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Water-energy nexus for urban water systems: A comparative review on energy intensity and environmental impacts in relation to global water risks

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HIGHLIGHTS

- This study quantifies the nexus as energy intensity and greenhouse gas potential.
- Baseline water stress and return flow ratio are identified as water risks.
- Source water accessibility significantly contributes to variations in the nexus.
- Water risks have little impact on the nexus of wastewater systems.
- Study on the nexus is suggested to be conducted at regional levels.

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ABSTRACT

The importance of the interdependence between water and energy, also known as the water-energy nexus, is well recognized. The water-energy nexus is typically characterized in resource use efficiency terms such as energy intensity. This study aims to explore the quantitative results of the nexus in terms of energy intensity and environmental impacts (mainly greenhouse gas emissions) on existing water systems within urban water cycles. We also characterized the influence of water risks on the water-energy nexus, including baseline water stress (a water quantity indicator) and return flow ratio (a water quality indicator). For the 20 regions and 4 countries surveyed (including regions with low to extremely high water risks that are geographically located in Africa, Australia, Asia, Europe, and North America), their energy intensities were positively related to the water risks. Regions with higher water risks were observed to have relatively higher energy and GHG intensities associated with their water supply systems. This mainly reflected the major influence of source water accessibility on the nexus, particularly for regions requiring energy-intensive imported or groundwater supplies, or desalination. Regions that use tertiary treatment (for water reclamation or environmental protection) for their wastewater treatment systems also had relatively higher energy and GHG emission intensities, but the intensities seemed to be independent from the water risks. On-site energy recovery (e.g., biogas or waste heat) in the wastewater treatment systems offered a great opportunity for reducing overall energy demand and its associated environmental impacts. Future policy making for the water and energy sectors should carefully consider the water-energy nexus at the regional or local level to achieve maximum environmental and economic benefits. The results from this study can provide a better understanding of the water-energy nexus and informative recommendations for future policy directions for the effective management of water and energy.

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1. Introduction

Water and energy are two key components in the global search for sustainable development [1] in response to the United Nations Sustainable Development Goals. Thus, the water-energy nexus is increasingly highlighted as an important issue for future sustainability planning and strategic policy considerations in literatures [2], especially since the two resources are highly vulnerable to the impacts of global climate change [3]. The Ontario Government has identified the water-energy nexus as deeply connected within the context of climate change [4]. In addition, a large share the global population is experiencing water scarcity due to climate change [5]. Lack of understanding of the interdependence between water and energy (or even among other key natural resource sectors) within a system may lead to overuse and mismanagement of resources [6]. Several countries have initiated projects to study the extent of their energy-water nexus and have sketched out future policy directions. In the USA, the Energy and Water Research Integration Act of 2011 was formulated (but was not enacted) to draw policy attention to integrate water considerations into energy-related research [7]. South Africa, Morocco, Mexico and China [8] are also working on incorporating water constraints into their energy plans [9]. However, most of the studies have been based on energy systems (e.g., because of the relative priority and importance of the impacts of energy production and usage) [10]. The energy requirement by water systems (i.e., energy for water) has been less studied [11] and the urban water systems are often managed separately [12].

Water systems are one of the major users of energy resources [13]. The level of energy requirement per unit of water (e.g., energy intensity) strongly depends on the processes involved and the water quality level before end use [14]. For instance, in Spain, the specific level of energy consumption per unit of delivered water is reported as 0.21, 0.34 and 0.56 kWh/m^3 for urban users, agriculture and wastewater treatment for recycling, respectively [14]. Fig. 1 summarizes the range of energy intensity at various stages of a typical urban water cycle using average values of benchmarking studies. Differences in these values also reflect the variety of boundary conditions of the studies, as well as other influential factors such as the type and quality of the source water and the efficiency of the water treatment and delivery system [15]. This suggests that greater focus on the energy requirement of the water systems will be a crucial point of the policy response to the sustainable management of the systems.

For sustainable management of urban water systems, a significant effort needs to be undertaken to improve water and energy use efficiency, which can consequently reduce their associated environmental impacts [16]. Jointly improving these efficiencies has been regarded as a win-win contribution to human well-being and environmental sustainability for current and future generations [6]. The experiences from the United States, Australia and several European countries (including Spain, Norway, Italy, etc.) have provided informative suggestions for future policy directions for resource management [17] and the need to study water-energy sustainability within an urban water system. A study of energy use by urban water systems in major Australian cities reported that electricity consumption could increase remarkably if alternative water sources such as desalination and wastewater recycling were implemented [18]. In an energy-water nexus analysis of water supply scenarios in coastal communities (Tampa Bay and San Diego) in the USA, maximizing water reclamation was found to be a better solution compared to desalination from embodied energy, greenhouse gas (GHG) emissions and energy cost perspectives [19]. However, a study of the energy requirements needed to deliver reclaimed water up-gradient of six watersheds indicated that the water needed for the energy exceeded the amount of water that would be pumped to the various delivery locations [20]. Another study considering the Middle East and North Africa showed a relatively weak dependence of energy systems on seawater (rather than fresh water) but a strong dependence of water systems (mainly abstraction and production) on energy [21].

While there is an increasing interest in understanding water-energy interdependences and associated management implications for urban water systems, most of the studies considered only a partial rather than full urban water system. Research on the water-energy nexus frequently

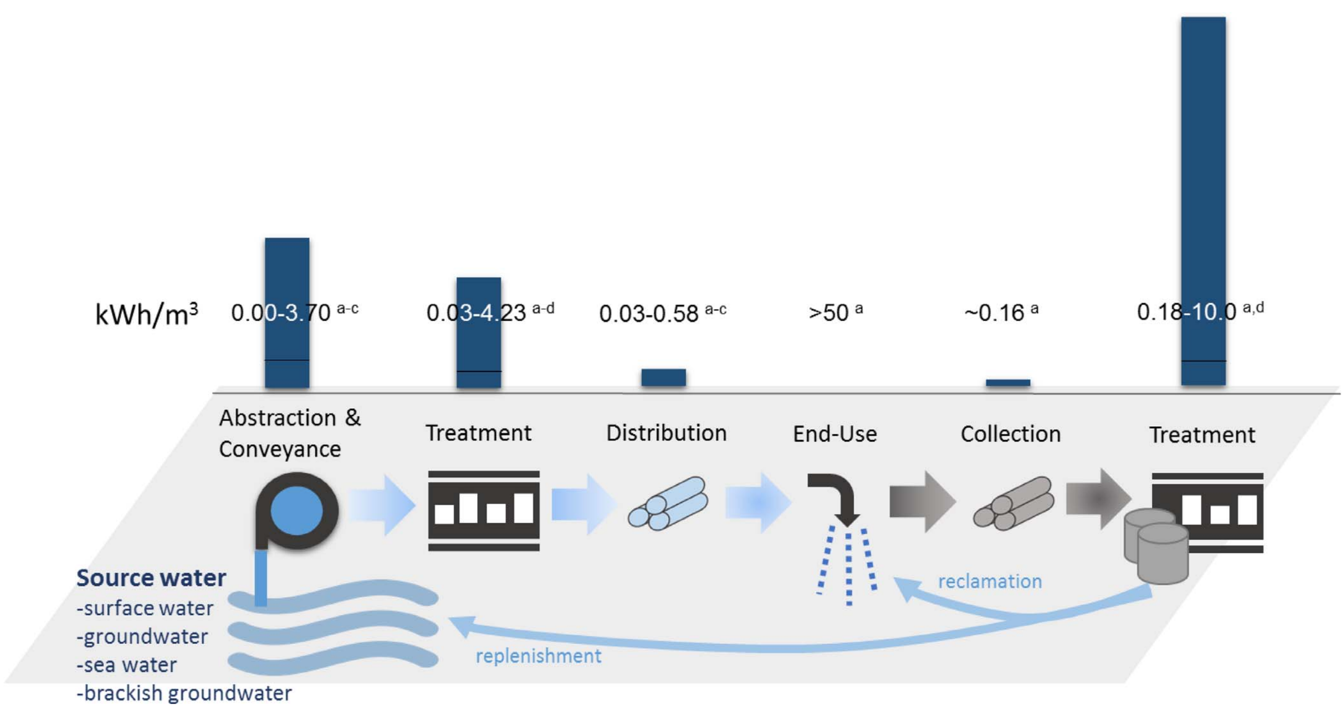

Fig. 1. Ranges of energy intensity within an urban water cycle using average values of benchmarking studies. This figure also illustrates selected urban water systems used in this study, including water abstraction and conveyance, potable water treatment, potable water distribution, wastewater collection and wastewater treatment, but excluding the end-use stage. Brackish groundwater or seawater desalination is included in the water treatment system. Data sources: [a] typical reported values for major regions of the USA, Australia, and Sweden [22]; [b] based on authors' calculations for California and Germany [23]; [c] based on a study conducted in California [24]; [d] based on a study conducted in the USA [25].

