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Global Water Challenges of Food and Energy Systems in the 21st Century

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Global Water Challenges of Food and Energy Systems in the 21st Century

By

Lorenzo Rosa

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Abstract

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Water is increasingly recognized as an important factor constraining humankind's ability to meet its burgeoning food and energy needs. Water is a major factor limiting crop production in many regions around the world. Irrigation can greatly enhance crop yields, but the local availability and timing of freshwater resources constrains the ability of humanity to intensify food production. Water plays an important role in the production of energy, including unconventional fossil fuels extraction. Water is also important to meet climate change targets. Carbon capture and storage is broadly recognized as a technology that could play a key role in limiting the net anthropogenic carbon dioxide emissions from industrial and energy systems. However, carbon capture and storage technologies are energy-intensive processes that would require additional power generation and therefore additional water consumption for the cooling process.

While substantial additional water will be required to support future food and energy production, it is not clear whether and where local renewable water availability is sufficient to sustainably meet future water consumption. The extent to which irrigation can be sustainably expanded within presently rain-fed cultivated land without depleting environmental flows remains poorly understood. It also remains unclear where and to what extent new water demanding technologies such as carbon capture and storage and hydraulic fracturing might generate or exacerbate water scarcity.

In this dissertation work, I used a global water balance model to determine at high spatio-temporal resolution local water demand and water availability for human societies. I was able to estimate if there is sufficient local water to sustainably meet future demand for water. I also determined the sustainability of these practices and the extent by which they deplete environmental flows and groundwater stocks.

I find that half of irrigation practices are currently unsustainable and that 15% of global unsustainable irrigation is embedded in international food trade. Despite widespread unsustainability from irrigation, I find that there is still substantial potential to increase food production by sustainably expanding irrigation over 140 Million hectares of croplands globally, potentially feeding 800 million more people. I also find that energy technologies such as hydraulic fracturing and carbon capture and storage will require substantial additional water, exacerbating water scarcity and creating a competition for the scarce local freshwater resources

among energy, industrial, and agriculture industries. I show that certain geographies lack sufficient water resources to meet the additional water demands of carbon capture technologies and hydraulic fracturing.

These findings shed light on the importance of freshwater in future decision making. The results of this dissertation have the potential to inform water, energy, and food security policies at global, regional, national, and local scales and to provide new insights to achieve global sustainability targets.

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INTRODUCTION

Fundamental to the functioning of the Earth's system and human societies, freshwater is a renewable natural resource that is available in finite amounts. Globally, irrigated agriculture accounts for ~85-90% of water consumption, followed by industrial (for example, water for cooling power plants) and domestic water use. Given the dependence of human societies on freshwater availability, water is increasingly recognized as an important factor constraining humankind's ability to meet its burgeoning agricultural, industrial, domestic, and energy needs.

Water plays an important role in the production of energy and is a major factor limiting crop production in many regions around the world. Irrigation can greatly enhance crop yields, but the local availability and timing of freshwater resources constrains the ability of humanity to intensify food production. At the same time, the twin costs of mitigating climate change and competing for water resources are vexing factors in managing climate mitigation technologies. For example, carbon capture and storage is broadly recognized as a technology that could play a key role in limiting the net anthropogenic carbon dioxide emissions from coal-fired power plants. However, CCS technologies are energy-intensive processes that would require additional power generation and therefore additional water consumption for the cooling process.

The emergent competition for water between food and energy systems is increasingly recognized in the study of the "water-energy-food nexus". The nexus between food and water is made even more complex by the globalization of agriculture and rapid growth in food trade, which results in a massive virtual transfer of water among regions and plays an important role in the food and water security of some areas. At the same time there are increasing concerns about global water scarcity. Recent studies have shown that some of the world's major agricultural baskets consistently exhibit unsustainable water consumption that is depleting groundwater stocks and environmental flows. Unsustainable water consumption raises significant threats to local and global water, energy, and food security. Given current societal trends in water use, in many regions of the world it will not be possible to achieve the Sustainable Development Goal Target 6.4, which consists of ensuring a sustainable use of water resources by 2030, while reducing the number of people suffering from water scarcity. Moreover, with longer dry spells and more erratic rainfalls, climate change will further increase the need for irrigation water as a critical input limiting global food production.

While substantial additional water will be required to support future food and energy production, it is not clear whether and where local renewable water availability will be sufficient to sustainably meet future water consumption needs. To sustainably meet future food demand without additional agricultural expansion, agriculture will likely need to expand irrigation into rain-fed croplands. The extent to which irrigation can be sustainably expanded within presently rain-fed cultivated lands without depleting environmental flows remains poorly understood. Although the spatial patterns of changes in future water availability have been widely investigated, it remains unknown how climate change will affect irrigated croplands. Moreover, while virtual water flows associated with the trade of agricultural commodities have been widely documented and the locations of unsustainably irrigated regions have been quantitatively determined, the globalized dimension of unsustainable irrigated food production and the associated unsustainable virtual water trade are poorly studied. It also remains unclear where and to what extent new water demanding energy projects such as carbon capture and storage and hydraulic fracturing might generate or exacerbate water scarcity. My dissertation addresses four main research questions:

1) *Where and to what extent can irrigation substantially increase food production without depleting groundwater stocks or environmental flows?*
2) *What is the fraction of unsustainable irrigation water consumption that is embodied in virtual water transfers? What countries and crops are responsible for unsustainable irrigation practices?*
3) *What are the water requirements of carbon capture and storage? Does the addition of carbon capture and storage impact water consumption enough to locally induce or exacerbate water scarcity?*
4) *Where and to what extent will hydraulic fracturing compete with other water needs and ecosystem functions?*

Table 1. Chapters and publications in the dissertation.

Chapter	Publication
1	Rosa, L., Rulli, M. C., Davis, K. F., Chiarelli, D. D., Passera, C., & D'Odorico, P. (2018). Closing the yield gap while ensuring water sustainability. *Environmental Research Letters*, *13*(10), 104002.
2	Rosa, L., Chiarelli, D. D., Tu, C., Rulli, M. C., & D'Odorico, P. (2019). Global unsustainable virtual water flows in agricultural trade. *Environmental Research Letters*, *14*(11), 114001.
3	Rosa, L., Chiarelli, D. D., Rulli, M. C., Dell'Angelo, J., & D'Odorico, P. (2020). Global agricultural economic water scarcity. *Science Advances*, *6*(18), eaaz6031.
4	Rosa, L., Reimer, J. A., Went, M. S., & D'Odorico, P. (2020). Hydrological limits to carbon capture and storage. *Nature Sustainability*, 1-9.
5	Rosa, L., Sanchez, D., Realmonte, G., Baldocchi, D., D'Odorico, P. The global water footprint of large-scale deployment of carbon capture and storage technologies under stringent climate policy. *Renewable and Sustainable Energy Reviews*. *Under review.*
6	Rosa, L., Rulli, M. C., Davis, K. F., & D'Odorico, P. (2018). The water-energy nexus of hydraulic fracturing: a global hydrologic analysis for shale oil and gas extraction. *Earth's Future*, *6*(5), 745-756.
7	Rosa, L., & D'Odorico, P. (2019). The water-energy-food nexus of unconventional oil and gas extraction in the Vaca Muerta Play, Argentina. *Journal of cleaner production*, *207*, 743-750.

Table 2. Other related publications resulting from this dissertation.

Publication
D'Odorico, P., Davis, K.F., **Rosa, L**., Carr, J.A., Chiarelli, D., Dell'Angelo, J., Gephart, J., MacDonald, G.K., Seekell, D.A., Suweis, S. and Rulli, M.C., 2018. The global food-energy-water nexus. *Reviews of Geophysics*, *56*(3), pp.456-531.
Rosa, L., Davis, K.F., Rulli, M.C. and D'Odorico, P., 2017. Environmental consequences of oil production from oil sands. *Earth's Future*, *5*(2), pp.158-170.
D'Odorico, P., Carr, J., Dalin, C., Dell'Angelo, J., Konar, M., Laio, F., Ridolfi, L., **Rosa, L.**, Suweis, S., Tamea, S. and Tuninetti, M., 2019. Global virtual water trade and the hydrological cycle: patterns, drivers, and socio-environmental impacts. *Environmental Research Letters*, *14*(5), p.053001.
Chiarelli, D.D., **Rosa, L.**, Rulli, M.C. and D'Odorico, P., 2018. The water-land-food nexus of natural rubber production. *Journal of cleaner production*, *172*, pp.1739-1747.

Borsato, E., **Rosa, L**., Marinello, F., Tarolli, P. and D'Odorico, P., 2020. Weak and Strong Sustainability of Irrigation: A framework for irrigation practices under limited water availability. *Front. Sustain. Food Syst. 4: 17*.
Chiarelli, D.D., Passera, C., Rulli, M.C., **Rosa, L**., Ciraolo, G. and D'Odorico, P., Hydrological consequences of natural rubber plantations in Southeast Asia. *Land Degradation & Development*.
Beltran-Peña, A.A., **Rosa, L**. and D'Odorico, P., 2020. Global food self-sufficiency in the 21st century under sustainable intensification of agriculture. *Environmental Research Letters*.
Graves A., **Rosa L.**,Nouhou A.M., Maina F., Adoum D., Avert catastrophe now in Africa's Sahel. *Nature* **575**, 282–286 (2019).

CHAPTER 1

Closing the yield gap while ensuring water sustainability

Reference: Rosa, L., Rulli, M. C., Davis, K. F., Chiarelli, D. D., Passera, C., & D'Odorico, P. (2018). Closing the yield gap while ensuring water sustainability. Environmental Research Letters, 13(10), 104002.

1.1 Abstract

Water is a major factor limiting crop production in many regions around the world. Irrigation can greatly enhance crop yields, but the local availability and timing of freshwater resources constrains the ability of humanity to increase food production. Innovations in irrigation infrastructure have allowed humanity to utilize previously inaccessible water resources, enhancing water withdrawals for agriculture while increasing pressure on environmental flows and other human uses. While substantial additional water will be required to support future food production, it is not clear whether and where freshwater availability is sufficient to sustainably close the yield gap in cultivated lands. The extent to which irrigation can be expanded within presently rainfed cropland without depleting environmental flows remains poorly understood. Here we perform a spatially explicit biophysical assessment of global consumptive water use for crop production under current and maximum attainable yield scenarios assuming current cropping practices. We then compare these present and anticipated water consumptions to local water availability to examine potential changes in water scarcity. We find that global water consumption for irrigation could sustainably increase by 48% (408 km^3 H_2O y^{-1}) – expanding irrigation to 26% of currently rain-fed cultivated lands and producing 37% more calories, enough to feed an additional 2.8 billion people. If current unsustainable blue water consumption and production practices were eliminated, a sustainable irrigation expansion and intensification would still enable a 24% increase in calorie production. Collectively, these results show that the sustainable expansion and intensification of irrigation in selected croplands could contribute substantially to achieving food security and environmental goals in tandem in the coming decades.

1.2 Introduction

Steady increases in crop production have supported marked population growth while substantially reducing incidences of malnourishment globally (Pingali 2012). This Green Revolution was made possible through the proliferation of high-yielding crop varieties, increased pressures on land and water, substantial nutrient inputs, and rising greenhouse gas emissions, making agriculture one of humanity's most profound environmental burdens. A continuation of these practices is expected to be constrained by the limited water resources of the planet (Postel et al 1996, Gleick and Palaniappan 2010) and be insufficient to sustainably ensure future food security in the long term (Wackernager et al 2002, Rockström et al 2009, Hoekstra and Wiedmann 2014, Galli et al 2014, Steffen et al 2015).

Recent work has devoted substantial focus to examining the avenues by which humanity can feed more people and minimize the environmental impacts of agriculture, including reducing food waste, improving resource use efficiencies, and shifting diets (Rost et al 2009, Godfray et al 2010, Tilman et al 2011, Foley et al 2011, Ray and Foley 2013, Cassidy et al 2013). Receiving the bulk of the attention among these promising solutions is the opportunity to enhance crop yields on current croplands, thereby ensuring that agricultural inputs are used efficiently by a crop as well as preventing agricultural expansion into biodiversity-rich ecosystems (Phalan et al 2011, Pretty et al 2011, Tscharntke et al 2012, Mueller et al 2012, Garnett et al 2013, Davis et al 2017). Known as 'agricultural intensification' such an approach entails closing (or at least narrowing) crop yield gaps – the difference between potential yield (or water-limited yield potential) and the actual yield that a farmer currently achieves (Cassman 1999, Lobell et al 2009, van Ittersum et al 2013, Gobbett et al 2017). A common benchmark used in studies estimating maximized crop production (e.g., Foley et al 2011, Mueller et al 2012), potential yield is defined as the yield of a crop cultivar when grown in an environment to which it is adapted, with non-limiting water and nutrient supplies, and with pests, weeds, and diseases effectively controlled (Evans 1993). While water and other inputs will likely be used more efficiently under higher yields (i.e., more crop per drop), additional irrigation will be needed in many places in order to close the yield gap and to maximize food production (Gerten et al 2011, Tilman et al 2011, Pfister et al 2011, Davis et al 2017a, Okada et al 2018).

Global crop production depends on water received both as precipitation (or "green water") and irrigation (or "blue water") from surface water bodies and aquifers (Rockström et al 2009). Through irrigation, it is possible to reduce crop exposure to water stress, and therefore enhance productivity. Particularly in regions frequently affected by crop water stress, irrigation represents a major pathway to the intensification of crop production and yield gap closure (Mueller et al., 2012). In some regions, the development of irrigation is limited by the availability of blue water resources. In other places, water withdrawals that exceed renewable water availability can affect environmental flows that support aquatic habitats (Poff et al 1997, Dudgeon et al 2006) and deplete groundwater resources (Konikow and Kendy 2005, Wada et al 2012). Recent work has assessed water scarcity under current levels of crop production (Mekonnen and Hoekstra 2016, Brauman et al 2016, Liu et al 2017), showing that many important agricultural regions maintain water consumptions that consistently exceed local freshwater availability. Yet it remains unclear where and to what extent local water resources will be sufficient to sustainably close the yield gap (i.e., achieve potential yields globally). Here we perform a global spatially distributed biophysical analysis of irrigation water demand under current and maximized crop production within the world's existing croplands. We then compare these demands to local renewable freshwater availability – accounting for environmental flows – to identify regions of the world where irrigation can be expanded into currently rainfed croplands without threatening freshwater ecosystems. We conclude our analysis by estimating the additional calories and protein that can potentially be produced while ensuring water sustainability. This study can ultimately help prioritize agricultural initiatives that can achieve food security and environmental goals together.

1.3 Methods

We evaluated the availability of freshwater resources for irrigation and the extent to which their consumption may affect environmental flows. We considered both current irrigation conditions and a possible scenario of yield gap closure on currently cultivated lands. Irrigation water use under yield gap closure accounts for both the intensification of irrigation and its expansion into rainfed croplands where yields are currently limited by precipitation availability in many places. This analysis allowed us to estimate where and to what extent water consumption for agriculture is sustainably accommodating local environmental needs and where crop production is or will be constrained by locally available renewable surface and groundwater resources. Our analysis examined only currently cultivated lands and did not consider cropland expansion, crop switching or increased cropping frequencies enabled by additional irrigation. Our hydrological analysis considers all renewable (blue) water resources (including both surface water and groundwater).

We used a process-based crop water model to estimate irrigation water consumption under current crop production and under yield gap closure for 16 major crops. This model was coupled with a daily soil water balance and integrated over each crop's growing season to determine spatially explicit, crop-specific irrigation water requirements (mm yr^{-1}) (Davis et al 2017). These blue crop water requirements were then multiplied by their respective irrigated areas – and combined with estimates of local blue water consumption for other human activities (BWC) (i.e., municipal and industrial uses) (Hoekstra and Mekonnen 2012) – to determine total blue water consumption in each grid cell under current levels of crop production and under maximized crop production. Following Rosa et al (2018) we calculated the renewable blue water availability (BWA) using estimates of renewable blue water flow (Fekete et al 2002) and a flow accumulation algorithm. Finally by combining estimates of the availability and consumption of renewable blue water resources (including both surface water and groundwater), we identified current and future areas of sustainable irrigation water consumption as those places where BWC < BWA.

1.3.1 Rainfed and irrigation water consumption assessment

We follow the methods in Davis et al (2017) and Davis et al (2018) to calculate the crop water requirement at yield gap closure. A crop's water requirement is the amount of water needed by a crop to satisfy its evapotranspirative demand and to avoid a water-stressed condition. This demand can be satisfied by precipitation (i.e., green water) and supplemented through irrigation (i.e., blue water) if precipitation is insufficient. We considered 16 major crops (barley, cassava, groundnuts, maize, millet, oil palm, potatoes, rapeseed, rice, rye, sorghum, soybeans, sugar beet, sugar cane, sunflower, and wheat) which account for 73% of the planet's cultivated areas and 70% of global crop production (Food and Agricultural Organization of the United Nations 2017).

1.3.2 Current and yield gap closure water consumption

The current (year 2000) extent of crop-specific irrigated areas and planting and harvesting dates came from Portmann et al (2010). The current blue water consumption for a crop in a given pixel was then calculated as the product of the blue water requirement of that crop and its respective irrigated harvested area (Portmann et al 2010). For each of the 16 major crops, we also calculated the additional volumes of blue water required to close the crop yield gap (i.e., to reach the maximum attainable yield) (Mueller et al 2012). In this yield gap closure scenario, given the uncertainty in determining where and to what extent cropping frequency can be increased through irrigation expansion, we assumed that current cropping practices will be implemented. This analysis was carried out for all cultivated lands around the world in which irrigation could substantially improve yields. In many humid areas where most of the crop water requirements can be met by precipitation, investments in irrigation infrastructures would not be justified by the modest increase in crop production induced by irrigation. Therefore, in these places we assume that farmers will likely continue to focus their efforts on rain-fed agriculture. With this in mind, we assumed that a given crop and pixel will be irrigated under yield gap closure if the ratio between the blue and the total crop water requirements (units: mm yr^{-1}) was greater than a critical value of 0.10 [i.e., Blue Water / (Blue Water + Green Water) > 0.10] (Dell'Angelo et al

2018) (Figure S1). This assumption is based on the rationale that in the wettest environments the development of irrigation infrastructure will not be economically justifiable given the marginal increases in yield it would likely bring. We also used thresholds of 0.00 and 0.20 to examine the sensitivity of our results to this threshold assumption (Table 1).

For each pixel (5 arcminute), current blue water consumption for irrigation was then summed with estimates of annual municipal and industrial freshwater consumption (Hoekstra and Mekonnen 2012) to determine the current total blue water consumption of humanity. This analysis was repeated for blue water consumption under yield gap closure – where we assumed constant consumption from municipal and industrial uses – to calculate the total blue water consumption of humanity under yield gap closure. The blue water consumption (BWC) of humanity at a 5 × 5 arcminute resolution was then aggregated to a 30 × 30 arcminute resolution, the resolution of the global renewable blue water availability analysis (see following section of methods).

Table 1. Sensitivity analysis of water consumption and harvested areas under yield potential scenario. We considered three ratios (0, 0.1, and 0.2) between the blue and the total crop water requirements [i.e., BW / (BW + GW)].

	WATER	**LAND**
Ratio BW/(BW+GW)	**Irrigation (Rainfed) (km^3 per year)**	**Irrigated (Rainfed) ($\times 10^6$ km^2)**
0	1722 (6151)	9.66 (3.4)
0.1	1607 (6151)	7.35 (5.71)
0.2	1460 (6151)	5.29 (7.77)

1.3.3 Renewable blue water availability

The global distribution of annual renewable blue water availability (BWA) (at 30 arcminute resolution) was calculated following the methods by Mekonnen and Hoekstra (2016), whereby the value of BWA in a grid cell was expressed as the sum of the local blue water availability in that cell (BWA_{loc}) and the net blue water flow from the upstream grid cells defined as the local renewable water availability in the upstream cells (BWA_{up}) minus the blue water consumption BWC of human activities (i.e., agriculture, municipal, and industrial) in the upstream cells (BWC_{up}). The net renewable blue water flows (combined surface and subsurface) were calculated using the upstream-downstream routing "flow accumulation" function in ArcGIS®, where the subscript *i* denotes the cells upstream from the cell *j* under consideration:

$$\mathrm{BWA}_{\mathrm{j}} = \mathrm{BWA}_{\mathrm{loc,j}} + \sum_{\mathrm{i=1}}^{\mathrm{n}} \left(\mathrm{BWA}_{\mathrm{up,i}} - \mathrm{BWC}_{\mathrm{up,i}}\right) \tag{1}$$

Local renewable blue water availability (surface + groundwater) was calculated as the local blue water flows generated in that grid cell minus the environmental flow requirement. Local blue water flows are calculated in every grid cell as the difference between precipitation and evapotranspiration − using estimates by Fekete et al (2002) − and therefore they account for surface and subsurface runoff generated in that cell as well as for aquifer recharge. We assumed that a fraction (y) of blue water flows is allocated to maintain environmental flows and that the remaining fraction (1-y) is considered blue water locally available for human needs, BWA_{loc} (Pastor et al 2014, Steffen et al 2015). Environmental flow is defined as the minimum runoff that is required to sustain ecosystem functions. For irrigation to be sustainable, these minimum flow requirements need to be met even during dry season and low flow conditions (Pastor et al 2014, Richter et al 2012). Three flow regimes were considered: low, intermediate, and high corresponding to less than 25th percentile, between 25th and 75th percentile, and greater than 75th percentile of annual runoff, respectively (Rosa et al 2018). Following Steffen et al. (2015) in each pixel the estimated blue water flow (Fekete et al 2002) was multiplied by the environmental flow fraction, y, associated with the corresponding flow regime (Pastor et al 2014) to calculate the environmental flows. Environmental flows were then subtracted from the local blue water flows to calculate the local blue water availability (BWA_{loc}). Thus, BWA_{loc} accounts only for renewable blue water resources that can be sustainably used for human activities and excludes both environmental flows and the (unsustainable) depletion of groundwater stocks.

To calculate the upstream to downstream water availability we used the flow direction raster (at 30 arcminute resolution) from the World Water Development Report II (Vörösmarty et al 2000 a-b). Runoff estimates were obtained from the Composite Runoff V1.0 database (Fekete et al 2002). Finally, we defined unsustainable irrigation as occurring when BWC is equal to or exceeds BWA, a condition that would imply the depletion of either environmental flows or groundwater stocks (or both).

1.3.4 Calorie and protein production from cultivated lands

For each of the 16 crops, calorie and protein production under current (Monfreda et al 2008) and yield gap closure (Mueller et al 2012) scenarios were assessed as the product of the crop yield value (tonne ha^{-1}), the crop harvested area (ha) (Monfreda et al 2008), and the calorie or protein content (kcal tonne^{-1}; tonne protein tonne^{-1}). Caloric content for each crop was taken from D'Odorico et al (2014), and crop-specific protein content was assessed as the ratio of per capita protein supply (g protein cap^{-1} day^{-1}) to per capita food supply (g cap^{-1} day^{-1}) from FAOstat (Food and Agricultural Organization of the United Nations 2017). The number of people that can be potentially fed was assessed according to a previous global average estimate of 3343 vegetal kcal cap^{-1} day^{-1} (Davis et al 2017).

1.3.5 Uncertainties, limitations, and assumptions

Our results are based on the assumption that in the yield gap closure scenario a given crop and pixel will be irrigated if the ratio between the blue and the total crop water requirements is greater than a critical value of 0.10 (Dell'Angelo et al 2018). This assumption is based on the fact that, if blue water is needed to meet less than 10% of the crop water requirements, farmers will likely decide that the cost of improving irrigation systems will exceed the cost of reduced agricultural production from crops that are slightly water stressed. This ratio certainly depends on a host of factors including crop type, cost of irrigation infrastructure, access to water, farmer access to capital or credit, additional revenue from higher crop yields, market incentives, government policies, food needs, and interannual rainfall variability. The 10% threshold is here used as a conservative estimate, based on the fact that for major staple crops irrigation is found to be typically developed in areas in which green water consumption contributes to at most 80% of the total water consumption (i.e., irrigation infrastructure is found in areas where more than 10% of crop water requirements come from irrigation because it doesn't rain enough) (Rost et al 2008, Tuninetti et al 2015). Moreover, we found that the ratio between the blue and the total crop water requirements is less than 0.10 for only 6% (1.51×10^5 km^2) of currently irrigated lands. We also performed a sensitivity analysis to analyze how our results would vary with different BW/(BW+GW) threshold values and we found that there is only a modest ±10% change in blue water consumption in the yield gap closure scenario when this ratio is reduced to zero or increased to 0.20 (Table 1). Thus our estimates of irrigation water use in the yield gap closure yield scenario are robust with respect to the assumption that irrigation is not performed in areas where rainfed agriculture undergoes a water deficit smaller than 10%.

Our assessment is based on temporal averages and does not account for inter-annual and seasonal variability in river discharge and crop water requirements. For example, some irrigated areas might only experience unsustainable irrigation water demand in dry years or during dry periods of the year (Brauman et al 2016). Our model considers only short-range transport (~50 km) of freshwater, without accounting for interbasin freshwater transfer projects like the South-to-North Water Diversion Project in China (Zhao et al 2017), the California State Water Project, and the Great Man Made River Project in Libya (Sternberg 2016). It also does not consider large water supply networks within the same basin that distribute water across hundreds of kilometers (at distances greater than the 50 km resolution of our model) such as the Nile and Indus basin channel networks. Moreover, because information on actual irrigation water use is limited, our model may produce instances where blue water demand in current irrigated lands cannot be fully met as a result of inadequate infrastructure or insufficient irrigation pumping capacity. The goal of this study is to provide biophysical estimates of crop water requirements that can be used to understand a farmer's average water needs. Our model also does not account for future potential changes in cropping frequency and crop types that could be enabled by additional irrigation infrastructure (Ray and Foley 2013, Rufin et al 2018).

