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Water quality: the missing dimension of water in the water–energy–food nexus

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Abstract The role of water quality, particularly its impact on health, environment, and wider well-being, are rarely acknowledged in the water–energy–food (WEF) nexus. Here we demonstrate the necessity for including water quality within the water dimension of the WEF nexus to address complex and multi-disciplinary challenges facing humanity. Firstly, we demonstrate the impact of water quality on the energy and food dimensions of the WEF nexus and vice versa at multiple scales, from households to cities, regions and transboundary basins. Secondly, we use examples to demonstrate how including water quality would have augmented and improved the WEF analysis and its application. Finally, we encourage hydrological scientists to promote relevant water quality research as addressing WEF nexus challenges. To make tangible progress, we propose that analysis of water quality interactions focuses initially on WEF nexus "hotspots", such as cities, semi-arid areas, and areas dependent on groundwater or climate change-threatened meltwater.

Keywords: health; nexus hotspots; scale; Sustainable Development Goals; well-being

1 Introduction

The water–energy–food (WEF) nexus is a framework increasingly used by researchers and promoted for policy making to address complex grand challenges facing humanity that require a multi-disciplinary approach (Liu et al. 2017, McGrane et al. 2019). An example is the United Nations Sustainable Development Goals, SDGs, (United Nations 2015) which include zero hunger (SDG 2), good health and well-being (SDG 3), clean water and sanitation (SDG 6), affordable and clean energy (SDG 7), sustainable cities and communities (SDG 11), combating climate change and its impacts (SDG 13) and life below water (SDG 14). All these goals are underpinned individually and also interlinked by water (UN-Water 2016, United Nations 2018, Connor et al. 2020). Detailed analyses of water quality interlinkages with the SDGs (Alcamo 2019) and the synergies between SDGs 2, 6 and 7 (Fader et al. 2018), have demonstrated how cross-sectoral efforts to improve water quality can bring mutual benefits in addressing other SDG targets. However, the synergies between goals relating to economic, energy, food and water security, and the essential role of water quality in these, are less prominent in national policies and laws that principally drive water quality management.

National and regional policies and laws relating to water management rarely mention the WEF nexus specifically, such as the EU Water Framework Directive (WFD) (Venghaus and Hake 2018). Nevertheless, interdependencies between water and other sectors are considered in some key policy instruments. For example, to improve the ecological status of waterbodies (which includes water quality) the WFD requires the establishment and implementation of River Basin Management Plans and Programmes of Measures over a 6-year planning cycle (EU Directive on Water Policy 2000). To ensure compliance with the U.S. Federal Water Pollution Control Act (often referred to as the

Clean Water Act, CWA; U.S. Government 1972), the U.S. Environmental Protection Agency (EPA) has established various programmes or regulations, such as the National Pollutant Discharge Elimination System to control point sources discharging pollutants and the CWA Section 319 Program to prevent nonpoint-source pollution from agricultural and residential areas, livestock and husbandry. In China, the "3 Red Lines" water policies implemented in 2012, includes a target to reduce industrial water use such as in energy generation, and the Water Pollution Prevention and Control Action Plan ("10-Point Water Plan", April 2015) aims to improve surface and groundwater quality (Qin et al. 2015), including by ecological protection and controlling agricultural nonpoint source pollution (Han et al. 2016). In India, the National Water Policy (Government of India 2012) recognises the need for integrated water resources management to meet different water needs, particularly domestic, agricultural and ecological, and the draft National Water Framework Bill (Government of India 2016) specifically mentions wastewater reuse and the consideration of water- and energy-efficiency in agricultural crop production. Both water quality and the WEF nexus are recurrent themes within these and other policy frameworks, yet they are rarely integrated explicitly with each other.

Water quality is defined as "the physical, chemical, biological and organoleptic properties of water" (WMO 2012), which includes water temperature. Here we also consider the processes that control these properties, operating over time and space. When examining the interaction of water with the other dimensions of the WEF nexus, the focus is often on water quantity (van Vliet et al. 2017). Yet, food and energy production need water of a suitable quality as well as in sufficient quantity. Conversely, energy and food production can damage water quality. In this paper, we consider "food" to encompass food produced by agriculture as well as by aquaculture and fisheries in rivers, lakes, and freshwater-influenced estuaries and coastal zones. Although fish can be considered as both a food and a component of water quality within ecological assessment schemes, here we highlight food production and security aspects of agriculture, aquaculture and fisheries because: (1) they are directly affected by water quality and (2) effects of water quality on human health through bioaccumulation in the food chain are primarily expressed in these two pathways.

The role of water quality, and in particular water quality aspects of health, environment, and wider well-being, are rarely acknowledged specifically in discussions of the WEF nexus (Varis and Keskinen 2018). However, the adverse effects of poor water quality are increasingly well documented worldwide. Inadequate drinking water caused an estimated 485,000 deaths due to diarrhoea in low- and middle-income countries in 2016 (Prüss-Üstün et al. 2019). GDP growth is reduced by a third in regions downstream of heavily polluted rivers, and use of saline water for crop watering results in a yield reduction equivalent to feeding 170 million people each year (Damania et al. 2019). Contributions from the hydrological sciences to addressing WEF challenges include hydrological tools and models (Liu et al. 2017, Scanlon et al. 2017, Cudennec et al. 2018) and combined socio-hydrologic modelling approaches (e.g. van Emmerik et al. 2014) to operationalising the WEF nexus, but do not explicitly encompass "water quality". Greater contributions from the hydrological sciences are anticipated through ongoing initiatives led by the International Association of Hydrological Sciences, notably the "Panta Rhei - Everything Flows" Scientific Decade 2013-2022 (McMillan et al. 2016, Montanari et al. 2013) and the identification of 23 "Unsolved Problems in Hydrology" (UPHs) (Blöschl et al. 2019), which include water quality and wider WEF nexus issues.

Even though water quality has been included in the conceptualisation of WEF interlinkages in three large Asian drainage basins (Keskinen et al. 2016), attempts to describe the issues quantitatively were limited to water quantity indicators. This highlights a very important reason for the lack of attention to water quality, i.e. that appropriate surface and groundwater global water quality data sets are not yet available to support such assessments (UNEP 2016). A further reason for the lack of attention to water quality in the WEF nexus is that many different conceptualisations of the nexus exist, including the number of sectors considered in it, the scale of application, and the aim of using a nexus framework (Keskinen et al. 2016). Within different conceptualisations of the WEF nexus, water quality is often used interchangeably with considerations of ecosystems, and human and ecosystem health. Nevertheless, UNECE uses a Water-Food-Energy-Ecosystems nexus to analyse transboundary issues, which does include consideration of water quality (de Strasser et al. 2016, UNECE 2018).

In this paper we aim to demonstrate how explicit inclusion of water quality in the water dimension of the WEF nexus enhances the value of the WEF approach for policy making and helps advance water quality research. This is particularly important for low- and middle-income countries such as those in Africa, Asia and Latin America, which are often disproportionately affected by pressures on the WEF nexus (Schlör et al. 2018). It will also become increasingly urgent under global change, including climate change, accelerated extreme events (Diffenbaugh 2020) and increased demands for water, energy and food (Ceola et al. 2016, Yillia 2016).

2 Identification of water quality as a key component of the different dimensions of the WEF nexus

In this section we discuss examples of water quality issues with respect to energy, food production, water quantity, and tripartite water–energy–food interactions. Due to the breadth and depth of water quality research in many areas - such as biogeochemical cycling, eco-hydrology, limnology - we do not attempt to include all water quality-related research within the current analysis, but note the availability and relevance of this foundational literature to support water quality integration into WEF research. Informed by an initial review of published papers (described in the Supplementary Material and summarised in Table S1), we identified examples of water quality interactions with the

WEF nexus dimensions and classified these by spatial scale (Fig. 1). This exercise revealed that water quality research was only referred explicitly to the WEF framework nexus in a few themes (numbers in black italic font in Fig. 1). We therefore refer to a large number of existing studies that are relevant, but do not make this connection explicit, to demonstrate how water quality is embedded in all three dimensions of the WEF framework. First, water quality aspects of each WEF nexus dimension are examined sequentially in three sub-sections (water quality-energy, water quality-food, water quality-water quantity). Within each sub-section we first discuss how that WEF dimension affects water quality (e.g. how energy production affects water quality) and then how water quality affects that dimension (e.g. how water quality affects energy production). Next, examples are presented of more complex water quality interactions with all three dimensions of the WEF nexus. The section concludes by highlighting water quality-WEF nexus interactions within and across spatial scales. The examples selected from the literature rarely mention WEF explicitly, but they demonstrate the significant impact of water quality included within the "water" dimension on the energy, food and water dimensions of the WEF nexus and vice versa.

[insert Figure 1 here]

2.1 Water quality and energy

2.1.1 Energy impacts on water quality

All forms of energy production have impacts on water quality (WWAP 2014). Water quality can be affected at every step in the extraction, refining and combustion of fossil fuels, including shale gas development using hydraulic fracking, through both regular operations and accidental releases, such as leaks from surface underground storage tanks. Approximately 15-18 billion m^3 of freshwater resources worldwide are affected by fossil fuel production every year (Allen et al. 2012), with significant implications for ecosystems and communities dependent on this water. For example, national demand for coal in the USA has been met partly by mountaintop mining with valley fills in Appalachian Kentucky resulting in contamination of domestic supply wells and selenium concentrations exceeding the threshold for toxic bioaccumulation in more than 90% of streams surveyed (Table S1 #9; Palmer et al. 2010). Atmospheric emissions from thermal power plants can include mercury, sulfur and nitrogen oxides which when transported and deposited across a range of scales may impair water quality and threaten aquatic ecosystems and human health (Peterson et al. 2007). Several pathways of surface water contamination are associated with unconventional shale gas development and hydraulic fracking (Table S1 #9; Vengosh et al. 2014). They involve flowback fluids (fluids returned to the surface after hydraulic fracturing) and produced waters (fluids extracted during production together with the natural gas), which are typically hypersaline and contain elevated concentrations of barium, strontium, radioactive radium and total dissolved organic carbon. Examples of deteriorating surface water quality downstream of treated wastewater disposal from shale gas operations include: increased concentrations of total trihalomethanes (THMs), especially brominated THMs, in municipal drinking water which might compromise disinfection processes (States et al. 2013); and potential environmental risks of radium bioaccumulation in downstream sediments in which ^{226}Ra concentrations may be ~200 times greater than background (Warner et al. 2013).

While other sources of energy, such as hydropower, solar (both photovoltaic and concentrating solar power plants), or wind, have significantly lower greenhouse gas emissions than fossil fuel power plants, they are not without consequences for the environment and water quality. Aside from manufacturing of materials and disposal of old units, maintenance and operation of energy production units can also impact water quality (e.g. application of pesticides and cleaning agents). Their very existence impacts on the land use and thermal balance in the local area. For example, wind farm developments may necessitate forest felling, resulting in a reduction in downstream river water quality due to increased soluble reactive phosphorus concentrations (Heal et al. 2020).

Hydropower reservoirs completely change the characteristics of water bodies from fast flowing to mostly stagnant waters upstream of dams, and their storage and release of water changes the hydrological regime and sediment and nutrient supply, and hinders fish movement. The principal effects on water quality are: an altered thermal regime; lowered dissolved oxygen but increased concentrations of the reduced forms of iron and hydrogen sulfide, limiting the capacity for pollutant breakdown; altered availability of the nutrients phosphorus, nitrogen and silicon; and habitat alteration (Table S1 #10; Winton et al. 2019). Water quality and environmental effects can be damaging and cumulative in river networks, particularly for biodiversity, which underpins fisheries in rivers, lakes, and freshwater-influenced estuaries and coastal zones and livelihoods in many low- and middle-income countries. The additional ~450 dams planned for the Amazon, Mekong and Congo river basins, which together contain one-third of the world's freshwater species, pose a threat to freshwater fish diversity and migration and thus to communities reliant on wild fish (Winemiller et al. 2016).

