 and 2020 values are assumed equal to 2008 values.

2020, respectively. Per-capita household consumption is expected to follow the actual declining trend and to have reduced by approximately 5% in 2015 (Hillerød and Watertech, 2006). We modeled per-capita drinking water consumption of 48 m^3/(person·year) and 44 m^3/(person·year) for 2015 and 2020 respectively, with the current water consumption rate at 52 m^3/(person·year). Even if the building of new facilities for water supply and treatment in Hillerød is not planned, the sewer system is expected to be extended to connect new users (Hillerød and Watertech, 2006). We assumed a decrease in leakages for the public and private drinking water suppliers from the current level of 7% to a future level of 6%. The municipal wastewater plan encourages the collection of rainwater by households, in particular to be used in toilets and washing machines (Hillerød and Orbicon, 2009). According to the Danish Environmental Protection Agency (Miljøstyrelsen, 2002), the annual consumption for toilet flushing and laundry in households is approximately 18 m^3/(person·year). We assumed that the percentage of total population that will be able to use rainwater for toilet flushing and laundry will be 10% in 2015 and 15% in 2020 respectively. The estimated water amounts of 89,441 m^3 and 140,248 m^3 are added to the 2008 value of the flow t_{I4} to obtain the updated flows for the years 2015 and 2020 respectively, as reported in Table 1. Since the collected rainwater replaces drinking water to be used for the same purpose, we also subtracted the estimated future amount of rainwater collected by households from the estimated future amount of water distributed to households by PBWS and PVWS. This modeling approach allows accounting for the reduction of groundwater abstraction achievable with the collection of rainwater without affecting the amount of wastewater produced by HH (which is not dependent on the source used by HH to get the water). Additional calculation details are provided in the SI (Table S5). During heavy rain events, excess rainwater is stored temporarily in a series of sedimentation basins distributed within the territory of Hillerød, thus reducing the intensity of the water flow into the sewer system and risk of exceeding its capacity. As mentioned in Section 2.1 urban areas of Hillerød have in fact been flooded in recent years due to water amounts exceeding the capacity of the sewer system. In addition to reducing the intensity of water flows in to the sewer system, Hillerød municipality plans to increase the capacity of the existing basins by approximately 32,000 m^3 (Hillerød and Orbicon, 2009). Even if this is not explicitly mentioned in the wastewater plan, we hypothesize an optimal scenario where a volume of water corresponding to the 32,000 m^3 rainwater is reused yearly in ID (e.g., in horticulture, energy production and even in the pharmaceutical industry given additional processing to improve water quality).

For some water flows, like e.g. the rainwater flow to WWT, we were not able to estimate quantitative changes for the 2015 and 2020 scenarios. Such flows were kept equal to those reported in 2008; details are given in Table 1.

2.4. Network Analysis and system-level indices

In this section we provide an overview of the NA methodology and of the so-called "system-level indices". Equations and algorithms for the computation of the system-level indices are available in peer-reviewed literature (Allesina and Bodini, 2008; Latham, 2006; Scotti, 2008; Ulanowicz, 2004). In the terminology of NA each user constitutes a *node*, the resource exchanged between the nodes is the *currency* (in the present study the currency is water, measured in m^3/year), and the whole system of interconnected nodes is represented as a *network*. Networks are therefore the mathematical transposition of human systems or ecosystems. For each configuration of the WMS depicted in Fig. 1, water flows (m^3/year) have been organized in an input–output flow matrix $\mathbf{T}$ [t_{ij}] (see Table 2 and Table S1 in SI).

In NA, $\mathbf{T}$ is the basis for the calculation of the system-level indices. We computed the following system-level indices of environmental performance:

1) *Total System Throughput* (*TST*) provides information about system growth. *TST* measures the total amount of currency that is exchanged in the network and is calculated as the sum of all flows. As such *TST* is an extensive index (it depends on the size of the network).
2) *Average Mutual Information* (*AMI*) provides information on the level of system development by looking at the structure of the network. *AMI* relates the configuration of the flows in the network to the predictability of the pathways (see Fig. S1 in SI): networks composed of many redundant connections display lower *AMI* values than networks characterized by more rigid and linear-like structures. *AMI* is an intensive index (it does not depend on the amount of currency exchanged in the network but only on the structure of the network).
3) *Ascendency* (*A*) is defined as the product of *TST* and *AMI* ($A = TST \cdot AMI$) and is therefore an extensive index that combines information on both growth and development. *A* measures the fraction of currency that the network distributes in an efficient way. Since "efficiency" may be crudely defined as the ratio of output to input, in the case of NA we consider that the most efficient way to transfer a specific quantity of currency from one node to another is by minimizing the dissipation and avoiding unnecessary exports. Efficient networks are expected to minimize the difference between total currency in circulation and the consumption needs of the nodes.
4) *Development Capacity* (*C*) is the maximum value that can be attained by *A* given the total amount of currency exchanged. *C* is defined as the product of *TST* and the maximum measurable *AMI* ($C = TST \cdot AMI_{max}$) and is therefore an extensive index. *C* is used to normalize the ascendency and obtain A_{ratio} ($A_{ratio} = A/C$), an intensive index. In extremely developed systems the value of *A* will approach the value of *C* as the corresponding networks tend to eliminate redundant flows to maintain their structure well organized. Therefore, the higher the A_{ratio}, the more developed, efficient, and organized is the system.
5) *Overhead* (*O*) measures the degree of freedom in the organization of the flows within the network. It is an extensive index, estimated from the difference between development capacity and ascendency ($O = C - A$). *O* is usually scaled by dividing its value with *C* to obtain the intensive index O_{ratio}. A_{ratio} and O_{ratio} are complementary and sum to one. Path multiplicity and low levels of flow organization lead to higher levels of *O*. *O* can be divided into four separate contributions, each related to a certain form of pathway multiplicity: overhead on import (O_I), overhead on export (O_E), dissipative overhead (O_D), and internal transfer overhead, also called redundancy (*R*); $O_I + O_E + O_D + R = O$. These can be normalized by *C* to obtain the intensive indices $O_{I\text{-}ratio}$, $O_{E\text{-}ratio}$, $O_{D\text{-}ratio}$, R_{ratio}; $O_{I\text{-}ratio} + O_{E\text{-}ratio} + O_{D\text{-}ratio} + R_{ratio} = O_{ratio}$.
6) The *Finn Cycling Index* (*FCI*) was introduced by Finn (Finn, 1980) to measure the amount of currency recycled relative to the total currency that is processed in the network. In contrast with *A*

Table 2
Example of input–output matrix $\mathbf{T}$ illustrating the WMS configuration for the year 2008. Water users: (1) PBWS, Water Supplier, Public; (2) PVWS, Water Supplier, Private; (3) PS, Public Service; (4) HH, Households; (5) ID, Industry; (6) EP, Energy Production; (7) WWT, Waste Water Treatment. I, Import; E, Export; D, Dissipation. Water flows are measured in m^3/year.

2008	I	1	2	3	4	5	6	7	E	D
I	0	2184653.76	1131022.83	42,000	21,000	247,500	0	3708423.80	0	0
1	0	0	0	270,541	1,642,568	110,624	3335	4660	152925.76	0
2	0	0	0	20,811	978,108	41,622	0	0	90481.83	0
3	0	0	0	0	0	0	0	333,352	0	0
4	0	0	0	0	0	0	0	2,622,358	0	20,397
5	0	0	0	0	0	0	0	106512.20	86736.60	206497.20
6	0	0	0	0	1079	0	0	200	607	1449
7	0	0	0	0	0	0	0	238,272	6,771,460	4046
E	0	0	0	0	0	0	0	0	0	0
D	0	0	0	0	0	0	0	0	0	0

which relates the amount of currency processed to the network configuration, *FCI* provides a crude measure of efficiency. For instance, a network where a single node reuses 10% of the total currency circulating would have the same *FCI* of another network recycling the same amount by means of all the nodes. However, based on *A* the latter configuration would be classified as less constrained and characterized by a higher structural efficiency (as it preserves multiplicity of pathways). *FCI* is an intensive index.

We calculated system-level indices for seven network configurations: years 2004–2008 and future scenarios of years 2015 and 2020. All the system-level indices have been computed using the software R (R Development Core Team, 2005).

2.5. Comparison of human systems and ecosystems

Since NA allows to assess different systems with the same intensive indices, no matter their size, architecture or currency, we compared past and future configurations of the Hillerød WMS with 24 networks describing human systems for the management of water (Bodini and Bondavalli, 2002; Li et al., 2009; Li and Yang, 2011; Zhang et al., 2010) and other resources (Scotti and Vedres, 2012; Zhang et al., 2009).