Crop choice and cropping frequency are decisions primarily driven by economics, and farmers will likely decide to produce the most profitable crop under a change to irrigated conditions. Indeed, it remains difficult to estimate where and to what extent cropping frequency can be increased. Because we assume current cropping patterns and frequencies, our estimates of water consumption for certain areas may be conservative, as the expansion of irrigation may allow for an additional cropping season. On the other hand, our assumption of 100% yield potential in non-water stressed conditions might overestimate water consumption under yield gap closure. Indeed, producers do not necessarily attempt to avoid water stress, but maximize their profit. This is usually not achieved by fully removing water and nutrient limitations to crop growth (i.e., maximum yield) because it may require an inefficient application of inputs (including water) that are not compensated by yield increases (Cassman 1999). Moreover, our biophysical model does not consider potential additional water demand from losses from irrigation infrastructure (e.g., losses from irrigation canals) and water use to control soil salinity. Future water consumption in agriculture will also be affected by climate change, which will alter both water availability and crop evapotranspiration (e.g., Katul et al 2012, Elliott et al 2014).

Lastly, the dataset on crop production under yield gap closure (Mueller et al 2012) relied on a statistical climate-binning approach to estimate the extent to which crop yields could be increased under improved management practices and inputs. Following the approach of Monfreda et al (2008) – the data we used to estimate current (circa 2000) crop production – Mueller et al (2012) developed gridded crop-specific maps of current crop yields and controlled for rainfall and temperature in order to develop yield distributions and estimate attainable yields. Because this yield gap closure dataset – which we also used here – relies on year 2000 yield data to estimate potential yields, it is likely that our estimates of potential crop production are conservative to a certain extent, as yields have been (slowly) increasing since the turn of the century because of new crop cultivars and improved management (Ray et al 2013), though, climate change could have a negative impact on potential yield growth in many regions of the world (Urban et al 2017).

1.4 Results

1.4.1 The status of current irrigation

We estimate that the current irrigation water consumption for major crop production is 847 $km^3 y^{-1}$ (Table 2). This agrees well with previous estimates by Siebert and Döll (2010) (1180 $km^3 y^{-1}$) and Hoekstra and Mekonnen (2012) (899 $km^3 y^{-1}$). Our assessment shows that 40% of this volume of irrigation water is currently consumed at the expense of environmental flows (i.e., the minimum flows needed to sustain ecosystem functions in streams and rivers) (Jägermeyr et al 2017) or groundwater stocks. Not surprisingly, some of the world's major agricultural baskets such as the U.S. High Plains and California's Central Valley, the North China Plain, the Murray-Darling Basin of Australia, and the Indo-Gangetic Basin consistently exhibit unsustainable water use, where blue water consumption exceeds its local availability (see Figure 1a). In these regions, irrigation is depleting groundwater stocks (Wada et al 2010, Gleeson et al 2012, Konikow and Kendy 2005, Scanlon et al 2012, Famiglietti 2014, Rodell et al 2018) and diminishing environmental flows (Mekonnen and Hoekstra 2016, Brauman et al 2016, Jägermeyr et al 2017).

Table 2. Water consumption, irrigated extent, and calorie production under current and yield potential scenarios. Sustainable irrigation is practiced in areas where blue water consumption (BWC) does not exceed renewable blue water availability (BWA), which accounts also for environmental flows. Irrigation is "unsustainable" when BWC≥BWA (i.e., it sacrifices environmental flows, requires non-renewable groundwater resources, or inter basin water transport). Values in parentheses correspond to rainfed croplands.

	WATER	LAND	CALORIES
	Irrigation (Rainfed) (km^3 per year)	**Irrigated(Rainfed) ($\times 10^6 km^2$)**	**Irrigated (Rainfed) ($\times 10^{15}$ kcal per year)**
Current			
Sustainable	511	1.69	1.69
Unsustainable	336	1.13	1.19
Total current	**847 (6151)**	**2.82 (10.24)**	**2.88 (6.35)**
Additional at yield gap closure			
Sustainable (expansion of irrigation)	336	2.67	2.33
Sustainable (intensification of irrigation)	72	-0.14	1.05
Unsustainable (expansion of irrigation)	261	1.86	0.91
Unsustainable (intensification of irrigation)	91	0.14	0.71
Total yield gap closure	**1607 (6151)**	**7.35 (5.71)**	**7.88 (6.35)**

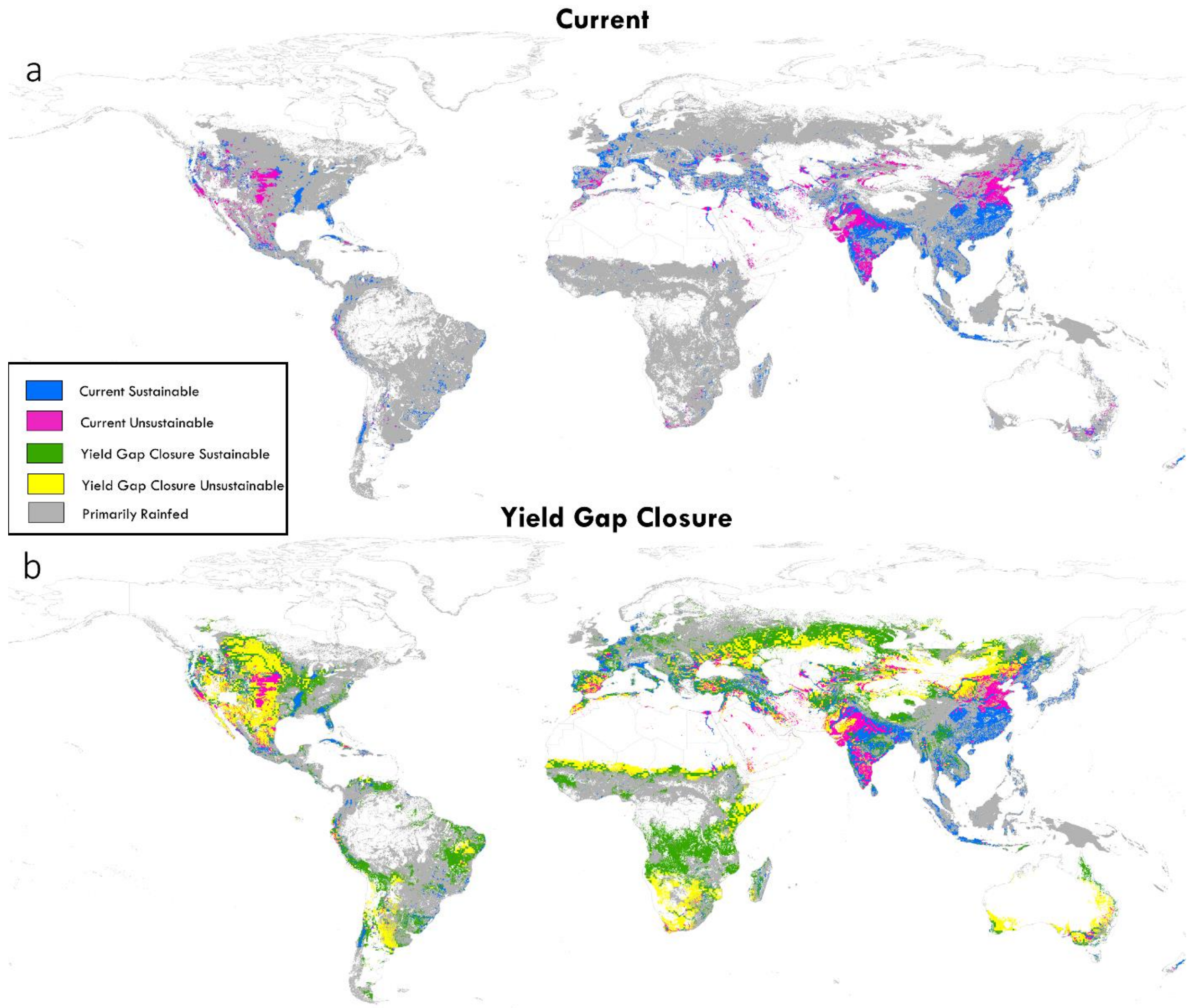

Figure 1. The extent of sustainable irrigation over cultivated lands. We define sustainable irrigation when water consumption for human activities remains below the limit imposed by environmental flow requirements (BWC<BWA). Blue (fuchsia) areas represent sustainable (unsustainable) irrigation water consumption over current (year 2000) irrigated lands. In the yield gap closure scenario green (yellow) areas show the potential for the sustainable (unsustainable) expansion and intensification of irrigation water consumption over currently underperforming cultivated lands (rainfed or irrigated). For current production, unsustainable irrigation consumption occurs on 40% of irrigated lands (1.13×10^6 km^2). In the yield gap closure scenario we estimate that irrigation needs to be expanded by 4.53×10^6 km^2 and that only half (56%) of this additional area has the potential for sustainable crop yield gap closure (See Table 2).

1.4.2 Sustainable yield gap closure through irrigation

To enhance crop yields, irrigation will need to expand into primarily rainfed agricultural areas where productivity is constrained by access to water resources. Investments in irrigation infrastructure, however, can only be justified if they induce a substantial increase in production, a condition that can be expressed by ratios between irrigation water and total water requirements greater than a critical value, typically taken equal to 0.10 (Dell'Angelo et al 2018). We estimate that this condition is met in 44% (i.e., 4.53×10^6 km^2) of the cultivated lands currently used for rainfed agriculture (Figure 1). To expand and intensify irrigation over these lands, global blue water consumption for agriculture would need to increase by 760 km^3 y^{-1}. Doing so would enhance global food calorie production by 54% (5.00×10^{15} kcal y^{-1}) – consistent with earlier estimates (Foley et al 2011) – and increase vegetal protein production by 51% (121×10^6 tonnes y^{-1}) (Table 2).

In many places, however, these irrigation water requirements and the associated increases in food production can be sustainably met without depleting environmental flows and groundwater stocks. For only those rainfed areas with adequate freshwater resources, sustainable irrigation expansion would require an additional 408 km^3 y^{-1}. Though more limited in extent, this sustainable yield gap closure would still realize large increases in calorie (+37%, or 3.38×10^{15} kcal y^{-1}) and protein (+34%, or 82×10^6 tonnes protein y^{-1}) production – enough to feed an additional 2.77 billion people. Overall, this means that 54% of the water needed to expand and intensify irrigation and 68% of the associated increase in calorie and protein production could be attained sustainably within the limits of renewable blue water availability (Table 2). In this scenario of sustainable yield gap closure, half of the calorie production would rely on irrigation water. In addition, we find that opportunities to close the yield gap by sustainably expanding irrigation exist only in 26% of currently rainfed cultivated lands (green areas in Figure 1b). Maximizing yields by sustainably expanding irrigation into these primarily rainfed lands (as opposed to intensifying irrigation in currently irrigated croplands) would require 82% of the sustainable additional blue water consumption, while contributing to 69% of the potential increase in calorie and protein production (Table 2).

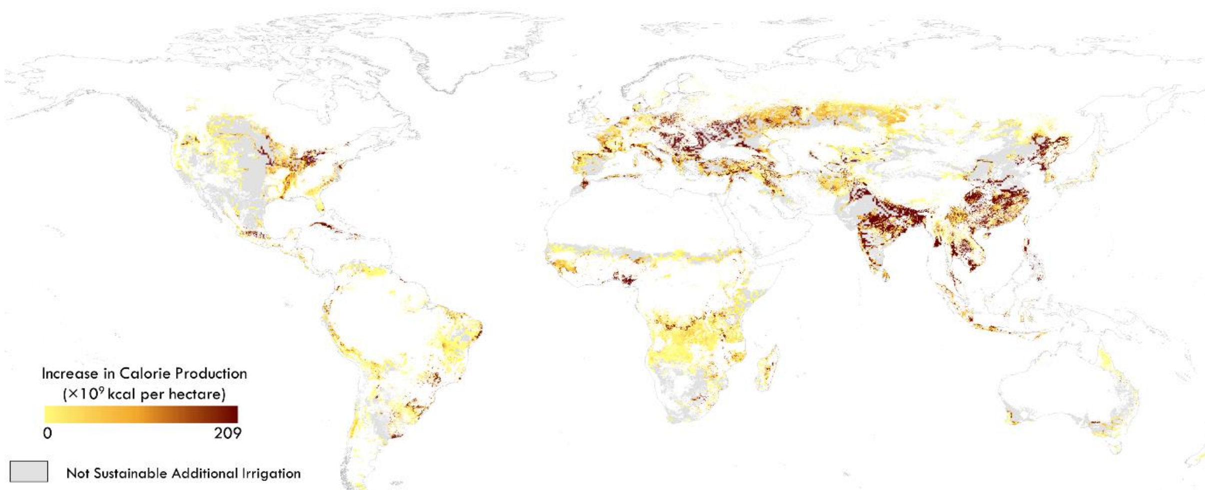

Figure 2. Sustainable increases in calorie production. The map represents the additional calories that can be produced through sustainable irrigation intensification (blue areas where additional irrigation is sustainable in Figure 1) and expansion (green areas in Figure 1). These additional calories would be enough to feed 2.77 billion more people.

If current unsustainable blue water consumption (336 km^3 y^{-1}) and crop production (1.19×10^{15} kcal y^{-1}) practices were eliminated, sustainable irrigation expansion and intensification (336 km^3 y^{-1} and 72 km^3 y^{-1}, respectively) would still enable a substantial net increase in sustainable calorie (+24%, or 2.19×10^{15} kcal y^{-1}) and protein (+22%, or 53×10^{6} tonnes protein y^{-1}) production (Table 2). In this scenario, total sustainable blue water consumption for irrigation would reach 919 km^3 y^{-1} (551 km^3 y^{-1} from current irrigation with additional 408 km^3 y^{-1} from irrigation intensification and expansion). Moreover, an intensification of irrigation over currently irrigated lands would shift 0.14×10^{6} km^2 of irrigated croplands from sustainable to unsustainable water consumption practices.

Under sustainable yield gap closure, we found at least a doubling of calorie production for 50 countries, 29 of which are in Africa (e.g., Nigeria, Ethiopia, Eritrea, Democratic Republic of Congo, Tanzania, and Mozambique) (Figure 2). We also found at least a doubling of protein production for 54 countries – most of which occur in the developing world (examples include 30 African countries, Mongolia, Cambodia, and Afghanistan). Collectively, China, the United States, India, Russia, Brazil, and Nigeria can contribute to about 46% of the global increase in food calorie production associated with the sustainable intensification and expansion of irrigation (Figure 3). China, the world's top food calorie producer, has the greatest potential to sustainably increase crop production by intensifying and expanding irrigation, thereby feeding an additional 382 million people. India and Russia also have great opportunities to sustainably increase calorie production to feed 261 and 222 million people, respectively (Figure 3). Africa, currently only sparsely irrigated (Burney et al 2013), currently produces enough calories to feed 400 million people – making it the continent with the largest gap between crop production and demand (van Ittersum et al 2016). An increase in yields through investments in irrigation expansion could sustainably feed an additional 450 million people and substantially reduce the continent's dependence on food imports.

Sustainable irrigation could increase national food self-sufficiency in countries that today meet large fractions of their domestic food demand through international trade (D'Odorico et al 2014). For example, net food importing countries (such as Mexico, Iran, Germany and Italy), would experience a greater than 15% increase in calorie production. This in turn could reduce their exposure to economic and environmental shocks to the global food system that occur beyond their borders (Suweis et al 2015, Oki et al 2017).

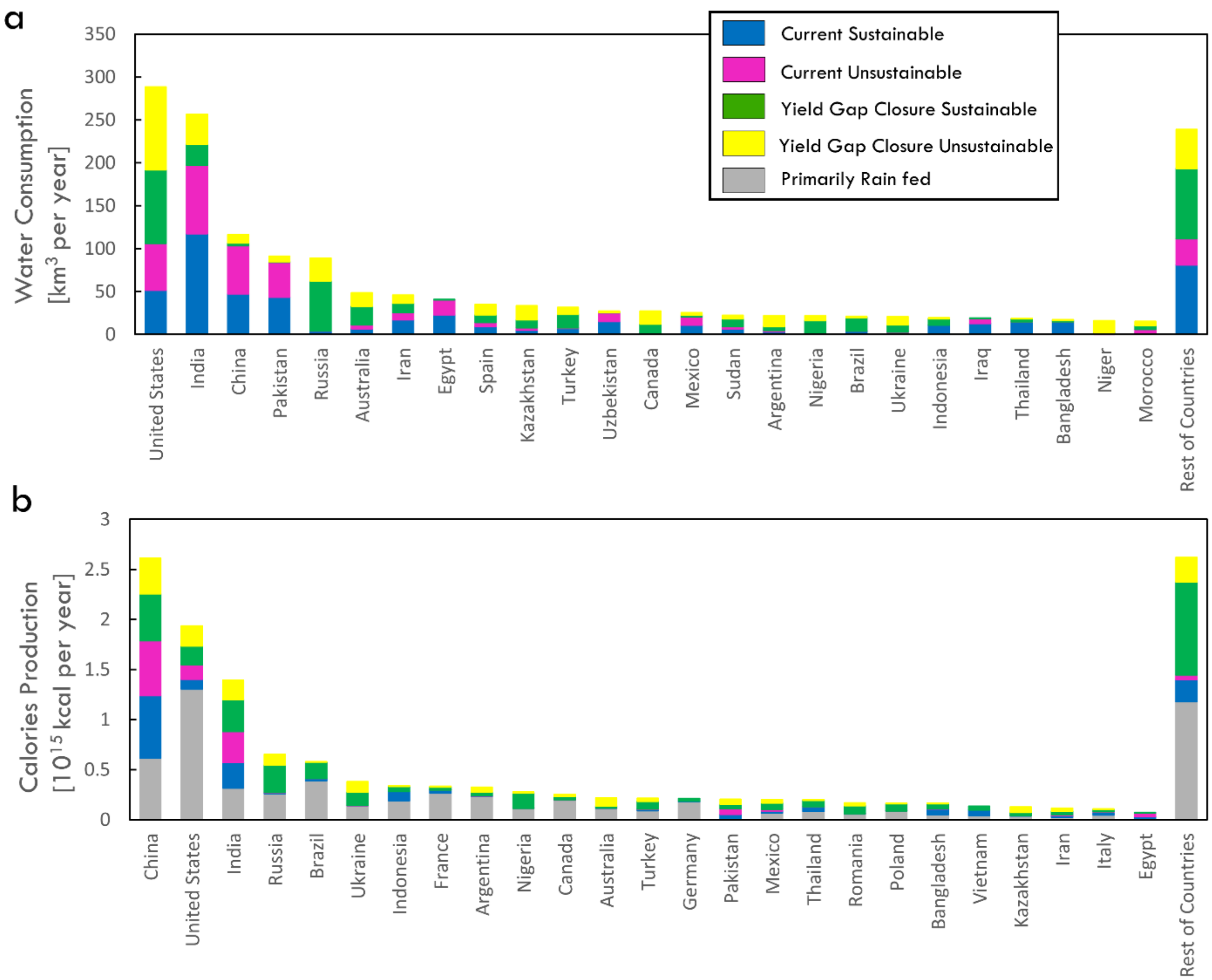

Figure 3. Potential increases in freshwater demand and calorie production. Panels show the extent to which (a) irrigation water consumption, and (b) calorie production can sustainably occur under current and maximized levels of crop production. Note that countries are listed in descending order based on potential irrigation water consumption, and calorie production at yield gap closure, respectively.

1.5 Discussion

Our study identifies where and to what extent crop production can be sustainably intensified through irrigation expansion in currently cultivated lands without inducing major losses of aquatic habitat, groundwater depletion, or changes in other (nonagricultural) water uses. This increase in food production would also aid in minimizing the expansion of agriculture into land that is presently not cultivated, thereby avoiding human appropriation of water resources (both green and blue) that are currently used by natural systems.

In the case of irrigation intensification and expansion, the current dominance of rainfed calorie production would be superseded by irrigated production (Table 2). However, the green water volumes would remain almost four times greater than those of irrigation water consumption, showing not only the huge water savings potential from making green water more productive in agriculture (Rockström et al 2009, Molden et al 2010, Davis et al 2017) but also the importance of green water in global food production and international food trade (Aldaya et al 2009).

An increase in irrigation would also draw humanity closer to the planetary boundary for freshwater (Rockström et al 2009), estimated to be on average 2,800 $km^3 y^{-1}$ of freshwater (with a range of uncertainty estimated to be between 1,110 $km^3 y^{-1}$ and 4,500 $km^3 y^{-1}$) (Table 3) (Gerten et al 2013). With current (1995 to 2000 period) blue water consumption estimated to be around 1,800-2,270 $km^3 y^{-1}$ (Shiklomanov et al 2003, Hanasaki et al 2010, Wada et al 2011), a sustainable expansion and intensification of irrigation would require an additional 408 $km^3 y^{-1}$ of freshwater, and may in some places increase competition for freshwater resources with other human activities, such as the industrial and energy sectors (Rosa et al 2017, Chiarelli et al 2018, Rosa et al 2018, D'Odorico et al 2018). Therefore, there is an urgent need to adopt water conservation strategies (Rost et al 2009, Jägermeyr et al 2016, Davis et al 2017, Davis et al 2018) and to reassess where irrigated agriculture currently occurs (Figure 1a) – especially in water-stressed areas – in tandem with sustainable irrigation expansion in order to increase the water productivity of food systems.

Our biophysical modelling results show that if current unsustainable blue water consumption and production practices were eliminated, sustainable irrigation expansion and intensification would still enable a net increase in food production while keeping a safe distance from the planetary boundary for freshwater, restoring environmental flows, and reducing reliance on irrigation from non-renewable freshwater resources. Our results show that targeted policy and farming decisions could achieve important reductions in unsustainable irrigation demand in many regions of the world, while sustainably increasing calorie production of 24% globally.

However, additional irrigation infrastructure availability needs to be accompanied by other changes in management practices in order to achieve maximum yields. Indeed, Mueller et al (2012) showed that in many regions of the world achieving yield gap closure requires an improvement in nutrient supply through fertilizer application. Other practices that might be changed in response to the expansion of irrigation infrastructure are a switch to crops with higher productivity, the introduction of an additional cropping season, or the storage of water from the rainy to allow for its use during the dry seasons.

Table 3. Planetary boundary of freshwater, current and projected blue water consumptions. Planetary boundary of freshwater is defined as the consumptive water left to be used for human activities (Rockström et al 2009).

	Blue Water (km^3 per year)	**Source**
Planetary Boundary of Freshwater	1,100-4,500	Gerten et al 2013
Current Total Blue Water Consumption	1,800-2,270	Shiklomanov et al 2003 Hanasaki et al 2010 Wada et al 2011

Blue Water Consumption for Irrigation		
Current	847	This Study
Additional for Sustainable Yield Gap Closure	408	

While the sustainable irrigation yield gap closure scenario investigated in this study accounts for the need to protect environmental flows that are crucial to the health of freshwater ecosystems, it does not evaluate other environmental and economic impacts associated with the irrigation of cultivated lands (e.g., changes in microclimate, habitat, and land use (Sacks et al 2009), energy costs and associated greenhouse emissions (Burney et al 2010), and infrastructure development (Blanc and Strobl 2013)) which require further investigation. Future research is also required to analyze in which areas additional irrigation water can exacerbate water stress and intensify a competition for water between food and energy production (Scanlon et al 2017, Rosa et al 2018). Our results are based on a biophysical model and on assumptions that are always necessary in any global modelling study. There are many factors that our model cannot predict (change in cropping patterns and harvest frequencies, rates of yield increase, rates of implementation of expanded irrigation, the influence of climate change, and changes in management practices such as improved fertilizer application) that depend on economic, institutional, and other non-biophysical factors that will need to be examined in much greater depth in future studies.

Ultimately, while the biophysical capacity of irrigation expansion to increase food production is an essential consideration, this is only one of the factors that influence governments' decisions to invest in agricultural infrastructure. Our results show that there is a great potential for the sustainable expansion of irrigation infrastructure in China, India, and Iran among many other places (Figure 1). This study does not however account for socioeconomic factors that will determine whether irrigation expansion will occur and to what extent it will change cropping practices (e.g., the use of multiple cropping seasons, their length, and type of crops). For example, in India irrigation water use and crop choice is largely driven by state-level economic incentives (Davis et al 2018) and states with very similar climates and water availability may have very different irrigation water demands. Targeted, location-specific analyses are therefore required to fully understand the potential for sustainable irrigation expansion to meet future food demand.

1.6 Conclusions

This study investigated the extent to which irrigation can be expanded within presently rainfed cultivated lands without depleting environmental flows. This question is central to the debate on water use in agriculture, food security, and sustainability policies. Our analysis shows where and whether available freshwater resources can accommodate a sustainable increase in irrigation for crop production. A sustainable expansion of irrigation into cultivated areas that are presently rainfed would allow for major increases (+37%) in food production (Table 2). Our results also confirm previous literature findings about the urgent need to adopt water management strategies to restore environmental flows and reduce the reliance on irrigation from non-renewable freshwater resources. Our results show that adequate and informed investments in irrigation infrastructure can help to feed billions more people, avoid agricultural expansion into natural habitats, and safeguard local boundaries of freshwater allocation for human and natural systems. In addition to investments, comprehensive policies that support the construction and maintenance of irrigation infrastructure and that implement monitoring systems for responsible and transparent water use will be essential. By examining food demand and resource availability together, our approach establishes a framework to assess the water sustainability of future crop production decisions in the coming decades.