A number of energy-water quality interactions may be exacerbated by climate change and measures to mitigate against it. For example, warmer temperatures due to climate change can exacerbate water temperatures increased by cooling waters from thermoelectric plants and impoundment of water for hydropower production (Table S1 #26; Lugg and Copeland 2014) and lead to biological impairments (Stewart et al. 2013). The recent increase in large hydropower dams planned principally in low- and middle-income countries is partly attributed to the Paris Agreement of December 2015, with the aim of supporting low greenhouse gas emissions (Hermoso 2017). Increased forestry operations to produce biomass as a substitute for fossil fuels can amplify the transfer of mercury from soil to water, enhancing mercury bioaccumulation in food webs (Eklöf et al. 2016).

Another reported impact on water quality of energy policies is the effect of promoting biomass production on soil erosion and nonpoint source pollution. For example, if the US congressional biofuel production mandate is met without enacting nutrient management measures, a 45% increase in total phosphorus loading in the Upper Mississippi River basin is projected to occur, resulting in declining river water quality and an increase in size of the Gulf of Mexico hypoxic zone (Table S1 #20; Demissie et al. 2012). Conversely, in a study focusing on nitrate, which behaves very differently from phosphorus, crop choice was found to have positive benefits for water quality. For example, modelling of a watershed in Illinois, USA, indicated that adoption of second-generation biofuel crops (e.g. *Miscanthus*) over corn and soybean would help reduce stream nitrate load, although irrigation water demand was predicted to increase (Table S1 #2; Ng et al. 2011). However, the local water quality improvement arising from replacing food-related crops with biofuel crops could be offset by increased food production and associated degradation in water quality in other regions, particularly where freshwater resources are limited and rainfall is insufficient for rainfed crops (Healy et al. 2015).

2.1.2 Water quality impacts on energy

Water quality, or the need to meet certain minimum water quality requirements, also affects the production of and demand for energy. Water temperature can act as a constraint on power production, with elevated water temperatures resulting in reduced power plant efficiency and shutdowns when water temperatures approach threshold values for environmental regulation (Miara et al. 2013). Measures taken to reduce the negative impacts of hydropower dams on fish, such as dam removals, the installation of fish passages, and turbine shutdowns during fish migration periods, result in lower electricity generation (Song et al. 2019). Partly because of more stringent water quality regulations for drinking water and wastewater treatment, the total electricity demand for water and wastewater treatment in the USA increased by more than 50% between 1996 and 2013 (Table S1 #30; EPRI and Water Research Foundation 2013).

Water scarcity in arid, coastal and island locations often necessitates the use of lower quality water sources, such as wastewater and saline water, requiring more extensive treatment or desalination. Desalination not only has high energy requirements to meet water quality use standards, but also produces hypersaline brine that can have negative environmental impacts if not disposed of properly. Currently, desalinated water production for human use totals ~100 million m^3 day^{-1} worldwide and generates ~140 million m^3 day^{-1} of brine (Jones et al. 2019).

The drive for water sensitive practices in cities may also lead to increased concentrations of sediment, nutrients, and organic matter in sewers due to water efficiency measures, resulting in higher treatment costs (Table S1 #7; Murali et al. 2019). However, cost-benefit analyses for 13 wastewater treatment plants in Spain showed that the higher energy costs of treating wastewater can be economically viable when external benefits (e.g. reduced nitrogen and phosphorus pollution of receiving waters) are considered (Table S1 #15; Molinos-Senante et al. 2011).

2.2 Water quality and food

2.2.1 Food impacts on water quality

Since agriculture accounts for the majority of water withdrawals worldwide (70%, FAO 2013), it is not surprising that food production in agriculture and aquaculture has major impacts on all aspects of water quality, including the transfer of emerging contaminants. As the global demand for food (including fish protein from aquaculture) continues to increase at an unprecedented pace there is significant pressure on farmers to intensify production, resulting in continued growth in demand for fertilisers (FAO 2017) and antimicrobials in livestock and aquaculture systems (Van Boeckel et al. 2015). Increased antimicrobial use, including antibiotics, in these sectors results in rising concentrations in surface waters and groundwater due to excretion and manure application (Charuaud et al. 2019), contributing to spread of antimicrobial resistance and human health impacts (Table S1 #22, Baquero et al. 2008). Increasing use of plastic mulch films in many regions of the world as an agricultural practice to enhance crop yield, quality and water-use efficiency, amongst other benefits, also increases concentrations of micro(nano)plastics in agricultural runoff which additionally transport adsorbed micropollutants, such as pesticides and PTEs, into surface and groundwaters (Steinmetz et al. 2016). For example, runoff from tomato plots covered with polyethylene film contained 3 times the sediment concentration and 19, 6, and 9 times the total load of the pesticides chlorothalonil and alpha- and beta-endosulfan, respectively, compared to a vegetative-mulched control (Rice et al. 2001).

Meeting the water quantity demands of food production in agriculture and aquaculture is frequently associated with deteriorating water quality, particularly salinisation. Over-pumping of groundwater for crop irrigation and mismanagement of irrigation water often has undesirable impacts on potable water supplies and crop yield if water and soil salinisation occurs, and on freshwater ecosystems due to drying up of rivers (Table S1 #23; Currell et al. 2012, Jayasekera et al. 2011, Libutti et al. 2018, Qureshi et al. 2010, Singh 2016, Singh 2018). However, in some situations the sources of contaminants in groundwater are geological, e.g. arsenic and fluoride in aquifers of the Indo-Gangetic basin, which can pose a greater water quality threat to groundwater use than over-pumping (MacDonald et al. 2016). Use of saline water in aquaculture can have similar impacts; rapid expansion of shrimp farming using saline water in Bangladesh led to the increase of average soil salinity by 6 to 15 times from 1984 to 2014 (Table S1 #24;

Islam and Tabeta 2019). In the same study, yields of other crops and livestock were reduced significantly due to salinity intrusion caused by shrimp farming.

Food-water quality interactions also occur through the greywater component of water "footprinting" (defined in this use as the amount of water needed to dilute pollution; Liu et al. 2019) and virtual water trade. The few studies that have explored this suggest that food trade may contribute to reducing the global greywater footprint (i.e. reduce total water pollution worldwide) because major food exporting countries had lower pollutant loss intensity than food importers (Liu et al. 2019).

2.2.2 Water quality impacts on food

The use of water of inappropriate quality in crop agriculture is the principal way in which water quality impacts food quality and security, since irrigation with contaminated water reduces crop yields and poses a risk to human health. However, there are also opportunities for wastewater reuse in crop production (when done properly) to enhance yields and reduce pollution of aquatic ecosystems, particularly in areas facing an acute water crisis. Recovery of phosphorus from wastewater can have a strong positive impact on water quality and (reduced) nutrient loads, and substitute for manufactured mineral fertilisers in crop production (Table S1 #14; Schoumans et al. 2015). Due to the sheer volumes involved, wastewater utilisation is crucial for meeting the irrigation and fertiliser requirements of food crops since it contains many nutrients, but the practice also raises environmental and public health concerns (Table S1 #18; Hanjra et al. 2012). The reduction of the water footprint of crop production is a clear benefit, but there are potential risks to health and environment, that require adequate policies.

In addition to PTEs such as chromium, cadmium, copper, arsenic and zinc, effluents from wastewater treatment plants contain micropollutants, including *o*-phenylphenol, chlordane, polychlorinated biphenyls, atrazine, and dioxins, which are endocrine disruptors that can be transferred to soil and subsequently to aquatic environments via wastewater irrigation. Both treated and untreated wastewater contain concentrations of microplastics in ranges of 0-125,000 and 1000-627,000 items m^{-3} that could result in the estimated application of 2×10^6-10^9 ha^{-1} plastic items per cropping season to wastewater-irrigated cabbage and maize crops (Bläsing and Amelung 2018). Although uptake of microplastics into the roots and translocation to shoots of edible plants has been demonstrated, the risk to human health of ingestion of plastics from this exposure pathway is not yet known (Qi et al. 2020). Wastewater is also an important source of antibiotic resistant bacteria (ARB) and antibiotic resistant genes (ARG), which have become a serious public health concern worldwide (Berglund 2015). A high diversity of ARGs is reported in municipal wastewater treated effluent that can serve as a potential hotspot for horizontal gene transfer and selection of ARBs (e.g. Agga et al. 2015, Rizzo et al. 2013). Reuse of wastewater in agricultural irrigation can increase soil concentrations of antibiotics and ARGs. For example, irrigation with untreated wastewater has resulted in soil accumulation of pharmaceuticals and sulfonamide resistance genes in the Mezquital Valley, central Mexico (Table S1 #18; Dalkmann et al. 2012). Contamination of food plants with pathogens can occur through surface contact with contaminated irrigation water, livestock and wildlife faeces, or soil during harvest, posing a health risk if the plant is not properly washed (Allard et al. 2019). Although uptake of pathogens from irrigation water into plant internal tissues has been demonstrated in greenhouse studies (Solomon et al. 2002), uptake of ARGs and faecal pathogens was not evident in plants grown in soils in central Mexico irrigated with untreated wastewater (Table S1 #18; Broszat and Grohmann 2018). The risk to human health from pharmaceutical and personal care products (PPCPs) consumed in the edible plant tissues from crops that were irrigated with wastewater has been suggested to be minimal, although the risk from mixtures of PPCPs requires further assessment (Prosser and Sibley 2015).

A further, more indirect, impact of water quality on crop production may occur when environmental flows set to improve water quality result in lower yields through reduced irrigation water allocations. This is exemplified in the Murray-Darling basin, Australia, which provides 40% of Australia's agricultural production by value and almost all of Australia's rice production. In this case, proposed reductions in the average annual water use to allow more environmental flows were projected to cause annual agricultural production to fall by 13-17% (Table S1 #25; Leblanc et al. 2012).

2.3 Water quality and water quantity

Although we have so far highlighted water quality interactions with the energy and food dimensions of the WEF nexus, we discuss below some examples of the interaction between water quality and water quantity to demonstrate that they are inextricably coupled (Nilsson and Renöfält 2008).

2.3.1 Water quantity impacts on water quality

Dams constructed on river systems for the purpose of water abstraction have similar impacts on downstream water quality and river health as hydropower dams, already discussed in Section 2.1.1. Urban water management is another example of how management for water quantity impacts water quality. In many parts of the world in the 20th century, urban water management initially focused on flood risk reduction within cities through combined or separately-sewered piped systems, which managed flood risk by transporting stormwater runoff rapidly downstream, but resulted in water quality deterioration through combined sewer overflows or washoff of pollutants from urban surfaces. In the early 21st century nonpoint source stormwater runoff was identified as the leading source of water pollution in the USA (Lee et al. 2007). The more recent adoption of distributed source control approaches to urban flood risk reduction in many parts of the world under various names (including: Best Management Practices – BMPs, Low Impact Development – LID, Water Sensitive Urban Design – WSUD, Sustainable Urban Drainage Systems – SUDS, and Blue Green Infrastructure - BGI) has provided co-benefits in improving water quality as well as water savings (Barbosa et al. 2012, Ossa-Moreno et al. 2017, Alves et al. 2019).

2.3.2 Water quality impacts on water quantity

The quality of water also defines the quantity of water available for a particular use, including potable water supply, toilet flushing, car washing, fire extinguishing, and managed aquifer recharge. For example, elevated nitrate concentrations in Vermilion River in Illinois, USA, limit the usability of river water for drinking water supply for the city of Pontiac, requiring use of an off-stream reservoir blending system (Hecht et al. 2014). The growing threat of cyanobacteria dominance in rivers and reservoirs can also limit use of water that would otherwise be available (He et al. 2016). Ever more stringent water quality regulations in many countries are also impacting water quantity, both in constraining the availability of water of sufficient quality, but also in increasing the demand for water to dilute contaminated flows and/or provide environmental flows (Pittock and Lankford 2010). Thus, increasingly water resource management options are being investigated which resolve both water pollution and water scarcity issues. For example, managed aquifer recharge with treated wastewater and urban river water has been demonstrated as a feasible way to address growing water demands and the lack of treatment of wastewater in cities such as Addis Ababa, Ethiopia (Table S1 #17; Abiye et al. 2009) .