We also examined the environmental performance of the Hillerød WMS with respect to patterns estimated for 12 ecosystems (Almunia et al., 1999; Baird et al., 1998; Baird and Milne, 1981; Baird and Ulanowicz, 1989; Krause, 2004; Monaco and Ulanowicz, 1997; Patricio et al., 2004; Ulanowicz, 1986; Ulanowicz et al., 1997, 1998, 1999, 2000). These ecosystems portray predator-prey relationships within freshwater, marine, or terrestrial environments. The ecosystems are described by networks of energy/matter flows where different kinds of currency are used. The size of these ecosystems ranges from 13 to 125 nodes, with each node representing a species, a trophospecies (a set of species with similar diet and predators), or a non-living compartment (e.g., detritus). Data on ecosystems were extracted from the websites of ATLSS and of the Prof. Robert E. Ulanowicz (see "Web Reference list") representing the most reliable repositories of predator-prey networks (i.e., these networks were constructed using the same criteria and data are consistent also if they refer to different ecosystems).

3. Results

3.1. System-level indices

The input–output **T** matrix relative to year 2008 is provided in Table 2, whereas all the **T** matrices calculated in this study – one for each network configuration corresponding to the WMS in years 2004–2008, and one for each of the 2015 and 2020 WMS scenarios – are presented in the Supporting information (Table S1, SI).

The entire set of system-level indices calculated for the WMS is reported in Table 3, together with lower, average, and upper values of intensive system level indices calculated for a set of 12 ecosystems. We observed that the WMS has a high A_{ratio} value (and consequently a low O_{ratio} value). This means that the system is close to its maximum level of organization of flows and tends to eliminate redundancy. For ecosystems, increasing values of A_{ratio} define a tendency to become "*rigidly linked and almost mechanical*" (Ulanowicz, 2004), "*brittle*" and "*vulnerable to collapse*" (Holling, 1985), whereas increasing O_{ratio} values can instead be interpreted as a measure of system inefficiency in processing a resource. Nevertheless, at times of stress and perturbation the presence of redundant routes for currency circulation can represent an advantage in terms of resilience and system adaptability. This interpretation applies perfectly to the case of Hillerød where the WMS has a constrained structure that works efficiently under normal conditions, but has difficulties in managing heavy rain events due to a limited potential to create alternative pathways for the transport and use of rainwater.

Table 3 shows that the values of the system-level indices estimated for the future scenarios (2015 and 2020) change as expected: *AMI* and A_{ratio} decreases due to the presence of additional flows (t_{75}) and thus resulting in a less constrained structure; *TST* decreases because of a decline in water consumption (less currency circulating in the system); recycling increases (*FCI* grows after the minimum values recorded for the years 2006–2007). However, the differences observed between indices calculated for future scenarios and for the other configurations (2004–2008) are statistically not significant (Kolmogorov–Smirnov test, $p >> 0.05$).

3.2. Comparison with other human systems and with ecosystems

Table 3 shows that intensive indices computed for the Hillerød WMS do not compare with the average values calculated for

Table 3
Extensive (*TST, C, A, O,* O_I, O_E, O_D, *R*) and intensive (*AMI*, AMI_{max}, A_{ratio}, O_{ratio}, $O_{I\text{-}ratio}$, $O_{E\text{-}ratio}$, $O_{D\text{-}ratio}$, R_{ratio}, *FCI*) system-level indices estimated for the WMS relatively to past configurations (2004–2008) and future scenarios (2015, 2020). Size (# nodes), number of interactions (# links) and currency are specified. Lower (L), average (A), and upper (U) values of intensive system level indices calculated for a set of 12 ecosystems are reported.