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CHAPTER 2

Global unsustainable virtual water flows in agricultural trade

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2.1 Abstract

Recent studies have highlighted the reliance of global food production on unsustainable irrigation practices, which deplete freshwater stocks and environmental flows, and consequently impair aquatic ecosystems. Unsustainable irrigation is driven by domestic and international demand for agricultural products. Research on the environmental consequences of trade has often concentrated on the global displacement of pollution and land use, while the effect of trade on water sustainability and the drying of over-depleted watercourses has seldom been recognized and quantified. Here we evaluate unsustainable irrigation water consumption (UWC) associated with global crop production and determine the share of UWC embedded in international trade. We find that, while about 52% of global irrigation is unsustainable, 15% of it is virtually exported, with an average 18% increase between year 2000 and 2015. About 60% of global virtual transfers of UWC are driven by exports of cotton, sugar cane, fruits, and vegetables. One third of UWC in Mexico, Spain, Turkmenistan, South Africa, Morocco, and Australia is associated with demand from the export markets. The globalization of water through trade contributes to running rivers dry, an environmental externality commonly overlooked by trade policies. By identifying the producing and consuming countries that are responsible for unsustainable irrigation embedded in virtual water trade, this study highlights trade links in which policies are needed to achieve sustainable water and food security goals in the coming decades.

2.2 Introduction

In the last decade, many studies have shown that some of the world's major agricultural baskets rely on unsustainable water use for irrigation (Konikow, 2011; Gleick and Paliniappan, 2010; Gleeson et al., 2012; Scanlon et al., 2012; Kummu et al., 2016; Mekonnen and Hoekstra, 2016). Irrigation practices are classified as unsustainable when their water consumption exceeds local renewable water availability. In these conditions, irrigation uses water that should be allocated to environmental flows and therefore contributes to environmental degradation and groundwater depletion (Rosa et al., 2018a). About 40% of global irrigation water use is at the expenses of environmental flow requirements (Jägermeyr et al., 2017) with detrimental effects on aquatic habitats, riparian biodiversity, and ecosystem services (Richter et al., 2012; Dudgeon et al., 2006; Palmer and Ruhi, 2019). Moreover, about 20% of irrigation water use worldwide is from non-renewable groundwater abstractions (Wada et al., 2012). Indeed, irrigation water withdrawals may also deplete freshwater stocks (both surface water bodies and aquifers) when their abstraction rates exceed those of natural recharge (Wada et al., 2010; Famiglietti, 2014; Richey et al., 2015; AghaKouchak et al., 2015; Rodell et al., 2018; Wang et al., 2018).

The reliance of food production on unsustainable irrigation threatens local and global water and food security (Aeschbach-Hertig and Gleeson, 2012; Turner et al., 2019). The problem is worsened when the nexus between irrigation-dependent agricultural production and food consumption occurs through distant interconnections resulting from the globalization of food and water resources through trade or international investments (Hoekstra and Chapagain, 2008; D'Odorico et al., 2018). About 20-24% of water resources embedded in food production – or 'virtual water' – are internationally traded (D'Odorico et al., 2014). While the virtual water trade associated with agricultural commodities (Konar et al., 2011; Hoekstra and Mekonnen, 2012), and the regions of unsustainable irrigation have been extensively investigated and mapped (Rosa et al., 2018a), the globalized dimension of unsustainable irrigation, the loss of environmental flows, and the associated unsustainable virtual water trade are poorly studied (D'Odorico et al., 2019).

Research on virtual water has often focused on quantitative analyses of water flows, with no consideration of the environmental impacts of virtual water transfers and of how trade affects the sustainability of irrigation practices (Gawel and Bernsen, 2013). Virtual water transfers have been related to groundwater depletion in the United States (Marston et al., 2015) and globally (Dalin et al., 2017). As important as these studies are, the extent to which virtual water trade contributes to the loss of environmental flows, remains poorly understood. It is not clear to what extent unsustainable surface water consumption for irrigation contributes to the desiccation of rivers and the loss of environmental flows to sustain agricultural production for the export market. Because surface water accounts for about 60% of total consumptive irrigation water use (Siebert et al., 2010), there is a direct association between agricultural production and patterns of streamflow depletion (Richter, 2014) that was not considered in previous analyses of unsustainable irrigation and virtual water trade (Martson et al., 2015; Dalin et al., 2017). Indeed, many rivers around the world are so strongly depleted that minimum flow requirements to sustain aquatic habitat are not met and in some cases the flow does not even reach the ocean anymore (Richter, 2014; Jägermeyr et al., 2017).

Here, we provide a comprehensive global analysis of unsustainable irrigation water consumption for crop production, accounting for the depletion of both freshwater stocks (including surface and ground water bodies) and environmental flows. We quantify the associated virtual water flows through trade (or 'unsustainable virtual water trade') in year 2000 and 2015 through a global spatially-distributed biophysical analysis of irrigation water consumption considering 130 primary crops or 26 crop classes (Portman et al., 2010). We quantify the amount of irrigation water that is unsustainably consumed by comparing irrigation water consumption to local renewable freshwater availability, calculated through a process-based water balance model accounting for environmental flows. We then determine the unsustainable water footprint of crop production and trade (FAOSTAT; http://www.fao.org/faostat/en/#data) to quantify the unsustainable irrigation water consumption embodied in the international trade of agricultural commodities.

This analysis sheds light on the water unsustainability of crop trade. It identifies the 'culprits' of unsustainable irrigation water consumption in terms of both production regions and consumer countries, and determines the associated virtual water flows. These results can assist in the development of consumption based decision-making tools aiming at meeting Sustainable Development Goals by ensuring sustainable use of water resources to reduce the number of people suffering from water scarcity (Vanham et al., 2018).

2.3 Methods

2.3.1 Assessment of unsustainable irrigation water consumption

We quantified unsustainable water consumption from irrigation worldwide (at 5×5 arcminute resolution) in years 2000 and 2015 for 26 crop classes, based on the MIRCA2000 dataset (Portmann et al., 2010). For year 2000 (the reference year for the MIRCA2000 global agricultural datasets), we used a global process-based crop water model to assess irrigation water requirements. This model has been extensively used to assess irrigation water requirements (Rosa et al., 2018a,b). The model calculates spatially explicit crop-specific irrigation water requirements (mm yr^{-1}) using a daily soil water balance during each crop's growing season. To assess irrigation water consumption, we then multiplied crop-specific irrigation water requirements by the irrigated harvested area of that crop in the year 2000 (Portmann et al., 2010).

Because there are no up-to-date global datasets of crop-specific irrigated harvested area, we estimated the change in irrigation water consumption between year 2000 and 2015 as proportional to the change in country-specific irrigation water withdrawals (from FAO's AQUASTAT) and crop-specific change in production (from FAO's FAOSTAT). Specifically, national crop-specific estimates of volumes of irrigation water consumption for year 2000 were scaled to year 2015 based on national agricultural water withdrawal and crop-specific production data. To quantify the sustainability of irrigation water consumption in year 2015, we assumed that the variation in irrigation water consumption between years 2000 and 2015 is proportional to country-specific sustainable and unsustainable irrigation expansion potentials. For example, if a country increases irrigation water consumption but has no potential to do so sustainably, all the additional irrigation volumes are assumed to be consumed unsustainably (at the expenses of environmental flows and surface- and ground- water stocks). Country specific values of the sustainability and unsustainability of irrigation expansion were taken from Rosa et al., 2018a. Crop-specific production in years 2000 and 2015 were taken from FAO's FAOSTAT database (see section: Assessment of unsustainable virtual water trade). Agricultural water withdrawals in years 2000 and 2015 were taken from FAO's AQUASTAT database.

To determine total blue water consumption (WC) in each grid cell, crop-specific irrigation water consumption values were summed with municipal and industrial water consumption estimates (for the 1996-2005 period) (Hoekstra and Mekonnen, 2012). By combining total blue water consumption and renewable blue water availability (WA) we assessed unsustainable irrigation practices. We identified areas of unsustainable irrigation water consumption as those where local renewable blue water resources are less than local total water blue consumption (WC > WA). This methodology to evaluate water sustainability has been extensively validated in studies aiming at analyzing the influence of energy and agricultural production on water resources (Rosa et al., 2018a,b; Rosa and D'Odorico, 2019). Renewable blue water availability (30 × 30 arcminute resolution, or ~50km at the Equator) was assessed following Mekonnen and Hoekstra (2016) and was calculated as the difference between blue water flows generated in that grid cell and environmental flow requirements. Renewable blue water availability accounts for surface- and ground- water volumes that are replenished through the annual hydrological cycle (Rosa et al., 2018a,b). This methodology explicitly links irrigation water consumption to unsustainable irrigation practices. Long term (circa year 2000) blue water flows were assessed from local runoff estimates (Fekete et al., 2002) and were calculated using the upstream-downstream routing "flow accumulation" function in ArcGIS®. Total blue water consumption (WC) at a 5 × 5 arcminute resolution was aggregated to the 30 × 30 arcminute resolution of the global available water (WA) dataset. Following previous global analyses we assumed that 80% of annual blue water flows should be allocated for environmental flows preservation (i.e., remain unavailable to human consumption) (Richter et al., 2012; Mekonnen and Hoekstra, 2016; Flörke et al., 2018).

We considered 26 crop classes or 130 primary crops (or nearly 100% of global crop production) (wheat, maize, rice, barley, barley, rye, millet, sorghum, soybeans, sunflower, potatoes, cassava, sugar cane, sugar beets, oil palm, rapeseed, groundnut, cassava, groundnuts, pulses, citrus, date palm, grapes, cotton, cocoa, coffee, other perennials, fodder grasses, other annual) based on the MIRCA2000 dataset (Portmann et al., 2010). Crop classes 'other annual' and 'other perennials' are labelled as 'Fruits & Vegetables' in Figure 2 and 3. 'Sugar Crops' include 'sugar beet' and 'sugar cane'; 'Other Grains' considers aggregated values of 'barley', 'rye', 'millet', and 'sorghum'. Water consumption from industrial and domestic sectors were taken from Hoekstra and Mekonnen, 2012 and were assumed to be constant between year 2000 and 2015. Possible inaccuracies in water consumption estimates for domestic and industrial uses are difficult to evaluate with the available data, however, they are expected to have limited impacts on our results. In fact, water consumption from industrial and domestic sectors accounts for just ~7% of total water consumption. Moreover, previous global studies of irrigation water consumption provided estimates that range from 847 km^3 to 1180 km^3 (Siebert et al., 2010; Hoekstra and Mekonnen, 2012). This range of uncertainty, by far, exceeds total water consumption from the domestic and industrial sectors (80 km^3; Hoekstra and Mekonnen, 2012).

2.3.2 Assessment of unsustainable virtual water trade

We used international trade matrices and national production data (FAOSTAT) (http://www.fao.org/faostat/en/#data) for 302 food commodities to assess the trade T of a food commodity x from a country y to z in year n (2000 and 2015) noted as $T(y, z, x, n)$. Because international trade and national crop production are dynamic and vary year by year, for international trade and national production data we used a five-year average around years 2000 and 2015 to smooth out this variability. For trade matrix data we used FAO's import data, because import reporting is more reliable than export reporting owing to custom reports at the port of entry (Dalin et al., 2017). Import data are expected to be more accurate, because customs have an incentive to collect data for tax purposes. Exports data are generally poorer, as very few countries tax exports. Moreover, the variation between import and export data of the FAO's dataset has a relatively small effect on the quantification of virtual water flows (Dalin et al., 2017). Because FAOSTAT's trade data are at the country and annual scales, our analysis is performed at annual and country levels.

We aggregated international trade matrices and national production data for 302 food commodities x into their 130 primary crops c dividing the quantity of produced or traded commodity $R(x)$ by its primary product extraction rate E_x:

$$\sum_{x \in c} \frac{R(x)}{E_x}$$

E_x was taken from FAO's technical conversion factors for agricultural commodities (http://www.fao.org/fileadmin/templates/ess/documents/methodology/tcf.pdf), and $R(x)$ is the quantity of produced or traded commodity x, and c is the set of commodities based on the same primary crop. Primary product extraction rate (E_x) is the fraction of the processed product obtained from the processing of the primary product. For example, FAOSTAT's commodities 'sunflower seed', 'sunflower oil', and 'sunflower cake' are based on the primary crop 'sunflower' and they have an extraction rate E equal to 1, 0.47, 0.49, respectively. To assess production and trade of the primary crop 'sunflower' we divided each commodity by its extraction rate and summed the results to obtain aggregated international trade matrices and national production data for sunflower in year n. The international trade matrices of the 130 primary crops where then aggregated into the 26 MIRCA2000 crop classes (*M*) to obtain trade fluxes *T* from a country *y* to *z* in year *n* of crop class *M*, *T* (*y*, *z*, *M*, *n*). The same procedure was followed for FAOSTAT's production data to obtain production data *P* of country *y* in year *n* of crop class *M*, *P(y, M, n)* (Dalin et al., 2017). Crop-specific primary product extraction rates are kept constant among countries.

We then assessed unsustainable irrigation water consumption intensity (*UWCI*) of each MIRCA2000 crop class (*M*) in each country (*y*) and year (*n*), as follows:

$$UWCI_{(y,M,n)} = \frac{UWC_{(y,M,n)}}{P_{(y,M,n)}}$$

Where *UWC* is the unsustainable irrigation water consumption of crops in class *M*, country *y*, and year *n* (expressed in m^3 of water). *UWC* has been assessed by aggregating irrigation blue water consumption (WC) at the country level. *P* is the aggregated production (expressed in tons) of crops belonging to class *M* in country *y* and year *n*.

We then used *UWCI* to convert the trade fluxes *T* (expressed in tons) into unsustainable virtual water flows (expressed in m^3):

$$UWCT_{(y,z,M,n)} = UWCI_{(y,M,n)} \times T_{(y,z,M,n)}$$

Where the trade *T* of a food commodity *x* from a country *y* to *z* in year *n* (2000 or 2015) is noted as *T* (*y*, *z*, *x*, *n*).

2.4 Results

2.4.1 The unsustainability of irrigation

We find that 52% (569 km^3) of global irrigation practices are unsustainable because they deplete freshwater stocks and/or environmental flows (Figure 1). About 70% of the global unsustainable water consumption for irrigation (hereunder UWC) is contributed by India (28%), China (16%), Pakistan (13%), and the United States (12%) alone (Figure 2a). In many countries a big share of irrigation water consumption is unsustainable as in the case of India (54% of national irrigation water consumption or 157 $km^3\ y^{-1}$), China (66% or 91 $km^3\ y^{-1}$), Pakistan (61% or 71 $km^3\ y^{-1}$), and the United States (62% or 69 $km^3\ y^{-1}$).

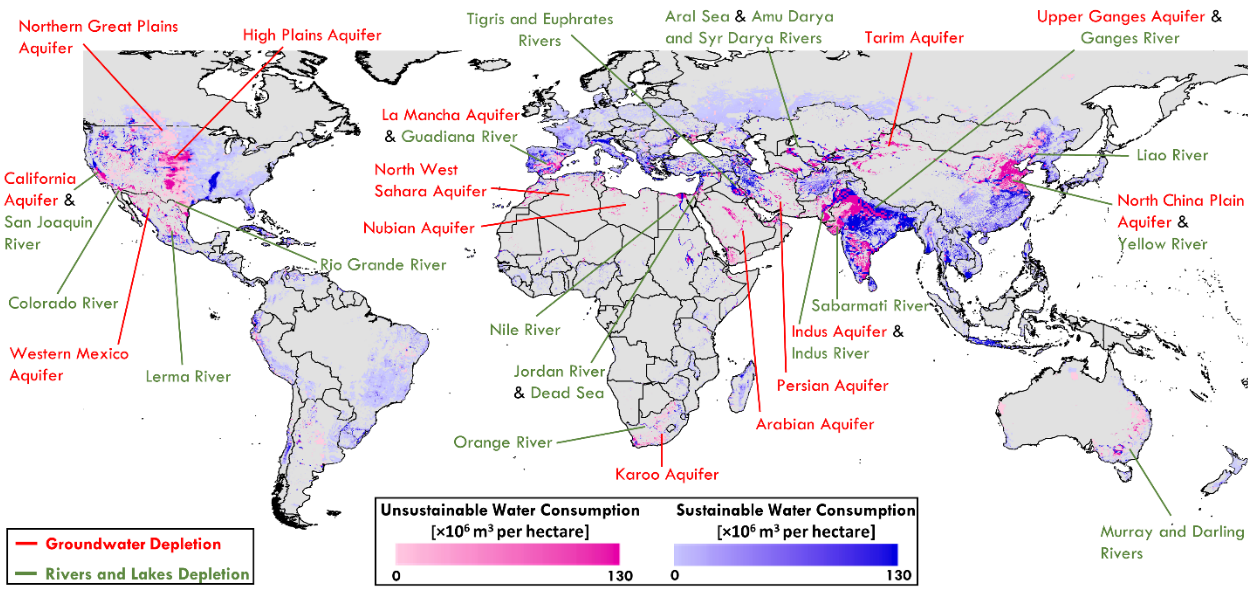

Figure 1. Global hotspots of unsustainable water consumption for irrigation. The map shows sustainable and unsustainable irrigation water consumption volumes and lists some of the freshwater stocks (aquifers, rivers, lakes) that are being depleted to grow crops (Richter, 2014; Richey et al., 2015; Jägermeyr et al., 2017).

The impact of agriculture on UWC strongly varies with crop type and geographic location with wheat, maize, rice, cotton, and fruits & vegetables collectively contributing to 73% (or 417 $km^3 y^{-1}$) of global UWC (Figure 3). While in India and Pakistan, wheat is the major contributor to UWC (32% and 38%, respectively), in China most of the UWC is from rice (33%) and in the United States from maize (29%) (Figure 2b).

UWC increased by 8% in fifteen years, from 525 km^3 in year 2000, to 569 km^3 in 2015 (Figure 2a), mostly because of irrigation expansion in India (+32 km^3), Pakistan (+6 km^3), Mexico (+2.5 km^3), China (+1.9 km^3), South Africa (+1.7 km^3), and Spain (+1.1 km^3). At the same time, UWC decreased in the United States (-7.2 km^3), Uzbekistan (-1.9 km^3), and Australia (-1.8 km^3). In this period, most of the increase in global UWC was contributed by irrigation expansion for maize (+23 km^3), wheat (+10 km^3), and cotton (+7 km^3) production, while most of the decrease in global UWC practices was from fodder grasses (-7.1 km^3), fruits & vegetables (-3.84 km^3), and sorghum (-1.24 km^3) (Figure 3).

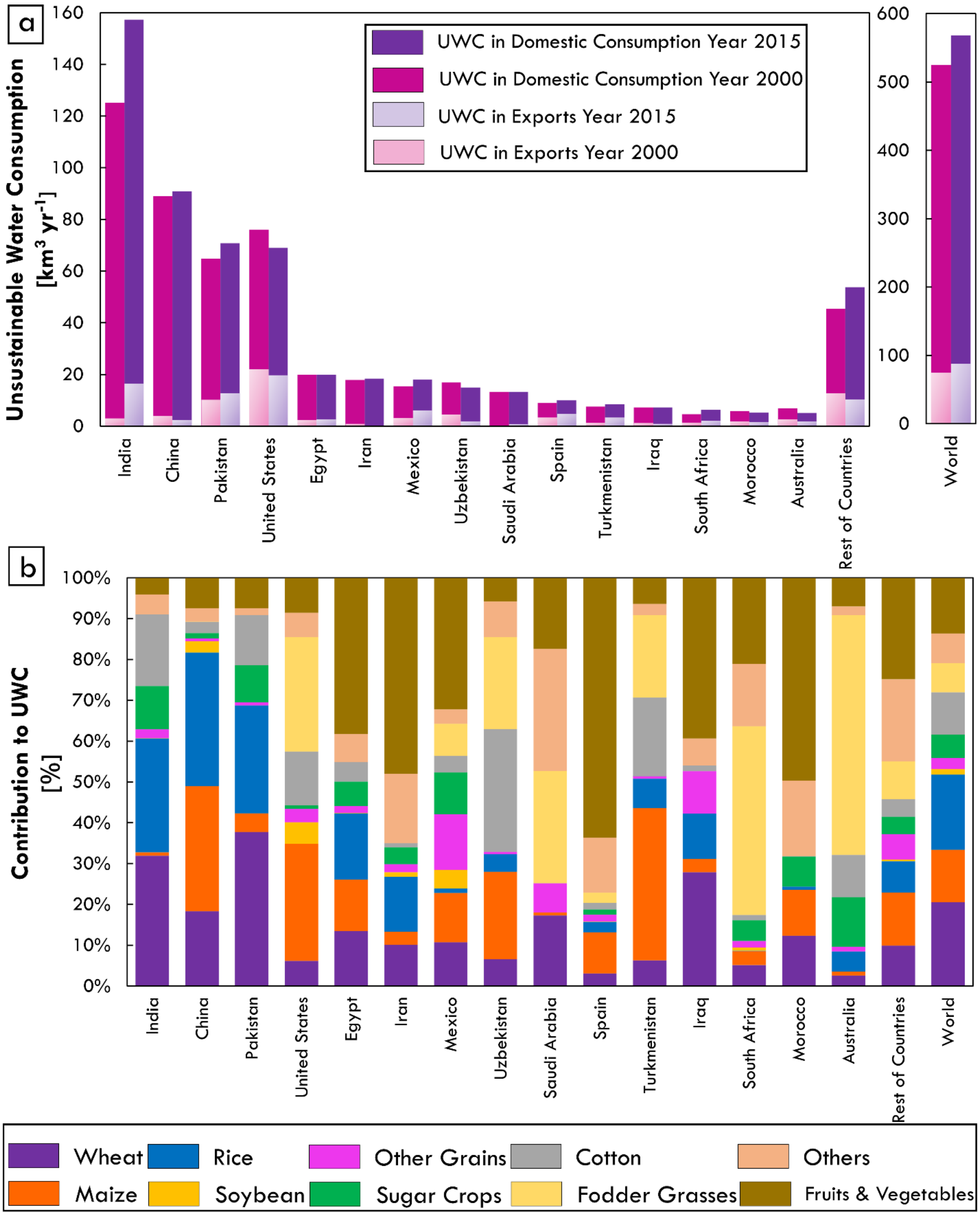

Figure 2. Unsustainable irrigation water consumption (UWC) embodied in domestic consumption and exports for 15 countries with the highest UWC. (a) Countries contributing the most to UWC for internal consumption and exports in year 2000 and 2015. (b) Crop-specific contribution to UWC by country in year 2015.

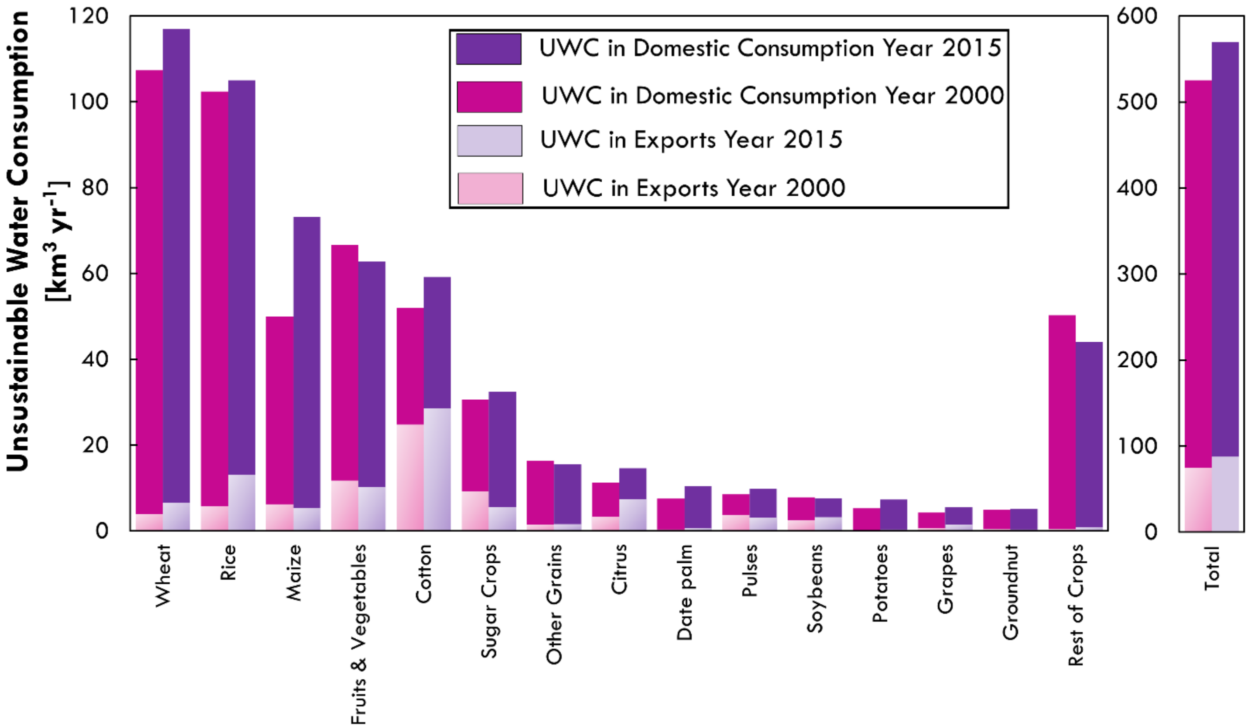

Figure 3. Crop-specific contribution to unsustainable irrigation water consumption (UWC) for domestic consumption and exports in year 2000 and 2015. Crops contributing the most to global UWC are wheat (21% or 117 km^3 of global UWC), rice (18% or 105 km^3), maize (13% or 73 km^3), fruits & vegetables (11% or 63 km^3), cotton (10% or 59 km^3), and fodder grasses (7% or 41 km^3). Crops contributing the most to international UWC trade are cotton (33% or 29 km^3), rice (15%), fruits & vegetables (12%), citrus (8%), wheat (7%), maize (6%), sugar cane (6%), and soybeans (4%). In the 2000-2015 period, UWC embedded in traded rice, wheat, citrus, cotton, and soybean has increased by 130%, 66%, 125%, 15%, 31%, respectively.