2.4 Water quality and tripartite water–energy–food interactions

Water quality can be embedded in complex tripartite interactions involving all three dimensions of the WEF nexus, as demonstrated at a range of spatial scales in the case-studies presented below.

Lightly polluted water (e.g. from showers, baths, and hand-basins, also referred to as "greywater") can offer another unconventional option for non-potable water reuse at the household/community scale (Table S1 #4; Boyjoo et al. 2013) and is suitable for on-site treatment, reducing demands on water resources and energy requirements of wastewater treatment systems. It is important to develop low energy and low maintenance greywater treatment systems that provide critical amenities and micro-climate benefits to cities facing water crises (Connor et al. 2020, Wong and Brown 2009). Biofiltration systems, such as green walls which require only a small land area, potentially represent a low cost, low maintenance on-site greywater treatment approach for use in dense urban areas (Pradhan et al. 2019). A number of studies have demonstrated the effectiveness of green walls for greywater treatment (Table S1 #4, e.g. Masi et al. 2016), with choice of plants (Table S1 #4; Fowdar et al. 2017), growing media and maintenance shown to be important considerations for optimal performance. Despite the reported contamination of greywater, health risks of bacterial pathogens from greywater are negligible where treatment recommendations have been followed (Table S1 #4; Benami et al. 2016). Furthermore, studies in Australia and Israel investigating if greywater use increases the risk of gastrointestinal illness, identified no significant differences in risk between greywater-exposed and control groups (Table S1 #4; Busgang et al. 2015).

At the larger drainage basin and regional/national scales, activities targeted to boost one or more dimensions of the WEF nexus and improve livelihoods can have negative consequences for the other dimensions, including water quality deterioration. For example, in the semi-arid area of Pernambuco, Brazil, tilapia fish farming has been encouraged in the Itacuruba reservoir dammed for hydropower production, to meet the growing demand for fish protein in Brazil and improve the economic situation of local people affected by the dam construction (Table S1 #24; Marques et al. 2018). However, phosphorus-containing effluent from intensive tilapia cage fish farms in the reservoir poses a risk to the already scarce water resources through eutrophication of the dam waters. In Egypt, the country's water availability has been increased by 20% by reusing agricultural drainage water, but only by increased energy demand for large pumping stations and small diesel pumps to return water from drainage ditches to irrigation canals. Moreover, water availability has been achieved at the expense of water quality deterioration in the country's irrigation network due to the increased concentration of salinity and pollutants in drainage water (Table S1 #29; Barnes 2014).

Water quality can also be the vector of conflicting interactions between the individual dimensions of the WEF nexus in drainage basins within countries, and in transboundary basins. For example, reservoirs constructed in the Upper Mahaweli Catchment, Sri Lanka, generate ~27% of the country's electricity through hydropower and provide irrigation water for 3000 km^2 of agricultural land (Table S1 #21; Diyabalanage et al. 2017). However, high soil erosion rates in the catchment (up to 700 t km^{-2} y^{-1}, 100 times greater than background soil production rates) attributed to intensive cultivation of crops on steep land have caused rapid sedimentation within the reservoirs and reduced water storage capacity to 56% over 17 years after impoundment in one reservoir and 72% over 3 years in another (Hewawasam 2010). At the transboundary scale, it is estimated that the construction of the 11 proposed dams on the main stem of the Mekong River primarily for hydropower energy production, but also for increased irrigation and flood control, would result in a 75% reduction in nutrient loading which supports the productivity of floodplain agriculture and fisheries downstream (Table S1 #31; ICEM 2010). A 21-42% reduction in fisheries production is projected, which could cause nutritional deficiencies in an additional 2-8 million people in the Lower Mekong Delta by 2030 (Golden et al. 2019).

2.5 Scale is a key consideration in water quality interactions in the WEF nexus

As has become apparent from the examples discussed above and summarised in Table S1, water quality interactions within the WEF nexus play out across a range of spatial scales. Water quality-WEF nexus interactions not only occur from the very local (individual households and fields) to international transboundary drainage basins, but complications occur because the impacts of these interactions may cross scales. Activities at the household/farm scale can affect WEF-water quality interactions at larger scales. For example, farmer decisions to replace food crops with biofuel crops are predicted to improve water quality locally, but displace intensive food crop production and associated water quality deterioration to other locations (Table S1 #2; Zhong et al. 2018). Conversely, national WEF-related policies can cause localised water quality impacts, as exemplified by national demand for coal in the USA resulting in significant surface and groundwater quality impairment in Appalachian Kentucky (Scott et al. 2011). Furthermore, there can be transboundary physical effects propagated from upstream to downstream, e.g. when management of hydropower dams impacts downstream water quality and river health, or virtual water embedded in commodities consumed in one country impact on water availability and quality in other countries (O'Bannon et al. 2014, Liu et al. 2019). The multi- and cross-scale nature of water quality-WEF interactions suggests that different data and analysis approaches are needed for different scales.

3 Perspective and outlook to better integrate water quality research into WEF issues

3.1 Including water quality in the WEF nexus helps to achieve SDG targets

Applying a water quality lens to the WEF nexus can help to identify appropriate solutions required to achieve the SDG targets. Environmental problems associated with poor water quality generally lead to other social and economic issues (Ghodsvali et al. 2019), driving complex feedbacks between social cohesion, land and water resource management and environmental conditions (Thanh et al. 2020). The myriad of water quality issues, and diversity of pollutants, leads to highly site-specific manifestations of these feedbacks, making them difficult to generalise. Here two of the most prominent water quality issues to the WEF nexus are discussed: eutrophication and sediments. We focus on these for the sake of brevity, noting that similar reasoning can be applied to <u>other important water issues such as salinisation, micropollutants and hazardous substances, plastics, pathogens and antimicrobial resistance, and aquatic ecosystem health</u> more broadly.

Eutrophication is a particularly instructive issue to analyse in this context as it spans several WEF nexus axes, is relevant across multiple SDGs, and in many cases is multi-scale and cross-boundary (Biermann et al. 2016, Le Moal et al. 2019). Existing annual fluxes of nitrogen and phosphorus already exceed the “planetary boundary” values estimated to define “a safe operating space for humanity” (Steffen et al. 2015). A primary driver has been the food production system based on crop and livestock agriculture that releases organic and inorganic nutrients into aquatic and terrestrial ecosystems. A call to end world hunger (SDG 2) through increasing agricultural food productivity amplifies eutrophication pressures if mitigation approaches are not adopted. Within cities, urban stormwater drainage practices and inadequate sanitation and wastewater treatment also significantly contribute to eutrophication, most notably in low- and middle-income countries. On the other side of the WEF nexus, controlling and reducing these nutrient releases into the environment consumes energy. Construction, maintenance, and operation of conventional centralised wastewater treatment systems contribute to achieving SDG Target 6.3 (improved water quality), but lead to increasing energy demand. The increasing adoption of blue green infrastructure, such as reducing wastewater at source, designing facilities to generate their own energy, and decentralised treatment technologies, such as constructed wetlands (Alcamo 2019), is helping to alleviate this increased energy demand.

Another water quality issue recognised at a global level, and as a defining feature of the Anthropocene, is altered river sediment regimes (Syvitski and Kettner 2011). Human activities have resulted in an estimated 160% increase in land-ocean sediment flux, whilst reservoirs trap 66% of the sediment flux within river systems (Walling 2008). Accelerated soil erosion has been linked to human activities, primarily agriculture, although other causes can be locally important (e.g. deforestation or construction). The loss of nutrient-rich top soil through erosion reduces agricultural productivity, thereby increasing demands for resources to sustain food production (FAO 1996). Dams, many of which are used for hydropower production or water supply for drinking or irrigation, also upset the river sediment balance. Sediments deposit in reservoirs due to the long residence time and low velocities encountered behind dams, slowly diminishing the capacity of the reservoir and reducing water security. While sediment management in reservoir systems is possible (Annandale et al. 2003), it can be difficult in practice and requires energy (Coker et al. 2009). In addition, flows from reservoirs may become sediment-starved (Kondolf 1997), leading to increased channel erosion or beach erosion and a loss of productive agricultural land or fisheries downstream of dams (Owusu et al. 2017). Unfortunately, downstream effects of dams (as also discussed above in Sections 2.1.1 and 2.4) are not always fully considered during the evaluation of dam proposals. Accordingly, river sediment regimes should be considered in devising appropriate solutions to several SDGs, such as zero hunger (SDG 2), clean water and sanitation (SDG 6), affordable and clean energy (SDG 7), combating climate change and its impacts (SDG 13) and life below water (SDG 14).

Although our recommendation is that water quality should become intrinsic to all WEF studies, it is recognised that this is seriously hindered by the limited availability of suitable water quality data in many situations (discussed in Section 2.3.2). This is especially a problem at smaller scales where management efforts are often targeted, since national, regional, or global estimates from statistical analyses, remote sensing, or coarse-scale hydrological models, must be relied upon for setting policy at larger scales. Therefore, for the scientific community to make tangible and rapid progress towards achieving the SDGs, we propose that efforts to collect water quality data and analyse water quality interactions with WEF should initially target nexus "hotspots" where WEF-related stresses are most severe and are anticipated to grow most rapidly. Such hotspots have been identified as: (1) urban areas (e.g. Grimm et al. 2008); (2) areas of intensive groundwater use for agricultural food and energy production (e.g. the North China Plain – Zheng et al. 2010); (3) semi-arid and arid areas where climate change and the use of scarce water resources threaten water quality (e.g. the Mediterranean region, where nexus projects are already planned – Global Water Partnership – Mediterranean 2019; Egypt – Abdel-Dayam 2011; Australia – Davis et al. 2015; and Texas, USA – Daher et al. 2019, Mohtar and Daher 2019); and (4) other areas particularly vulnerable to climate change, such as drainage basins heavily dependent on meltwater from snow and glaciers (Huss et al. 2017). Explicit inclusion of water quality in the WEF nexus is also essential for addressing transboundary issues, for which UNECE has proposed a water-food-energy-ecosystems nexus methodology (de Strasser et al. 2016, UNECE 2018).

3.2 The need for coordinated and open water quality datasets

In order to better integrate water quality research within WEF issues, there is a clear need for water quality data suited to addressing nexus challenges (Cudennec et al. 2020, Scanlon et al. 2017). Relevant water quality monitoring is required to reflect changing environmental, climate and anthropic conditions, including emerging contaminants. In water quality-WEF interactions related to human and ecosystem health, this monitoring must be rapid and reliable, for example, to ensure the safety of food crops irrigated with recycled wastewater. However, despite several efforts to compile water quality data at a global scale (e.g. GEMStat (https://gemstat.org/), the Global River Chemistry Database (GLORICH, Hartman et al. 2014), and the Water Quality Database of the Global Open Data Index (https://index.okfn.org/dataset/water/), water quality data are sparse in many parts of the world where the pressures are the greatest. One of the aims of the newly established World Water Quality Alliance co-ordinated by UNEP, within the framework of UN Water, is to build upon the Global Environment Monitoring System for Water (GEMS/Water) that includes water quality measurements in groundwater, rivers, lakes and reservoirs, and wetlands, to improve the availability of data that is fit-for-purpose for water quality assessment (UNEP 2016). Opportunities for this may be enhanced by increasing use of remote sensing for some parameters (e.g. chlorophyll-*a* to indicate eutrophication in lakes and large rivers; Alikas et al. 2015), and water quality monitoring using affordable and wireless sensors and smart phones including by citizen scientists (e.g. Liao et al. 2020); though this approach is not without challenges (Buytaert et al. 2014). Large-scale models can be augmented to include water quality components to harvest the information from monitored basins for estimating water quality in ungauged catchments as estimates of long-term values (e.g. Cohen et al. 2013) or at a finer temporal resolution using hydrodynamic models (e.g. Bartosova et al 2019).