	2004	2005	2006	2007	2008	2015	2020	Ecosystems		
# Nodes	7	7	7	7	7	7	7	L	A	U
# Links	14	14	14	14	14	15	15			
Currency	Water [m^3/year]	Water [m^3/year]	Water [m^3/year]	Water [m^3/year]	Water [m^3/year]	Water [m^3/year]	Water [m^3/year]			
TST	2.03E+07	1.88E+07	2.03E+07	2.34E+07	2.10E+07	1.990E+07	1.923E+07			
AMI	1.395	1.431	1.432	1.372	1.420	1.378	1.359	1.264	1.755	2.250
C	6.25E+07	5.90E+07	6.27E+07	6.82E+07	6.44E+07	6.116E+07	5.889E+07			
AMI_{max}	3.080	3.135	3.085	2.909	3.062	3.073	3.062	3.154	4.442	5.457
A	2.83E+07	2.70E+07	2.91E+07	3.21E+07	2.99E+07	2.742E+07	2.614E+07			
A_{ratio}	0.453	0.457	0.464	0.472	0.464	0.448	0.444	0.331	0.400	0.526
O	3.42E+07	3.21E+07	3.36E+07	3.60E+07	3.45E+07	3.374E+07	3.275E+07			
O_{ratio}	0.547	0.544	0.536	0.529	0.536	0.552	0.556	0.474	0.600	0.669
O_I	1.56E+07	1.46E+07	1.56E+07	1.71E+07	1.61E+07	1.602E+07	1.581E+07			
$O_{I\text{-}ratio}$	0.249	0.247	0.249	0.251	0.249	0.262	0.269	0.029	0.098	0.197
O_E	3.91E+06	3.76E+06	3.84E+06	3.86E+06	3.89E+06	3.381E+06	2.982E+06			
$O_{E\text{-}ratio}$	0.063	0.064	0.061	0.057	0.060	0.055	0.051	0.004	0.028	0.088
O_D	4.95E+05	4.93E+05	5.07E+05	5.20E+05	5.26E+05	5.446E+05	5.505E+05			
$O_{D\text{-}ratio}$	0.008	0.008	0.008	0.008	0.008	0.009	0.009	0.085	0.166	0.260
R	1.42E+07	1.33E+07	1.36E+07	1.45E+07	1.41E+07	1.380E+07	1.340E+07			
R_{ratio}	0.228	0.225	0.218	0.213	0.218	0.226	0.228	0.206	0.308	0.453
FCI	0.026	0.023	0.017	0.016	0.017	0.019	0.020	0.018	0.134	0.507

ecosystems (Kolmogorov–Smirnov test, $p << 0.01$). We measured higher values of AMI/AMI_{max}, and lower values of O_{ratio} and FCI than in ecosystems. Networks for Hillerød WMS are characterized by abundant input resulting in a higher $O_{I\text{-}ratio}$. Water is linearly distributed (lower R_{ratio}) and unevenly exploited (lower $O_{D\text{-}ratio}$), whereas the majority of this resource is returned back to environment (higher $O_{E\text{-}ratio}$).

Fig. 2 compares AMI with AMI_{max} for each of 43 networks: 12 ecosystems and 31 human systems (the seven related to the Hillerød WMS – configurations during the period 2004–2008 and future scenarios for 2015 and 2020 – and 24 other human systems). In all seven configurations the Hillerød WMS lies between natural and human systems in terms of AMI, whilst AMI_{max} scores are lower than those of ecosystems but comparable to those of other human systems.

In Fig. 2 a clear separation between the group of ecosystems (located in the center of the chart) and the group of human systems (closer to the origin of the chart) can be observed. This can be interpreted as the tendency of human systems to have low structural complexity, high levels of organization, and prevalence of almost linear pathways. The AMI/AMI_{max} ratios estimated for the WMS of Hillerød are higher than the majority of the values measured for ecosystems; when the regression between AMI and AMI_{max} is computed for the scores of the 12 ecosystems, the points that refer to the WMS score lie below the regression line (see the two-dashed line in Fig. 2). Although the relative development of the WMS displays patterns consistent with ecosystems (i.e., the normalized outcomes are close to what can be observed for natural systems; see also A_{ratio} in Table 3), a clear difference emerges when considering AMI_{max}. The WMS of Hillerød preserves the potential for better exploiting the available water (this is mainly due to the large amount of rainwater). However, in absolute terms the upper limit to the development of the WMS is much more constrained than in ecosystems (lower AMI_{max}). In general, networks with a lower AMI/AMI_{max} ratio can be considered to have lower efficiency in terms of currency exchange, but are more flexible in their structure as they possess redundant links (i.e., there is a multiplicity of pathways for energy/matter flows). These alternative connections are important to achieve resilience, i.e. to be able to withstand extreme events or situations of stress (for the case of water management, these could be the collapse of pipes in the water distribution system or an excess of rainwater). Number of nodes and links, information about the currency circulating, and values of system-level indices calculated for 24 human systems and 12 ecosystems are reported in the Supporting information (Tables S2 and S3 respectively, SI).