noted the need for increased local production and consumption of energy and water but rarely paid adequate attention to regional-scale consequences and impacts [2]. Moreover, the relationship between the nexus and water stress or water risk has not been generally reported. Therefore, this study aimed to explore the quantitative results of the nexus in terms of energy intensity and environmental impacts (mainly GHG potentials) on existing water infrastructures within urban water cycles, followed by an investigation of the relationship between the nexus and water risk indicators of baseline water stress (a water quantity indicator) or return flow ratio (a water quality indicator) at regional levels. In total, 20 regions and 4 countries with low to extremely high water risks, geographically located in Africa, Australia, Asia, Europe, and North America, were surveyed in this study. Our analysis attempted to characterize the influence of water risk on the water-energy nexus. In-depth discussions on the energy and environmental implications highlighted in this study can therefore provide a better understanding and critical information for future policy making for the effective management of water and energy.

2. Methods

Characterization of the influence of water risks on the water-energy nexus for urban water systems was conducted by investigating the relationship between risk and nexus at regional levels. This study particularly looks into the relationship between energy intensity of water systems and the water risk indicators (i.e., BWS and RFR). Exclusion of outliers from the study groups were made prior to the analysis (Fig. A1). One thing to note is that San Diego was constantly removed from the relationship analysis as its water supply depends substantially on non-reliable imported water sources [19]. Detailed framework of the analysis is illustrated in Fig. 2.

Energy intensity and GHG emissions were selected as the major measurements for the water-energy nexus, while baseline water stress (BWS) and the return flow ratio (RFR) were employed as water risk indicators. The study boundary within the urban water cycle (Fig. 1) considered only water abstraction and conveyance, potable water treatment, potable water distribution, wastewater collection and wastewater treatment while excluding the end-use stage.

2.1. Quantification of the water-energy nexus

Quantification of the water-energy nexus is usually characterized in resource use efficiency terms [2]. In this study, the nexus term was quantified as energy intensity, which was defined as energy consumption (in kWh) per unit volume (in m^3) of product water. Additionally, GHG potentials as GHG emissions (in g CO_2 eq) per unit volume (in m^3) of product water were used to represent the environmental impacts associated with the nexus. The GHG emissions, which mainly considers emissions of CO_2, CH_4 and N_2O, is a climate change indicator as defined by Intergovernmental Panel on Climate Change (IPCC) [26] and it is highly related to energy use of a system [27].

The nexus, in terms of energy intensity and associated environmental impacts (mainly GHG emissions) of water infrastructure systems, were extracted from empirical and theoretical studies in the literature. In total, 20 samples at regional levels and 4 samples at country levels (Fig. 3), which included detailed life cycle inventories and discussions on energy demand and GHG potential at urban water system scales, were surveyed and compared, if applicable. In this study, we intended to include regions or countries with relatively high water annual consumption (China and India), high per capita water footprint (USA) or under high water stress (Sydney and Melbourne in Australia). Their major water sources and wastewater treatment priorities are summarized in Table 1. It should be noted that the assessed quantitative values provided in this study may not strictly be compared directly; however, this information can be useful for analogous comparisons between different regions.

2.1.1. Role of life cycle assessments (LCAs) in the water-energy nexus

LCA is a widely used tool to evaluate the environmental performance of urban water systems [28]. It allows the quantification of direct energy requirements during the use phase as well as the embodied energy demand associated with the utilization of chemicals and materials. Use of LCAs also enables identification of environmental hot spots for system improvement [12] and helps to avoid potential pollution transfers between impact categories or life cycle stages [29].

Quantification of the nexus was conducted for the whole urban water system, including the majority of the urban water cycle from water supply to wastewater treatment (Fig. 1), based on the LCA framework for the surveyed studies. Recovery of energy from water (e.g., energy recovery from the organic content in wastewater) was considered in this study. In developed countries such as the USA and Germany, a remarkable share of the electricity required for the wastewater treatment plants (WWTPs) can be achieved though energy efficiency or energy harvesting (i.e., biogas) [30]. Most of the heat requirements at the WWTPs can be met using on-site biogas as reported by a study conducted in Oslo, Norway [31]. Energy consumption at the water end-use stage was excluded from the analysis, as it is strongly influenced by water use behavior and knowledge [32], and it was considerably high compared to other stages of the urban water cycle [33]. However, we acknowledge that water heating usually accounts for a significant percentage of the total household energy demand [34].

The functional unit for the selected studies was defined as the provision and treatment of 1 m^3 of product water (for the user) or the equivalent. Due to insufficient information in the literature on the environmental impacts associated with the fabrication and transportation of chemicals and materials used in water and wastewater treatment, the associated embodied energy requirements were not fully incorporated in this study. Impacts associated with the transportation of materials and electricity to the study site are usually negligible compared with the impacts during the operational stage of water systems. The GHG emissions were either directly calculated from electricity consumption or were reported as overall emissions from life cycle perspectives.

Since the water-related energy intensity and/or GHG emissions were seldom reported as a whole for urban water systems in Asia (China, Japan and India), national averages were considered but excluded from the relationship analysis. It should be noted that the nexus values reported in this study were assumed to be the lowest estimates for their corresponding systems, in part because of differences in the studied scope boundary, data availability and methodological approaches.

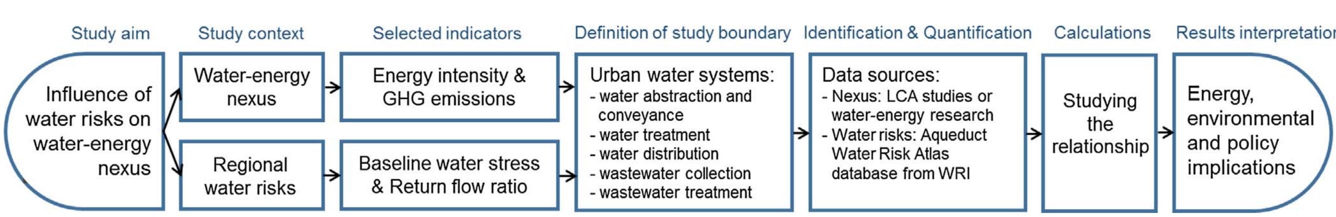

Fig. 2. Analysis framework of the relationship between the water-energy nexus and regional water risks for urban water systems.