In addition to data describing ambient water quality, the quality of municipal and industrial wastewater effluents, agricultural return flows with irrigation water sources and types, drinking water sources, wet atmospheric deposition, and urban stormwater require specific attention to generate additional information critical to understanding water quality issues within the WEF nexus. New indices and analyses are also needed to fully harness the rich data provided by ever-increasing water quality monitoring (Hipsey et al. 2015). These include development of holistic water quality and river/lake/groundwater health indicators supported by enhanced remote sensing capabilities, novel *in situ* water quality sensors, and the necessary hydro-informatics approaches associated with data analytics (Tauro et al. 2018). Many new indicators are being developed to assess progress towards multi-faceted and interlinked challenges, such as the SDG indicators, water footprinting and virtual water, or the quantification of ecosystem services. A challenge for hydrologists is to articulate conventional water quality assessments with these new WEF related use requirements.

3.3 Moving beyond case-studies: research needs for including water quality within WEF nexus analyses

We have used a number of examples to demonstrate how including water quality within the WEF context would have augmented and improved the analysis and its application. The case-studies demonstrate that the WEF nexus cuts across scales (household/field, drainage basin, region, and impacts at a distance through trade and transportation), is highly trans-disciplinary, and displays dynamics that can evolve over time. There is no doubt this complexity will continue to create seemingly endless opportunities for studies into local issues and solutions, but the pressing challenge is to develop generalised conceptualisation and analysis frameworks that allow us to transcend the anecdotal. Indeed, this challenge is reflected in the recent compilation of "Unsolved Problems in Hydrology" (Blöschl et al. 2019).

The need for more integrated WEF nexus analyses is increasingly pressing with growth in demands for water, energy and food. Methods for identifying an optimum solution space in the nexus – that also consider water quality constraints – are necessary to understand the trade-offs and synergies that exist in the system. Prior approaches to this problem for allocation of water quantity in drainage basins to multiple users include multi-objective optimisation models (Roozbahani et al. 2014), analytical methods such as bankruptcy rules (Mianabadi et al. 2014) and game theory (Madani 2010). The goals are to maximise the net benefit or production of allocated water to agriculture (food), domestic and industrial users (water supply), and hydropower (energy), whilst minimising water shortage, sewage drainage, and the amounts of polluted water (Liu et al. 2012). Game theory provides a framework for the study of the strategic behaviours of individual decision makers with which to develop more broadly acceptable solutions (Madani 2010), and has been integrated into a general systems analysis for water resources exploring solutions with both cooperative and non-cooperative actors (Madani and Hooshyar 2014, Roozbahani et al. 2014, Wei et al. 2010). Further research is required to develop these methods for diverse application contexts and extend their ability to account for the changing drivers of water, energy and food, such as population and economic growth, urbanisation, improvements in living standards, and changes in diets and consumption patterns (Yillia 2016).

A further practical challenge remains in identifying suitable metrics for summarising the water quality attributes that drive change; for example, how do we quantify aquatic system "health". Risk assessment methods represent one approach to quantify in a tangible way the impacts of water quality within the WEF nexus, translating diverse water quality data to human health or ecological risks. These methods relate the quantity of the water quality substance of interest, its characteristics, and its use, to a quantifiable dimensionless risk that can be evaluated alongside economic indicators for impact assessment and comparison of alternatives across substances. For example, the U.S. EPA recommends four steps to assess health risks (U.S. EPA 2014): (1) exposure assessment that quantifies pathways, duration, frequency, and other factors of exposure; (2) hazard identification that specifies the type of impact that can be associated with the substance (e.g. cancer, birth defects); (3) dose-response assessment that defines a relationship between the exposure dose and the probability of the hazard; and (4) risk characterisation that quantifies the magnitude and uncertainty of risk. These general steps are also applicable to assessing ecological risks. The risk-based assessments can evaluate impacts of different water quality substances and express the impacts in a comparable way, although significant information about the substances needs to be available for such quantifications.

The WEF nexus is not static, and needs to consider the changing nature of global change drivers, which affect demands for water, energy and food, individually as well as their interactions. The effects of climate change in particular increase the relevance of water quality in shaping key WEF nexus issues. For example, the decade-long drought in California in 2007-2016 caused an increasing reliance on groundwater resources for irrigation. The intense pumping of groundwater in the Central Valley to sustain crop yields resulted in seawater intrusion into aquifers near the coastal zone, thus reducing the quality and availability of potable water resources (Goebel et al. 2017). Climate change will also contribute to increased cyanobacterial dominance in polluted waterways (Carey et al. 2012), further challenging water supply systems and river health, with implications for water quantity, treatment costs and reuse. Further, increasing water temperatures due to climate change are projected to impact freshwater fish populations, though the impacts can be spatially variable due to the nonlinear relationships between fish survival and water temperature and their interaction with water management activities, such as reservoir operation (Zhang et al. 2019). Whilst climate change and other global drivers for food and energy tend to intensify water quality related problems, there is also evidence that this can drive innovation, resulting in increased overall resilience of the nexus to stressors (e.g.

Low et al. 2015). Identifying opportunities to integrate WEF analyses with research on new and emerging water quality management technologies can assist decision makers to undertake cost-benefit calculations and identify economically viable solutions.

The non-stationarity in global, regional and local drivers of water, energy and food sectors, plus the myriad of complex feedbacks linking communities with their environment requires continued advancement in integrated earth-human system modelling frameworks (e.g. Elshafei et al. 2014, Di Baldassarre et al. 2019). The need for WEF nexus analysis of multi-disciplinary, multi-scale and multi-model frameworks integrating physical, chemical, ecological and socio-economic processes and feedbacks has already been identified (Liu et al. 2017) with examples increasingly emerging (Howells et al. 2013, van Emmerik et al. 2014, Zhang et al. 2019, Zhang et al. 2020). Great opportunities exist to expand these earth-human system modelling frameworks to more coherently include water quality dynamics and feedbacks. In parallel, there have been advances in local and regional to global scale water quality modelling (Hofstra et al. 2019, Hipsey et al. 2020). Efforts to improve integration of these modelling approaches to account for energy and food drivers will allow for more comprehensive tools able to operate over a range of scales relevant to decision making within the nexus.

The complex external drivers and internal processes within these models also create challenges for addressing uncertainty, such as in model inputs, structures and validating datasets, which will further grow given the future myriad of potential scenarios and plausible trajectories (Maier et al. 2016). Convincing relevant stakeholders of the utility of WEF analyses and model predictions requires transparent communication of the risks and uncertainties in the recommended solutions that aim to balance competing interests. New practical approaches dealing with the treatment of uncertainty relating to water quality within the various axes of the WEF nexus will help facilitate the uptake of these analyses by decision makers.

4 Conclusions

Through discussion of a number of examples, we have highlighted the significant impact of water quality on the energy, food and water dimensions of the WEF nexus and vice versa. We advocate that including water quality explicitly will increase the ability of the WEF framework to generate integrated policies and management approaches, not only for water quality improvement, but most importantly for balancing and meeting demands for water, energy, food, and ecosystem needs. Including water quality adds themes of health, well-being and environment that are not often explicitly covered within the nexus, and helps to address some of the power and social inequalities in the WEF nexus conceptualisation (Wiegleb and Bruns 2018).

Ultimately, societal needs have driven the development of hydrological science (Sivapalan and Blöschl 2017) and, given the importance of water quality in the Sustainable Development Goals (SDG 6 in particular), addressing this need may be an overlooked opportunity for the hydrological sciences community. For example, meeting the SDG Targets 6.3 (improve water quality) and 6.5 (implement integrated water resource management) requires the understanding of pollution sources and mobilisation through hydrological pathways into rivers, lakes and aquifers that the hydrological sciences can provide. Given that only ~30% of the journal papers cited in this paper are in the "Water Resources" journal subject category of Web of Science, we conclude that water quality-WEF studies in water-related journals are under-represented and suggest that hydrological scientists could do more to ensure that their research contributes to addressing complex nexus challenges. So that water quality does not remain an overlooked component of the WEF nexus, we encourage hydrological scientists to label and promote relevant water quality research as addressing WEF nexus challenges during both the research design and publication stages.

We have demonstrated that water quality within the WEF nexus cuts across scales (households and fields, cities, drainage basins, regions, transboundary and distant through trade and transportation). This scale-independency demonstrates the generic appeal of the WEF nexus approach and its flexibility when put into practice for generating innovative and impactful measures to improve water quality for addressing grand challenges in different settings. Nevertheless, the generic approach to considering the role of water quality in the WEF nexus taken here should be implemented within specific country or drainage basin contexts so that resource constraints and other local conditions are considered, such as cultural practices and ecosystem services relating to water.

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References

Abdel-Dayem, S. 2011. Water quality management in Egypt. *International Journal of Water Resources Development,* 27(1), 181-202.

Abiye, T. A., Sulieman, H. and Ayalew, M. 2009. Use of treated wastewater for managed aquifer recharge in highly populated urban centers: a case study in Addis Ababa, Ethiopia. *Environmental Geology,* 58(1), 55-59.

Agga, G. E.*, et al.* 2015. Antimicrobial-resistant bacterial populations and antimicrobial resistance genes obtained from environments impacted by livestock and municipal waste. *Plos One,* 10(7), e0132586.

Alcamo, J. 2019. Water quality and its interlinkages with the Sustainable Development Goals. *Current Opinion in Environmental Sustainability,* 36, 126-140.

Alikas, K.*, et al.* 2015. Satellite-based products for monitoring optically complex inland waters in support of EU Water Framework Directive. *International Journal of Remote Sensing,* 36(17), 4446-4468.

Allard, S. M., *et al.* 2019. Creek to Table: Tracking fecal indicator bacteria, bacterial pathogens, and total bacterial communities from irrigation water to kale and radish crops. *Science of the Total Environment,* 666, 461-471.
Allen, L., *et al.*, 2012. Fossil fuels and water quality. *In:* Gleick, P. H. ed. *The World's Water Volume 7, the Biennial Report on Freshwater Resources.* Washington, DC: Island Press/Center for Resource Economics, 73-96.
Alves, A., *et al.* 2019. Assessing the co-benefits of green-blue-grey infrastructure for sustainable urban flood risk management. *Journal of Environmental Management,* 239, 244-254.
Annandale, G. W., *et al.*, 2003. *Reservoir Conservation: Economic and Engineering Evaluation of Alternative Strategies for Managing Sedimentation in Storage Reservoirs: RESCON Approach.* Washington DC: The World Bank.
Baquero, F., Martinez, J. L. and Canton, R. 2008. Antibiotics and antibiotic resistance in water environments. *Current Opinion in Biotechnology,* 19(3), 260-265.
Barbosa, A. E., Fernandes, J. N. and David, L. M. 2012. Key issues for sustainable urban stormwater management. *Water Research,* 46(20), 6787-6798.
Barnes, J. 2014. Mixing waters: The reuse of agricultural drainage water in Egypt. *Geoforum,* 57, 181-191.
Bartosova, A., *et al.* 2019. Future socioeconomic conditions may have a larger impact than climate change on nutrient loads to the Baltic Sea. *Ambio,* 48(11), 1325-1336.
Benami, M., Gillor, O. and Gross, A. 2016. Potential microbial hazards from graywater reuse and associated matrices: A review. *Water Research,* 106, 183-195.
Berglund, B. 2015. Environmental dissemination of antibiotic resistance genes and correlation to anthropogenic contamination with antibiotics. *Infection Ecology & Epidemiology,* 5(1), 28564.
Biermann, F., *et al.* 2016. Down to Earth: Contextualizing the Anthropocene. *Global Environmental Change,* 39, 341-350.
Bläsing, M. and Amelung, W. 2018. Plastics in soil: Analytical methods and possible sources. *Science of the Total Environment,* 612, 422-435.
Blöschl, G., *et al.* 2019. Twenty-three unsolved problems in hydrology (UPH) - a community perspective. *Hydrological Sciences Journal,* 64(10), 1141-1158.
Blöschl, G. and Sivapalan, M. 1995. Scale issues in hydrological modelling: A review. *Hydrological Processes,* 9(3-4), 251-290.
Boyjoo, Y., Pareek, V. K. and Ang, M. 2013. A review of greywater characteristics and treatment processes. *Water Science and Technology,* 67(7), 1403-1424.
Broszat, M. and Grohmann, E., 2018. Antimicrobial resistance spread mediated by wastewater irrigation. *In:* Keen, P. L. and Fugère, R. eds. *Antimicrobial Resistance in Wastewater Treatment Processes.* Hoboken, NJ: Wiley, 207-217.
Busgang, A., *et al.* 2015. Epidemiological study for the assessment of health risks associated with graywater reuse for irrigation in arid regions. *Science of the Total Environment,* 538, 230-239.
Buytaert, W., *et al.* 2014. Citizen science in hydrology and water resources: opportunities for knowledge generation, ecosystem service management, and sustainable development. *Frontiers in Earth Science,* 2, 26.
Carey, C. C., *et al.* 2012. Eco-physiological adaptations that favour freshwater cyanobacteria in a changing climate. *Water Research,* 46(5), 1394-1407.
Ceola, S., *et al.* 2016. Adaptation of water resources systems to changing society and environment: a statement by the International Association of Hydrological Sciences. *Hydrological Sciences Journal,* 61(16), 2803-2817.