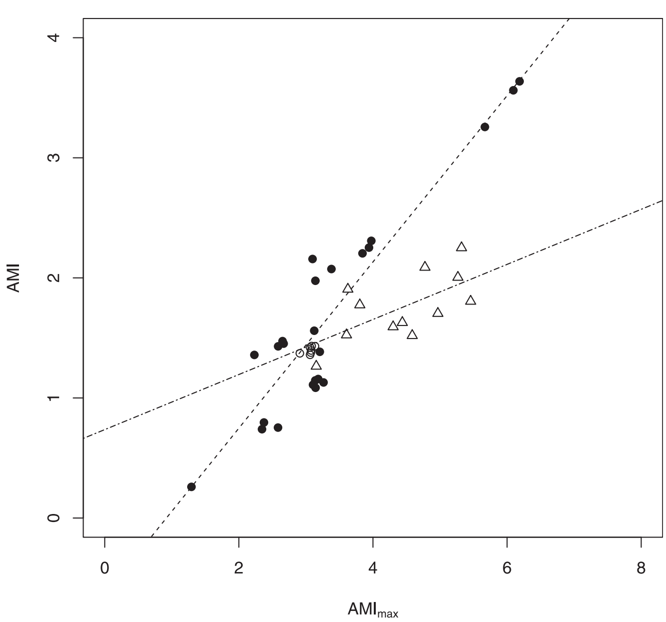

Fig. 2. *AMI* (unitless) versus AMI_{max} (unitless). Empty circles stand for WMS (configurations 2004–2008 and future scenarios 2015, 2020); Solid circles correspond to the 24 human networks; Empty triangles represent the 12 ecosystems. Regression lines obtained from: 24 human networks only (dashed line, adjusted R-squared: 0.8530, $p << 0.001$); 12 ecosystems only (dot-dashed line, adjusted R-squared: 0.3374, $p = 0.028$).

4. Discussion

4.1. System performance

If compared to the intricate architecture of ecosystems, the WMS is simpler in terms of flow organization (lower AMI); in human systems the redundant pathways are scarce, while ecosystems are characterized by less organized structures. The WMS is more efficient than ecosystems because a high quantity of resource can be controlled and exchanged in the network through a limited number of pathways (higher A_{ratio}). In the WMS the vertical hierarchy of the nodes is shorter than in ecosystems and pathways are almost linear (lower value of AMI_{max}). The circularity of the currency in the network is minimal (lower value of FCI), implying that water is not recycled inside the WMS but is mostly imported, used, and exported to the environment.

The analysis shows that future interventions and actions planned by Hillerød municipality point towards increased sustainability. In fact, they reduce the use of resource and increase the number of connections, thus bringing the WMS performance closer to that of ecosystems. However, these interventions (1) do not significantly modify the performance of the WMS compared to past configurations and (2) do not make the WMS comparable with ecosystems. Indeed, the WMS preserves high levels of efficiency at the expense of flexibility (A_{ratio} still high compared to ecosystems), and a drastic reduction in the risk of flooding appears unlikely. To improve sustainability and achieve higher resilience, thus yielding a system that is both efficient and able to withstand stress events, several strategies should be pursued: (1) the skewed distribution of water circulating in the system should be reduced by promoting the reuse of rainwater (this would further increase the O_{ratio}); (2) pathway redundancy should be fostered by creating new connections (e.g., through the construction of a separate sewage system) that permit to redirect or discharge water at times of excessive rainfall; (3) an increase in FCI should be achieved by recycling water in the system, e.g. from WWT to HH, ID, EP. Since these strategies can be effective only with the simultaneous involvement of several water users (household, public sector, industry), the joint and combined action of different economic sectors appears essential for planning a sustainable development; such a point of view is shared with previous studies on sustainability (Shin et al., 2008; Ukidwe and Bakshi, 2004).