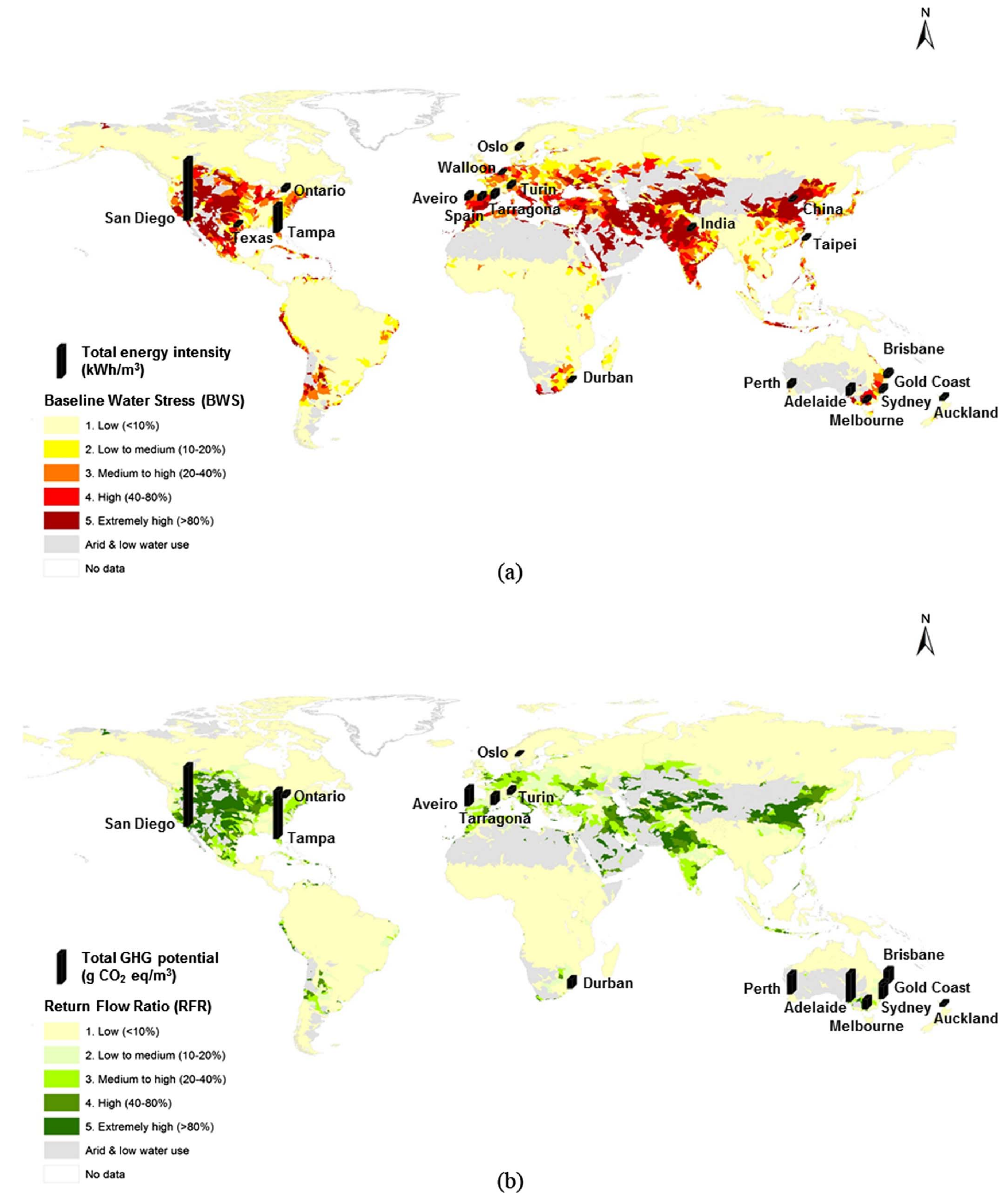

Fig. 3. Quantification of the water-energy nexus in terms of (a) energy intensity and (b) environmental impacts of GHG potentials from benchmarking studies. This figure also depicts the spatial distributions of water risk indicators of (a) baseline water stress (for water quantity) and (b) return flow ratio (for water quality) obtained from Aqueduct Global Maps 2.1 [41].

2.2. Identification of water risks

While there are several measurements that can possibly capture the concept of water risk, such as access to freshwater or available drinking water [35], water stress and water availability [36] are expected to provide the most comprehensive coverage at regional or national levels. The level of water stress mainly depends on population growth, climate change [37] and elevated atmospheric CO_2 concentration [38], which

Table 1
Summary of the major water sources and wastewater treatment priorities for the studied regions.

Region	Major water source[a]	Wastewater treatment level[b]	Ref.
Low to medium risk regions			
Auckland, New Zealand	S	3	[18,42]
Gold Coast, Australia	S + D	3	[18,43,44]
Toronto, Canada	S	2	[45]
Medium to high risk regions			
Brisbane, Australia	S	3	[18]
Oslo, Norway	S	2	[31]
Perth, Australia	S + D	2	[18]
Turin, Italy	G	2	[46]
High risk regions			
Aveiro, Portugal	G	2	[47]
Durban, South Africa	S	3	[48]
Taipei, Taiwan	S	2	[49]
Tampa Bay, Florida, USA	G	3	[19,50]
Tarragona, Spain	S	2	[51]
Wallon region, Belgium	G	2	[52]
Extremely high risk regions			
Adelaide, Australia	S	2	[18]
Melbourne, Australia	S	3	[18]
Qingdao, China	S	2	[53]
San Diego, California, USA	I	3	[19,54]
Sydney, Australia	G or S	2	[11,18]

[a] Water source: S represents surface water; G represents groundwater; D represents desalination of seawater; I represents importation.
[b] Wastewater treatment level: 2 for secondary treatment; 3 for tertiary treatment.

can be determined by water availability (supply), consumption (demand) and quality.

Among 12 global water risk indicators identified by the Aqueduct Water Risk Atlas published by the World Resources Institute (WRI) [39], baseline water stress (BWS) and the return flow ratio (RFR) were selected as physical quantity and quality risk indicators, respectively. The water risk indicators were developed based on withdrawal-to-availability ratios [40], the well-acknowledged approaches in expressing water stress. Direct use of the indicators developed by the WRI minimize the uncertainty in determining the indicators across regions. Scores and categorization of the indicators for the studied regions were accessed using Aqueduct Global Maps 2.1 Data [41]. These indicators were categorized into 5 major groups, namely low ($<10\%$, scored 0–1), low to medium (10–20%, scored 1–2), medium to high (20–40%, scored 2–3), high (40–80%, scored 3–4) and extremely high ($>80\%$, scored 4–5), as defined in the report of Aqueduct Global Maps (Fig. 3) [39]. For regions with multiple water risk scores, the mean value was calculated and reported.

2.2.1. Baseline water stress (BWS)

BWS was chosen for evaluation of the potential risk and stress associated with maintaining a sustainable water supply. This indicator measures the ratio of total annual water withdrawals, accounting for all upstream consumptive uses, to mean available annual renewable supply (blue water), expressed as follows [39]:

$$\mathrm{BWS} = \frac{\textit{total annual withdrawals}}{\textit{mean of available blue water}} \tag{1}$$

where available blue water is the summation of water flowing into a catchment from upstream and runoff in the catchment minus upstream water consumption (surface water availability subtracts water consumed upstream). Higher BWS values indicate more competition among users (including users from municipal, industrial and agricultural sectors).

2.2.2. Return flow ratio (RFR)

RFR was used to determine the potential risk of low water quality. This indicator calculates the percent of available water previously used and discharged upstream as wastewater and is expressed as follows [39]:

$$\mathrm{RFR} = \frac{\textit{upstream non-consumptive use}}{\textit{mean of available blue water}} \tag{2}$$

where non-consumptive use is the remainder of water withdrawals that is not consumed and returns to ground or surface water bodies, including post-treatment of consumptive use (e.g., wastewater) that is available for reuse. Higher RFR values suggest a higher percentage of non-consumptive use that requires appropriate treatments and has a potentially higher risk of lower water quality in areas without sufficient treatment infrastructures and policies [39].

3. Results and discussion

3.1. Characterization of water-energy nexus profiles

Quantification of the water-energy nexus in terms of energy intensity and environmental impacts (Table 2) in various regions is discussed in this section. These regions are classified into four groups: low to medium, medium, medium to high and extremely high water risk regions, based on their mean score of water risks (BWS and RFR). Variations in the values were partially influenced by their water supply characteristics and water treatment technologies. Risks associated with water availability and water quality also contributed to the variation in the values.