2.4.2 Unsustainability embodied in international food trade

We find that 15% of global UWC (88 km^3) is embedded in international crop trade. In the 2000-2015 period, global unsustainable virtual water trade increased by 18%, from 75 km^3 to 88 km^3 (Figure 2a), while the amount of food traded increased by 65%. Over this period, UWC in agricultural exports increased fourfold (+13.4 km^3) for India, followed by a 25% increase for Pakistan, Egypt (+9%), Mexico (+89%), and Spain (+42%). At the same time, the United States decreased their unsustainable exports of virtual water trade by 11% (-2.3 km^3), followed by China (-38%), Iran (-65%), and Uzbekistan (-61%).

In year 2015, the United States, India, Pakistan, Mexico, and Spain account for two thirds of UWC embodied in food trade. The United States is the largest exporter, with 22% (19.7 km^3) of global unsustainable virtual water transfers, followed by India (19%), Pakistan (14%), Mexico (7%), and Spain (5%). China is the largest importer of UWC-based crops, followed by the United States, Turkey, Mexico and Japan (Figure 4).

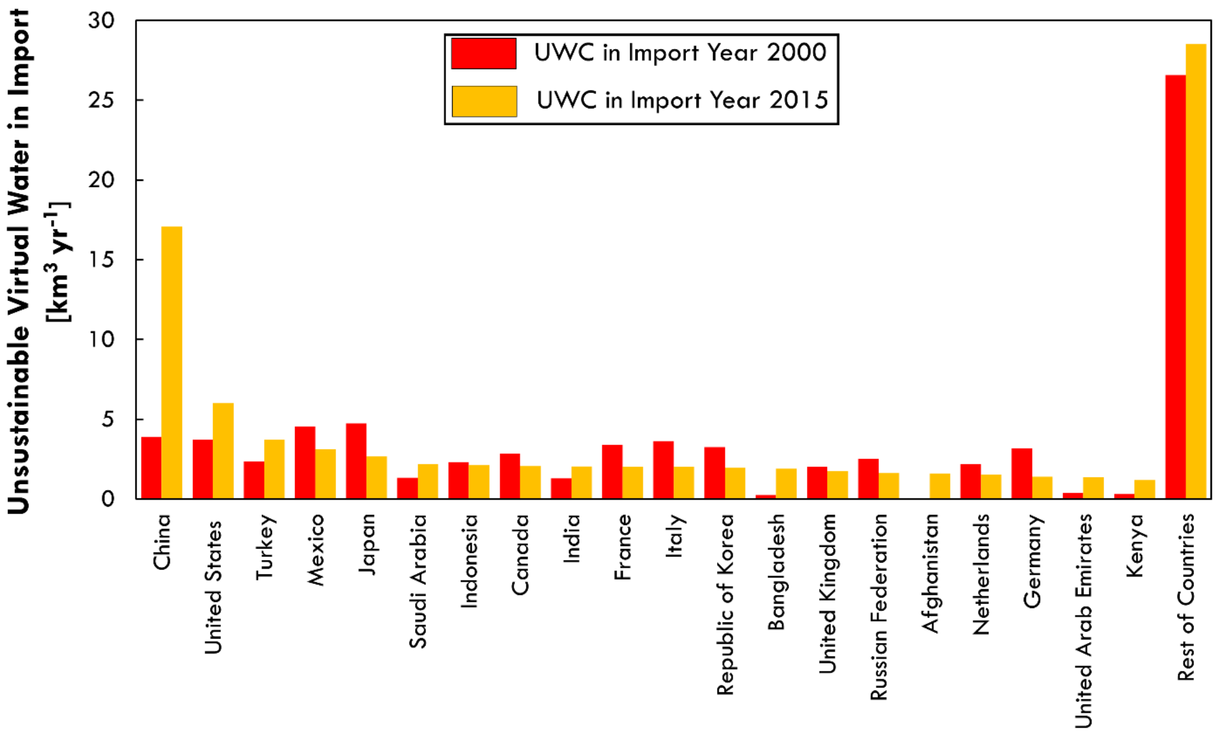

Figure 4. World largest unsustainable virtual water importers. The figure shows 20 countries with the largest dependency on unsustainable virtual water imports.

India, the United States, Pakistan, Spain, Turkmenistan, Egypt, Uzbekistan, and Australia consistently act as net exporters of UWC-based crops (Figure 5), while Canada, United Kingdom, France, Germany, Italy, China, Turkey, Russia, and Indonesia act as net importers of UWC-based crops. In the 2000-2015 period, Iran, Peru, Libya, Algeria, and Ethiopia have switched from being net exporters to net importers of unsustainable virtual water. Other countries such as Mexico, Tunisia, and Mozambique have become net exporters. Figure 5 also shows recent history of international food trade with the increasing presence of China as major importer, while it also shows the rise of India and Pakistan as major exporters of non-sustainable agricultural commodities.

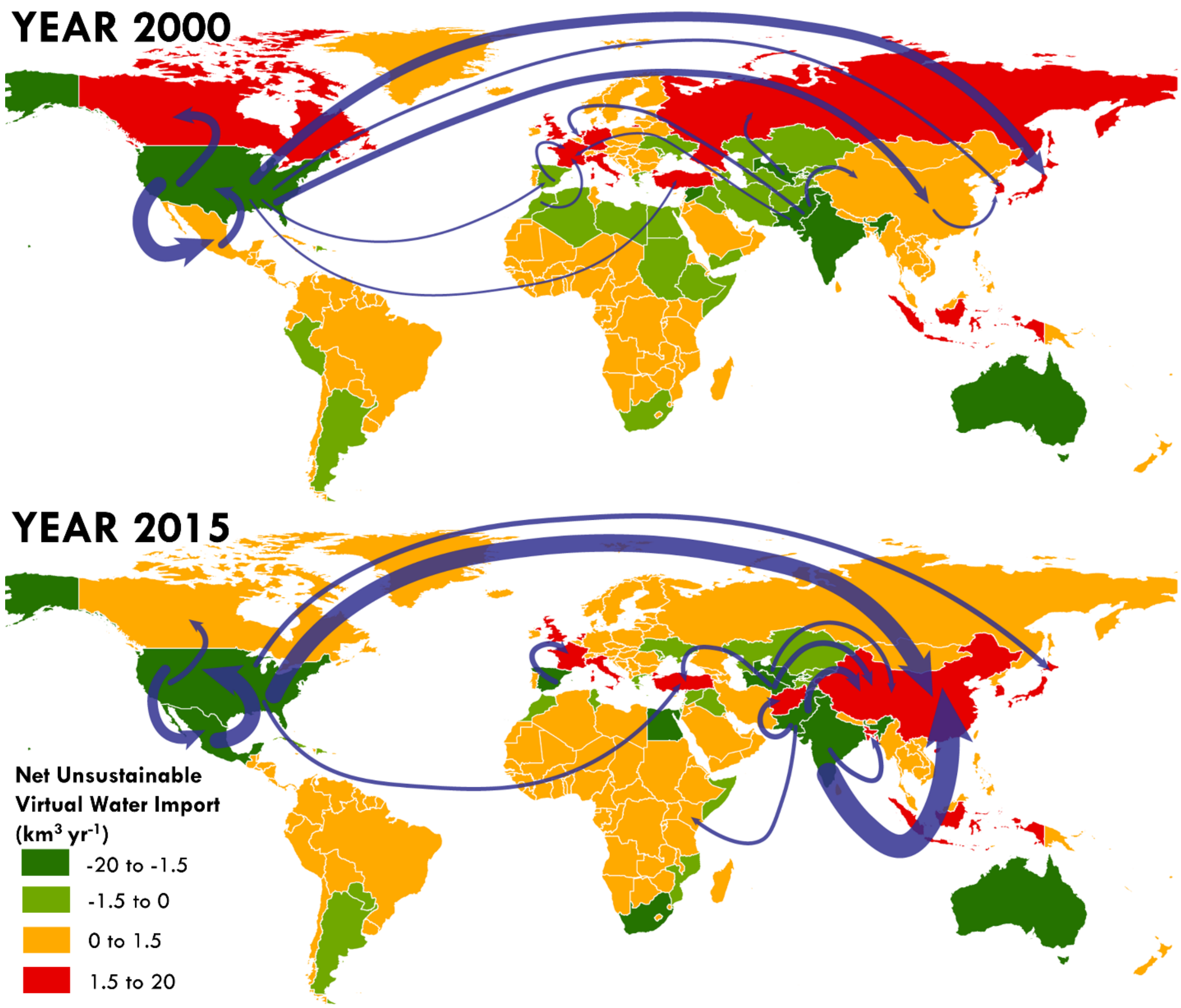

Figure 5. UWC embodied in international food trade in year 2000 and 2015. Net exporters are in light and dark green. Net importers are in orange and red. Arrows indicate the relative sizes of the fifteen largest global unsustainable virtual water flows (see Table 1).

In the United States 29% of the UWC is due to crops for export markets (Figure 2a). India and Pakistan are the second and third largest exporters of UWC-based crop production, even though they keep 90% and 82% of UWC for domestic consumption, respectively. China is the world largest net importer of UWC-based crops (19% of the global virtual trade of UWC). Moreover, China keeps 97% (89 km^3) of its UWC-based crops for domestic consumption.

In year 2015, 42% (8.2 km^3) of the Unites States' unsustainable virtual water trade was embedded in cotton export mainly to China, Mexico, Canada, Japan, and Turkey (Table 1 and Figure 5). Maize and soybeans accounted for 17% and 11% of the United States' unsustainable virtual water exports to China, Mexico, and Japan (Table 1). India exported unsustainably produced cotton and rice to China and Bangladesh. Mexico is a major exporter of unsustainably produced citrus and fruits & vegetables to the United States. UWC embedded in rice production accounts for 70% of Pakistan's unsustainable virtual water exports, mainly to China, Afghanistan, and Kenya. Uzbekistan and Turkmenistan export unsustainably produced cotton to China and Turkey. Spain and Morocco are major exporters of fruits & vegetables to European countries.

Table 1. Global largest unsustainable trade relationships in year 2000 and 2015. The table shows importers, exporters, volume of unsustainable virtual water traded (UWT) per trade link, and the main crops contributing to each unsustainable virtual water trade link.

	YEAR 2000				YEAR 2015			
Rank	**Importer**	**Exporter**	**UWT (km³)**	**Crops mainly traded**	**Importer**	**Exporter**	**UWT (km³)**	**Crops mainly traded**
1	Mexico	United States	4.5	Cotton (53%); Sorghum (16%); Maize (12%).	China	India	6.9	Cotton (90%).
2	Japan	USA	2.9	Maize (38%); Cotton (16%); Wheat (10%).	China	USA	5.2	Cotton (50%); Soybeans (20%).
3	Canada	USA	2.5	Cotton (36%); Other Annual (34%).	USA	Mexico	4.6	Citrus (50%); Fruits and Vegetables (20%); Sugar Cane (13%).
4	USA	Mexico	2.2	Cotton (62%); Sugar Cane (12%).	Mexico	USA	3.0	Cotton (47%); Maize (20%).
5	China	USA	2.0	Cotton (37%).	China	Pakistan	2.1	Rice (68%); Cotton (17%).
6	China	Pakistan	1.1	Cotton (47%); Sugar Cane (29%).	China	Turkmenistan	1.7	Cotton (98%).
7	Netherlands	Pakistan	1.0	Sugar Cane (91%).	Canada	USA	1.7	Fruits & Vegetables (44%); Cotton (19%).
8	South Korea	USA	0.9	Maize (32%); Cotton (42%).	France	Spain	1.6	Citrus (26%); Fruits & Vegetables (20%).
9	Russia	Uzbekistan	0.9	Cotton (96%).	Afghanistan	Pakistan	1.6	Wheat (36%); Rice (31%); Sugar Cane (26%).
10	South Korea	China	0.9	Maize (70%).	Japan	USA	1.5	Maize (41%); Cotton (16%).
11	France	Spain	0.8	Fruits & Vegetables (40%); Citrus (42%).	Bangladesh	India	1.4	Cotton (70%); Wheat (15%); Rice (12%).
12	France	Morocco	0.8	Fruits & Vegetables (80%).	Turkey	Turkmenistan	1.3	Cotton (100%).
13	France	Pakistan	0.7	Sugar Cane (67%); Cotton (30%).	China	Uzbekistan	1.3	Cotton (96%).
14	Spain	USA	0.7	Cotton (27%); Maize (16%).	Turkey	USA	1.1	Cotton (94%).
15	Turkey	USA	0.7	Cotton (66%); Maize (30%).	Kenya	Pakistan	1.1	Rice (100%).

2.5 Discussion

Virtual water trade is fundamental to achieve food security in water scarce regions of the world, however, it establishes a disconnection between consumers and the water resources they rely on. This ultimately leads to a loss of ecosystem stewardship (D'Odorico et al., 2019) and the enhancement of environmental degradation associated with the drying of rivers and loss of aquatic habitat (Soligno et al., 2017, 2018). Research on the environmental impacts of trade and trade policies (Peters et al., 2011; Zhang et al., 2017) suggests that production is expected to shift to regions of the world with weaker environmental regulations (Dean et al., 2005). In the case of agricultural production and trade, however, the focus has often been on environmental pollution, land use change, and labor rights, while the environmental impacts of unsustainable irrigation and their displacement through trade have remained poorly understood and have just started to be recognized and quantified.

Our results shed light on crops, country, and trade relationships that rely on unsustainable irrigation practices in production and consumption. More than 30% of unsustainable irrigation practices of Mexico, Spain, Turkmenistan, South Africa, Morocco, and Australia are embedded in food exports, while, in India, China, Iran, and Saudi Arabia 90% of unsustainable irrigation volumes are embodied in domestic food consumption. We also find that 60% (53 km^3) of global unsustainable virtual water trade is driven by cash crops (cotton, sugar cane, fruits & vegetables). In particular, cotton alone is responsible for 33% of UWC embedded in international crop trade. These findings show important trade-offs between the economic benefits and the environmental consequences of unsustainable irrigation practices.

Not surprisingly, the fact that only 30% of the increase in virtual water trade (in the 2000-2015 period) is contributed by unsustainable irrigation, confirms previous studies that quantified the increasing reliance of international markets on cropland expansion and rain-fed agriculture, including soybean production in Brazil and Argentina, and oil palm production in Indonesia and Malaysia (Aldaya et al., 2010; D'Odorico et al., 2019).

This study improves our previous assessment of unsustainable irrigation water consumption (Rosa et al., 2018a), where we found that in year 2000 about 40% (336 km^3) of global irrigation was unsustainable, based on 16 major crops that account for 70% of global crop production. Here, we considered 130 primary crops (or nearly 100% of global crop production) and found that 51% (525 km^3) of global irrigation volumes are unsustainable. Moreover, here we also provide crop-specific and country-specific analyses of unsustainable irrigation and evaluate extent to which it is contributed by international trade.

2.5.1 Uncertainty, limitations, and assumptions

The complexity of a global analysis often requires the adoption of suitable assumptions. We used a well-established methodology to assess irrigation water consumption based on existing maps of irrigated areas for the year 2000 (Siebert et al., 2010). However, it is important to note that the estimation of irrigated areas would change significantly using different input data and statistics (Meier et al., 2018). Because, to our knowledge, there are not global datasets providing crop-specific irrigated harvested areas after year 2000, we used country-scale statistics to assess irrigation water consumption in year 2015. While this is a limit of global studies aiming at an assessment of irrigation water consumption, our results are in good agreement with recent country-specific changes in irrigation water consumption as available for Australia (-1.3 km^3 from 2000 to 2015) (Australian Bureau of Statistic, 2018), and India (+26 km^3 from 2000 to 2009) (Davis et al., 2018).

Our assessment is based on temporal averages and does not account for inter-annual variability in river discharge and crop water requirements. We performed a sensitivity analysis of irrigation water consumption with respect to changes in climate conditions between 2000 and 2015. We run our crop water model with year 2015 climate forcing, while keeping the same spatial extent of irrigated area as in the MIRCA2000 dataset. We find that there is little sensitivity of irrigation water consumption between the two years (1025 km^3 for year 2000 versus 1035 km^3 for year 2015).

Because fodder grasses (Alfalfa, clover, and grasses) are not present in FAOSTAT's trade data, we excluded them from our unsustainable virtual water trade analysis. It is important to notice that fodder grasses are mainly used for domestic consumption as feed to livestock and therefore they are not commonly traded among countries. Nevertheless, we find that fodder grasses account for 7% of global unsustainable irrigation water consumption (Figure 2b).

Our results show little sensitivity to different environmental flow thresholds, as previously highlighted also by Mekonnen and Hoekstra, 2016. With the current assumption that 80% of blue water flows should be allocated to environmental flows preservation, we find that, in year 2000, 51% (525 km^3) of global irrigation water consumption is unsustainable. When we assume that environmental flows account for 90% and 60% of blue water flows, unsustainable irrigation water consumption in the same year becomes 536 km^3 and 523 km^3, respectively. As already stressed by Mekonnen and Hoekstra (2016), this low sensitivity to the threshold used to define environmental flows is due to the huge spatiotemporal mismatch between water consumption and availability.

2.6 Conclusions

Policymakers and major corporations are broadening the scope of their actions to meet the increasing consumer demand for sustainable commodities and improve corporate social responsibility. For example, there has been a recent commitment to purchase or produce deforestation-free products (Carlson et al., 2017; Curtis et al., 2018). In an increasingly water scarce world, governments could take specific actions targeting unsustainable irrigation practices by penalizing the associated imports. By identifying the producing and consuming countries that are responsible for unsustainable virtual water trade, this study highlights trade links in which policies are needed to achieve sustainable water and food security goals in the coming decades. Future studies should examine socio-economic implications, such as the feasibility to reduce unsustainable virtual water trade through the adoption of adequate policies.

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CHAPTER 3

Global agricultural economic water scarcity

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3.1 Abstract

Water scarcity raises major concerns on the sustainable future of humanity and the conservation of important ecosystem functions. To meet the increasing food demand without expanding cultivated areas, agriculture will likely need to introduce irrigation in croplands that are currently rain-fed but where enough water would be available for irrigation. 'Agricultural economic water scarcity' is here defined as lack of irrigation due to limited institutional and economic capacity instead of hydrologic constraints. To date, the location and productivity potential of economically water scarce croplands remain unknown. We develop a monthly agro-hydrological analysis to map agricultural regions affected by agricultural economic water scarcity. We find these regions account for up to 25% of the global croplands, mostly across Sub-Saharan Africa, Eastern Europe, and Central Asia. Sustainable irrigation of economically water scarce croplands could feed an additional 840 million people, while preventing further aggravation of blue water scarcity.

3.2 Introduction

The global growth in food demand is placing unprecedented pressure on the land and water resources of our Planet. Water and nutrients are important key biophysical factors determining food production (*1*). While, advances in technology have allowed humanity to economically produce massive quantities of nitrogen fertilizers (*2*), water still remains a critical input limiting global food production (*3*). To halt agricultural expansion and meet the increasing demand for food commodities, agricultural production will likely have to intensify by expanding irrigation to water-limited croplands that are currently rain-fed (*4*). In some regions of the world the expansion of irrigation will likely put under additional stress water bodies and aquifers that are already depleted (*5*), rising concerns about the Earth's ability to feed humanity with its limited freshwater resources (*6*).

Water scarcity refers to a condition of imbalance between freshwater availability and demand where freshwater demand exceeds availability (*7*). Water scarcity represents a multidimensional state of human deprivation characterized by lack of access to affordable and safe water to satisfy societal needs or a condition in which such needs are met at the expenses of the environment (*8*). While water scarcity may affect entire regions, it is the most vulnerable and poor people that suffer the most severe consequences (*9*). This fact points to the strong role played by economic and institutional factors as determinants of water scarcity. Therefore, water scarcity is generally considered both from the perspective of its *physical* constraints and *economic* determinants.

Physical water scarcity affects both blue and green water (i.e., water from water bodies or aquifers and soil moisture, respectively; see Box 1). In the case of crop production, *green water scarcity* corresponds to a condition in which the rainfall regime is unable to meet the crop water requirements (Box 1). In other words, for at least part of the year irrigation is needed to prevent water-limited crop growth. *Blue water scarcity* occurs in croplands facing green water scarcity if the available renewable blue water resources are not sufficient to meet the irrigation water requirements. In this context, renewable blue water resources are defined as the water resources that can be withdrawn from aquifers and surface water bodies without causing either groundwater depletion or loss of environmental flows – the stream flows that need to be maintained to preserve aquatic habitats (*10-12*). In case of blue water scarcity, farmers can either practice sustainable irrigation without completely meeting the crop water requirements (i.e., "deficit irrigation"), or meet such requirements through unsustainable irrigation practices at the expenses of environmental flows and/or groundwater stocks (Box 1).

Blue water has been at the center of the water scarcity debate because it underlies the emerging competition between water uses for societal and environmental needs (*13-19*). Indeed, blue water scarcity is increasingly perceived as a global socio-environmental threat (*20*) that has been associated with questions about food security and energy security (*3*). Moreover, Target 6.4 of the Sustainable Development Goals (SDGs) explicitly addresses blue water scarcity with the goal of ensuring adequate blue water resources for humans and ecosystems. Conversely, green water scarcity has received much less attention even though ≈65% of global crop production is contributed by green water (*21-24*). Interestingly, a management plan for green water is still missing in the SDGs agenda (*25*). Even less studied is the case of economic water scarcity.

While green and blue water scarcity refer to conditions of *physical water scarcity* associated with insufficient freshwater availability to meet human needs (*8,26*), *economic water scarcity* has been defined as the condition in which renewable blue water resources are physically available, but lack of economic and institutional capacity limits societal ability to use that water (*8,27,28*). An early definition of economic water scarcity is one that described countries that have adequate renewable water resources to meet current and projected water requirements but need to make massive improvements in their water development programs to be able to utilize their freshwater resources (*27*). The technocratic, hydraulic engineering perspective that has dominated the 'hydraulic mission' of the 20th century has pushed infrastructural development as the main determinant of water development (*29*). As such the lack of infrastructural development has been at the center of the conceptualization of economic water scarcity (*30*). However, this "old water governance" approach has been exposed for its inability to deal with fast-changing socio-hydrological conditions and often criticized for doing more harm than good to the environment and the society. Emerging research agendas on adaptive water governance (*31*), the political ecology of water (*32*), water justice (*33*), and community (*34*) debate how institutional, political, and power dynamics are ultimately affecting the relationship between access and restriction; and possession and dispossession of water resources. As such the understanding of economic water scarcity needs to consider the variety of socio-political factors that interact at different scales. For example, maintaining a focus on the global scale, we see that there is a fundamental gap in the way the notion of economic water scarcity has been integrated in agricultural development so far (*27*).

We here define and introduce the original concept of '*agricultural economic water scarcity'* as the condition whereby croplands exposed to green water scarcity are not irrigated even though a sufficient amount of renewable blue water resources for irrigation is locally available. These conditions occur for instance as a result of a variety of socio-economic and political factors that impede irrigation. To date, little attention has been given to the analysis of this phenomenon and its role in the global geography of water scarcity.

Here, we firstly develop and apply a monthly agro-hydrological model to quantify and map croplands affected by agricultural green, blue, and economic water scarcity. By doing so we firstly provide a comprehensive, spatially explicit, global assessment of agricultural economic water scarcity (Figure 1). We, first, identify croplands affected by green water scarcity and estimate their irrigation water requirements with an evapotranspiration model coupled with a soil water balance analysis. We use a simple comparison between irrigation water requirements and local water availability to investigate to what extent rain-fed croplands affected by green water scarcity are also prone to blue water scarcity. Because farmers might not always irrigate at maximum potential, in areas affected by blue water scarcity we also considered two deficit irrigation scenarios, where 80% and 50% of full irrigation water requirements are applied to crops (also named as 20% and 50% irrigation deficit, respectively). These deficit irrigation scenarios are investigated only if and where they do not entail the depletion of groundwater resources or environmental flows. We then identify economically water scarce lands as those rain-fed areas where irrigation water requirements can be sustainably met either completely or through deficit irrigation, but irrigation is still missing. Further, we calculate the maximum volume of renewable blue water resources that would be consumed to support crop production in cultivated lands affected by green water scarcity but not prone to blue water scarcity. This water includes current sustainable irrigation water consumption and the additional water that would be needed to expand irrigation into rain-fed areas affected by economic water scarcity. We finally estimate the additional calorie produced and number of people that can be fed from sustainable irrigation expansion over economically water scarce croplands.

Our results improve the understanding of how agricultural economic water scarcity impacts water and food security globally. The application of the concept of agricultural economic water scarcity has the potential to inform water and food security policies at global, regional, national, and local scales and provide new insights to achieve global sustainability targets.

Box. 1. Concepts and definitions about agricultural economic water scarcity.

Water Consumption: The volume of abstracted water that is evapotranspired.
Green Water: Root-zone soil moisture that is available for uptake by plants.
Blue Water: Freshwater in surface and groundwater bodies available for human use.
Green Water Scarcity (GWS): When green water is insufficient to sustain unstressed crop production and irrigation is needed to boost yields. *Green water scarcity* can be defined as the ratio between irrigation water requirement (or 'green water deficit') and the total crop water requirement (*4*).
Irrigated Agriculture: When there is GWS and crop production is enhanced by irrigation (blue) water.
Sustainable Irrigation (SI): When renewable blue water availability is sufficient to sustain crop production while preventing loss of environmental flows and depletion of freshwater stocks (*4, 10*).
Blue Water Scarcity (BWS): When irrigation is unsustainable (UI) and renewable blue water availability is insufficient to sustainably meet crop water requirements. In these cases, irrigation impairs environmental flows and depletes freshwater stocks. *Blue water scarcity* has been defined as the ratio between societal blue water demand and renewable blue water availability (*26, 35*).

Agricultural Economic Water Scarcity (EWS): When there is GWS but no BWS. There is renewable blue water to irrigate but lack of economic or institutional capacity. Agricultural economically water scarce croplands are underperforming rain-fed croplands suitable for sustainable irrigation expansion.
Total Water Scarcity (TWS): When there are GWS, BWS, and lack of economic or institutional capacity.
Deficit Irrigation: An irrigation practice whereby blue water supply is reduced below maximum levels and crops are grown under mild water stress conditions with minimal effects on yields.