4.2. Modeling uncertainties

There are a number of alternatives regarding the way of constructing a network. Although these have been discussed – in the literature on Network Analysis – for the specific case of ecosystems (Fath et al., 2007), there are presently no guidelines about how to model human systems (i.e., no clear rules exist regarding the optimal number of nodes and the type of users that should be included in the network). In principle, every natural compartment or user affected by anthropic activities (both directly and indirectly) could be included within a "human system". In modeling the

Hillerød WMS we adopted a parsimonious approach. Users represent economic sectors or services, rather than physical entities (e.g., a single wastewater treatment plant). Previous studies of water management, such as Kenway et al. (2011) and Lim et al. (2010), applied a comparable approach for the modeling of urban water systems, albeit differences in the level of resolution can be observed. Increasing the resolution of the analysis might allow for consideration of additional flows and user-enclosed sub-networks (e.g. the exchange of drinking water between different pumping stations inside the PBWS). However, this might lead to higher levels of parameter uncertainty due to missing data concerning sub-flows within the WMS.

In comparison to other studies of Network Analysis, there are some notable differences in the way we constructed the WMS network. Zhang and co-workers (Zhang et al., 2009, 2010) assembled more circular networks by considering "environment" to be a node, whereas we included in the WMS only activities related to economy and belonging to the techno-sphere (i.e., water users correspond to infrastructures and activities that are man-made or driven by humans). We believe that our choice allows for a more objective definition of system boundaries. Since human activities are limited by environmental constraints such as availability of resources and ecosystem services, we assert that including the environment as a component of the human system would concur to underestimate its role. Moreover, the matrices provided by Zhang and co-workers describe networks without cross-boundary flows. Since import, export, and dissipative flows are absent, their networks are almost fully connected; the *FCI* inevitably equals unity and it is not possible to calculate separately *O* and *R* (as well as their normalized versions).

We did not include nodes representing surface water bodies and reservoirs as was done by Bodini and Bondavalli (2002) and by Li and colleagues (Li et al., 2009; Li and Yang, 2011). This limits our possibility of addressing the excessive exploitation of freshwater and groundwater. On the other hand, uncertainties related to the exact quantification of exchanges between water bodies are avoided (e.g., yearly water load to freshwater and groundwater reservoirs, water interchange between streams and groundwater bodies). The exclusion of nodes representing components of the environment (e.g., water bodies) results in a reduction of the circular pathways within the network. This may influence the value of the system-level indices, in particular by reducing the *FCI*.

Since estimates related to drinking water supply and wastewater treatment were based on direct information from Hillerød, the uncertainty on these values is expected to be less than 10%. The reliability of the data in the industry sector is expected to carry the highest uncertainty due to the biological dissipation of water. However, maximal fluctuation in flow intensities should not exceed 20% of their value. Results show that for variations up to 20% of flow intensity the system-level indices are stable (i.e., no significant changes characterize their value – see Fig. S2, SI). The most impacted index is *C*, whereas *TST* is extremely stable. Although we did not estimate the uncertainty associated with the intensity of each flow component, simulations confirmed the robustness of our results (provided variation in flow intensity is less than 20%).

The use of STAN® allowed estimating the flows in the WMS when primary data were not available. The estimate of rainwater inflow (m^3/year) to the WWT cannot be calculated by multiplying the amount of local precipitation (mm/year) by the area of the municipality (km^2), because not all the rainwater is conveyed to the sewer system. Part of the rainwater falls over green areas (gardens, forest, and arable land) and blue areas (freshwater bodies such as rivers and lakes). This estimate has not been validated. Yet, the two parameters should be correlated. We found a significant correlation between actual precipitation and estimated rainwater inflow (R-squared $= 0.8183$; $p < 0.001$, see Table S4, SI); if not completely accurate, the calculation via STAN® is considered precise.

Some approximations were done in the analysis of future scenarios regarding the estimate of population growth, a parameter that influences the intensity of several flows within the WMS. Although the linear extrapolation may not be the best solution to infer population growth, it represents an acceptable approximation if we consider that the population of Hillerød increased, during the last 5 years, following a linear trend. Moreover, our linear extrapolation is reasonable as it pertains to the near future (a horizon of no more than 10 years) and does not refer to the far future (e.g., 100 years). These projections seem also realistic if we take into account the planned expansion of the city (Hillerød and Watertech, 2006). In the present study the analysis of future scenarios is primarily a research exercise to explore the potential of NA as a tool for sustainability assessment. For this reason, we argue that the level of accuracy provided by the linear approximation can be considered sufficient to reach the initial objectives.