3.1.1. Low to medium water risk regions

Among the surveyed regions, New Zealand, Canada and Australia were the only nations with low to medium water risks within their borders.

In Auckland, the energy requirements for water supply system has remained fairly constant (approximately 0.21 kWh/m^3), even though the population and water volume supplied have exhibited steady growth [18]. Higher energy intensity (0.84 kWh/m^3) was required for its wastewater treatment system, mainly due to the tertiary treatment of wastewater that accounted for about 60% of the total energy demand in its water system. The water system in Auckland also had relatively small carbon footprint (53.6 and 219.0 g CO_2 eq/m^3 for water supply and wastewater treatment systems, respectively) compared to other systems (Table 2), in part because of the presence of internal electricity generation by biogas.

The energy demand in the Province of Ontario ranged from 0.68 to 1.11 kWh/m^3 for water supply (surface and groundwater) and from 0.34 to 0.70 kWh/m^3 for wastewater treatment, whereas embedded energy from manufacturing of chemicals only accounted for 0.01 kWh/m^3 [4]. An LCA study of the water treatment facility in the City of Toronto determined that the pumping of raw water and water treatment were the most energy and GHG intensive processes, consuming over 75% of the energy intake of the plant. Chemical manufacturing for operation of the facility was responsible for 5% of total energy use (0.64–0.73 kWh/m^3) and 7% of GHG emissions (116–128 g CO_2 eq/m^3) [45]. The relatively low GHG emissions could be attributed to the low GHG emission factor in the Province of Ontario (270 g CO_2 eq/m^3) [4].

Gold Coast in Australia has a comparatively low energy intensity for water supply of 0.21 kWh/m^3, as the system employs a relatively simple water treatment procedure and uses gravitational transport from the main water source (Hinze Dam, from surface water). A relatively higher energy intensity (1.00 kWh/m^3) was observed for tertiary wastewater treatment [18]. The tertiary treatment was introduced in the Gold Coast to provide Class A+ recycled water for toilets and outdoor uses [44]. Gold Coast also owns a desalination plant, which is always in standby mode and is capable of supplying the region's drinking water needs

Table 2
Summary of energy intensity and greenhouse gas (GHG) potential of urban water infrastructures.

Region/locations	Water risk indicators		Water supply system[a]		Wastewater treatment system[b]		Ref.
	Baseline water stress (BWS)	Return flow ratio (RFR)	Energy intensity (kWh/m^3)	GHG potential* (g CO_2 eq/m^3)	Energy intensity (kWh/m^3)	GHG potential* (g CO_2 eq/m^3)	
Africa							
Durban, South Africa	High (40–80%)	High (40–80%)	0.19[c]	481**	0.58[c]	353**	[61]
Australia							
Adelaide, Australia	Extremely high (> 80%)	Extremely high (> 80%)	1.92	1884	0.69	674	[18]
Auckland, New Zealand	Low to medium (10–20%)	Low to medium (10–20%)	0.21	54	0.84	219	[18]
Brisbane, Australia	Medium to high (20–40%)	Medium to high (20–40%)	0.68	586	0.57	499	[18]
Gold Coast, Australia	Low to medium (10–20%)	Low (< 10%)	0.21	206	1.00	991	[18]
Melbourne, Australia	Extremely high (> 80%)	High (40–80%)	0.09	69	1.13	835	[18]
Perth, Australia	Medium to high (20–40%)	Medium to high (20–40%)	0.98	1030	0.71	748	[18]
Sydney, Australia	Extremely high (> 80%)	Extremely high (> 80%)	1.10	1044	0.45	423	[18]
Sydney, Australia	Extremely high (> 80%)	Extremely high (> 80%)	0.34	406–452**	0.48	672–719**	[11]
Asia							
China	–	–	0.43–0.67[d]	–	0.13–0.50[e]	–	[30,63,67]
India	–	–	0.17–0.71	–	0.05–1.40	–	[68]
Japan	–	–	–		0.30–2.07[f]		[69]
Qingdao, China	Extremely high (> 80%)	Extremely high (> 80%)	0.73–1.08	–	–	–	[53]
Taipei, Taiwan	High (40–80%)	Medium to high (20–40%)	0.26–0.51	–	0.39–0.44[g]	–	[49]
Europe							
Aveiro, Portugal	High (40–80%)	Medium to high (20–40%)	0.79	766–898**	0.84	659–851**	[47]
Oslo, Norway	Medium to high (20–40%)	Low to medium (10–20%)	0.39–0.44	70–80	0.67–0.81	55–62	[31]
Spain	High (40–80%)	Medium to high (20–40%)	0.23–6.99	–	0.41–0.72	–	[14]
Tarragona, Spain	High (40–80%)	Medium to high (20–40%)	0.85	736	1.09	128	[51]
Turin, Italy	Medium to high (20–40%)	Low to medium (10–20%)	0.85[h]	323**	0.47[h]	126**	[46]
Walloon region, Belgium	High (40–80%)	High (40–80%)	0.39	–	0.31	–	[52]
North America							
Tampa Bay, Florida, USA	High (40–80%)	Medium to high (20–40%)	3.00–3.60[i]	1585–1911	4.50–5.10[i]	2457–2739	[19]
Texas, USA	Medium to high (20–40%)	High (40–80%)	0.38–0.48	–	0.32–0.36	–	[16]
Toronto, Canada	Low (< 10%)	Low to medium (10–20%)	0.64–0.73[i]	111–128	0.47	22	[45]
Province of Ontario, Canada	Low (< 10%)	Low (< 10%)	0.68–1.11[j]	184–300	0.34–0.70	92–189	[4]
San Diego, California, USA	Extremely high (> 80%)	Extremely high (> 80%)	6.20–16.4[i]	1855–4903	3.50–7.50[i]	1041–2249	[19]

[a] The water supply systems include at least water abstraction, conveyance and treatment, unless stated otherwise.
[b] The wastewater treatment system includes collection and treatment, unless stated otherwise.
[c] Includes inputs from locally manufactured chemicals and construction of infrastructure (pipes and pumps).
[d] Includes water abstraction from surface water or groundwater and public water distribution.
[e] Based on an estimation from 5 interviewed municipal WWTPs [30]; the average value at the provincial level was 0.26 kWh/m^3.
[f] 0.44–2.07 kWh/m^3 for oxidation ditch plants, 0.30–1.89 kWh/m^3 for conventional activated sludge plants without sludge incineration.
[g] Excludes collection of wastewater.
[h] Excludes combined energy consumption (potable water and wastewater treatment) for chemical manufacturing (0.04 kWh/m^3) and transportation (0.03 kWh/m^3).
[i] Includes embodied energy demand, such as energy demand for chemicals and materials, as well as services during operation and maintenance.
[j] These values vary from the system's operational capacity. Embodied chemical energy was calculated as 0.01 kWh/m^3.
* Based on GHG emission factor for electricity generation; the values are strongly influenced by the unique proportion of fossil-fuel fired electricity generation.
** Life cycle-based GHG potentials.

(e.g., during existing water treatment plant upgrades in August 2016 or during flood events) [43], as well as a water pipeline network to distribute the aforementioned climate-resilient water sources (e.g., desalinated water and recycled water) across the region [55].

3.1.2. Medium to high water risk regions

Regions with medium to high water risk usually involve intensive urban growth, higher density in population and decreased climate-driven water availability [38]. For instance, Taipei, Taiwan, is experiencing high water risks due to the high-density population, improper water usage, an inefficient water distribution system and extreme climate events [49]. Perth and Brisbane in Australia, Turin in Italy and Oslo in Norway, are also listed as regions with medium to high water risks.