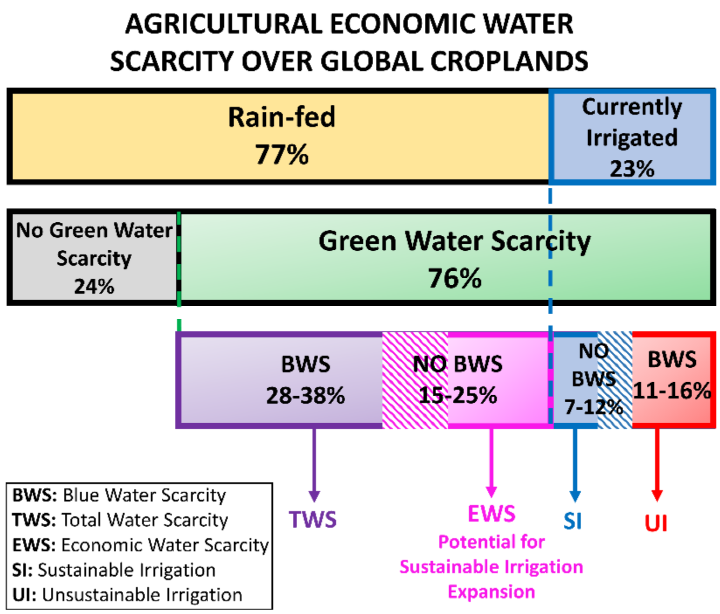

Figure 1. Conceptual framework and extent of agricultural economic water scarcity. Percentages represent fraction of the global cultivated area in each category. Shading indicates croplands affected by blue water scarcity (BWS) that can be sustainably irrigated with deficit irrigation. These areas are then reclassified as suitable for sustainable irrigation (i.e., with no blue water scarcity (NO BWS)), considering different deficit irrigation scenarios. Lack of irrigation in these areas is interpreted as agricultural economic water scarcity. See Box 1 for concepts and definitions about agricultural economic water scarcity.

3.3 Methods

3.3.1 Assessment of green, blue, and economic water scarcity

Water scarcity was assessed per month per grid cell at 5×5 arcminute resolution (or ~10km at the Equator). Monthly green water scarcity was expressed as the ratio between irrigation blue water requirements (or green water deficits) (*BWR*) and crop water requirements (*CWR*). Crops face green water scarcity when rainfed conditions can meet the crop water requirements. We define the green water (*GWS*) as the ratio (*4*)

$$GWS = \frac{BWR}{CWR}$$

Because areas with small levels of crop water deficit do not require irrigation, we classify as water scarce those regions with GWS>0.1.

Monthly blue water scarcity was calculated as the ratio between current blue water consumption (WC_{CURR}) and renewable blue water availability (WA). Blue water scarcity occurs when total blue water consumption exceeds blue water availability, or when the following ratio is greater than 1 (*17*).

$$BWS = \frac{WC_{CURR}}{WA} > 1$$

Monthly economic water scarcity was calculated over croplands currently not equipped for irrigation and facing green water scarcity but no blue water scarcity. Therefore, economic water scarcity (EWS) was defined as the ratio between total blue water consumption under yield gap closure (WC_{GAP}) and renewable blue water availability (WA).

$$EWS = \begin{cases} \text{currently not equipped for irrigation} \\ \text{facing green water scarcity} \\ \dfrac{WC_{GAP}}{WA} < 1 \end{cases}$$

3.3.2 Assessment of green and blue water consumption

We used a global process-based crop water model to calculate the crop water requirements (CWR) for 130 primary crops or 26 crop classes (or nearly 100% of global crop production) for the 1996-2005 period using monthly climate forcing, while keeping the spatial extent of global croplands fixed to the MIRCA2000 dataset (*49*). This model has been extensively used to assess spatially explicit CWR (*4, 10, 50*). CWR is the amount of water needed by a crop to satisfy its evapotranspirative demand and to avoid water-limited plant growth. CWR can be satisfied by precipitation (i.e., green water) and supplemented through blue water (or irrigation) (blue water requirement, BWR, or irrigation water requirement) if precipitation is insufficient to meet the entire CWR. The model calculates a crop-specific CWR (mm yr^{-1}) using a daily soil water balance during each crop's growing season (*4, 10, 50*).

In every grid cell the current irrigation water consumption ($WC_{IRR,CURR}$) was calculated multiplying crop-specific blue water requirement by the irrigated harvested area of that crop in the year 2000 (*49*). To assess green water consumption (WC_{GREEN}), we multiplied crop-specific green water consumption calculated by the model by the rain-fed harvested area of that crop in year 2000 (*49*). For each crop we also assessed irrigation water consumption at yield gap closure ($WC_{IRR,GAP}$) by multiplying crop-specific blue water requirements by the rain-fed harvested area of that crop in year 2000 (*49*). Water consumption at yield gap closure – the difference between current and attainable yields (*51*) – is the additional irrigation water necessary to avoid water-limited plant growth and therefore reach the maximum crop productivity (or 'close the yield gap') in rain-fed croplands (*4*). Yield gap closure can be achieved by avoiding biophysical deficiencies that constrain crop growth and are not addressed by current management practices, including irrigation and fertilizer applications (*1*). Yet, this study focuses on limitations arising from water scarcity.

Monthly current total blue water consumption (WC_{CURR}) was assessed by summing monthly current irrigation water consumption ($WC_{IRR,CURR}$) and monthly estimates of industrial and municipal blue water consumption. This analysis was repeated to assess monthly total blue water consumption under yield gap closure (WC_{GAP}) – where we assumed constant consumption from industrial and municipal uses. Industrial and municipal blue water consumption for the 1996-2005 period were taken from Hoekstra and Mekonnen (2012) (*52*). For each month of the year, we considered a ten-year average for the 1996-2005 period. WC_{CURR} and WC_{GAP} at 5×5 arc minute resolution were aggregated to 30×30 arc minute resolution, the resolution of the renewable blue water availability analysis (WA).

3.3.3 Assessment of renewable blue water availability

Renewable blue water availability (WA) (30 × 30 arcminute resolution) was evaluated following Mekonnen and Hoekstra (2016) (*17*) as the difference between blue water flows generated in that grid cell and environmental flow requirements. Renewable blue water availability accounts for surface- and ground- water volumes that are recharged through the hydrological cycle (*4*). Long term (circa year 2000) monthly blue water flows were assessed from local runoff estimates obtained from the Composite Runoff V1.0 database (*53*) and were calculated using the upstream-downstream routing "flow accumulation" function in ArcGIS®. Environmental flow requirements were assessed by using the Variable Monthly Flow (VMF) method (*54*). The VMF method estimates environmental flow requirements taking into account the seasonality of flow regimes. Once assessed, blue water scarcity at 30×30 arc minute resolution was disaggregated at 5×5 arc minute resolution, the resolution of the rain-fed and irrigated harvested areas datasets (*49*).

3.3.5 Assessment of calorie production

For each of the 26 crop classes, current and maximized calorie production were assessed as the product of crop yield (tons per hectare), crop calorie content (kcal per tons), and crop harvested area (hectares). Current and maximized crop yields were taken from Monfreda et al., 2008 and Mueller et al., 2012 (*55*, *1*), respectively. Calorie content for each crop was taken from D'Odorico et al., 2014 (*56*). Crop harvested areas were taken from Portman et al., 2010 (*49*). We considered a linear relation between crop yields and biophysical water deficit (*57*) assuming that irrigated production decrease by 20% and 50% under a 20% and 50% irrigation deficit scenario, respectively. We assessed the number of people that can be potentially fed considering a global average diet of 3343 vegetal kcal per capita per day (*4*).

3.3.6 Uncertainties, assumptions, and limitations

The complexity of a global analysis lends itself to a scenario-based approach and to the use of suitable assumptions. First, our model does not consider future potential changes in crops and cropping practices that could result from the development of irrigation infrastructure, nor does it consider the economic viability of new irrigation projects. For example, while it might be technically possible to expand irrigation over economically water scarce lands in Western Europe and North America, from the standpoint of economic evaluations it might be unfeasible because of the low return on investment relative to the cost of irrigation infrastructure (Figure 4). Increasing crop productivity might not always be the preferred option, considering other local socio-economic or environmental factors that our biophysical model is unable to account for (e.g., regional water and land management policies, transboundary water rights, and political instability). Second, irrigation infrastructure might also include new water storage to meet water demand during the dry season. Thus, the access to new water storage would affect our agricultural economic and blue water scarcity assessment at the monthly scale. Third, we assessed intra-annual agricultural water scarcity based on long term renewable water availability data. Inter-annual variations in water availability, however, may lead to year-to-year fluctuations in the global patterns of water scarcity that are not investigated in this study (*58*). Fourth, this study did not account for the fact that many of the assessed agricultural economic water scarce regions require not only additional irrigation water, but also an improvement in nutrient supply (e.g., through manure or industrial fertilizers) in order to achieve maximum yields (*1*). Fifth, we assumed that irrigated crop yields decrease linearly with the reduction in irrigation water applied under deficit irrigation scenarios. This approach is widely implemented in global studies aiming to assess changes in yields under deficit irrigation (*5*). However, we acknowledge that each crop variety has different responses to water-stressed crop growth. Moreover, the 20% and 50% deficit irrigation thresholds were chosen as an intermediate and extreme value of deficit irrigation that can be applied to crops. Sixth, water scarcity depends also on the quality of water resources, because water of poor quality is not suitable for irrigation. In this study we assessed agricultural economic water scarcity only as a function of the available water quantity without considering water quality. Seventh, our study assesses sustainable irrigation based on the amount of water evapotranspired by crops and therefore it does not need to account for the efficiency of the irrigation systems, which needs to be considered in studies that use water withdrawals in their analyses. Eighth, we assumed that staple crops and cash crops are all irrigated under the same conditions. However, we acknowledge that the flexibility in irrigation water applications varies between crops depending on the costs or effects associated with water-stressed crop growth (*19*). Lastly, given the global scope of this study, we assessed environmental flows using the Variable Monthly Flow method (*54*). However, we acknowledge that, depending on the scale of the analysis, environmental flow requirements could be defined differently to account for watershed-specific attributes of the hydrologic regime that are crucial to the maintenance of aquatic habitats. These are all assumptions, limitations, and uncertainties that can be accepted within the current study scale and objective, which is to introduce the idea of agricultural economic water scarcity, a method to measure it, and its potentials for global sustainable intensification of agriculture.

3.4 Results

3.4.1 Exposure to green and blue water scarcity

We develop a spatially-explicit integrated mapping of green, blue, and economic water scarcity across the global croplands for 130 primary crops (or nearly 100% of global crop production) for the 1996-2005 period using monthly climate forcing (Figure 2). The exposure to water scarcity strongly varies with geographic location and month of the year (Figure 3). We find that 76% of global croplands (or 69% of global rain-fed calorie production) face green water scarcity for at least one month a year and 42% experience green water scarcity for five months a year (Figure 1).

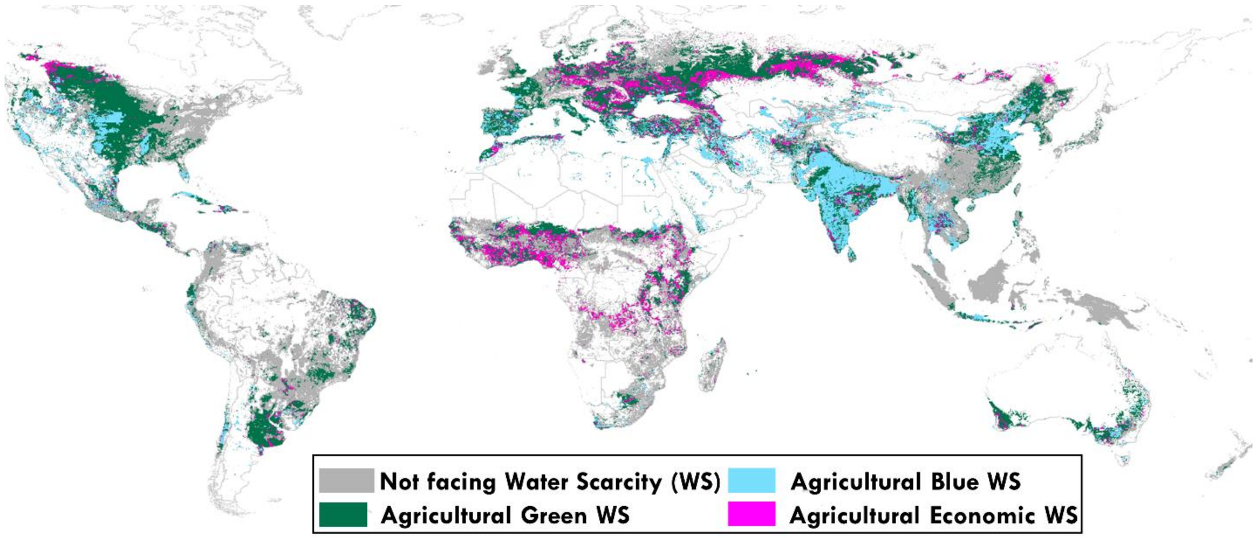

Figure 2. The geography of global agricultural water scarcity. The map shows the global distribution of agricultural green, blue, and economic water scarcity across global croplands. In the map are shown croplands facing at least one month of water scarcity per year.

We estimate that current green water consumption over croplands is 5,406 $km^3\ y^{-1}$. To avoid crop growth in water-stressed conditions as a result of green water scarcity, global croplands would require an additional 2,860 $km^3\ y^{-1}$ of blue water consumption. In other words this is the global irrigation water requirement without accounting for the limits imposed by sustainability needs. Presently, 23% of global cropland areas are irrigated, consuming 1,083 $km^3\ y^{-1}$ of blue water resources. Irrigation currently provides 34% of global calorie production (calculated as the difference between irrigated and rain-fed production over irrigated lands) or 40% if calculated as the total production from irrigated lands. Major irrigated regions in the United States (High Plans and the Central Valley of California), Mexico, Spain, North China, Australia (the Murray-Darling Basin), India, and Pakistan consistently face blue water scarcity for several months during their crops' growing seasons (Figure 3). In those months irrigation water requirements can only be met with an unsustainable use of water resources.

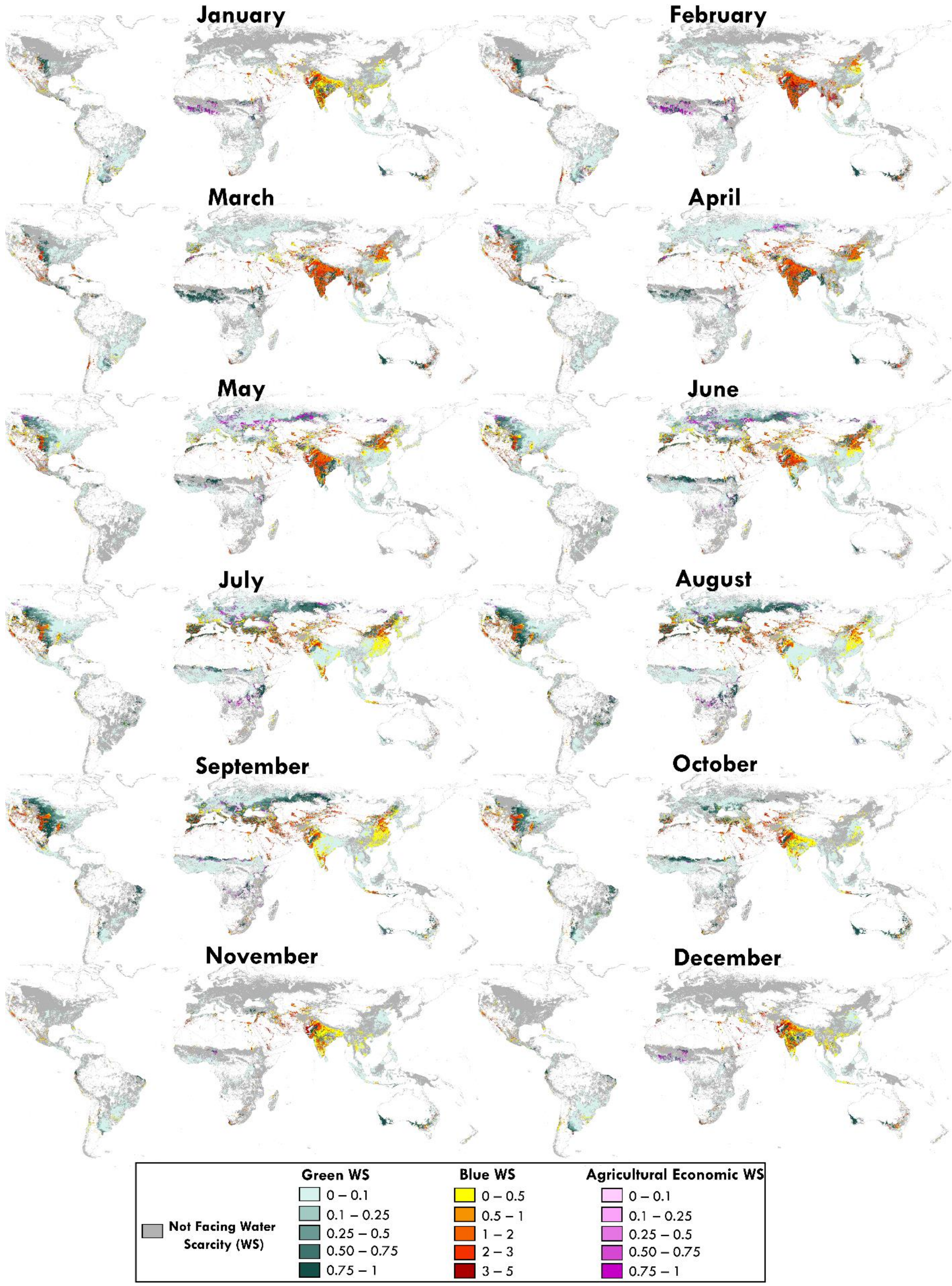

Figure 3. Monthly agricultural green, blue, and economic water scarcity over global croplands.

We find a widespread reliance of food production on irrigated regions affected by blue water scarcity. Indeed, 68% of the global irrigated croplands face blue water scarcity for one month a year and 37% experience blue water scarcity for five months during the year. We estimate that 22% of global calorie production is exposed to at least one month of blue water scarcity during the growing season and that 56% (611 $km^3 yr^{-1}$) of global irrigation volumes are applied on unsustainably irrigated lands (Figure 4). We also analyze to what extent deficit irrigation would be sustainable even in currently irrigated areas affected by blue water scarcity. We find that water applications with a 20% and 50% irrigation deficit, could be sustainably carried out in 7% (0.01 billion hectares) and 33% (0.05 billion hectares) of the currently irrigated lands affected by blue water scarcity, respectively (Figure 4).

3.4.2 Exposure to agricultural economic water scarcity

The widespread reliance on unsustainable irrigation, combined with longer dry spells, and more erratic rainfalls are of particular concern for local and global food security. The expansion of irrigation over economically water scarce lands could be an important adaptation strategy to climate change, contributing to a more reliable and resilient crop production.

We find that 15% (0.14 billion hectares) of global croplands are exposed to agricultural economic water scarcity, whilst, 16% of the cultivated lands are currently unsustainably irrigated. Considering current crop types and growing seasons, the expansion of irrigation to lands affected by economic water scarcity would increase global blue water consumption for irrigation by 10% (+105 $km^3 y^{-1}$), thereby allowing for a 6% increase in global calorie production (0.76×10^{15} kcal), which would be sufficient to feed 620 million people (Figure 4). Because rain-fed production usually allows for only one growing season per year, we find that 43% (0.06 billion hectares) of economically water scarce croplands face agricultural economic water scarcity for only one month in the course of its rain-fed growing season, and 86% is exposed to agricultural economic water scarcity for three months during its rain-fed growing season (Figure 3).

By applying a 20% and 50% irrigation deficit the extent of economically water scarce croplands would increase. With a 20% irrigation deficit, it is possible to further expand sustainable irrigation to an additional 5% of global croplands (+0.05 billion hectares) (Figure 4). This expansion of sustainable irrigation would feed an additional 160 million people, while increasing irrigation water consumption by 50 $km^3 y^{-1}$. By applying a 50% irrigation deficit, an additional 5% of global croplands could be irrigated sustainably to produce food for 60 million more people. Therefore, in a 50% irrigation deficit scenario, up to 25% of global croplands are found to be exposed to agricultural economic water scarcity (Figure 1). In this scenario, sustainable irrigation expansion over underperforming rain-fed (i.e., economically water scarce) lands could increase food production to feed about 840 million people.

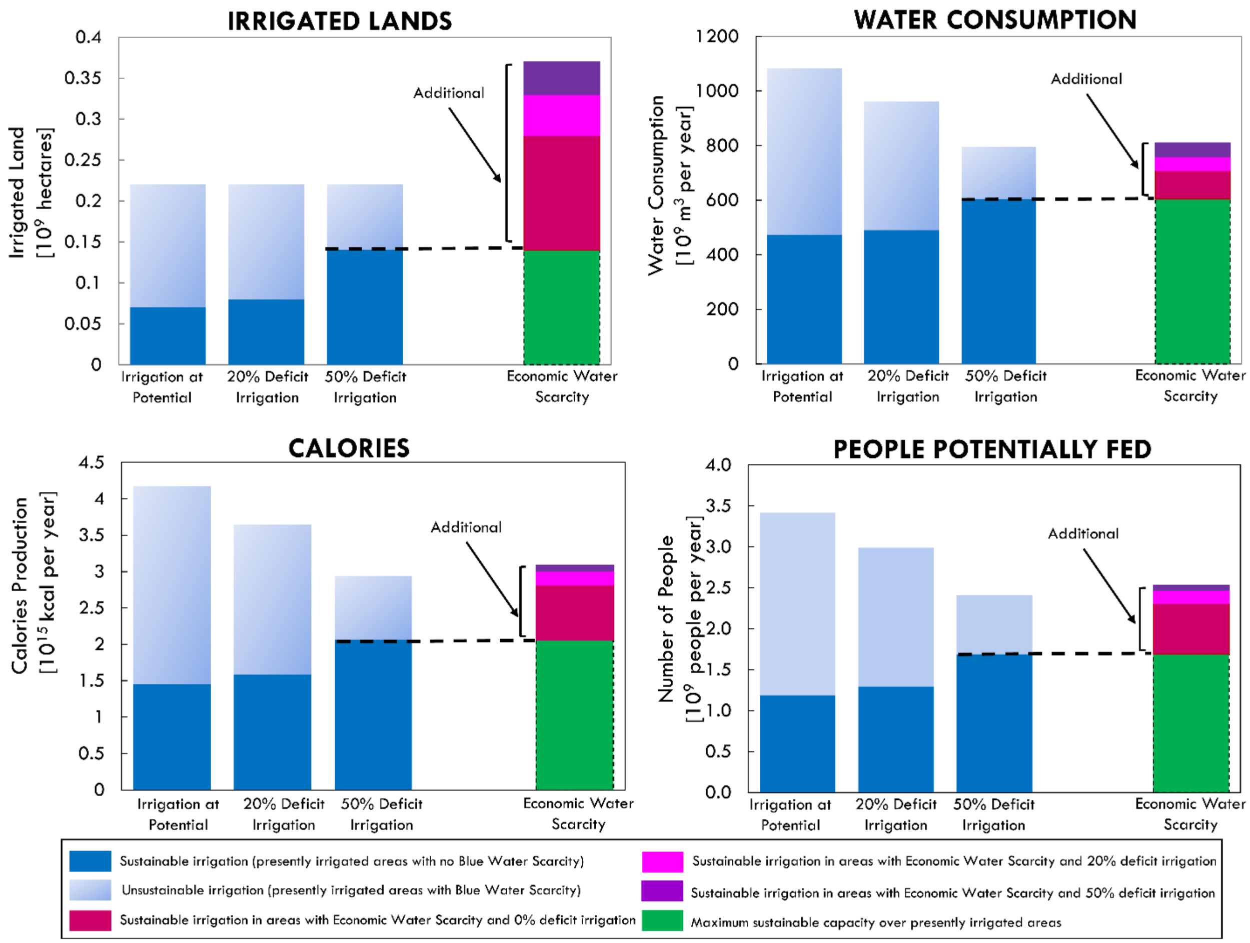

Figure 4. Global irrigated land, blue water consumption, calorie production, and people potentially fed in presently irrigated areas (see the three blue columns to the left), and in croplands facing agricultural economic water scarcity (column to the right). Maximum sustainable capacity over currently irrigated areas (green bars) are obtained in the 50% deficit irrigation scenario. Additional sustainable irrigation can be obtained by expanding irrigation to agricultural economic water scarce rain-fed areas and adopting deficit irrigation in rain-fed croplands affected by blue water scarcity.

We also determined the extent of croplands facing total water scarcity (Figure 1; Box 1). In these rain-fed croplands, irrigation expansion would be unsustainable (i.e., it contributes to groundwater depletion or loss of environmental flows) even in the two deficit irrigation scenarios discussed above. Depending on such scenarios, we find that 28-38% of global croplands are exposed to total water scarcity (Figure 1). Over these agricultural regions, trade-offs among the opportunity to increase food production through irrigation expansion, the cost of irrigation infrastructure, and the sustainable use of blue water resources should be evaluated.

3.4.3 Regional hotspots of agricultural economic water scarcity

Agricultural economic water scarcity tends to concentrate in low-income countries with large yield gaps, likely because of lack of capacity to invest in the irrigation infrastructure needed to meet crop water requirements using the available renewable blue water resources. Not surprisingly, in both high-income and in arid regions, there are less agricultural economically water scarce croplands where irrigation expansion can be used to increase food production (Figure 5).