Regarding uncertainty of results, a key issue is understanding whether the non-significant differences between past and future performance of the WMS may be due to the failure of system-level indices in capturing the changes concerning different scenarios, rather than depending from a real absence of change. We believe that the indices are robust in measuring the system performance for three reasons: (1) the large amount of rainwater flowing through WWT (t_{17a} is the largest contributor to *TST*) is masking the system-level consequences of managing strategies put in action for improving water use in future scenarios; (2) flows within a network are non-independent data and a non-parametric approaches must thus be used to test the significance between sample outcomes extracted from NA; in our case, however, we are dealing with a small sample (5 past configurations of the WMS are compared with 2 future scenarios) and this probably affects the predictive power of the Kolmogorov–Smirnov test; (3) planned changes appear too localized to produce a significant shift, in the direction of sustainability, at whole system-level (only one additional connection is created, few users are involved, and modest amounts of water are recycled compared to the *TST*).

4.3. NA as an environmental sustainability assessment tool

NA allows studying human systems with the consistent evaluation of different systemic properties (e.g., growth and development can be measured with *TST* and *AMI*, respectively). Insights on the level of sustainable use and exchange of resources within a human system can be obtained by evaluating the system's evolution (through the analysis of time series showing different states of the same network) according to different resource management strategies. For instance, in the WMS a raise in drinking water demand may be faced by increasing the water withdrawal and thus the pressure on the natural reservoirs (non-sustainable strategy) or by promoting the reuse of the resource (more sustainable strategy). The first option would be captured by analyzing trends of *TST* (and it would result in higher values of this index), while the second would be reflected by lower *AMI* and A_{ratio}, and by an increase of the *FCI*.

Also, sustainability can be evaluated by comparing human systems with ecosystems, using the same indices. The differences between human systems and ecosystems – in terms of structural complexity (AMI/AMI_{max}), efficiency of resource management (*FCI*), and capacity to withstand stress by maintaining redundant pathways (O_{ratio}) – can be quantified setting the performance of ecosystems as an absolute measure of sustainability (strong sustainability; consider that NA is a mathematic tool and does not provide any metrics for system sustainability *per se*).

It could be debated whether differences across the systems (human systems and ecosystems) used in our analysis may impede any evaluation about sustainability. Although these systems are of different type, the comparison is consistent as it was based on system-level indices that are not biased by network size, topology, currency, and time scales. Table 3 shows that average A_{ratio} in ecosystems is 0.4, a value that is clearly different from what is displayed by human systems. Despite differences regarding network construction (e.g., the inclusion/exclusion of natural compartments – cf. Section 4.2) human systems perform clearly different from ecosystems. We observe that the comparison of different systems appears more straightforward with NA than with other sustainability assessment tools like Life Cycle Assessment (LCA), where full equivalence of system boundaries between systems is required.

In our study we neither focused on the role played by single users in terms of dominance/dependency relationships (Bodini and Bondavalli, 2002) nor extracted trophic structures, i.e. unfold the network complexity in terms of primary/secondary producers and consumers – see Baird and Ulanowicz (1989) and Zhang et al. (2012). Although these features appear to be of little interest when analyzing a simple network such as the one presented in this study, they could be of great importance in the analysis of more complex configurations (i.e., networks obtained by increasing the number of nodes and interactions). We highlight two other shortcomings of NA. A weakness of NA may be the limited ability to take into account changes in the quality of the resource flows, as qualitative aspects are assessed only indirectly. For instance, in the Danish WMS water flows are grouped according to suitability for use defined according to quality criteria for individual use categories (e.g. rainwater may be used for toilet flushing but not for drinking). Thus, an improvement in water quality is accounted for only if this either induces a variation in the absolute value of one or more flows, e.g. by triggering an increase in the use of water through a certain pathway, or if it leads to the creation of new flows thus increasing circularity, e.g. recycling of cleaned wastewater in industry. Yet the advantage of discharging clean rather than polluted wastewater into the environment is not directly quantifiable by NA.