Perth is becoming one of the driest cities in Australia. In response to the water shortages, the Kwinana Desalination Plant was commissioned by the Western Australian State Government in 2006, which resulted in a substantial increase (75%) in energy demand for its water supply system from approximately 0.56 kWh/m^3 (between 2001 and 2006) to 0.98 kWh/m^3 (as of July 2006) [18]. The energy supply for Perth's water system is currently dominated by electricity, and only a small amount (less than 5%) is supplied by biogas generated by the WWTPs. Thus, the carbon footprint for the water system in Perth is relatively high. Brisbane in Australia has high water stress but a relatively low water-related energy intensity of 0.68 kWh/m^3 for water supply and 0.57 kWh/m^3 for wastewater treatment systems, despite the use of a tertiary treatment that represents approximately 40% of the total energy requirement. Since coal-fired power plants are the main energy source in Australia [28], their water-related carbon emissions are comparatively high.

In Turin, Italy, its wastewater treatment, including indirect energy consumption for chemical production and transportation, accounted for the largest share of the total energy demand, but nearly 40% of the demand could be satisfied by biogas produced on-site from wastewater sludge [46]. Therefore, the net electricity consumption was 0.85 kWh/m^3 for water supply and 0.47 kWh/m^3 for wastewater. In a longitudinal study on the energy consumption and environmental impacts of Oslo's urban water services from 2000 to 2006, it was observed that its specific energy consumption in the operations and maintenance (O & M) phase for the water supply system (0.39–0.44 kWh/m^3) increased by approximately 12.8%, and that for the wastewater system (0.67–0.81 kWh/m^3) increased by 20.9% [31]. This increase was primarily due to diesel generators used for water supply and electricity consumption for aeration to remove nitrogen during the wastewater treatment. Surprisingly, the GHG emissions for the wastewater system were approximately 22% lower on average than those for the water supply system. This was managed by avoiding the production of natural gas and by reducing the use of electricity for heating [31].

3.1.3. High water risk regions

Regions with high water risk might have experienced extremely weather events, in addition to the influence from urbanization (e.g., population growth or increase in economic activities). The occurrence of water scarcity and drought is increasing and is widespread in European countries, especially in the south such as Spain and Portugal [56].

Spain is encountering hydrological stress and is projected to have significantly increased water stress by 2040 [57], thus, desalination is used as an alternative water source (as of 2008 constituted approximately 2% of the total water supply). Approximately 5.8% of the total electricity demand in Spain was from the water sector, based on 2008 estimates [14]. Most of the water-related energy use at this time was for the water supply system (approximately 64%, with a mean value of 0.99 kWh/m^3), while wastewater treatment (mainly for urban and industrial sectors) only accounted for 16% (mean value of 0.58 kWh/m^3) [14]. A study of the environmental impacts of the water cycle in the water-stressed region of Segura Basin, Spain, revealed that seawater desalination had the highest energy use (4.10 kWh/m^3) followed by the Tajo-Segura water transfer (0.87–1.55 kWh/m^3) [58].

Another study conducted in Tarragona, Spain, showed that a significant fraction of the energy use within its urban water cycle (a total of 1.94 kWh/m^3) occurred during the water distribution stage (approximately 25–42% of the total), which was mainly attributed to the pumping processes [51]. Moreover, water abstraction by pumping (direct consumption of 1.2 m^3 of source water to supply of 1.0 m^3 of potable water) was found to be the key contributor to the Freshwater Ecosystem Impact, whereas the impact at other stages in the water cycle could be neglected [51]. The potable water treatment system had very little impact on the urban water system, as the system received nearly 75% of the total water supply from surface water (Ebro River), and the remainder originated from mines that only required chlorine treatment. An environmental assessment of urban water system was conducted in the high water risk region of Aveiro, Portugal [47]. Their results showed that water abstraction and treatment (766–898 g CO_2 eq/m^3) were the major contributors to the overall environmental impacts, which could be seen as a consequence of the intensive energy consumption associated with groundwater pumping. It is worth noting that in 2011, only 64% of the wastewater services in Aveiro complied with the discharge limits, which indicates a risk of poor water quality.

Several regions in the United States suffer from water stress and may endure variability in water availability [16]. Tampa, a coastal city in Florida, has aggressively searched for strategies and technologies (such as water reclamation or desalination) to augment its existing water supply. When considering the embedded energy demand associated with chemicals and materials, as well as services during operation and maintenance, relatively higher energy demands in the range of 3.0–3.6 for water supply and 4.5–5.1 kWh/m^3 for wastewater treatment were reported [19]. Tertiary wastewater treatment in Tampa was prioritized for reducing adverse impacts from discharge of biodegradable organic matter and nutrients than for increasing alternative water sources through reclamation [59]. It is worth mentioning that water reclamation in Tampa is still considered more sustainable than increasing the traditional water supply or desalinating seawater from energy-water nexus perspectives, despite that the energy consumption for desalination is relatively low due to the lower salinity (26,000 ppm) in the source saline water (brackish groundwater) [19].

South Africa is also expected to become a water-stressed region as indicated by water resource assessments and models [60]. A water recycling plant was commissioned in 2001 to produce industrial-grade water for a paper mill and an oil refinery that were originally supported by two dam systems (Nagle Dam and Inanda Dam) to satisfy the potable water supply gap. An environmental assessment study of the urban water systems was conducted in Durban, the core city in the eThekwini Municipality and the largest South African port city [61]. Their results indicated that the distribution of potable water had the highest contribution when water losses in the distribution network were considered, but treatment of wastewater (i.e., activated sludge process) was regarded as an energy-intensive and high environmental impact process. Inclusion of locally manufactured chemicals and materials (pipes and pumps) in the assessment contributed little to the overall impacts. Relatively higher impacts on climate change were reported in Durban, as its GHG emission factors (970 g CO_2 eq/kWh) for electricity were high, even with lower electricity consumption for the urban water systems [48].

3.1.4. Extremely high water risk regions

Qingdao in China, San Diego in the USA and several populated cities in Australia, including Sydney, Melbourne and Adelaide, are well known for their extremely high water risk in terms of water scarcity and water security.

China is expanding its groundwater abstraction and water transportation, as a consequence of the significant increase in water use for the agricultural sector [3]. A water-energy nexus study in China revealed that the agricultural sector uses the largest share of water in an inefficient way [62]. The energy for providing agricultural water contributes approximately 41% of total energy for water provision [63]. In addition, large-scale water transfer projects from south to north are being established to compensate for the shortage in water supply in northern China. It is expected that this project will involve intensive energy costs for construction, maintenance and pumping, as well as for additional treatment and monitoring of the water [64]. The case study for Qingdao city showed that water provision in northern China could be very energy intensive, in particular that nonconventional water supplies such as seawater desalination, inter-basin transfer and recycled wastewater were implemented as part of their water supply system [53].

Water is at the forefront of concerns in San Diego. San Diego relies on imported water for approximately 84% of its total supply, from the Colorado River and northern California [54]. The energy costs of water in San Diego are high due to the considerable transport distance from distant sources [65]. For instance, water transported from the Sacramento-San Joaquin Delta to southern California requires 2.43 kWh/m^3 on average [66], which is significantly higher than the energy requirement for groundwater pumping (< 1.0 kWh/m^3). Water transport to southern California accounts for approximately 3% of the state's total electricity usage [66]. Furthermore, under a low water table or dry hydrological conditions, more power is required to extract groundwater. Mitigating the potential risk of increased seawater intrusion during these dry periods also requires additional energy for injecting freshwater into the aquifers [65]. The energy-water nexus analysis for San Diego's urban water system revealed a relatively high embodied energy (9.7–23.9 kWh/m^3) but lower GHG emission coefficient (approximately 301 g CO_2 eq/kWh) for most of its water infrastructure, mainly due to a higher percentage of renewable energy in the electricity mix of California [19].