Two thirds of agricultural economically water scarce croplands are located in Sub-Saharan Africa, Eastern Europe, and Central Asia (Figure 5). In Sub-Saharan Africa, a region currently sparsely irrigated, irrigation expansion over economically water scarce croplands – combined with the adoption of sustainable deficit irrigation practices – would produce enough food to feed an additional 189-235 million people while requiring an additional 38-61 km^3 of irrigation water (≈24%-96% increase with respect to current irrigation water consumption). In Eastern Europe and Central Asia the expansion of irrigation in regions affected by economic water scarcity – combined with the adoption of sustainable deficit irrigation practices – would produce enough food to feed an additional 317-417 million people using 40-77 km^3 of irrigation water (Figure 5).

Opportunities for irrigation expansion differ dramatically by country. Maximizing crop production by expanding irrigation over economically water scarce croplands would increase by at least one third current total calorie production in nineteen low-income countries. About half of the increase in global calorie production associated with irrigation expansion over economically water scarce croplands would be contributed by only five countries – namely, Nigeria, Ukraine, Russia, Romania, and Kazakhstan – where vast cropland areas are affected by agricultural economic water scarcity. Nigeria, a country with rapid population growth, has the potential to increase food production and feed an additional 87-98 million people by expanding irrigation to agricultural economic water scarce areas. Ukraine, Russia and Romania also have good opportunities to increase food production for an additional 84-119 million, 67-88 million, and 33-39 million people, respectively. With an increase in food production in agricultural economic water scarce lands, net food importing countries, many in Sub-Saharan Africa, could reduce their reliance on international food trade and therefore their exposure to socio-environmental shocks in food supply systems (*36*).

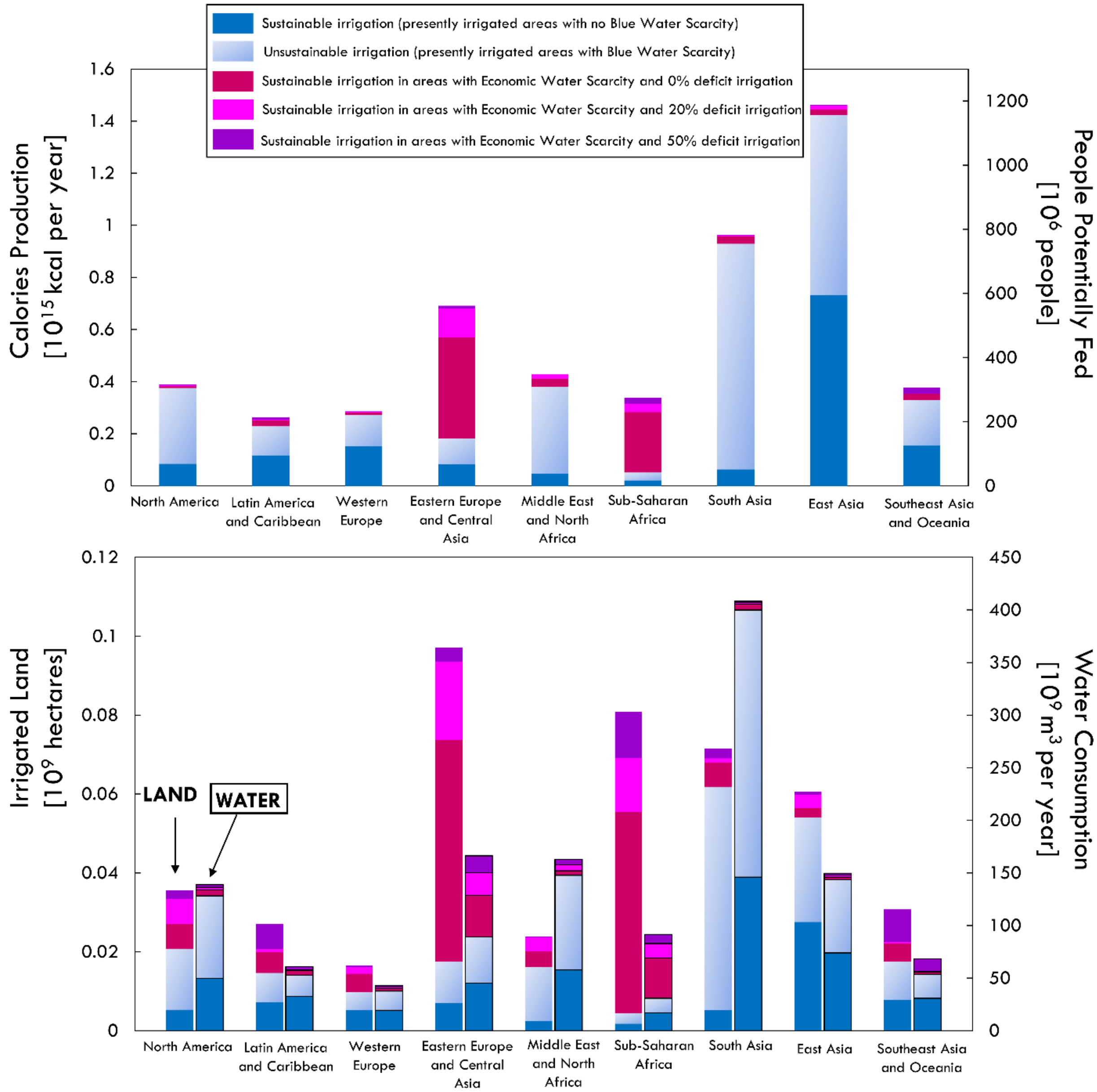

Figure 5. Regional distribution of calorie production, people potentially fed, irrigated land, and blue water consumption over agricultural economic water scarce croplands. The figure shows i) current (sustainable and unsustainable) land, water, and calorie produced in irrigated lands considering irrigation at maximum potential; ii) additional land, water, and calorie that could be sustainably produced in economically water scarce lands also considering deficit irrigation scenarios. Results are represented by considering croplands facing at least one month of blue water scarcity and agricultural economic water scarcity along the year. Note: calories production and people potentially fed are proportional.

3.5 Discussion

Building on previous efforts that assessed green and blue water scarcity (4, 10, *17, 24*), our study maps and quantifies the productivity potential of sustainable irrigation expansion into rain-fed croplands that are economically water scarce. Sustainable irrigation expansion has the potential to increase food production without degrading terrestrial and aquatic habitats by claiming uncultivated land or environmental flows. Sustainable irrigation is also an adaptation strategy to climate change, which creates more reliable and resilient food production than solely rain-fed croplands. Our monthly assessment allows us to estimate also the maximum amount of blue water resources that can be consumed by humanity across the global croplands. We estimate that, while at most 810 $km^3\ yr^{-1}$ of blue water resources can be consumed for sustainable irrigation worldwide, humanity is currently consuming 1,083 $km^3\ yr^{-1}$, thereby overshooting the planetary boundary for water (Figure 4). While, 0.10-0.15 billion hectares of agricultural land are facing unsustainable irrigation for at least one month per year, we find that 0.14-0.23 billion hectares of rain-fed croplands (mostly in Sub-Saharan Africa and Eastern Europe and Central Asia) are suitable for sustainable irrigation but they are not irrigated because of agricultural economic water scarcity.

The use of irrigation to complement green water deficits has boosted agricultural production in many regions worldwide making irrigation a crucial factor in global food security. However, this practice is largely exposed to blue water scarcity. We estimated that 2.23 billion people, corresponding to 22% (2.72×10^{15} kcal y^{-1}) of global food production, rely on unsustainable uses of blue water resources. If current unsustainable irrigation were to be totally eliminated, a combined adoption of sustainable irrigation deficit practices and sustainable irrigation expansion over economically water scarce croplands would contribute to 13% (1.64×10^{15} kcal y^{-1}) of global calorie production, or produce food enough to feed 1.34 billion more people (Figure 4). Because water availability and crop water demand have a large intra-annual variability, the construction of small and sometimes temporary reservoirs built to store excess run-off in the course of the year – could retain enough water to bridge seasonal water deficits. In fact, a previous study, at the annual scale and under the same assumptions, has shown that sustainable irrigation expansion into rain-fed croplands could produce 1.57×10^{15} kcal y^{-1} (or food for about 1.28 billion people) more than the monthly assessment (*4*). This means that in the presence of adequate water storage to mitigate the effects of seasonal blue water scarcity, there would be an increase in food production (3.21×10^{15} kcal y^{-1}), which would be enough to sustainably offset the loss of calorie production in the event unsustainable irrigation practices were eliminated.

Most likely, the construction of local water storages will allow intermediate conditions between these two limit scenarios to be achieved. Of course, the enhancement of agricultural productivity on underperforming croplands is only one of the possible options available to feed humanity while meeting environmental goals. On the consumption side, food waste reduction (*37*), moderating reliance on first generation biofuels, reducing meat consumption, and improving resource use efficiency (*38*) can be adopted to sustainably reduce food demand while improving water and food security without requiring an increase in production (*3*). Moreover, investing in girls' education and expanding people's access to family planning are other valuable strategies that could be adopted to limit population growth and reduce future food demand (*39*).

3.5.1 Opportunities to ease green water deficits

Nearly half of the economically water scarce croplands are exposed to green water scarcity for only one month a year. In these areas investments in irrigation might not be justified by the limited increase in crop production that would result from irrigation in such short water deficit periods. Therefore, it is important to consider less costly and environmentally more suitable "soft" approaches to reduce crops' exposure to water stress (*40*). These approaches are nature-based solutions that allow for a sustainable intensification of agriculture in target areas, while maximizing climate resilience and minimizing resource demands and environmental impacts (*41*). For instance, it is possible to retain more green water in the soil by reducing soil evaporation with appropriate low-cost land and water management options (*25, 42*). Contour stone-bund, pitting, and terracing are indigenous farming techniques that increase soil moisture by enhancing infiltration rates and reducing surface runoff (*25, 42*). Mulching and no-till farming can also improve infiltration of precipitation in the soil and reduce evaporation by lowering soil temperature due to shading (*43*). Agro-forestry and agrivoltaics – combing agriculture with forestry or solar panels – can decrease croplands exposure to sunlight and therefore reduce evaporation rates while increasing productivity (*39, 44*). Replacing water-intensive crops with less water consuming crops can as well reduce exposure to green water scarcity (*45*). The removal of weeds can further reduce non-productive green water consumption (*42*). The implementation of these approaches could provide enough rainwater to bridge a month-long green water scarcity.

For longer green water deficits, however, irrigation infrastructure is necessary to enhance crop productivity. In areas affected by only short periods of green water scarcity, the construction of small, decentralized water harvesting and storage facilities is often seen as an economically more viable option than the construction of large dams and centralized irrigation systems (*46*). In fact, collecting run-off in small human-made storage systems such as ponds and tanks, and in natural storage systems (e.g., managed aquifer recharge) can effectively alleviate green water deficits (*47*). Moreover, these solutions are more likely to serve small-scale farmers in economically water scarce lands by reducing the capital and operational costs of storage with respect to large centralized irrigation systems (*48*).

3.6 Conclusions

With continuing growth in food demand and limited potential for cropland expansion, sustainable irrigation becomes an increasingly important strategy to ensure a reliable and resilient global supply of food in a changing climate. This study maps global agricultural economic water scarcity at unprecedented spatial and temporal resolution. We determine agricultural economic water scarce lands where investments in sustainable irrigation have the possibility to increase food production by expanding irrigation over currently rain-fed croplands. We find that 22% global calorie production happens under conditions of blue water scarcity. While irrigation currently consumes 1,083 $km^3\ yr^{-1}$ of blue water resources, we estimate that only 810 $km^3\ yr^{-1}$ of blue water resources can be consumed sustainably by the global croplands. We estimate that cultivated lands affected by agricultural economic water scarcity account for 15%-25% of the global croplands and could be irrigated sustainably contributing to future food security. A sustainable irrigation expansion into these areas could increase global food production by 6-8% and feed an additional 620-840 million people, while avoiding agricultural expansion into natural ecosystems. The findings of this study show that wise agricultural governance and interventions have the potential to contribute to global food and water security without negatively impacting natural ecosystems. Investigating and explaining the nexus, interlinkages and tradeoffs between environmental sustainability and human wellbeing is fundamental to orientate rural development towards a more sustainable trajectory.

3.7 References

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CHAPTER 4

Hydrological limits to carbon capture and storage

Reference: Rosa, L., Reimer, J. A., Went, M. S., & D'Odorico, P. (2020). Hydrological limits to carbon capture and storage. Nature Sustainability, 1-9.

4.1 Abstract

Carbon capture and storage (CCS) remains a strategy to mitigate climate change by limiting carbon dioxide emissions from point sources such as coal-fired power plants (CFPPs). As decision makers seek to implement policies regarding CCS, the consequences of this added technology on water scarcity have not been fully assessed. Here we simulate the impacts on water resources that would result from retrofitting global CFPPs with four different CCS technologies. We find that 43% of the global CFPP capacity experiences water scarcity at least one month per year and 32% experiences scarcity for five or more months during the year. Although retrofitting CFPPs with CCS would not greatly exacerbate water scarcity, we show that certain geographies lack sufficient water resources to meet the additional water demands of CCS technologies. For CFPPs located in these water scarce areas, trade-offs between the climate change mitigation benefits and the increased pressure on water resources of CCS should be weighed. We conclude that CCS should be preferentially deployed at those facilities that would be least impacted by water scarcity.

4.2 Introduction

Globally, coal-fired power plants (CFPPs) account for 38% of electricity generation[1] and 19% of total carbon dioxide emissions[2]. Coal generation is also a primary source of toxic airborne emissions globally[3]. Despite the growing reliance on renewable energy and the recent policy efforts aimed at reducing the use of coal[4], the global coal dependence for power generation is the same as twenty years ago[1]. Since the turn of the 21st century, population growth, increasing affluence, and industrialization in developing countries have demanded an unprecedented growth in coal consumption (+57%)[1], leading to a boom in the construction of CFPPs[2]. Given that each new coal plant is at least a billion-dollar investment with a 30- to 50-year lifetime[5], currently operating CFPPs commit the energy sector to emissions above the levels compatible with 1.5-2° C limit on global temperature rise[6] and commit fresh water consumption to levels that potentially compete with natural ecosystems and other human uses[7-21]. These commitments compel increasing attention to global water scarcity[22] in the context of humankind's ability to meet its burgeoning food and energy needs[23].

A successful solution towards mitigating climate change will curtail CO_2 emissions and minimize unnecessary use of water resources in managed energy systems with minimum costs. Although renewable energy and other technologies that replace coal are necessary and increasingly viable, a portfolio of climate solutions must account for the existing assets and committed billion-dollar investments in coal[24,25]. Post-combustion carbon capture and storage (hereunder CCS) is a preferred, economically viable technology to reduce CFPP carbon emissions because it can be retrofitted to existing power plants without decommissioning them[26]. To date, however, a global assessment of the potential impacts of CCS on water resources – should the CFPPs existing around the world be retrofitted with CCS technologies – is missing. As we continue to evaluate the cost-effectiveness of different climate change mitigation technologies, the assessment of potential water limits to CCS can provide relevant and necessary insights.

We consider four prominent CCS technologies that can be deployed to retrofit CFPPs: absorption with amine solvents, membrane separation, and adsorption using solid sorbents with either pressure swing (PSA) or temperature swing (TSA) processes. While amine-based absorption is proven and commercially available, membrane-based and adsorption-based CCS systems are at lower stages of development[27]. All of these CO_2 capture technologies are energy-intensive processes[28] that would impose parasitic power demands on existing power plants and thus decrease their efficiencies[27]. The additional power generation required for CCS would result in additional water consumption from the CFPP cooling process[29]. Moreover, in most cases additional water is required as an integral part of the carbon capture processes[30]. Recent work has assessed that a post-combustion amine absorption process would nearly double a CFPP's water intensity, decrease net plant efficiency from 38% to 26%, and increase levelized cost of electricity by 75%[31].

Previous research has simulated water risks of power generation with CCS in the United States[32-35], Europe[36], and the UK[37]. These studies focused on regional-scale analyses of water requirements from the absorption process without considering other CCS technologies, however, and did not utilize a monthly hydrological model to quantify potential impacts on water resources. These studies fall short of elucidating whether CCS might induce or exacerbate water scarcity at specified times of the year, and what the different water intensity impacts are for the various CCS technologies. A limited hydrological understanding of the potential impacts of CCS adds uncertainties to the environmental consequences of the implementation of CCS worldwide.

Herein we present a global hydrological analysis of the potential impacts on water resources that would result from retrofitting large (> 100 MW of gross capacity) CFPPs with four CCS capture systems: amine absorption, pressure swing adsorption (PSA), temperature swing adsorption (TSA), and membrane separation. This analysis begins with a monthly, regional assessment of water scarcity experienced by current CFPPs. We assess the monthly water withdrawal and consumption for each CFPP using the Integrated Environmental Control Model (IECM Version 11.2)[38] and analyze its exposure to water scarcity. A comprehensive assessment of water withdrawals, consumption, and scarcity facilitates the development of sustainable water management practices and sheds light on regional hydrologic impacts of CCS. Our study promotes the understanding of the water requirements of CCS and provides relevant insights to mitigate carbon emissions from the electricity and industry sectors while preserving water resources.

Box 1a | Concepts and definitions about water systems.

Water Consumption is the volume of water that is used by human activities and returned to the atmosphere as water vapor. Therefore, this water becomes unavailable for short-term reuse within the same watershed.
Water Withdrawal is the total volume of water removed from a water body. This water is partly consumed and partly returned to the source or other water bodies, where it is available for future uses.
Water Consumption Intensity (m^3/MWh) is the volume of water consumed (m^3) per unit of net power produced (MWh). It is a measure of efficiency of water consumption.

Water Withdrawal Intensity (m^3/MWh) is the volume of water withdrawn (m^3) per unit of net power produced (MWh). It is a measure of efficiency of water withdrawal.
Blue Water Flows are freshwater flows associated with both surface and groundwater runoff.
Environmental Flows describe the quantity, timing, and quality of water flows required to sustain freshwater ecosystems.
Available Water is the water sustainably available for human uses. It is calculated as *Blue water flows* minus *Environmental Flows.*
Water Scarcity refers to the condition of imbalance between freshwater availability and demand. Here we define water scarcity based on whether the ratio between *Freshwater Consumption* and *Available Water* is greater than one[22]. Water scarcity corresponds to conditions in which the monthly available water resources are less than total water consumption, and freshwater requirements from coal-fired generation must therefore compete with water uses for domestic and irrigation needs, as well as environmental flow requirements.

Box 1b | Concepts and definitions about post-combustion carbon capture and storage technologies.

Post-Combustion Carbon Capture and Storage (CCS) consists of retrofitting existing power plants with carbon capture and storage units without having to modify the power plant itself. CO_2 is first separated from the flue gas of power plants. Once captured, CO_2 is compressed to its supercritical state and transported and injected into a safe geological formation.
Absorption is a CCS technology based on using a liquid solvent to dissolve (absorb) CO_2 molecules into a liquid solution such as an aqueous amine. The CO_2-enriched liquid solution is pumped to a regenerator where it is heated to liberate a stream of almost pure gaseous CO_2 and the lean solution is circulated back to the absorber.
Membrane Separation is a CCS technology that separates CO_2 from flue gas by its selective permeation through a membrane material. CO_2 permeates the membrane if its partial pressure is higher on one side of the membrane relative to the other side, which is accomplished by compression and/or vacuum.
Adsorption is a CCS technology based on adsorption of CO_2 molecules onto the surface of a solid material. The CO_2-enriched solid sorbent is regenerated using low pressure (***Pressure Swing Adsorption (PSA)***) or high temperature (***Temperature Swing Adsorption (TSA)***. Gaseous CO_2 is liberated and collected and can be compressed for storage and the lean solid sorbent is reused to capture CO_2.

4.3 Methods

This analysis begins with the identification (through aerial imagery) of cooling types (wet cooling tower, air-cooled condenser, and once-through systems) and the water source used as a cooling medium (seawater or freshwater) by 1,888 global CFPPs. We then run the IECM Model using the ‘Baseline Power Plant Configuration’, and considering power-plant specific monthly air temperature, cooling type, and gross power inputs, we assessed water consumption and water withdrawal intensities for each CFPP under each scenario. Third, for each CFPP and scenario we assessed its monthly water consumption and withdrawal. Finally, for each scenario, we assessed water scarcity by accounting for water consumption from CFPPs. A detailed description of the methods used in this study is presented in the following sections.

4.3.1 Global coal-fired plant database

Global Coal Plant Tracker (update as of July 2018)[48] provides an inventory of all the coal-fired plants with a capacity greater than 30 MW existing around the world. It reports information about location, status, capacity, operating company, plant name, and year of construction of global coal-fired units with a total global estimated operating capacity of 2,003 GW (as of July 2018). The status is classified as "announced", "pre-permit", "permitted", "in construction", "shelved", "cancelled", "operating", "mothballed", or "retired".

Here, we focus only on "operating" coal-fired units with a capacity greater than 100 MW, assuming that investments in CCS retrofitting would not be justified in the case of smaller units. Multiple units belonging to the same CFPPs were aggregated into a single power plant. The operating large CFPPs that meet the above criteria account for 1927 GW or 96% of total estimated operating capacity from coal-fired plants worldwide[48]. For all these CFPPs, we used aerial imageries from Google Earth® to identify cooling types (wet cooling tower, air-cooled condenser, and once-through systems) and the water source used as a cooling medium (seawater or freshwater). Determining cooling technology and cooling water source of CFPP by visual inspection using aerial images has been proved an effective way to fill gaps existing in available data on power plant cooling systems[16,49]. Wet cooling tower systems are equipped with cooling towers, air-cooled condenser are equipped with air-cooling islands, and once-through cooling systems do not have such cooling systems and are located close to large water bodies. Visual inspection results were also cross-checked when possible with information provided by the operating company listed in the Global Coal Plant Tracker[48].

4.3.2 Assessing water intensities of CFPP with and without CCS

We assessed water consumption intensity and water withdrawal intensity (m^3/MWh) from CFPPs using the Baseline Power Plant configuration of the Integrated Environmental Control Model (IECM Version 11.2) developed by Carnegie Mellon University for the U.S. Department of Energy's National Energy Technology Laboratory (USDOE/NETL)[38]. The IECM Model is a well-documented publicly available model that provides systematic estimates of performance and emissions for fossil-fueled power plants with or without CCS systems[29,38]. Water intensities in the IECM Model do account for the parasitic energy demand of the CCS process. Therefore, the Baseline Power Plant configuration in the model assumes that the additional power required to perform CCS is taken at the expenses of the plant efficiency and therefore less heat and power would be generated. Moreover, the Baseline Power Plant configuration in the IECM Model does consider that each CFPP is retrofitted with environmental control systems (selective catalytic reduction, electrostatic precipitator, and wet flue gas desulfurization). We considered the water use by these environmental control systems both in the scenarios with and without CCS.

For each coal-fired unit, water intensity was assessed considering 1) a current, and 2) four hypothetic future scenarios. In the current scenario, we assessed water intensity of each coal-fired unit considering its cooling system (wet cooling tower, air-cooled condenser, and once-through). In the future scenario we assumed that only CFPPs operating after year 2000 (1,093 CFPPs or 1018 GW) will be retrofitted with CCS units considering four different CCS technologies: absorption with amine solvents, membrane separation, and adsorption with pressure swing (PSA) and temperature swing (TSA) capture systems. For each scenario and for each unit we assessed water intensity considering local average monthly air temperature and its gross power input. Average monthly temperatures at 5 × 5 arcminute resolution were taken from Fick et al., (2017)[50]. Coal type (anthracite, lignite, bituminous, sub-bituminous), combustion technology (supercritical, sub-critical, ultra-supercritical), plant efficiency, plant size, environmental control systems (selective catalytic reduction, electrostatic precipitator, and wet flue gas desulfurization for removing nitrogen oxides, fly ash, and sulfur dioxide, respectively, from the flue gas), and CO_2 capture level are other factors that influence water intensity of a CFPP[33]. Because the Global Coal Plant Tracker database used in this study does not contain detailed information about these factors, we tested the sensitivity of our results to ±20% changes in monthly water consumption in each CFPP.

For each CFPP we assessed monthly water consumption and water withdrawals (m^3/month) by multiplying its monthly water intensity (m^3/MWh) times the coal-fired unit capacity by a 50% capacity factor and the number of hours in each month. The 50% capacity factor is a conservative assumption given that the global average capacity factor of coal-fired plants was 52.5% in year 2016 (ref. 13), and also considering that we are experiencing a reduction in coal use owing to natural gas conversion[51,52].

4.3.3 Water scarcity analysis

Monthly water scarcity (5 × 5 arcminute resolution) was assessed combining the monthly availability and consumption of freshwater resources. CFPPs are located in water scarce areas if the ratio between freshwater consumption (WC) and available water (WA) is greater than one[22].

$$WS = \frac{WC}{WA} > 1$$

This methodology to evaluate water scarcity has been extensively validated in studies aiming at analyzing the influence of energy and agricultural production on water resources[22,42-44, 53]. WC accounts for freshwater consumption for irrigation, domestic uses, and CFPPs. For this reason, CFPPs cooled with seawater were not considered in the water scarcity analysis, because they do not consume freshwater in their operations. Monthly available water (WA) (5 × 5 arcminute resolution, or ~10km at the Equator) was calculated as the difference between monthly blue water flows generated in that grid cell and the environmental flow requirement. Monthly blue water flows (2011-2015 period) were assessed by adding up for every cell routed river discharge and groundwater discharge. Discharge data were taken from PCR-GLOBWB-2 outputs[54,55]. Upstream water consumption and its unavailability for downstream uses were accounted for by considering - for every cell of the landscape - all water uses (agriculture, industrial, municipal, and environmental flows). Irrigation water consumption (at 5 × 5 arcminute resolution) was taken from Rosa et al. (2019)[44] and was assessed using a process-based crop water model that estimated irrigation water consumption for major crops. Domestic water consumption (at 5 × 5 arcminute resolution) was taken from Hoekstra and Mekonnen (2012)[56] and assessed using country-specific per capita values multiplied by the local population taken from population density maps. We assumed that CFPPs cooled with seawater face no water scarcity and only land-based water plants are at risk of water scarcity. Because the irrigation water consumption dataset used[44] was generated for a five-year time period, we here used the same five years of discharge data[54,55] to assess inter-annual variability of water scarcity. While this time period might be too short to capture a full range of extreme wet and dry periods, our results are robust and show little sensitivity to different environmental flow requirements, which are by far the largest factor affecting our results.