A similar concern has been expressed for the water footprint (Chapagain et al., 2006), where the consequences of polluting have been included by converting the amount of emitted chemicals into the dilution volume required for their assimilation in water bodies. For the case of NA, potential solutions could be either reducing the intensity of some flows, as a consequence of lower water quality, or changing the currency and using exergy (Chen and Ji, 2007; Patten, 1995) instead of water volume. Secondly, NA accounts for only a single currency, whereas other tools such as LCA can handle multi-level trade-offs. For instance, in LCA the consumption of energy and the emission of toxic substances arising from wastewater treatment can be evaluated against the potential of the resulting high-quality water product for being re-used elsewhere in the system (Houillon and Jolliet, 2005). The use of exergy seems a promising approach to account simultaneously for multi-currency flows in NA. When flows of different currencies are converted into exergy, they can be included in the same network. Examples of conversion of different material flows into exergy have been proposed for LCA (Finnveden and Östlund, 1997; Ukidwe and Bakshi, 2004) with the aim of measuring resource depletion, and such approach could be applied to NA; this may as well create new possibilities for the combination of the two tools.

We believe that the analysis here performed for the case of water could be extended directly to the study of other natural resources and human systems. The works of Scotti and Vedres (2012) and Zhang et al. (2012), addressing respectively natural gas distribution within an European network and multiple resource flows in China, show that NA can be applied to study human systems for the management of different resources at different geographical scales. By aggregating NA data from previous studies, we also showed that a comparison between human systems and ecosystems is possible regardless of the unit and scale adopted.

Resources which management is expected to become critical in Europe in the near future may represent interesting case studies. Examples are: (1) biodiesel, relevant for its role in energy security and its production-related environmental impacts (Demirbas, 2009); (2) electronic waste (e-waste), relevant for its mixed content in toxic and valuable metals (copper, gold, palladium, nickel) and for its growing management system covering the entire EU area (Widmer et al., 2005); (3) phosphorus, relevant because of the global scarcity of phosphate rock reserves, controlled by extra-EU countries (Arab Nations, China, and Russia) (Cordell et al., 2011), and for the increasing interest in the industrial recovery of phosphorus (Morse et al., 1998). For these resources, the application of NA for the sustainability assessment of existing and future management strategies may allow a more comprehensive appraisal, beyond traditional estimates of resource use efficiency. We argue that, by applying the ecosystem perspective, the systemic capacity to withstand perturbations such as extreme climatic events, economic crisis, or wars may be in fact evaluated.

5. Conclusions

In this study we addressed the topic of sustainable use of natural resources in human systems, and we studied the case of water use within a Danish municipal Water Management System. We proposed a sustainability assessment approach consisting in the quantitative comparison between human systems and ecosystems via common indices calculated by means of Network Analysis. In both kinds of system, resources are processed by a number of users within a network, but following the paradigm of strong sustainability we define ecosystems as the only ones showing sustainable patterns of growth and development. The results of the NA reveal that the WMS differs from ecosystems: it is highly efficient at processing the water resource (higher values of A_{ratio} than in ecosystems), but shows a rigid structure where water flows follow linear pathways (lower AMI_{max} and FCI compared to ecosystems). In ecosystems, these conditions lead to vulnerability in situations of stress. This interpretation describes perfectly the WMS, where flooding episodes occur in concomitance with heavy rain events due to the lack of alternative pathways to transport the excess of rainwater. The analysis of future scenarios showed a trend towards increased sustainability, but differences between past and expected future performance of the WMS are marginal. We suggest that future interventions should aim at improving structural flexibility within the WMS (measurable in terms of higher O_{ratio}). This could be achieved by creating alternative pathways for the rainwater circulation (e.g., by means of a separate system for collecting rainwater independently from sewage water, thus allowing to reuse rainwater in the WMS or to discharge it into natural water bodies).

We are aware that several methodologies need to be brought into play to address sustainability of human systems, depending on the goal and scope of a study. However, we do believe that NA is of added value for assessing sustainable use of natural resources within human systems by addressing sustainability-related systemic properties and by using patterns of resource management within well-characterized ecosystems as reference. Although the use of ecosystems as absolute metrics for measuring the sustainability of human systems has been proposed and debated in the literature at the theoretical level, quantitative approaches are scarce. Nevertheless, we presented an extended set of data for comparing the performance of human and natural systems in the use and exchange of resources. We thus believe that our study advances the state of the art in the field of sustainability assessment, as we showed explicitly how NA can be applied to perform quantitative comparisons between human and natural systems.

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Appendix A. Supporting information

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jenvman.2012.12.042.

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