The Millennium Drought in southeast Australia from 2001 to 2009 had a significant impact on the water resource management within the nation. Sydney [11], Melbourne and Adelaide are also listed as regions with extremely high water stress in Australia [18]. The energy intensity for the urban water system in these cities is highest for Adelaide, followed by Sydney and Melbourne [18]. Differences in energy intensities are mainly due to extensive pumping, water and wastewater treatment techniques, and most importantly, the presence of an energy recovery system in the WWTPs (sewage sludge recovery for biogas generation). An increase in water pumping for additional water storage contributes to higher energy intensity for water supply during drought periods. A significant amount of energy (1.92 kWh/m^3) was required to provide extra storage of water during the drought period in Adelaide, which accounted for nearly 74% of the total energy requirement of the water system (2.61 kWh/m^3) in 2006; its energy requirement for water supply strongly depended on the proportion of supply pumped from the Murray River [18].

The drought event in 2006 also required Sydney to incur extra energy for water pumping (1.10 kWh/m^3) from the Shoalhaven river system, which contributed to approximately 70% of the total energy requirement of the water system (3-fold more than in January 2000), whereas the remainder was mostly for wastewater treatment (0.45 kWh/m^3). This was also the main reason that the estimated energy intensity for the water supply system in Sydney was remarkably higher than the value (0.34 kWh/m^3) reported in an earlier study [11]. It is also expected that the coastal sewage treatment plants in Sydney will contribute a significant amount of total energy use and climate change impacts to meet the New South Wales Environmental Protection Authority Standards for wastewater [11].

Unlike the case in Sydney, the energy requirement for water supply in Melbourne was very low (0.09 kWh/m^3 in 2006), as most of the water is gravity fed from reservoirs. On the contrary, the energy requirement for the treatment and disposal of wastewater (1.13 kWh/m^3) was more than 10-fold higher than that for water supply [18]. This high energy intensity was mainly due to the implementation of tertiary treatment in its wastewater system. It is worth noting that most of the studied cities in Australia utilize biogas generated in the WWTPs to supply their water-related energy, which further reduces their wastewater GHG emissions. The internal generation of electricity using biogas from WWTPs or imported natural gas constituted nearly 40% of the energy demand of WWTPs in Melbourne [18].

3.2. Energy and environmental implications

Energy requirement and environmental impacts at different stages within an urban water system may be site specific [19], being influenced by the technologies used for water abstraction, water and wastewater treatment (including the level of treatment), transport distance, local orography, efficiency of the system and the extent of water losses [47].

In the 20 surveyed studies (with 20 at regional and 4 at country levels), the highest share of energy intensity was attributed to the water supply system (1.33 kWh/m^3 on average) followed by the wastewater treatment system (1.07 kWh/m^3 on average). Inclusion of the San Diego case that relied on imported water increased the average energy intensity for the water supply systems by approximately 0.55 kWh/m^3. Similarly, the environmental impacts of the GHG potential were the highest for the water supply system (0.80 kg CO_2 eq/m^3 on average) followed by the wastewater treatment system (0.64 kg CO_2 eq/m^3 on average). It is worth noting that the surveyed regions were not that similar to each other; however, some insights into the differences between the reported values can be provided by this study.

3.2.1. Energy implications related to the water supply system

A positive trend between the energy intensity of the water supply system and baseline water stress (BWS) can be observed in Fig. 4(a) and Fig. A2. This was mainly due to the accessibility of source water (e.g., surface water or groundwater) that had a strong impact on its energy demand. Commissioning of alternative water sources such as desalination and wastewater recycling would also increase the total urban energy requirement [18]. As the BWS measures the ratio between water withdrawals and availability, higher values suggests the need for alternative water sources to supplement the existing water supply.

Groundwater supplies typically require approximately 30% higher direct electricity consumption and 27% more embodied energy than surface water supplies [67]. Theoretically, lifting 1 m^3 of water up 1 m at 100% efficiency uses approximately 0.0027 kWh of energy [3], but the energy requirement for lifting groundwater also depends on the groundwater elevation, volume, and the efficiency of the pump [67]. The energy requirements for various water sectors at a global level are 0.0002–1.74 kWh/m^3 for surface water supply, 0.37–1.44 kWh/m^3 for groundwater pumping and 2.4–8.5 kWh/m^3 for desalination (using membrane-based technology) [67].

In this regard, San Diego (with extremely high risks with respect to both BWS and RFR) had the highest energy intensity associated with extensive electricity use for water importation. Adelaide and Sydney in Australia also had relatively high energy intensity values (> 1 kWh/m^3) for their water supply systems as a consequence of increased groundwater pumping, with extremely high water risks. Regions with medium to high water risks (scores of 2–4) normally used groundwater to augment the existing water supply, which resulted in a greater intensity of energy usage and substantial associated environmental impacts. The regions that relied on surface water (mainly those with low to medium BWS) had relatively lower energy intensity associated with their water supply system.

Although the abstraction of groundwater consumes a significant amount of energy, the treatment of groundwater typically requires less energy than that of surface water, as groundwater generally has a lower concentration of total dissolved salts (TDS) [70]. The cases from

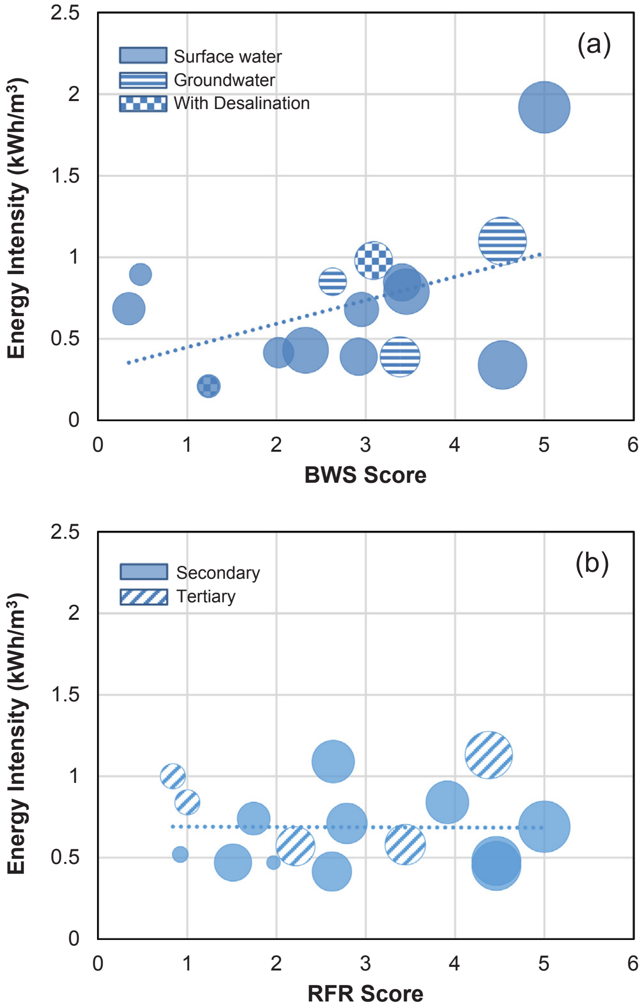

Fig. 4. Energy intensity of water systems in relation to water risk indicators. (a) Energy intensity of water supply systems and BWS; the bubble size is proportional to the score of RFR; (b) energy intensity of wastewater treatment systems and RFR; the bubble size is proportional to the score of BWS. The largest bubble represents the highest water risk score of 5. The different patterns in the legend indicate the major water sources (surface water, groundwater and inclusion of desalination) and type of wastewater treatment process (secondary and tertiary). Outliers such as San Diego were excluded from the analysis as discussed in the Methods section and Fig. A1.

Tarragona, Spain, and the Gold Coast, Australia, that utilized simple treatment processes also demonstrated that the source water quality greatly affected the level of treatment as well as the energy consumption associated with that treatment.

3.2.2. Energy implications of wastewater treatment systems

The energy intensity of the wastewater systems was independent from the water risks, as shown in Fig. 4(b). Small variations in the reported energy intensity were also observed. This suggested that a relatively high proportion of the energy intensity of the wastewater treatment appeared to reflect the source water quality, effluent quality, treatment level and water treatment technology. The water risks, on the other hand, seemed to have little impact on the energy intensity of the wastewater systems (Fig. A2(b) and Fig. A3(b)).