Charuaud, L., *et al.* 2019. Veterinary pharmaceutical residues in water resources and tap water in an intensive husbandry area in France. *Science of the Total Environment,* 664, 605-615.

Cohen, S., *et al.* 2013. WBMsed, a distributed global-scale riverine sediment flux model: Model description and validation. *Computers & Geosciences,* 53, 80-93.

Coker, E. H., Hotchkiss, R. H. and Johnson, D. A. 2009. Conversion of a Missouri River dam and reservoir to a sustainable system: Sediment management. *Journal of the American Water Resources Association,* 45(4), 815-827.

Connor, R., *et al.*, 2020. Chapter 9 Water-Climate-Energy-Food-Environment Nexus. *In:* WWAP ed. *The United Nations World Water Development Report 2020: Water and Climate Change.* Paris: UNESCO, World Water Assessment Programme, 118-125.

Cudennec, C., *et al.* 2020. Editorial – Towards FAIR and SQUARE hydrological data. *Hydrological Sciences Journal,* 65(5), 681-682.

Cudennec, C., *et al.* 2018. Epistemological dimensions of the water–energy–food nexus approach: reply to discussions of "Challenges in operationalizing the water–energy–food nexus". *Hydrological Sciences Journal,* 63(12), 1868-1871.

Currell, M. J., *et al.* 2012. Sustainability of groundwater usage in northern China: dependence on palaeowaters and effects on water quality, quantity and ecosystem health. *Hydrological Processes,* 26, 4050-4066.

Daher, B., *et al.* 2019. Towards bridging the water gap in Texas: A water–energy–food nexus approach. *Science of the Total Environment,* 647, 449-463.

Dalkmann, P., *et al.* 2012. Accumulation of pharmaceuticals, *enterococcus*, and resistance genes in soils irrigated with wastewater for zero to 100 years in Central Mexico. *Plos One,* 7(9), e45397.

Damania, R., *et al.*, 2019. *Quality Unknown: The Invisible Water Crisis.* Washington, DC: World Bank.

Davis, J., *et al.* 2015. When trends intersect: The challenge of protecting freshwater ecosystems under multiple land use and hydrological intensification scenarios. *Science of the Total Environment,* 534, 65-78.

de Strasser, L., *et al.* 2016. A methodology to assess the water energy food ecosystems nexus in transboundary river basins. *Water,* 8(2), 59.

Demissie, Y., Yan, E. and Wu, M. 2012. Assessing regional hydrology and water quality implications of large-scale biofuel feedstock production in the Upper Mississippi River Basin. *Environmental Science & Technology,* 46(16), 9174-9182.

Di Baldassarre, G., *et al.* 2019. Sociohydrology: Scientific challenges in addressing the Sustainable Development Goals. *Water Resources Research,* 55(8), 6327-6355.

Diffenbaugh, N. S. 2020. Verification of extreme event attribution: Using out-of-sample observations to assess changes in probabilities of unprecedented events. *Science Advances,* 6(12), eaay2368.

Diyabalanage, S., *et al.* 2017. Impact of soil and water conservation measures on soil erosion rate and sediment yields in a tropical watershed in the Central Highlands of Sri Lanka. *Applied Geography,* 79, 103-114.

Eklöf, K., Lidskog, R. and Bishop, K. 2016. Managing Swedish forestry's impact on mercury in fish: Defining the impact and mitigation measures. *Ambio,* 45(2), 163-174.

Elshafei, Y., *et al.* 2014. A prototype framework for models of socio-hydrology: identification of key feedback loops and parameterisation approach. *Hydrology and Earth System Sciences,* 18(6), 2141-2166.

EPRI and Water Research Foundation, 2013. *Electricity Use and Management in the Municipal Water Supply and Wastewater Industries USA.* Palo Alto, CA: Electric Power Research Institute.

EU Directive on Water Policy 2000. Directive 2000/60/EC of the European Parliament and of the Council of the 23 October 2000 establishing a framework for community action in the field of water policy. *Official Journal of the European Communities,* L327, 1-72.

Fader, M., *et al.* 2018. Toward an understanding of synergies and trade-offs between water, energy, and food SDG targets. *Frontiers in Environmental Science,* 6, 112.

FAO, 1996. *Control of Water Pollution from Agriculture.* Rome: Food and Agriculture Organization of the United Nations.

FAO, 2003. *Agriculture Food and Water.* Rome: Food and Agriculture Organization of the United Nations.

FAO, 2017. *World Fertilizer Trends and Outlook to 2020.* Rome: Food and Agriculture Organization of the United Nations.

Fowdar, H. S., *et al.* 2017. Designing living walls for greywater treatment. *Water Research,* 110, 218-232.

Ghodsvali, M., Krishnamurthy, S. and de Vries, B. 2019. Review of transdisciplinary approaches to food-water-energy nexus: A guide towards sustainable development. *Environmental Science & Policy,* 101, 266-278.

Global Water Partnership – Mediterranean, 2019. *Nexus in the Med: UfM Water Agenda & UN Env/MAP GEF MedProgramme* [online]. Available from: https://www.gwp.org/globalassets/global/gwp-med-files/list-of-programmes/see-nexus/regional-roundtable/2019/tirana-nexus-roundtable---nexus-in-the-mediterranean-gwp-med.pdf [Accessed 19/05/2020].

Goebel, M., Pidlisecky, A. and Knight, R. 2017. Resistivity imaging reveals complex pattern of saltwater intrusion along Monterey coast. *Journal of Hydrology,* 551, 746-755.

Golden, C. D., *et al.* 2019. Impacts of mainstream hydropower development on fisheries and human nutrition in the Lower Mekong. *Frontiers in Sustainable Food Systems,* 3, 93.

Government of India, 2012. *National Water Policy (2012)* [online]. Available from: Ministry of Jal Shakti, Department of Water Resources http://jalshakti-dowr.gov.in/sites/default/files/NWP2012Eng6495132651_1.pdf [Accessed 18/05/2020].

Government of India, 2016. *Draft National Water Framework Bill (2016)* [online]. Available from: Ministry of Jal Shakti, Department of Water Resources http://jalshakti-dowr.gov.in/sites/default/files/Water_Framework_May_2016.pdf [Accessed 18/05/2020].

Grimm, N. B., *et al.* 2008. Global change and the ecology of cities. *Science,* 319(5864), 756-760.

Han, D., Currell, M. J. and Cao, G. 2016. Deep challenges for China's war on water pollution. *Environmental Pollution,* 218, 1222-1233.

Hanjra, M. A., *et al.* 2012. Wastewater irrigation and environmental health: Implications for water governance and public policy. *International Journal of Hygiene and Environmental Health,* 215(3), 255-269.

Hartmann, J., Lauerwald, R. and Moosdorf, N. 2014. A brief overview of the GLObal RIver CHemistry Database, GLORICH. *Procedia Earth and Planetary Science,* 10, 23-27.

He, X. X., *et al.* 2016. Toxic cyanobacteria and drinking water: Impacts, detection, and treatment. *Harmful Algae,* 54, 174-193.

Heal, K., *et al.* 2020. Wind farm development on peatlands increases fluvial macronutrient loading. *Ambio,* 49(2), 442-459.

Healy, R. W., *et al.*, 2015. *The Water-Energy Nexus—An Earth Science Perspective: U.S. Geological Survey Circular 1407.* Reston, Virginia: U.S. Geological Survey.

Hecht, J. S., Cai, X. M. and Eheart, J. W. 2014. Operating rules for an off-stream blending reservoir to control nitrate in a municipal water system. *Journal of Water Resources Planning and Management,* 140(8), 04014015

Hermoso, V. 2017. Freshwater ecosystems could become the biggest losers of the Paris Agreement. *Global Change Biology,* 23(9), 3433-3436.

Hewawasam, T. 2010. Effect of land use in the upper Mahaweli catchment area on erosion, landslides and siltation in hydropower reservoirs of Sri Lanka. *Journal of the National Science Foundation of Sri Lanka,* 38(1), 3-14.

Hipsey, M. R., *et al.* 2020. A system of metrics for the assessment and improvement of aquatic ecosystem models. *Environmental Modelling & Software,* 128, 104697.

Hipsey, M. R., *et al.* 2015. Predicting the resilience and recovery of aquatic systems: A framework for model evolution within environmental observatories. *Water Resources Research,* 51(9), 7023-7043.

Hofstra, N., *et al.* 2019. Editorial overview: Water quality: A new challenge for global scale model development and application. *Current Opinion in Environmental Sustainability,* 36, A1-A5.

Howells, M., *et al.* 2013. Integrated analysis of climate change, land-use, energy and water strategies. *Nature Climate Change,* 3(7), 621-626.

Huss, M., *et al.* 2017. Toward mountains without permanent snow and ice. *Earth's Future,* 5(5), 418-435.

ICEM, 2010. *Strategic Environmental Assessment of Hydropower on the Mekong Mainstream. Summary of the Final Report.* Hanoi: International Centre for Environmental Management.

Islam, M. R. and Tabeta, S. 2019. Shrimp vs prawn-rice farming in Bangladesh: A comparative impacts study on local environments and livelihoods. *Ocean & Coastal Management,* 168, 167-176.

Jayasekera, D. L., Kaluarachchi, J. J. and Villholth, K. G. 2011. Groundwater stress and vulnerability in rural coastal aquifers under competing demands: a case study from Sri Lanka. *Environmental Monitoring and Assessment,* 176(1-4), 13-30.

Jones, E., *et al.* 2019. The state of desalination and brine production: A global outlook. *Science of the Total Environment,* 657, 1343-1356.

Keskinen, M., *et al.* 2016. The water–energy–food nexus and the transboundary context: Insights from large Asian rivers. *Water,* 8(5), 193.

Kondolf, G. M. 1997. Hungry water: Effects of dams and gravel mining on river channels. *Environmental Management,* 21(4), 533-551.

Le Moal, M., *et al.* 2019. Eutrophication: A new wine in an old bottle? *Science of the Total Environment,* 651, 1-11.

Leblanc, M., *et al.* 2012. A review of historic and future hydrological changes in the Murray-Darling Basin. *Global and Planetary Change,* 80-81, 226-246.

Lee, H., *et al.* 2007. Design of stormwater monitoring programs. *Water Research,* 41(18), 4186-4196.

Liao, J., *et al.* 2020. Wireless water quality monitoring and spatial mapping with disposable whole-copper electrochemical sensors and a smartphone. *Sensors and Actuators B: Chemical,* 306, 127557.

Libutti, A., Cammerino, A. R. B. and Monteleone, M. 2018. Risk assessment of soil salinization due to tomato cultivation in Mediterranean climate conditions. *Water,* 10(11), 1503.

Liu, D., *et al.* 2012. A macro-evolutionary multi-objective immune algorithm with application to optimal allocation of water resources in Dongjiang River basins, South China. *Stochastic Environmental Research and Risk Assessment,* 26(4), 491-507.

Liu, J., *et al.* 2017. Challenges in operationalizing the water–energy–food nexus. *Hydrological Sciences Journal,* 62(11), 1714-1720.