Environmental flow is here defined as the minimum freshwater flow that is required to sustain ecosystem functions. Environmental flow requirements were accounted for in our water scarcity analysis, assuming that 80% of the monthly blue water flows should be preserved for environmental flows protection (i.e., remain unavailable to human consumption) to maintain ecosystem functions[57]. We tested the sensitivity of our results to the less conservative Variable Monthly Flow (VMF) method[47], which account for intra-annual variability in discharge by classifying flow regimes into high-, intermediate-, and low-flow months.

4.3.4 Caveats

Even though a 100% adoption of CCS technology is not a realistic scenario of CCS adoption, this assumption allows us to assess the impacts of CCS retrofit on water resources. Moreover, this assumption is in line with the urgent need to drastically reduce global CO2 emissions from CFPPs in order to meet climate targets[39]. The goal of our study is to determine the water requirements and the exposure to water scarcity of CFPPs with and without CCS. We are not trying to determine future likely CCS adoption scenarios. The research question we want to answer is: Are there enough water resources for a massive adoption of CCS to curb emissions from coal fired power plants? Thus, our analysis is conservative because we now consider that all the coal plants (built after year 2000) will be retrofitted with CCS. Of course, the adoption of less “aggressive” socio-economic pathways can lead to different scenarios of CCS application to the electricity sector. A partial adoption of CCS technology would entail a lower pressure on the water system. We also stress that in this study we consider four different scenarios of CCS technologies (amine, membrane, solid sorbents PSA & TSA). These CCS scenarios are meant to be illustrative, rather than representative of future capacity expansion and CCS deployment.

Our results are based on a biophysical model and on assumptions that are always necessary in any global modelling study. First, decisions to retrofit existing plants with CCS are complicated and involve many factors such as plant age and size, economic viability, land restraints, and location close to geological formations suitable for carbon storage. The analysis of these factors falls outside of the scope of this work. We also do not consider the potential impacts that carbon dioxide storage could have on regional groundwater quality and therefore water availability[58,59]. Second, we assumed that current power plants cooled with seawater will also withdraw and consume seawater (in the same proportion) when retrofitted with CCS. Third, while our water balance model considers water consumption and accounts for the need to protect environmental flows that are crucial to the health of freshwater ecosystems, it does not evaluate other environmental and economic impacts associated with water withdrawals from coal-fired plants, which involve local effects that a global analysis fails to assess. Moreover, quantifying water scarcity using water withdrawals might overestimate water scarcity since return flows can be used multiple times. For example, water withdrawals in the Colorado River Basin exceed water availability because of substantial reuse of return flows. Therefore, we assessed water scarcity using water consumption. Fourth, because hybrid-cooling technology (wet cooling paired with air-cooling) is a relatively new technology, we did not consider this cooling technology in our analysis. Fifth, power plants located in water scarce areas are unlikely to remain water stranded in the sense that they are expected to continue their operation in months of water scarcity by sourcing water through inter-basin water transfers, artificial reservoirs, mining non-renewable groundwater, building desalination plants, or using water at the expenses of environmental flows. Alternatively, water stranding can be avoided by lowering power production or by retrofitting coal-fired plants with emerging technologies that have lower water intensity (e.g. air-cooled systems)[16], although, at the expense of increased energy consumption and economic costs[60,61]. Furthermore, there are also opportunities to use desalinated brine from saline carbon dioxide sequestration aquifers to provide alternative freshwater sources and offset the additional water requirements of CCS[34]. These are economic, institutional, and non-biophysical factors that our hydrological model were unable to take into account. Moreover, energy corporations can prevent a shut-down (and associated losses) during periods of water scarcity by buying water from other sectors (typically agriculture, in the presence of tradeable water rights) and paying more attention to water as a risk for their business operations[46]. Today, the reliability of coal-fired generators is quite high in the sense that they rarely experience power losses associated with water availability limitations[15,62]. Curtailments or shutdowns during dry periods are seldom due to constraints in water availability but to the ability to cool down water when its temperature exceeds environmental regulatory thresholds for discharge in water bodies[62,63]. Increased water temperatures have led to curtailments in power generation worldwide[12,17]. Future improvements in the assessment of the vulnerability of CCS can possibly be achieved by accounting for water temperatures as a constraint to CCS adoption.

Lastly, our analysis considers the possibility to retrofit global coal-fired power plants with post-combustion carbon capture and storage technologies. However, post-combustion carbon capture and storage is an emerging technology not just for coal-fired generation, but also for other industrial[64] and energy CO_2 sources[65,66]. Other technologies also could be deployed to capture carbon such as pre-combustion and oxy-combustion[27,67]. Another promising technology is to remove carbon dioxide from the atmosphere and generate negative emissions via Bioenergy with Carbon Capture and Storage (BECCS)[68] or Direct Air Capture (DAC)[69].

4.4 Results

4.4.1 Current water scarcity without CCS

Global hydrological models are powerful tools to simulate and quantify changes in water availability and consumption. Here we use water scarcity as an indicator of where, in what period of the year, and for how long CFPPs without CCS systems are vulnerable to risks of limited water availability. Our hydrological analysis uses a monthly biophysical water balance model that accounts for water consumption for irrigation, domestic, and coal-fired power generation needs, as well as for environmental flows required to maintain the health of aquatic ecosystems. Our water scarcity results are displayed considering long-term monthly average available water in the 2011-2015 period, although we have also analyzed inter-annual variability in water resources.

We find that 32% (625 GW) of CFPPs exhibit water scarcity for five or more months per year and 43% (830 GW) of the world's CFPPs face regional water scarcity at least one month per year. Of these 625 GW, 56% are located in China, 15% in India, and 11% in the United States. Other CFPPs facing water scarcity for at least five months per year are located in South Africa (34 GW), Australia (12 GW), Russia, (8 GW), Poland (8 GW), and Germany (7 GW).

Figure 1 shows the geographical distribution, water scarcity duration (in number of months), and cooling technology of CFPPs operating in 2018. CFPPs are typically built adjacent to lakes, rivers, or oceans where water availability is abundant. Year-round CFPPs that do not face water scarcity are located in the Great Lakes region in the North-Eastern United States, Europe, Russia, and South China. Other CFPPs not affected by water scarcity are located along the coasts as they use seawater as a cooling medium (we assumed that CFPPs currently cooled with seawater are not affected by water scarcity).

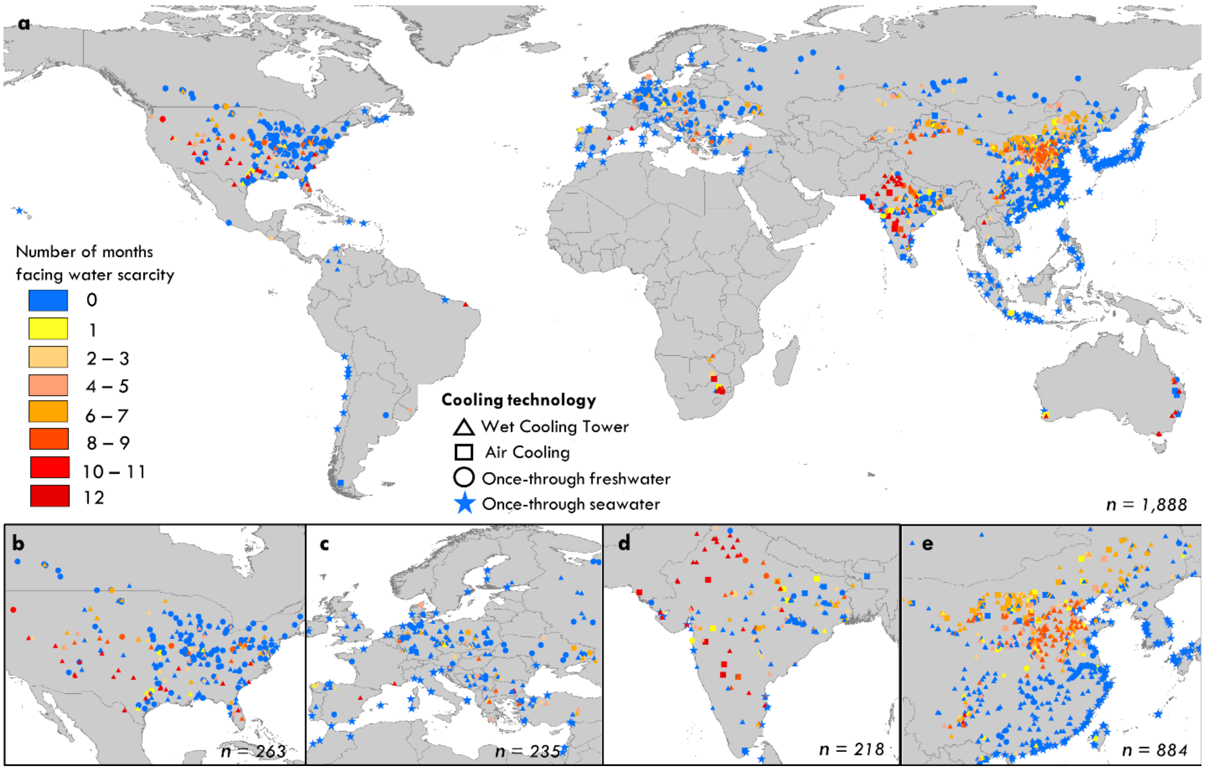

Figure 1. Geospatial distribution of coal-fired plants facing water scarcity in the 2011-2015 period. Detail **(a)** shows location, number of months per year facing water scarcity, and cooling technology of 1,888 coal-fired plants (*n*) worldwide. Details (**b-e)** show the four main regions where CFPPs are located (United States, Europe, India, and China). CFPPs facing water scarcity appear either in intensively irrigated areas (for example, High Plains in the United States), in high population density regions (Pretoria, Johannesburg conurbations), or in irrigated and populated areas (North China Plain, India). Water scarcity also occurs in arid regions with a well-defined dry season (Western United States, India, Australia, Xinxiang and Inner Mongolia provinces in China). Generating units with once-through cooling are shown distinguishing seawater and freshwater as a cooling medium.

The analysis of the share of CFPP capacity currently facing water scarcity in different regions of the world and months of the year shows that in China more than 30% of the installed capacity faces water scarcity from March to October (Figure 2a). In the United States, at least 20% of coal capacity faces water scarcity from April to November. A similar picture can be found in Europe, where at least 20% of coal capacity faces water scarcity from June to September. More than 40% of India’s coal capacity faces water scarcity in the dry season

(December to June). CFPPs located in other Asian countries are not particularly exposed to water scarcity because of high water availability and their construction along the coast using seawater as a cooling medium. It is worth noting that for those global CFPPs that use fresh water for cooling, the predominant cooling technologies are wet cooling towers (60% of total capacity), followed by once-through systems (35%), and air-cooling (5%) (Figure 2b). Air-cooling is a relatively new technology and 90% of its capacity is located at new plants in China and India. About 22% of global coal-fired operating capacity is cooled using seawater, while the remaining 78% uses freshwater.

The analysis of the CFPP capacity facing water scarcity by cooling technology shows that 60% (728 GW) of the units cooled with wet cooling towers face water scarcity for at least one month per year. Because of their lower water intensity (Figure 3), air-cooled systems are usually implemented in newly built units located in arid and/or water scarce areas. In fact, we find that 72% (67 GW) of CFPP cooled using air-cooled systems are facing water scarcity. These air-cooled CFPPs are located in regions that are so dry that even the little amount of water they use is depleting environmental flows and groundwater stocks. While 56% (360 GW) of the once-through cooled capacity uses seawater as a cooling medium and therefore is not affected by water scarcity. Only 6% (36 GW) of once-through generating capacity is exposed to water scarcity. China has 62% (403 GW) and 74% (53 GW) of its wet cooled and air-cooled CFPPs, respectively, exposed to at least one month of water scarcity per year (Figure 2b). The United

States and India have 60% (89 GW) and 63% (113 GW) of their wet cooled coal-fired units exposed to water scarcity for at least one month per year.

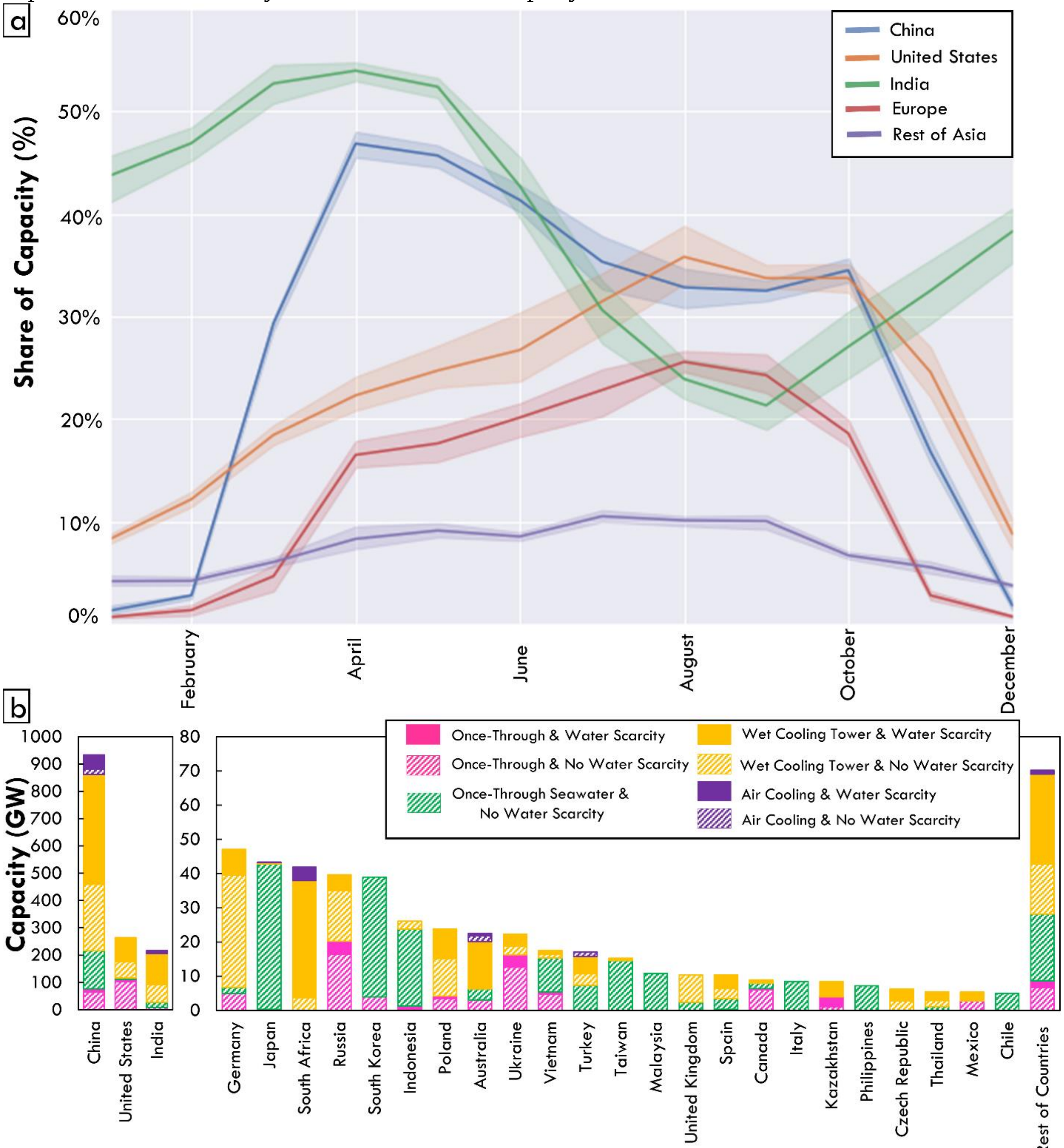

Figure 2. The exposure of coal-fired plants to water scarcity. Panel (a) shows regional share of coal-fired operating capacity facing water scarcity each month of the year. Solid lines represent average water scarcity in the 2011-2015 period, shaded areas show inter-annual variability of water scarcity in the years from 2011 to 2015. Panel (b) shows coal-fired capacity facing average water scarcity (for at least one month per year) by cooling technology. Panel (b) shows the current installed coal-fired capacity and respective cooling systems by country (or region).

4.4.2 Future water scarcity with CCS

Using the water balance approach described above, we turn to an important aspect of future decisions regarding CCS, namely to what extent the available freshwater resources would allow for the adoption of CCS as a means to curb carbon emissions from existing CFPPs. Meeting humanity's burgeoning energy and water demand while avoiding an increase in anthropogenic CO_2 emissions and protecting environmental flows is one of the most pressing challenges of this century.

Given that old, small (less than 100 MW), and low-efficiency CFFPs without environmental control systems will likely be shut down before being retrofitted with expensive CCS technologies, we assumed that only 1,093 large (>100 MW) CFPPs operating since year 2000 will be retrofitted with CCS. We assume that these CFPPs will capture 90%[26] of their CO_2 emissions by 2020. Because of this relatively short timeframe, we assume that water availability and coal-fired generation would not substantially change compared to current conditions. This scenario allows us to establish an upper bound on the potential impacts of CCS retrofit on water resources. Moreover, this assumption is likely a conservative scenario compared with the urgent need to drastically reduce global CO_2 emissions from CFPPs in order to meet climate targets[39]. This analysis provides the estimated additional water withdrawals and consumption from coal-fired generators considering 1) current 1,888 CFPPs, and 2) four hypothetical scenarios where the 1,093 CFPPs built after year 2000 are retrofitted with CCS units.

4.4.3 Water requirements of CCS

Our estimates show that the water intensity of CFPPs with and without CCS technologies strongly vary with the type of cooling system and CCS technology (Figure 3). Interval bars show that water intensity from air-cooling and once-through cooling technologies can differ by up to 4% with different air temperatures, relative humidities, and gross power inputs, while for wet cooling it can vary up to 20%. CFPPs with wet cooling towers retrofitted with CCS units have the highest water consumption intensity, while CFPPs with once-through cooling technology have the highest water withdrawal intensity. Independent of the cooling system, the least water intensive CCS technologies are solid sorbent PSA and membrane systems.

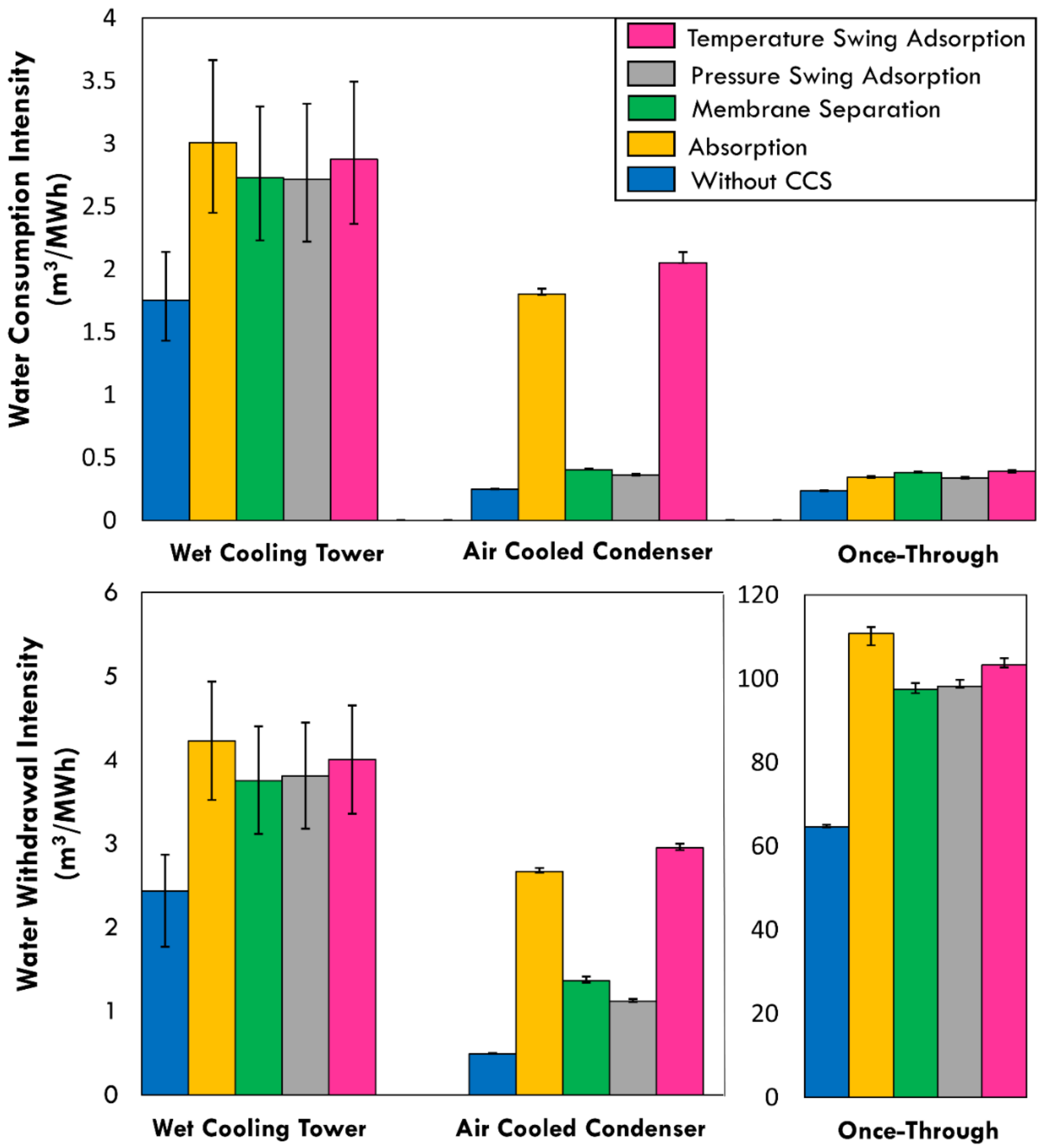

Figure 3. Water consumption and withdrawal intensities of coal-fired plants with and without CCS. The figure was generated running the Integrated Environmental Control Model (IECM Version 11.2)[38] and considering the different monthly air temperatures, relative humidity, and gross power inputs of the 1,888 CFPPs considered in this study. Interval bars represent maximum and minimum values of water intensities. Note that water withdrawal intensity with once-through cooling technology is shown using a different scale. Water intensities are expressed in terms of net power generation.

An analysis of water consumption by CFPPs with and without four different retrofitted-CCS technologies shows a substantial increase in water consumption. Current total global water consumption from CFPP is 9.66 $km^3\ y^{-1}$, of this volume 88% is sourced from freshwater, while the remaining 12% is sourced from seawater (Figure 4). China, with 48% of world's CFPP capacity, has also the greatest share in freshwater consumption (53%), followed by India (16%), and the United States (13%). By retrofitting CFPPs built after year 2000 with the off-the-shelf amine absorption technology[27,40], global water consumption by CFPPs would increase by 50%

(4.81 km^3 y^{-1}). If CFPPs were all retrofitted with membranes, water consumption would increase by 31% (3.00 km^3 y^{-1}). Water consumption would increase by 32% (3.13 km^3 y^{-1}) and 42% (4.07 km^3 y^{-1}) if CFPPs were retrofitted with solid sorbent PSA, and solid sorbent TSA, respectively. Assuming that current CFPPs cooled with seawater will use seawater when retrofitted with CCS, 0.69-1.10 km^3 y^{-1} of this additional water consumption would come from seawater, while the remaining fraction (2.31-3.71 km^3 y^{-1}) would be consumed from freshwater bodies. Similar results can be found in terms of water withdrawals (Figure 4).