The operational phase of WWTPs is generally reported to be the greatest contributor to energy consumption [45]. The energy requirement for the collection of wastewater is usually negligible since the water transfer is mostly gravity driven. Primary wastewater treatment typically focuses on the physical removal of solids and is therefore less energy intensive (0.003–0.37 kWh/m^3) compared with other treatment procedures [67]. Tertiary treatment is considered an energy intensive procedure for nutrient removal and/or disinfection, with energy demand ranging from 0.40 to 0.50 kWh/m^3 [14] and even reaching 3.74 kWh/m^3 [69]. A study on energy requirements for wastewater treatment systems in Australia and New Zealand reported that the energy intensity doubled between primary and secondary treatment and then doubled again between secondary and tertiary treatment [18]. Thus, regions considered to have advanced or tertiary wastewater treatment systems have relatively higher energy intensities, as depicted in Fig. 4(b).

Effluent quality is also deemed as a major component in determining the magnitude of the energy demand and the environmental burdens in an urban water system. As the Urban Wastewater Treatment Directive (for European countries) aims to protect the environment from adverse effects of urban wastewater discharges, as well as discharges from certain industrial sectors, most of the participating countries have attempted to maximize their wastewater reuse and have consequently promoted the construction of tertiary WWTPs [71]. Other coastal regions such as Tampa Bay, Florida, further utilize advanced wastewater treatment to improve water quality (e.g., remove nutrients) prior to discharge. Consequently, the energy intensity of these wastewater systems is relatively high. This also implies that the tertiary treatment of wastewater, as for now, is a matter of political or environmental concern rather than a physical concern (e.g., to increase water supply through reclamation).

The energy requirements for decentralized wastewater treatment systems were generally reported to be greater than that for centralized systems [28]. In India, the average energy intensity for decentralized WWTPs (small-scale institutional plants, 4.87 kWh/m^3) was reported to be much greater than that for centralized plants (large-scale municipal plants, 0.40 kWh/m^3) [72]. They also found that there was a clear correlation between the electricity consumption and operating capacity of the plants [72]. The reported average energy intensity in India was lower than that in the UK (0.46 kWh/m^3) [72], owing to the preference for low-energy requirement technologies such as upflow anaerobic sludge blanket (UASB) reactors [68] or use solar heat for sludge drying [72] in India. Providing treated water and disposing of wastewater in the USA represents an approximate average of 3% of total energy use but could be as high as 20% (e.g., in California). The 3% energy use for water corresponds to a specific energy use of approximately 2.26 kWh/m^3 [73].

3.2.3. Environmental implications

The GHG potentials (as a dominant indicator of environmental impacts) were positively related to the water risks (Fig. 5). As expected, the GHG emissions from the urban water systems were highly influenced by the electricity consumption and the corresponding energy grid mix [12]. Regions that generate electricity mainly from fossil fuels typically have the highest GHG impacts [47]. Sydney [11] was therefore found to have relatively high GHG emissions compared to Turin [74], even with similar energy requirements for the wastewater treatment systems, as a consequence of their electricity generation portfolio. Thus, utility suppliers such as water companies often attempt to comply with government regulations to monitor and reduce GHG emissions through energy conservation or energy-use efficiency programs [3], as well as on development of low-energy demand water treatment technologies [75]. Changes in the energy grid are expected to provide direct opportunities to reduce carbon emissions from urban water systems. It is worth noting that electricity consumption may also pose energy-related environmental impacts such as abiotic depletion, acidification, ozone layer depletion, human toxicity and photochemical oxidation [76].

Electricity consumption during the operational phases of water infrastructures was the main contributor to the total environmental impacts. Indirect impacts of chemical and material utilization during water or wastewater treatment should not be ignored when assessing water infrastructures. Process chemicals, such as ferrous chloride for the removal of phosphorus or the control of odorous sulfur compounds,

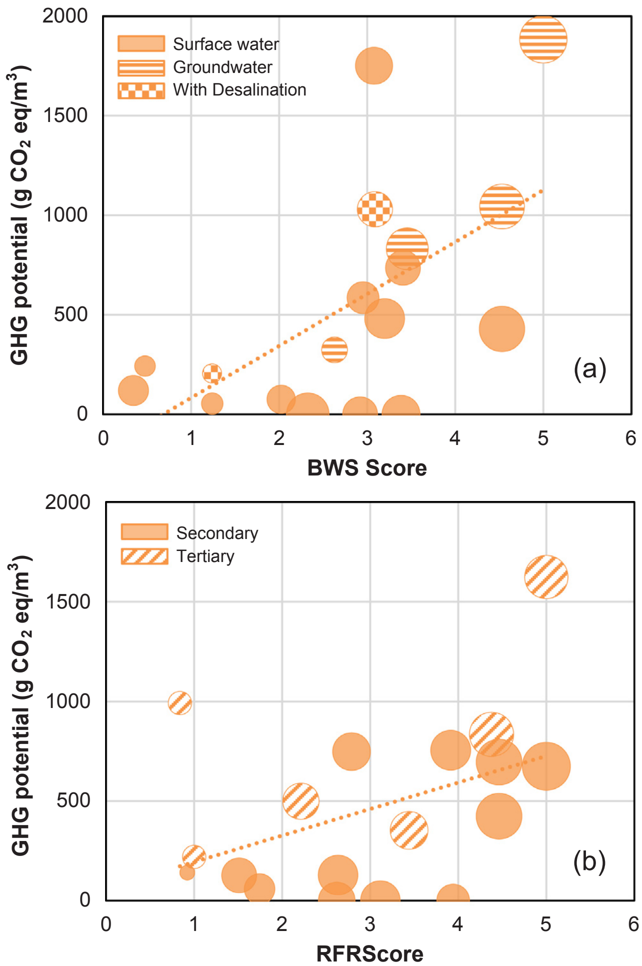

Fig. 5. Greenhouse gas (GHG) potentials of water systems in relation to water risk indicators. (a) GHG potentials of water supply systems and BWS; the bubble size is proportional to the score of RFR; (b) GHG potentials of wastewater treatment systems and RFR; the bubble size is proportional to the score of BWS. The largest bubble represents the highest water risk score of 5. The different patterns in the legend indicate the major water sources (surface water, groundwater and inclusion of desalination) and type of wastewater treatment process (secondary and tertiary). Outliers such as San Diego were excluded from the analysis as discussed in the Methods section and Fig. A1.

used for wastewater treatment, could possibly contribute greatly to the total energy demand and overall environmental impacts such as climate change and photochemical oxidant formation potential [11].

Environmental impacts of a wastewater treatment system are influenced by the quality of the source water (influent) and discharge (effluent), and efficiency of treatment technology [77]. High chemical oxygen demand (COD) in the influent results in increase in environmental impacts from larger aeration periods and other advanced treatment. But the high influent load does not necessary share high effluent load if a high removal efficiency water treatment technology is applied. Discharge of wastewater (treated as well as untreated overflows) contributes to eutrophication impacts, mainly due to the release of nitrogen and phosphorus to the environment [78]. The wastewater treatment stage (primarily for secondary treatment) had the highest eutrophication potential (~75%), due to excess nutrients in the sewage sludge used for agricultural applications. A study based in the Walloon region, Belgium, also reported that wastewater discharge generated significant acidification and eutrophication impacts from energy consumption and chemical usage (e.g., lime, iron chloride and polymers) [52]. All of these findings indicate that proper treatment of wastewater is required to reduce the environmental impacts resulting from the discharge of pollutants into the environment.

3.3. Policy actions and future management implications

While the need for integrated system design is global, the high variability in fresh water distribution necessitates a regional and local level assessment for identifying the most appropriate policy directions and technologies [21]. Several water-energy nexus studies also pointed out that site-specific factors should be considered for sustainable water and energy management at regional scales [19]. Direct measurement and reporting of energy requirement in regional urban water systems would provide a more appropriate basis for planning, management and policy. Energy-intensive sections in the water system should also be optimized [67].