Liu, W. F., *et al.* 2019. Savings and losses of global water resources in food-related virtual water trade. *WIREs Water,* 6(1), e1320.

Low, K. G., *et al.* 2015. Fighting drought with innovation: Melbourne's response to the Millennium Drought in Southeast Australia. *WIREs Water,* 2(4), 315-328.

Lugg, A. and Copeland, C. 2014. Review of cold water pollution in the Murray-Darling Basin and the impacts on fish communities. *Ecological Management & Restoration,* 15(1), 71-79.

MacDonald, A. M., *et al.* 2016. Groundwater quality and depletion in the Indo-Gangetic Basin mapped from *in situ* observations. *Nature Geoscience,* 9(10), 762-766.

Madani, K. 2010. Game theory and water resources. *Journal of Hydrology,* 381(3), 225-238.

Madani, K. and Hooshyar, M. 2014. A game theory–reinforcement learning (GT–RL) method to develop optimal operation policies for multi-operator reservoir systems. *Journal of Hydrology,* 519, 732-742.

Maier, H. R., *et al.* 2016. An uncertain future, deep uncertainty, scenarios, robustness and adaptation: How do they fit together? *Environmental Modelling & Software,* 81, 154-164.

Marques, E. A. T., *et al.* 2018. Assessing the negative impact of an aquaculture farm on effluent water quality in Itacuruba, Pernambuco, Brazilian semiarid region. *Water Science and Technology,* 78(7), 1438-1447.

Masi, F., *et al.* 2016. Green walls for greywater treatment and recycling in dense urban areas: a case-study in Pune. *Journal of Water Sanitation and Hygiene for Development,* 6(2), 342-347.

McGrane, S. J., *et al.* 2019. Scaling the nexus: Towards integrated frameworks for analysing water, energy and food. *Geographical Journal,* 185(4), 419-431.

McMillan, H., *et al.* 2016. Panta Rhei 2013–2015: global perspectives on hydrology, society and change. *Hydrological Sciences Journal,* 61(7), 1174-1191.

Mianabadi, H., *et al.* 2014. A new bankruptcy method for conflict resolution in water resources allocation. *Journal of Environmental Management,* 144, 152-159.

Miara, A., *et al.* 2013. Riverine ecosystem services and the thermoelectric sector: strategic issues facing the Northeastern United States. *Environmental Research Letters,* 8(2), 025017.

Mohtar, R. H. and Daher, B. 2019. Lessons learned: Creating an interdisciplinary team and using a nexus approach to address a resource hotspot. *Science of the Total Environment,* 650, 105-110.

Molinos-Senante, M., Hernandez-Sancho, F. and Sala-Garrido, R. 2011. Cost-benefit analysis of water-reuse projects for environmental purposes: A case study for Spanish wastewater treatment plants. *Journal of Environmental Management,* 92(12), 3091-3097.

Montanari, A., *et al.* 2013. “Panta Rhei—Everything Flows”: Change in hydrology and society—The IAHS Scientific Decade 2013–2022. *Hydrological Sciences Journal,* 58(6), 1256-1275.

Murali, M. K., *et al.* 2019. The development and application of improved solids modelling to enable resilient urban sewer networks. *Journal of Environmental Management,* 240, 219-230.

Ng, T. L., *et al.* 2011. An agent-based model of farmer decision-making and water quality impacts at the watershed scale under markets for carbon allowances and a second-generation biofuel crop. *Water Resources Research,* 47, W09519.

Nilsson, C. and Renöfält, B. M. 2008. Linking flow regime and water quality in rivers: a challenge to adaptive catchment management. *Ecology and Society,* 13(2), 18.

O'Bannon, C., *et al.* 2014. Globalization of agricultural pollution due to international trade. *Hydrology and Earth System Sciences,* 18(2), 503-510.

Ossa-Moreno, J., Smith, K. M. and Mijic, A. 2017. Economic analysis of wider benefits to facilitate SuDS uptake in London, UK. *Sustainable Cities and Society,* 28, 411-419.

Owusu, K., Obour, P. B. and Nkansah, M. A. 2017. Downstream effects of dams on livelihoods of river-dependent communities: the case of Ghana's Kpong Dam. *Geografisk Tidsskrift-Danish Journal of Geography,* 117(1), 1-10.

Palmer, M. A., *et al.* 2010. Mountaintop mining consequences. *Science,* 327(5962), 148-149.

Peterson, S. A., *et al.* 2007. Mercury concentration in fish from streams and rivers throughout the Western United States. *Environmental Science & Technology,* 41(1), 58-65.

Pittock, J. and Lankford, B. A. 2010. Environmental water requirements: demand management in an era of water scarcity. *Journal of Integrative Environmental Sciences,* 7(1), 75-93.

Pradhan, S., Al-Ghamdi, S. G. and Mackey, H. R. 2019. Greywater recycling in buildings using living walls and green roofs: A review of the applicability and challenges. *Science of the Total Environment,* 652, 330-344.

Prosser, R. S. and Sibley, P. K. 2015. Human health risk assessment of pharmaceuticals and personal care products in plant tissue due to biosolids and manure amendments, and wastewater irrigation. *Environment International,* 75, 223-233.

Prüss-Üstün, A., *et al.* 2019. Burden of disease from inadequate water, sanitation and hygiene for selected adverse health outcomes: An updated analysis with a focus on low- and middle-income countries. *International Journal of Hygiene and Environmental Health,* 222(5), 765-777.

Qi, R., *et al.* 2020. Behavior of microplastics and plastic film residues in the soil environment: A critical review. *Science of the Total Environment,* 703, 134722.

Qin, Y., *et al.* 2015. China's energy-water nexus – assessment of the energy sector's compliance with the "3 Red Lines" industrial water policy. *Energy Policy,* 82, 131-143.

Qureshi, A. S., Gill, M. A. and Sarwar, A. 2010. Sustainable groundwater management in Pakistan: challenges and opportunities. *Irrigation and Drainage,* 59(2), 107-116.

Rice, P. J., *et al.* 2001. Runoff loss of pesticides and soil: A comparison between vegetative mulch and plastic mulch in vegetable production systems. *Journal of Environmental Quality,* 30(5), 1808-1821.

Rizzo, L., *et al.* 2013. Urban wastewater treatment plants as hotspots for antibiotic resistant bacteria and genes spread into the environment: A review. *Science of the Total Environment,* 447, 345-360.

Roozbahani, R., *et al.* 2014. A Multi-objective Approach for Transboundary River Water Allocation. *Water Resources Management,* 28(15), 5447-5463.

Scanlon, B. R., *et al.* 2017. The food-energy-water nexus: Transforming science for society. *Water Resources Research,* 53(5), 3550-3556.

Schlör, H., *et al.* 2018. Deliberations about a perfect storm - The meaning of justice for food energy water-nexus (FEW-Nexus). *Journal of Environmental Management,* 220, 16-29.

Schoumans, O. F., *et al.* 2015. Phosphorus management in Europe in a changing world. *Ambio,* 44, S180-S192.

Scott, C. A., *et al.* 2011. Policy and institutional dimensions of the water-energy nexus. *Energy Policy,* 39(10), 6622-6630.

Singh, A. 2016. Hydrological problems of water resources in irrigated agriculture: A management perspective. *Journal of Hydrology,* 541, 1430-1440.

Singh, A. 2018. Alternative management options for irrigation-induced salinization and waterlogging under different climatic conditions. *Ecological Indicators,* 90, 184-192.

Sivapalan, M. and Blöschl, G. 2017. The growth of hydrological understanding: Technologies, ideas, and societal needs shape the field. *Water Resources Research,* 53(10), 8137-8146.

Solomon, E. B., Yaron, S. and Matthews, K. R. 2002. Transmission of *Escherichia coli* O157:H7 from contaminated manure and irrigation water to lettuce plant tissue and its subsequent internalization. *Applied and Environmental Microbiology,* 68(1), 397-400.

Song, C., *et al.* 2019. Managing dams for energy and fish tradeoffs: What does a win-win solution take? *Science of the Total Environment,* 669, 833-843.

States, S., *et al.* 2013. Marcellus Shale drilling and brominated THMs in Pittsburgh, Pa., drinking water. *Journal American Water Works Association,* 105(8), E432-E448.

Steffen, W., *et al.* 2015. Planetary boundaries: Guiding human development on a changing planet. *Science,* 347(6223), 1259855.

Steinmetz, Z., *et al.* 2016. Plastic mulching in agriculture. Trading short-term agronomic benefits for long-term soil degradation? *Science of the Total Environment,* 550, 690-705.

Stewart, R. J., *et al.* 2013. Horizontal cooling towers: riverine ecosystem services and the fate of thermoelectric heat in the contemporary Northeast US. *Environmental Research Letters,* 8(2), 025010.

Syvitski, J. P. M. and Kettner, A. 2011. Sediment flux and the Anthropocene. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences,* 369(1938), 957-975.

Tauro, F., *et al.* 2018. Measurements and Observations in the XXI century (MOXXI): innovation and multi-disciplinarity to sense the hydrological cycle. *Hydrological Sciences Journal,* 63(2), 169-196.

Thanh, H. T., Tschakert, P. and Hipsey, M. R. 2020. Tracing environmental and livelihood dynamics in a tropical coastal lagoon through the lens of multiple adaptive cycles. *Ecology and Society,* 25(1).

U.S. EPA, 2014. *Framework for Human Health Risk Assessment to Inform Decision Making. EPA/100/R-14/001.* Washington DC, USA.

U.S. Government, 1972. *33 U.S. Code §1251 et seq. Federal Water Pollution Control Act* [online]. Available from: https://www.govinfo.gov/content/pkg/USCODE-2018-title33/pdf/USCODE-2018-title33-chap26.pdf [Accessed 19/05/2020].

UN-Water, 2016. *Water and Sanitation Interlinkages across the 2030 Agenda for Sustainable Development.* Geneva: UN-Water.

UNECE, 2018. *Methodology for Assessing the Water-Food-Energy-Ecosystems Nexus in Transboundary Basins and Experiences from its Application: Synthesis.* Geneva: United Nations Economic Commission for Europe.

UNEP, 2016. *A Snapshot of the World's Water Quality: Towards a Global Assessment.* Nairobi: United Nations Environment Programme.

United Nations, 2015. *Transforming our World: The 2030 Agenda for Sustainable Development.* New York: United Nations, A/RES/70/1.

United Nations, 2018. *Sustainable Development Goal 6 Synthesis Report 2018 on Water and Sanitation.* New York: United Nations.

Van Boeckel, T. P.*, et al.* 2015. Global trends in antimicrobial use in food animals. *Proceedings of the National Academy of Sciences of the United States of America,* 112(18), 5649-5654.

van Emmerik, T. H. M.*, et al.* 2014. Socio-hydrologic modeling to understand and mediate the competition for water between agriculture development and environmental health: Murrumbidgee River basin, Australia. *Hydrology and Earth System Sciences,* 18(10), 4239-4259.

van Vliet, M. T. H., Florke, M. and Wada, Y. 2017. Quality matters for water scarcity. *Nature Geoscience,* 10(11), 800-802.

Varis, O. and Keskinen, M. 2018. Discussion of "Challenges in operationalizing the water–energy–food nexus". *Hydrological Sciences Journal,* 63(12), 1863-1865.

Venghaus, S. and Hake, J. F. 2018. Nexus thinking in current EU policies – The interdependencies among food, energy and water resources. *Environmental Science & Policy,* 90, 183-192.

Vengosh, A.*, et al.* 2014. A critical review of the risks to water resources from unconventional shale gas development and hydraulic fracturing in the United States. *Environmental Science & Technology,* 48(15), 8334-8348.

Walling, D. E., 2008. The changing sediment loads of the world's rivers. *In:* J. Schmidt*, et al.* eds. *Sediment Dynamics in Changing Environments, IAHS Publication 325.* Wallingford, UK: IAHS Press, 323-338.

Warner, N. R.*, et al.* 2013. Impacts of shale gas wastewater disposal on water quality in Western Pennsylvania. *Environmental Science & Technology,* 47(20), 11849-11857.