The selection of water source based on the corresponding specific water quality requirement may result in considerable water and energy demand trade-offs [28]. For water-stressed regions in coastal areas, seawater is considered as a stable water source (for water supply or power plant cooling water) compared to fresh water. However, relying on seawater for water supply and increasing the expansion of desalination capacity may also pose environmental implications such as thermal pollution and ecotoxicity to the marine environment. Reclamation of wastewater, on the other hand, is a favorable choice from both economic and environmental perspectives [48]. But Stokes and Horvath [79] noted that water recycling can be more harmful than desalination.

Understanding the water-energy nexus can lead researchers and decision-makers to explore new opportunities to conserve water, energy and costs, as well as achieve maximum benefits. The nexus between water and energy policies could possibly result in spillover effects on both resources. Introducing an energy tax could lead to co-benefits of reducing air pollution and water conservation, whereas water conservation policies might contribute to higher energy consumption [80]. It is also well recognized that the price structure of water and energy is a critical component affecting the interaction between the two. This has motivated efforts toward one strategy that has the greatest benefit. For example, a water conservation program in Portland, Oregon, aggressively targeted summer water use to optimize the energy savings from using less pumped water [15].

In response to the strong dependence of water systems on energy, future integrated system designs may favor water conservation or reuse in end-use sectors (e.g., agricultural sectors) as opposed to the expansion of energy intensive water reclamation or desalination systems. It is encouraging that researchers and policy makers are recognizing the potential of efficient water and energy use to achieve environmental, social and economic [66] benefits [4]. Improving water efficiency will reduce power demand, and improving energy efficiency will reduce water demand. On-site energy recovery may be used on-site to reduce overall electricity demand and its associated carbon footprint or to be exported to the national grid. But energy derived from biogas might contribute to acidification impacts, if hydrogen sulfide is not removed from the sour biogas [81].

4. Limitations and uncertainty management

Two main sources of uncertainty associated with quantifying the water-energy nexus were identified: choice of water treatment technology and operational parameters (e.g., operation capacity or water quality standards). The choice of wastewater treatment technology was considered as the main source of uncertainty in this study, and the ranges in electricity consumption for typical wastewater treatment processes are compared in Fig. 6. Regardless of the differences in the technologies applied to the wastewater treatment processes, the energy intensity of wastewater systems was similar for regions with the same treatment levels (secondary or tertiary), as shown in Fig. 4(b). The operational parameters were also responsible for the uncertainty in estimating the nexus. For instance, old water infrastructures might have high water leakage, which creates an extra energy burden on the water

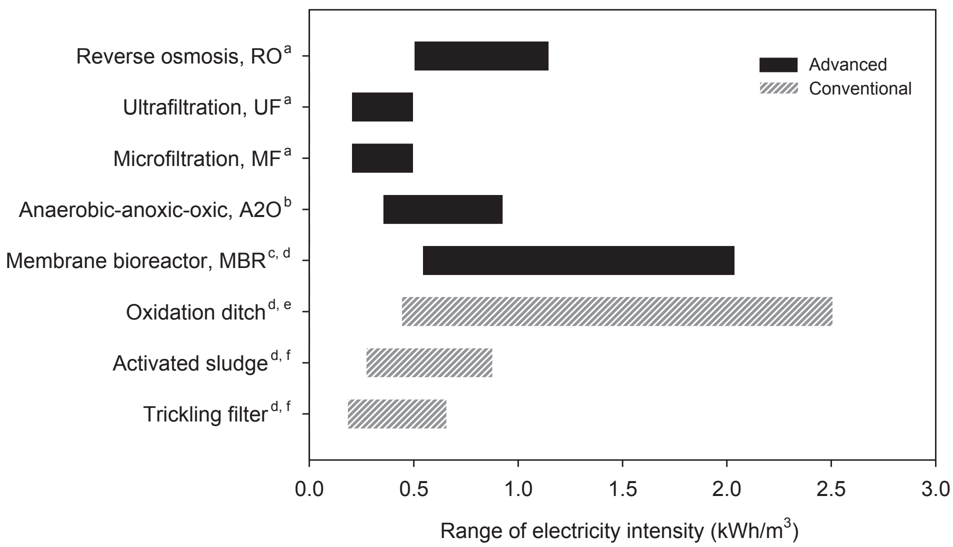

Fig. 6. Comparison of electricity intensity for typical wastewater treatment processes. Trickling filter, activated sludge, oxidation ditch, MBR and A2O are biological wastewater treatment processes, whereas MF, UF and RO are membrane-based separation processes. The energy intensities for MF, UF and RO are based on treatment of wastewater with total dissolved solid concentration between 800 and 1200 mg/L. Data sources: [a][82]; [b][83]; [c][84]; [d][30]; [e][69]; [f][16].

supply systems, and large treatment facilities might exhibit relatively lower impacts [16]. These uncertainties could be avoided by clearly defining the study scope and system boundary (Table 2). The uncertainty associated with facility management (e.g., environmental awareness) was usually negligible, as this impact might be averaged out during calculation. These findings are comparable to those presented by Wakeel et al. [67].

It is well acknowledged that consistency issues may occur while using a combination of data from various origins. To minimize this, the same level of detail was sought while collecting the energy and GHG emission intensities data from different studies. That is, in the surveyed studies, quantification of the intensities should be conducted for the whole urban water system rather than only one part of the system (e.g., water supply or wastewater system). It should be noted that the intensity values reported in this study were assumed to be the lowest estimates for their corresponding systems, in part because of differences in the studied scope boundary, data availability and methodological approaches [3].

5. Conclusions

The water-energy nexus, a general term used to detail the interdependence between water and energy, was characterized as energy intensity and GHG emission potential in this research. We conducted a systematic quantification of the water-energy nexus, and an analysis of the influence of water risks (BWS and RFR) on the nexus for urban water systems, including water supply and wastewater treatment systems, for 20 regions and 4 countries around the world. The energy intensities of water supply systems revealed a positive relationship with the water quantity risk (BWS), whereas the intensity of wastewater systems was independent from the water quality risk (RFR).

For water supply systems, accessibility of water source was identified as the key factor for the energy intensity as well as GHG potentials in the surveyed studies. Regions with higher water risks generally required energy-intensive water supply options such as groundwater pumping or desalination for augment their existing water supply. The case in San Diego served a great example in that regard, it suffered from extremely high BWS and RFR risks that consequently required extensive electricity use for water importation. On the other hand, the regions with low to medium BWS, such as Auckland or Toronto, relied on surface water supplies and had relatively lower energy intensities within their water systems.

For wastewater treatment systems, the level of wastewater treatment and treatment technology were regarded as major influential factors for the energy intensity in the surveyed studies. The water risks seemed to have little impact on the nexus, as little variation was observed for the energy intensities. The addition of tertiary treatment to existing wastewater treatment systems, either for water reclamation or environmental protection purposes, resulted in relatively higher energy and GHG emission intensities in the systems. This implied that effluent quality could determine the magnitude of the energy requirement as well as environmental burdens in the wastewater treatment systems.

Similarly, the environmental impacts of the urban water systems were highly dependent on their electricity consumption and implemented wastewater processes. On-site energy recovery (e.g., biogas or waste heat) in the wastewater treatment systems offered a great opportunity for reducing overall energy demand and its associated environmental impacts. Changes in the energy grid are also expected to provide direct opportunities to reduce carbon emissions and associated environmental impacts from the urban water systems. Proper treatment of wastewater is also required to reduce the overall environmental impacts of the wastewater treatment systems.

This study further confirmed that investigation of the water-energy nexus should be conducted at a regional level to ensure better understanding of the use of water and energy. Energy-intensive sections in urban water systems should also be optimized based on the water-energy nexus perspective with consideration of the tradeoff between environmental protection and economic growth.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.apenergy.2017.08.002.

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