Wei, S.*, et al.* 2010. Game theory based models to analyze water conflicts in the Middle Route of the South-to-North Water Transfer Project in China. *Water Research,* 44(8), 2499-2516.

Wiegleb, V. and Bruns, A. 2018. What is driving the water–energy–food nexus? Discourses, knowledge, and politics of an emerging resource governance concept. *Frontiers in Environmental Science,* 6, 128.

Winemiller, K. O.*, et al.* 2016. Balancing hydropower and biodiversity in the Amazon, Congo, and Mekong. *Science,* 351(6269), 128-129.

Winton, R. S., Calamita, E. and Wehrli, B. 2019. Reviews and syntheses: Dams, water quality and tropical reservoir stratification. *Biogeosciences,* 16(8), 1657-1671.

WMO, 2012. *International Glossary of Hydrology.* Geneva: World Meteorological Organization.

Wong, T. H. F. and Brown, R. R. 2009. The water sensitive city: principles for practice. *Water Science and Technology,* 60(3), 673-682.

WWAP, 2014. *The United Nations World Water Development Report 2014: Water and Energy.* Paris: UNESCO, World Water Assessment Programme.

Yillia, P. T. 2016. Water–energy–food nexus: framing the opportunities, challenges and synergies for implementing the SDGs. *Österreichische Wasser-und Abfallwirtschaft,* 68(3), 86-98.

Zhang, X.*, et al.* 2020. River Regulation Alleviates the Impacts of Climate Change on U.S. Thermoelectricity Production. *Journal of Geophysical Research: Atmospheres,* 125(4), e2019JD031618.

Zhang, X.*, et al.* 2019. On the variable effects of climate change on Pacific salmon. *Ecological Modelling,* 397, 95-106.

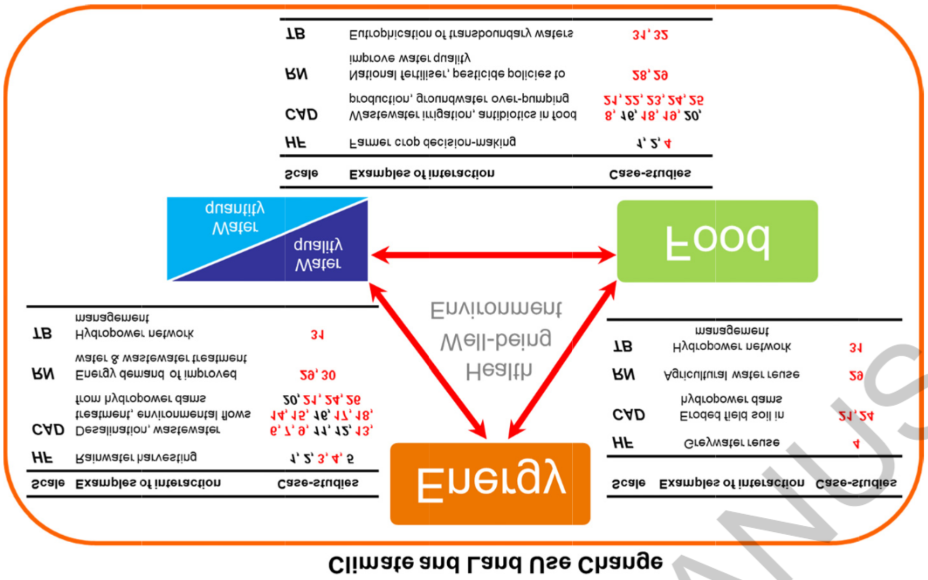
Zheng, C.*, et al.* 2010. Can China Cope with Its Water Crisis?—Perspectives from the North China Plain. *Groundwater,* 48(3), 350-354.

Zhong, J., *et al.* 2018. Effect of land use change for bioenergy production on feedstock cost and water quality. *Applied Energy,* 210, 580-590.

Figure caption

Figure 1. Overview of some themes that are relevant to water quality issues within the WEF framework. Numbers refer to case-study themes in Table S1. Themes in which the WEF framework is mentioned (***black italic font***) are distinguished from themes where it is not explicitly mentioned (**red font**). Studies are classified by scale in tables along the WEF dimensions using spatial scale definitions for hydrology which are loosely based on the four scales identified in Blöschl and Sivapalan (1995). "HF", "CAD", "RN" and "TB" refer to the scales of "*Household/Farm field plot*", "*City/Aquifer/Drainage basin*", "*Region/Nation*" and "*Transboundary*" respectively. The figure format is based on Fig. 1 in Liu et al. (2017), with the orange envelope around the WEF nexus indicating that WEF interactions occur within (and also influence) climate and land use change.

Fig 1



Scale	Examples of interaction	Case-studies
HF	Famer crop decision-making	1, 2, 4
CAD	Wastewater irrigation, antibiotics in food production, groundwater over-pumping	8, 16, 18, 19, 20, 21, 22, 23, 24, 25
RN	National fertiliser, pesticide policies to improve water quality	28, 29
TB	Eutrophication of transboundary waters	31, 32

Scale	Examples of interaction	Case-studies
HF	Rainwater harvesting	1, 2, 3, 4, 5
CAD	Desalination, wastewater treatment, environmental flows from hydropower dams	6, 7, 9, 11, 12, 13, 14, 15, 16, 17, 18, 20, 21, 24, 26
RN	Energy demand of improved water & wastewater treatment	29, 30
TB	Hydropower network management	31

Scale	Examples of interaction	Case-studies
HF	Greywater reuse	4
CAD	Eroded field soil in hydropower dams	21,24
RN	Agricultural water reuse	29
TB	Hydropower network management	31

Supplementary Material

Water quality: the missing dimension of water in the water–energy–food nexus

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Description of the initial review of published papers

The starting point for researching this opinion piece was a literature search in Web of Knowledge in November 2017 using topic "water quality and energy food nexus" for the period 1900-2017. This initial literature review helped to identify the case-study themes in Fig. 1, and was followed by further literature review targeting these themes. The initial search identified 32 results, all from 2011 or later, apart from one result in 1995. Of these, only seven journal papers explicitly related water quality to the WEF nexus. These are indicated by the case-study numbers in black italic font in Fig. 1. The numbers in red font in Fig. 1 represent other published studies that are relevant to water quality issues within the WEF framework, but in which the WEF framework is not explicitly mentioned. From the dominance of red numbers in Fig. 1, it is clear that water quality issues are not widely acknowledged or well integrated into WEF research, despite the obvious significance of water quality issues and volume of relevant studies in non-WEF literature.

A high proportion of the existing WEF studies that explicitly mention water quality were focused on urban areas, which were identified as nexus hotspots because of the pressures of meeting large and concentrated demands for water, energy and food. Investigating the relative effects of future water quality and climate change scenarios on energy use in two water treatment plants, life-cycle analysis showed that energy use increased when lower quality water inputs were used, due to increased treatment chemical usage (Table S1 #12; Stang et al. 2018). However, in the water treatment plant located in a humid continental climate, this increase may be partially offset by future global warming, which would reduce the energy demand for heating the plant. Miller-Robbie et al. (2017) (Table S1 #16) analysed wastewater treatment and reuse for irrigation of food crops in the city of Hyderabad, India, through a WEF lens. One of their conclusions was that, due to the dilute nature of wastewater effluent, to fully utilise the nutrients contained in treated wastewater, large areas of cropland are required within or near to the city, which may not be available due to the other land demands of a rapidly growing urban population. Surprisingly, additional energy invested in treating wastewater before reuse did not have the full expected benefit in reducing pathogen contamination of food crops in this study. This was attributed to crop contamination arising from farmer harvesting practices, in which there was frequent contact between the harvested crop and the pathogen-contaminated soil, and harvested crops were placed under sacking moistened with wastewater to prevent wilting. However, other water quality issues associated with urban wastewater or sludge, such as potentially toxic elements (PTEs), pharmaceuticals, polycyclic aromatic hydrocarbons or plastics, to name just a few, and any long-term impacts from their application on agricultural land were not evaluated.

Agricultural landscapes are also critical geographies for the WEF nexus, as they produce food and energy crops that both require water resources and also have a number of impacts for surrounding aquatic environments, particularly in regions where freshwater resources are limited and rainfall is insufficient for rainfed crops (Healy et al. 2015). In all regions the application of pesticides, herbicides and fertilisers (including as manures) to both food and energy crops can result in subsurface leaching and runoff of nutrients and agro-chemicals during rainfall events, contaminating surface water bodies and groundwater. This hinders their capacity to be used in supporting downstream demands such as drinking water provision, irrigation water or sustaining aquatic habitats (Table S1 #1; Jayasekera et al. 2011, Yao et al. 2018). In extreme situations, the excess runoff of nutrient-laden sediment can result in eutrophication and the development of harmful algal blooms (HABs) in both freshwater and coastal environments, posing a significant threat to aquatic habitats (Table S1 #20; Reddy et al. 2018, Le Moal et al. 2019). The choice of crop type can affect the water quality outcomes for downstream users and ecosystems. For example, economic modelling showed that conversion of food-related crops to switchgrass for cellulosic ethanol biofuel in west Tennessee, USA, would reduce nitrate loading to groundwater locally (Table S1 #2; Zhong et al. 2018). However, whilst promotion of energy over food crops may result in local improvement in water quality, the concern remains that “more polluting” crops could be displaced to other areas.

A notable feature of Fig. 1 is that several water quality case-studies (4, 21, 24, 29, 31) occur in all three dimensions of the WEF framework, demonstrating how water quality can be embedded in complex tripartite nexus interactions. The large number of existing studies that are relevant to water quality in a WEF framework, but do not make this connection explicit, demonstrates that these water quality dimensions occur across scales, from individual farms to cities and their hinterlands, and link not only surface water and groundwater, but also upstream and downstream users in drainage basins.

Table S1. Alignment of water quality issues with the WEF nexus dimensions, in order of increasing spatial scale at which the issue occurs. *Italics* represent case-studies where the connection between water quality and the WEF nexus is mentioned explicitly in the reference. The items listed are examples that are not meant to be all-inclusive. WEF dimension notation: W = water, E = energy, F = food.

Increasing spatial scale

Case-study number	Case-study theme	WEF dimensions	Examples of interaction with water quality issues and further comment	References
Household / Farm field plot scale (~10s-100s m)				
1	*Fertiliser, pesticide, herbicide inputs to food and bioenergy crops*	*F→W* *E→W*	*Can compromise local surface and groundwater sources*	Jayasekera et al. (2011), *Yao et al. (2018)*
2	*Farmer crop decision-making*	*F→W* *E→W*	*Crop choice has impacts (can be both positive and negative) on water quality and water demand. Shift from food crops to biofuel feedstock crops may reduce nitrate loading to freshwaters, but may have other local and regional consequences.* Shift from apples to more water-intensive berry crops in Monterey County, California, has contributed to groundwater over-abstraction, resulting in saline intrusion and threats to agricultural production.	*Ng et al. (2011), Zhong et al. (2018)* Rudestam et al. (2018)
3	Rainwater harvesting	W→E	Rainwater harvesting decreases energy required for water transport, but changes water quality risk profile	Amos et al. (2016)
4	Greywater re-use	W→W W→E W→F	Greywater re-use for irrigation (crops, landscape), toilet flushing, car washing, fire extinguishing, decreases demand on centralised water supply system and energy associated with wastewater treatment. Human health concerns about handling contaminated greywater, but limited epidemiological studies so far have found evidence of this.	Boyjoo et al. (2013), Busgang et al. (2015), Benami et al. (2016), Masi et al. (2016), Fowdar et al. (2017), Prodanovic et al. (2017)
5	*Household water treatment*	*W→E*	*Localised water treatment increases energy demand of householders*	*Hussien et al. (2018)*

City/Aquifer/Drainage basin scale (~10s km)				
6	Stormwater management to reduce nutrient and contaminant loads	W→E	Reducing urban flood risks through storm water networks reduces water quality; conversely WSUD (Water Sensitive Urban Design) initiatives improve water quality	Bell et al. (2017), Carstens and Amer (2019)

Increasing spatial scale

7	Sewer network blockages	W→E	Water savings measures in cities drive increasing sewer solids concentrations, increasing energy and water requirements for network management	Murali et al. (2019)
8	Contamination reducing water supply	F→W	Industrial contamination of rivers and aquifers reduces the water quality available for potable supply or for irrigation use	Burri et al. (2019)
9	Coal and coal seam gas (CSG) extraction; hydraulic fracking for shale gas	E→W	Extraction of coal creates issues associated with acid mine drainage and CSG pollutes aquifers. Water use for hydraulic fracking for shale gas can lead to water shortages in areas with drier climate and/or already limited water availability, resulting in water quality degradation. It also has the potential to reduce quality of groundwater and surface water.	Palmer et al. (2010) Vengosh et al. (2014)
10	Water supply downstream impacts	W→W	Dams for water supply impact downstream water quality and river health	Winton et al. (2019)
11	*Water transport*	*W→E*	*Energy demand of water transport to overcome water quality constraints in local source waters (salinity, contamination)*	*Sanders and Webber (2012)*
12	*Water treatment to reduce pathogens and contaminants*	*W→E*	*Energy demand associated with water treatment to reduce health risk. Lower quality water inputs to treatment plants increase energy use.*	*Stang et al. (2018)*
13	Desalination	W→E	Energy demand of desalination of high salinity water to overcome water quality constraints in local source waters (salinity, contamination)	Elimelech and Phillip (2011)
14	Wastewater treatment to reduce pollution and recover nutrients	W→E	Energy demand associated with conventional wastewater treatment to reduce point source pollution. However, if recovered nutrients substituted for chemical fertilisers used in agriculture avoids energy used in fertiliser production.	Schoumans et al. (2015), Alcamo (2019)
15	Wastewater treatment to recover water	W→E W→W	Energy costs associated with treatment of wastewater to suitable standards for reuse. Cost-benefit analyses showed that water reuse at 13 Spanish wastewater treatment plants was economically viable when external benefits (e.g. reduced nitrogen and phosphorus pollution of receiving waters) were included.	Molinos-Senante et al. (2011)

16	*Reuse wastewater in urban farming to recover nutrient and water resources and*	*W→F* *W→E*	*Although wastewater reuse brought water cycling benefits, land constraints in urban areas may limit the nutrient recovery from wastewater. Wastewater reuse can compromise food safety (pathogen content) due to crop harvesting practices.*	*Miller-Robbie et al. (2017)*

Increasing spatial scale

	reduce energy use in wastewater treatment			
17	Managed aquifer recharge with effluent from wastewater treatment	W→E W→W	Benefits are reduced water scarcity, groundwater overabstraction reduced, lower energy use in water treatment, improved water quality through avoided wastewater effluent discharge	Abiye et al. (2009)
18	Wastewater irrigation	W→F W→E W→W	Many benefits, including: higher crop yields, energy savings in fertiliser production, reduced pollutant discharge into waterways, reduced water footprint of agriculture. However, risks need to be addressed, e.g. soil and groundwater contamination, public and farmer health, human health risk via food consumption from contaminated irrigation water. Accumulation of pharmaceuticals and antimicrobial resistance genes in soils in central Mexico irrigated with untreated wastewater, but no evidence of plant uptake sufficient to cause a health risk.	Hanjra et al. (2012). Many examples in Otoo and Drechsel (2018) Dalkman et al. (2012), Broszat and Grohmann (2018)
19	Irrigation return flows	F→W	Returned water from irrigation districts is often of poor water quality and re-enters river systems	Alcamo (2019)
20	*Eutrophication*	*F→W* *E→W*	*Accumulation of diffuse pollution from cropping for food and biofuel production and animal husbandry for food production leads to hypoxia, harmful algal blooms and other challenges in downstream receiving waters.* e.g. Increased biofuel production in Upper Mississippi River Basin (to meet US congressional mandate to produce 36 billion gallons of biofuel per year by 2022) projected to result in 12% and 45% increases in annual suspended sediment and total phosphorus loadings, respectively, but a 3% decrease in nitrogen loading. Increased phosphorus loadings may increase the size of the Gulf of Mexico hypoxic zone if nutrient management measures are not enacted.	*Reddy et al. (2018)* Demissie et al. (2012)
21	Soil erosion from agriculture	F→W→E	Soil erosion from intensive crop cultivation generates high soil erosion, resulting in rapid sedimentation and reduced capacity of hydropower reservoirs	Diyabalanage et al. (2017)

22	Use of antibiotics in livestock farming, freshwater aquaculture, mariculture	F→W	Increase in antimicrobial use for animal husbandry and aquaculture results in increased concentrations in surface and groundwater due to leaching and excretion, with spread of antimicrobial resistance and human health impacts.	Baquero et al. (2018) Li et al. (2018)

Increasing spatial scale

			e.g. These activities are the main source of antibiotics measured in water and sediment in the Pearl River Delta, south China. Concern that can compromise water quality and ecology, but in this study most antibiotics assessed to pose insignificant risk to fish and green algae, and insignificant to medium risk to *Daphnia*.	
23	Groundwater extraction causing salinisation	F→W	Continued extraction of groundwater for irrigation compromises drinking and irrigation water supply through saline intrusion and concentrates salts in the soil profile, resulting in reduced crop yield. Reduced cultivation of high water-demanding crops (e.g. rice) and small changes in conjunctive use of irrigation water supplied from groundwater and canals modelled to prevent further groundwater logging and soil salinisation.	Qureshi et al. (2008), Rudestam et al. (2015) Singh (2018)
24	Aquaculture	F→W E→F→W	Beneficial protein production from aquaculture but can compromise local water quality and, if eutrophication occurs in reservoirs for hydropower production, algal blooms could clog intakes. P-rich effluent from fish farms poses eutrophication risk to receiving waters, especially where dammed for hydropower provision. Salinisation of soil and groundwater has occurred from saline water shrimp farming in Bangladesh, reducing crop and livestock production.	Marques et al. (2018) Islam and Tabeta (2019)
25	Environmental flow provision	W→F W→W	Government supported environmental flow releases to maintain water quality and river health reduces water supply for food production, e.g. Murray-Darling basin, Australia. Reservoir releases dilute upstream pollutants to provide downstream potable water	Leblanc et al. (2012) Yoon et al. (2015)
26	Energy supply downstream impacts	E→W	Management of hydroelectric dams impact downstream water quality and river health	Lugg and Copeland (2014)
27	Reservoir releases	W→W	Reservoir operation to supply downstream water quantity demands causes deterioration in reservoir water quality for potable and irrigation uses, e.g. the Tuyamuyun Hydroengineering Complex, Amu Darya watershed, Central Asia	Froebrich et al. (2007)
Region/Nation scale (~100s km)				

28	Fertiliser, herbicide and pesticide policy	W→F	Policies to reduce eutrophication and improve water quality impact on the competitiveness of the food production sector	Larsson and Granstedt (2010)

Increasing spatial scale

29	Policy to reuse agricultural drainage water to increase water availability	W→E W→W W→F	Agricultural water reuse requires more energy for water pumping. It also results in increasing salinity and pollutant concentrations in recycled water, which can reduce crop yields, e.g. Egypt.	Barnes (2014)
30	Stricter regulations on drinking water quality	W→E	Energy demand for water and wastewater increases for improved treatment processes to ensure quality standards are met for drinking water supplies (human health driven) and wastewater discharges (ecosystem health driven), e.g. USA	EPRI and Water Research Foundation (2013)
Transboundary scale (~1000s km)				
31	Hydropower network optimisation impacts river water quality and water supply for food production	E→W→F	Managing hydropower dams and releases across a network, e.g. Mekong, Euphrates-Tigris, Nile	ICEM (2010)
32	Agricultural policies, activities and runoff resulting in	F→W	Intense agricultural practices can result in eutrophication across international borders, where different governance structures overlap, e.g. the Great Lakes region in North America and the Baltic Sea in Europe.	Jetoo (2018)

References (additional to References in the paper text)

Amos, C. C., Rahman, A. and Gathenya, J. M. 2016. Economic analysis and feasibility of rainwater harvesting systems in urban and peri-urban environments: A review of the global situation with a special focus on Australia and Kenya. *Water,* 8(4), 149.

Bell, C. D*., et al.* 2017. Characterizing the effects of stormwater mitigation on nutrient export and stream concentrations. *Environmental Management,* 59(4), 604-618.

Burri, N. M*., et al.* 2019. A review of threats to groundwater quality in the anthropocene. *Science of the Total Environment,* 684, 136-154.

Carstens, D. and Amer, R. 2019. Spatio-temporal analysis of urban changes and surface water quality. *Journal of Hydrology,* 569, 720-734.

Elimelech, M. and Phillip, W. A. 2011. The future of seawater desalination: Energy, technology, and the environment. *Science,* 333(6043), 712-717.

Froebrich, J*., et al.* 2007. Water quantity and quality dynamics of the THC - Tuyamuyun Hydroengineering Complex and implications for reservoir operation. *Environmental Science and Pollution Research,* 14(6), 435-442.

Hussien, W. A., Memon, F. A. and Savic, D. A. 2018. A risk-based assessment of the household water-energy-food nexus under the impact of seasonal variability. *Journal of Cleaner Production,* 171, 1275-1289.

Jetoo, S. 2018. Barriers to effective eutrophication governance: A comparison of the Baltic Sea and North American Great Lakes. *Water,* 10(4), 400.

Larsson, M. and Granstedt, A. 2010. Sustainable governance of the agriculture and the Baltic Sea - Agricultural reforms, food production and curbed eutrophication. *Ecological Economics,* 69(10), 1943-1951.

Le Moal, M*., et al.* 2019. Eutrophication: A new wine in an old bottle? *Science of the Total Environment,* 651, 1-11.

Li, S*., et al.* 2018. Antibiotics in water and sediments of rivers and coastal area of Zhuhai City, Pearl River estuary, south China. *Science of the Total Environment,* 636, 1009-1019.

Miller-Robbie, L., Ramaswami, A. and Amerasinghe, P. 2017. Wastewater treatment and reuse in urban agriculture: exploring the food, energy, water, and health nexus in Hyderabad, India. *Environmental Research Letters,* 12(7), 075005.

Otoo, M. and Drechsel, P., eds., 2018. *Resource Recovery from Waste: Business Models for Energy, Nutrient and Water Reuse in Low- and Middle-Income Countries.* Oxon, UK: Routledge – Earthscan.

Prodanovic, V*., et al.* 2017. Green walls for greywater reuse: Understanding the role of media on pollutant removal. *Ecological Engineering,* 102, 625-635.

Reddy, V. R., Cunha, D. G. F. and Kurian, M. 2018. A Water–Energy–Food Nexus Perspective on the Challenge of Eutrophication. *Water,* 10(2), 101.

Rudestam, K., Brown, A. and Langridge, R. 2018. Exploring “Deep Roots”: Politics of place and groundwater management practices in the Pajaro Valley, California. *Society & Natural Resources,* 31(3), 291-305.

Rudestam, K., Langridge, R. and Brown, A. 2015. "The commons" as a dynamic variable in understanding strategic alliances of scale: A groundwater case study in Pajaro Valley, California. *Environmental Science & Policy,* 52, 33-40.

Sanders, K. T. and Webber, M. E. 2012. Evaluating the energy consumed for water use in the United States. *Environmental Research Letters,* 7(3), 034034.

Stang, S*., et al.* 2018. Influences of water quality and climate on the water-energy nexus: A spatial comparison of two water systems. *Journal of Environmental Management,* 218, 613-621.

Winton, R. S., Calamita, E. and Wehrli, B. 2019. Reviews and syntheses: Dams, water quality and tropical reservoir stratification. *Biogeosciences,* 16(8), 1657-1671.

Yao, Y. X., Martinez-Hernandez, E. and Yang, A. D. 2018. Modelling nutrient flows in a simplified local food-energy-water system. *Resources Conservation and Recycling,* 133, 343-353.

Yoon, T., Rhodes, C. and Shah, F. A. 2015. Upstream water resource management to address downstream pollution concerns: A policy framework with application to the Nakdong River basin in South Korea. *Water Resources Research,* 51(2), 787-805.