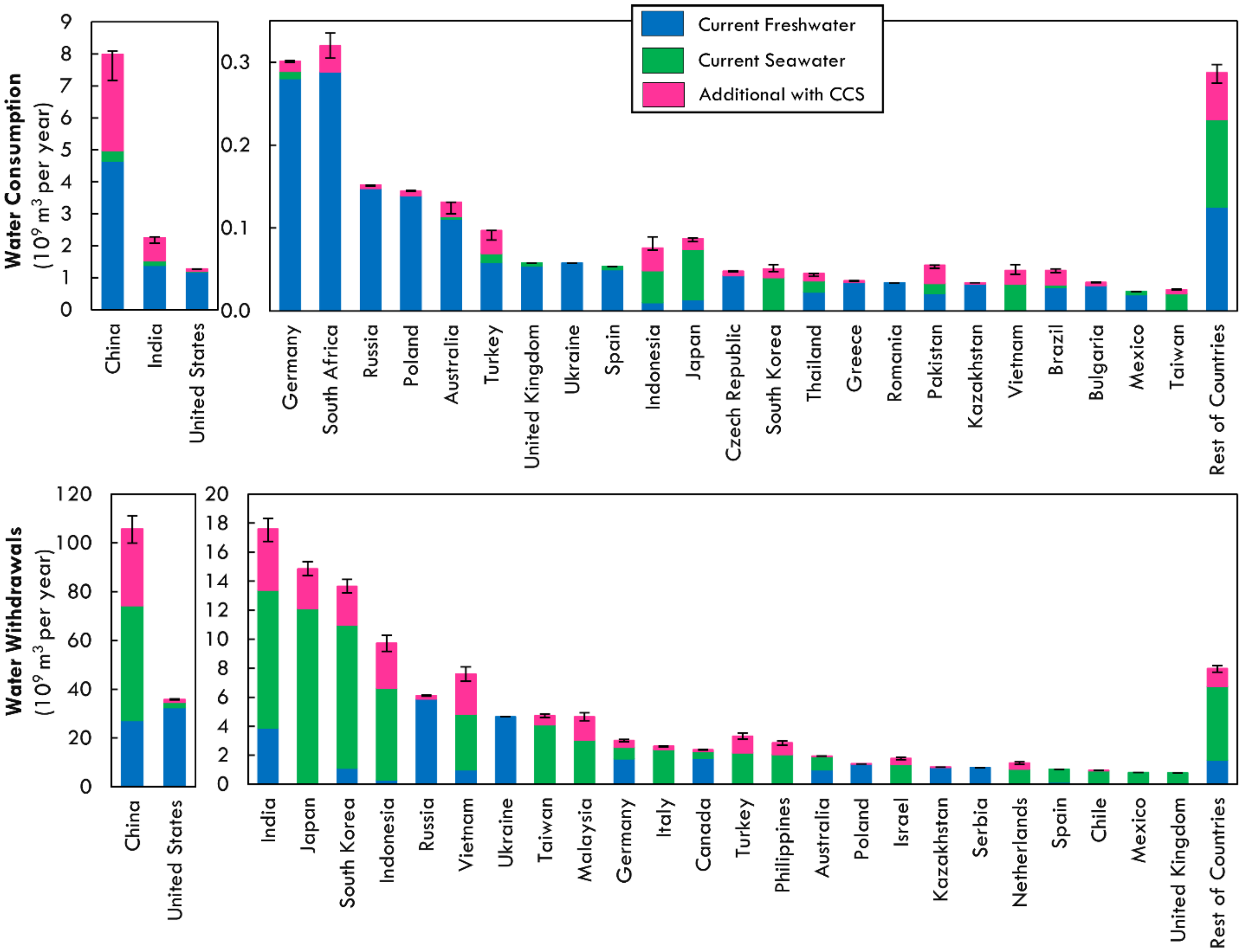

Figure 4. Water consumption and withdrawals of coal-fired plants with and without CCS. Current water consumption and withdrawals from 1,888 CFPPs are differentiated between freshwater and seawater. Additional water consumption and withdrawals from the 1,093 CFPPs (in operation since year 2000) include both freshwater and seawater. Note that countries (or regions) are listed in descending order of current water consumption and withdrawals by CFPPs. Interval bars represent the maximum and minimum values of water consumption and withdrawals (seawater and freshwater combined) considering the four CCS scenarios assumed in this study. Current water withdrawals from CFPPs total 204 km^3 y^{-1}. Of this volume, 43% is sourced from freshwater, while the remaining 57% is sourced from seawater.

4.4.4 Exposure to water scarcity with CCS

Retrofitting CFPPs with CCS units would create or exacerbate water scarcity conditions compared to current operations. Amine absorption and solid sorbents TSA technologies would most significantly impact water resources. By retrofitting CFPPs built after year 2000 with these two technologies, an additional 13 GW (1%) of CFPP capacity would face water scarcity. Moreover, an additional 23% (232 GW) of CFPP capacity would be exposed to water scarcity for at least one additional month a year (Figure 5). Because of their lower water intensities, membranes and solid sorbents PSA would increase water scarcity for only 18% and 20% of CFPP capacity, respectively. If CFPPs in China and India were retrofitted with the commercially available amine absorption technology, an additional 168 GW and 52 GW of coal fired capacity would be exposed to longer periods of water scarcity every year (Figure 5b). In other words, in China and India 23% and 37% of CFPPs built after year 2000, respectively, would be vulnerable to longer periods of water scarcity with CCS installed.

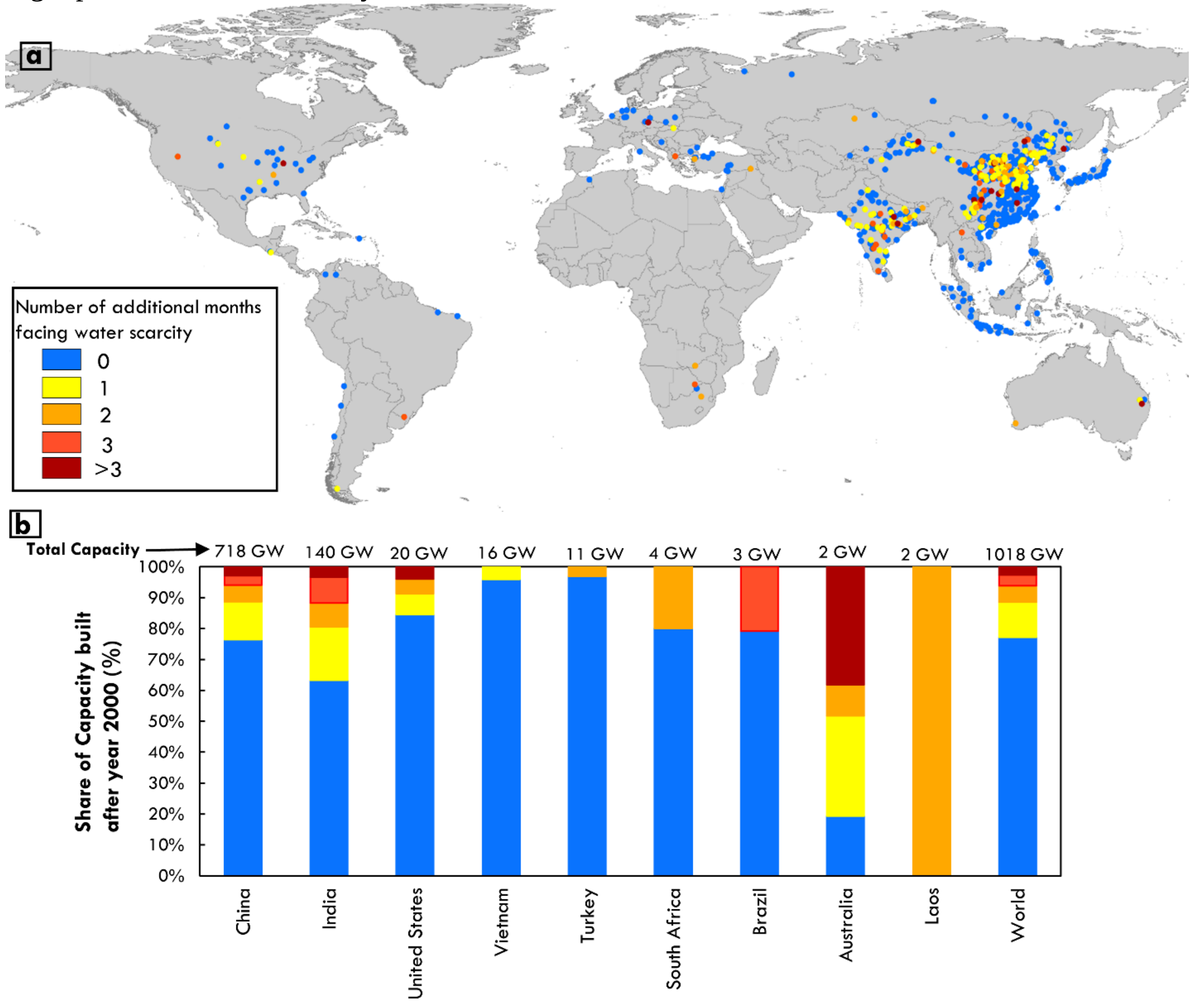

Figure 5. Additional water scarcity with carbon capture amine absorption technology. The figure shows the number of additional months of water scarcity per year that CFPPs built after year 2000 would face in the event they were retrofitted with the commercially available amine absorption technology. Detail **(a)** shows the geographical distribution of CFPPs built after year

2000 and the number of months of additional water scarcity they would face if retrofitted with amine absorption, **(b)** shows country-specific share of coal fired capacity built after year 2000 that would face additional months of water scarcity if retrofitted with amine absorption. Countries are listed in descending order based on additional capacity facing water scarcity.

4.4.5 Tradeoffs between climate mitigation and water resources

This study highlights the water impacts of coal-fired power generation and the potential water scarcity that would result from the adoption of CCS to address the associated CO_2 emissions. Our results show that cooling systems and CCS technologies have different water requirements, in terms of both consumption and withdrawal. For CFPPs located in water scarce areas, the additional water consumption that would be required by CCS (Figure 4) could create a competition for local water resources with other human activities[41,42] and/or generate unsustainable water consumption at the expenses of aquatic ecosystems and freshwater stocks[43,44]. Therefore, the choice of cooling and CCS technologies is fundamental to avoid a competition for freshwater with other local human activities and ecosystem health, and at the same time reduce water consumption. Worldwide the additional water requirements of CCS are dwarfed by freshwater demand from irrigation in the agriculture sector. Modest improvements in the efficiency of irrigation would free up enough freshwater for aquatic habitats and other human uses such as CCS.

The finding that 32% of CFPPs are exposed to water scarcity for at least five months per year shows that these coal-generating units might not be well-suited for retrofitting with CCS if alternative water sources are not implemented. If CFPPs were to be retrofitted with CCS, it would mainly take place in India and China (Figure 5), where 80% (858 GW) of global CFPPs capacity has been built after year 2000 and where 309 additional GW are planned or under construction[25]. We find, however, that in these two countries already a vast share of CFPPs capacity is exposed to water scarcity, and the addition of CCS would further exacerbate the vulnerability to water scarcity and potentially even strand CCS operations. Decision makers, energy corporations, and investors will have to consider the tradeoffs between the climate change mitigation benefits of CCS and the increased demand it places on local water scarce resources.

4.5 Discussion

Constraints on water availability already influence the location of power plants planned for the near future and the choice of cooling technologies. In China, the need to adapt to growing water scarcity has resulted in fewer water intensive cooling systems in new power plants and the refurbishment of existing ones[16, 45]. Investors are also becoming increasingly concerned with the effects of water scarcity. For instance, because wind and solar power production require less water than once-through coal-fired plants, UBS, a global leading investment firm, is recommending its investors to buy low water intensive wind power assets and sell coal-fired assets to avoid exposure to risks associated with water scarcity[46]. Moreover, energy corporations and investors should pay more attention to water as a risk for their business operations when they plan for investments in coal-fired power plants. As such, our findings have important implications for future investments in the global coal power sector.

We tested the sensitivity of our results to different environmental flow requirements, which are by far the largest factor affecting our results. With the current assumption that 80% of the available water needs to be allocated to environmental flows, we find that 43% and 32% of global CFPP capacity faces water scarcity for at least one and five months per year, respectively.

By adopting the less conservative Variable Monthly Flow (VMF) method[47], the fraction of CFPPs capacity facing at least one and five months of water scarcity decreases to 39% and 23%, respectively.

In attempting a global analysis like the one presented in this study, some approximations need to be made, and data limitations are inevitable. Water consumption of CFPPs can vary up to 20%, depending on coal type, combustion technology, plant efficiency, plant size, and environmental control systems[33]. Because Global Coal Plant Tracker – the dataset containing the CFPPs inventory used in this study – does not provide information on these factors, we tested the sensitivity of our water scarcity analysis by increasing and decreasing monthly water consumption estimates of each CFPP by 20%. We find that our results show little sensitivity to this change in water consumption by CFPP. When we increase water consumption, we find that 44% and 34% of global CFPPs capacity would face water scarcity for one to five months per year, respectively. By reducing monthly water consumption of each CFPP by 20%, we find that 42% or 30% of global CFPP capacity would by exposed to water scarcity for one to five months per year, respectively.

4.6 Concusions

The twin costs of mitigating climate change and competing for water resources are vexing factors in managing energy systems. In an increasingly water scarce and carbon-enriched world, governments will take specific actions targeting CO_2 emissions and water intensive technologies, and investors may want to know whether new environmental policies could reduce viability of coal-fired power generation with CCS systems. Our results enable a more comprehensive understanding of water uses by coal-fired plants and can better inform the management and policy decisions that are critical for a sustainable allocation of water resources in energy production. For coal-fired plants located in water scarce areas, tradeoffs between the climate change mitigation benefits and the increased pressure on water resources of CCS should be weighed. This study shows that the water requirements of CCS technologies should be taken into account while evaluating future CCS scenarios because it is crucial to mitigate emissions from the energy sector without compromising on the sustainable use of water resources. Because refineries, natural gas power plants, steel and concrete factories can also be retrofitted with CCS, the analysis presented in this study can be extended beyond the case of coal-fired power plants.

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CHAPTER 5

The global water footprint of large-scale deployment of carbon capture and storage technologies under stringent climate policy

5.1 Abstract

Carbon capture and sequestration (CCS) is an important technology to reduce fossil CO_2 emissions and remove CO_2 from the atmosphere. Scenarios for CCS deployment consistent with global climate goals involve gigatonne-scale deployment of CCS within the next several decades. CCS technologies typically involve large water consumption during their energy-intensive capture process. Despite potential concerns, the water footprint of large-scale CCS adoption consistent with stringent climate change mitigation has not yet been explored. Here, we quantify the water footprints (m^3 water per tonne CO_2 captured) of four prominent CCS technologies: Post-combustion CCS, Pre-combustion CCS, Direct Air CCS, and Bioenergy with CCS. Depending on technology, the water footprint of CCS ranges from 0.74 to 575 m^3 H_2O/tonne CO_2. Bioenergy with CCS is the technology that has the highest water footprint per tonne CO_2 captured, largely due to the high water requirements during biomass cultivation. The widespread deployment of CCS to meet the 1.5ºC climate target would almost double anthropogenic water footprint. Consequently, this would likely exacerbate and create green and blue water scarcity conditions in many regions worldwide. Climate mitigation scenarios with a diversified portfolio of CCS technologies have lower impacts on water resources than scenarios relying mainly on one of them. We suggest that water footprint assessment of CCS is a crucial factor in evaluating these technologies. Water-scarce regions should prioritize water-efficient CCS technologies in their mitigation goals. We conclude that the most water-efficient way to stabilize the Earth's climate is to rapidly decarbonize our energy systems and improve energy efficiency.

5.2 Introduction

Carbon capture and storage (CCS) is an important technology to reduce CO_2 emissions from electricity and industrial sectors, as well as to remove CO_2 from the atmosphere. Depending on the origin of CO_2, there are different technologies to realize CCS. Emissions pathway scenarios for carbon capture technologies deployment consistent with global climate goals show that it will be required to remove an additional 640-950 billion tonne of CO_2 from the atmosphere by the end of the century in order to stabilize global temperatures at or below 1.5ºC above preindustrial temperatures [1,2]. By removing CO_2 from the atmosphere and decarbonizing energy and industrial systems, CCS is one of the technologies that can play a key role in meeting climate change targets [3]. Since natural climate solutions are not large or fast enough to mitigate climate [4,5], CCS is receiving an increasing interest not only from the scientific community, but also from the international political community and the corporate world. For example, some major corporations are pledging to be carbon neutral and committing to sequester their historical CO_2 emissions in the next few decades [6]. As CCS seems ever more necessary [7], technology developers and policymakers should ensure these approaches reliably sequester CO_2 emissions and minimize unnecessary environmental impacts [8].

The twin challenges of managing climate change and water scarcity cannot be considered independently. For example, recent low carbon energy policies have had the unintended consequence of exacerbating tensions between food and energy systems with increased water requirements for biofuels production [9,10], hydropower generation [11,12], and afforestation for carbon sequestration [13-16]. Water is also becoming an increasingly important issue for low-carbon electricity generation [17-21]. Therefore, water is starting to be considered a major factor that will constrain humanity's ability to meet future societal needs while also managing climate change mitigation [22-23]. The expected adoption of CCS technologies [24,25] generates the need for more detailed information about their water footprints and how they will interplay in the water-energy-food-climate nexus.

CCS systems are energy- and water-intensive technologies that, if adopted, will commit humanity to additional water use, further compelling attention to water scarcity [26]. CCS technologies use water during the cooling process at the power-plant level [27] and require additional water as an integral part to the carbon capture processes [28]. For example, it has been estimated that retrofitting a coal-fired power plant with post-combustion CCS would increase the power-plant water intensity by 55%, while decreasing the net plant efficiency by 45% [29]. Notably, bioenergy with CCS requires water during the carbon capture process at the power-plant level, but also additional water during biomass cultivation via evapotranspiration. Previous studies have assessed the water footprint of direct air CCS [30], bioenergy with CCS [8,31], and post-combustion CCS [26,28,29,32]. We, here, provide more comprehensive and detailed estimates of water footprints (Box 1) from a broad portfolio of carbon capture technologies, considering direct air CCS and bioenergy with CCS in addition to pre-combustion- and post-combustion- CCS technologies.

A successful solution towards mitigating climate change will curtail CO_2 emissions and minimize use of freshwater resources, especially in water-scarce regions. Despite the mounting concerns about global water scarcity, the water requirements of CCS are often overlooked. As we continue to evaluate the cost-effectiveness of different climate change mitigation technologies, the assessment of the water footprints of different CCS technologies can provide relevant insights to inform policy makers about the implication of alternative scenarios.

Here, we give a comprehensive overview of the water footprint (m^3 of fresh water per tonne CO_2 captured) of the four most prominent CCS technologies: (1) post-combustion CCS; (2) pre-combustion CCS; (3) Direct Air Capture Capture and Storage (DACCS); (4) Bioenergy with Carbon Capture and Sequestration (BECCS) (Box 2). Then, using future CCS adoption scenarios consistent with 1.5 and 2ºC climate targets [33], we estimate projected global water consumption associate with carbon dioxide removal by CCS throughout the 21st century.

The assessment of the water footprint of a broad range of CCS technologies can generate well-informed policies aiming to capture CO_2 in the most water-efficient way. This study provides insights into how CCS adoption consistent with 1.5ºC and 2ºC climate policies will influence the water footprint of humanity in the 21st century.

Box 1 | Concepts and definitions about water systems.

WATER CONSUMPTION is the volume of net water extracted. This water is evapotranspired and becomes unavailable for short-term reuse within the same watershed.

WATER WITHDRAWAL is the volume of water abstracted from a water body. This water is partly consumed and partly returned to the source or other water bodies, where it is available for future uses.
WATER FOOTPRINT is the volume of fresh water consumed to produce goods or services during their life cycle [34,35]. Based on the source of the water, the water footprint can be divided in green and blue water footprint.
GREEN WATER: Root-zone soil moisture that is available for uptake by plants. Biomass plantations use green water during the photosynthesis process.
BLUE WATER: Freshwater in surface and groundwater bodies available for human use. All CCS technologies use blue water during the CO_2 capture process at the power-plant level.
GREEN WATER FOOTPRINT refers to water from the unsaturated root zone of the soil profile that is used by plants and soil microorganisms. It is relevant for the assessment of the water footprint of BECCS because of the evapotranspiration of water by biomass feedstock.
BLUE WATER FOOTPRINT refers to water from surface and groundwater bodies, it is relevant for the assessment of the water footprint of DACCS, and pre- and post-combustion CCS because of the evaporation of water at the power plant level during the capture and sequestration process.

Box 2 | Concepts and definitions about carbon capture and storage technologies.

CARBON CAPTURE AND STORAGE (CCS) is the process of trapping carbon dioxide (CO_2) produced by anthropogenic activities and storing it in such a way that it is unable to affect the atmosphere [41,42]. CCS is a critical technology for climate change mitigation, but most of these technologies are commercially immature [3]. CCS technologies typically involve large water consumption during their energy-intensive capture process.

TECHNOLOGY	**TECHNOLOGY READINESS LEVEL** [ref. 25] **(from *1* to *9*; low to high maturity level)**
DIRECT AIR CAPTURE AND STORAGE (DACCS) capture and permanent sequestration of CO_2 directly from the atmosphere [30,36]. Proposed processes entail using solid or liquid sorbents to capture CO_2. DACCS uses blue water during the energy-intensive capture process.	***8***. Small-scale of direct air capture technologies have found niche markets for greenhouses and synthetic fuels [37]. ***7***. Large-scale solid sorbent technologies have been built at demonstration-scale in Squamish, BC, Canada. Only one DACCS project exists, in Iceland [38].
BIOENERGY WITH CARBON CAPTURE AND STORAGE (BECCS) capture and permanent sequestration of biogenic CO_2 during energy conversion from biomass [39], including post-combustion and pre-combustion technologies. BECCS uses blue water during the energy-intensive capture process,	***9.*** CCS from corn ethanol production has been practiced at commercial scale, both for enhanced oil recovery, and permanent geologic storage [39]. ***6-7.*** Several plants are under development to produce transportation fuels from lignocellulosic biomass in or near California, United States [40].

and green water during biomass feedstock cultivation.	
POST-COMBUSTION CARBON CAPTURE AND STORAGE capture and permanent sequestration of CO_2 after the combustion process has taken place [41,42]. This process uses blue water during the energy-intensive capture process.	***8.*** Post-combustion capture and sequestration is practiced at commercial scale at Boundary Dam Power Station in Saskatchewan, Canada. It is not yet in widespread commercial use.
PRE-COMBUSTION CARBON CAPTURE AND STORAGE there are two different processes: *Integrated gasification combined cycle* is a process that converts coal and biomass into syngas, capturing and sequestering CO_2 before the combustion process has taken place. *Oxycombustion* is the process of burning coal and biomass in pure oxygen, capturing and sequestering a pure stream of CO_2 after the combustion process has taken place [41,42]. These processes use blue water during the energy-intensive capture process.	***7.*** Electricity generation via integrated gasification combined cycle with CCS was attempted, but ultimately abandoned, at the Kemper County energy facility in Mississippi, United States.

CARBON SEQUESTRATION EFFICIENCY is the fraction of carbon in the biomass feedstock that is captured and sequestered through a CCS supply chain (**Figure 1**).

5.3 Methods

The production of food, fiber, feed, and energy depends on the uptake and consumption of soil moisture (or green water) supplied by rainfall and freshwater from surface water bodies and aquifers (or blue water) (Box 1). Here we assess the total water consumption from CCS. While pre-combustion CCS, post-combustion CCS, and DACCS use solely blue water in their processes, BECCS uses green water to produce biomass feedstock and then blue water in the capture and sequestration of carbon dioxide at the power plant. In the following section, we describe how we calculated the water footprint of four CCS processes.

5.3.1 Calculation of the water footprint of post-combustion and pre-combustion CCS

We assessed blue water footprints of post-combustion- and pre-combustion- CCS using the Baseline Power Plant configuration of the Integrated Environmental Control Model (IECM Version 11.2) developed by Carnegie Mellon University for the U.S. Department of Energy's National Energy Technology Laboratory (USDOE/NETL) [43]. The IECM Model is a well-documented publicly available engineering model that provides systematic estimates of water uses of coal fired- and natural gas fired- power plants with or without CCS systems. CCS processes are energy-intensive technologies [44] that would impose additional energy demands on existing power plants and thus require additional water for cooling processes. Water footprints vary depending on atmospheric temperature, relative humidity, cooling technology, and power plant capacity [26]. We run the IECM Model generating an ensemble of water footprints considering a range of atmospheric temperatures (from 0º C to 30º C), relative humidity (from 25% to 75%), power plant capacities (from 100 MW to 2500 MW), and cooling technologies (wet-cooling, air-cooling, once-through, and hybrid cooling). We also run the IECM model considering four post-combustion CCS processes (amine absorption, pressure swing adsorption, pressure swing adsorption, and membrane separation) and two pre-combustion CCS processes (oxycombustion and integrated gasification combined cycle) (Box 2).

5.3.2 Calculation of the water footprint of DACCS

Water loss in DACCS processes come from the sorbent-air contacting process [25]. The blue water footprint of DACCS varies in function of temperature, relative humidity, and sorbent molarity [30]. The water footprint was assessed using the definitions and assumptions of Socolow et al., 2011 [45] (Page 40) and considering a range of temperatures (from 0º C to 30º C), relative humidity (from 25% to 75%), and two sorbent molarities (5M and 10M).

5.3.3 Calculation of the water footprint of BECCS

The water footprint of BECCS was assessed considering the water required to produce the biomass feedstock (or green water) and the water use in the carbon dioxide capture process (or blue water). To estimate the water required to produce biomass feedstock, we compiled an inventory of water use efficiencies (gH_2O per gCO_2) of different dedicated feedstock from existing studies (Table 1). Water use efficiency is a measure of the amount of water required by a biomass feedstock to sequester a certain amount of carbon dioxide [46,47]. Water use efficiency is dependent on climate, phenology, latitude, available water [48-51]. Blue water used to capture CO_2 in the combustion process of biomass was assessed using the IECM Model and considering the water footprint of integrated gasification combined cycle.

We consider two technology cases for BECCS: an efficient carbon supply chain, and an inefficient supply chain. Estimates of carbon sequestration efficiency were first estimated by Smith and Torn in 2013 [52] (Figure 1a). They model an indirectly heated biomass integrated gasification combined cycle-CCS facility with relatively little heat integration [53], and assume very high losses of CO_2 during transport and injection [54]. In total, they estimate that 47% of carbon in the biomass feedstock is captured and sequestered in the integrated gasification combined cycle and CCS process [52].

We expect commercial applications of BECCS for power generation to achieve higher carbon sequestration efficiencies. In our efficient scenario, we model a carbon-efficient integrated gasification combined cycle facility with 90% CO_2 capture, and adjust losses during transport and injection to 1.8%. This figure is the low-range estimate of Brandt et al., 2014 [55], a comprehensive review of methane (CH_4) leakage rates. Large-scale CO_2 transportation and injection may incur similar losses to existing CH_4 systems. In total, we estimate a carbon sequestration efficiency of 81%. Both scenarios are shown in Figure 1.

While the water footprint of pre-combustion CCS, post-combustion CCS, and DACCS is solely from blue water, the water footprint of BECCS is from both green water and blue water. Feedstock biomass growth uses both green water and in many cases blue water supplied by irrigation [31]; blue water is also used in the capture and sequestration process during the integrated gasification combined cycle. Here we assume that feedstock biomass is solely rain-fed and therefore only green water is used in the production of biomass.

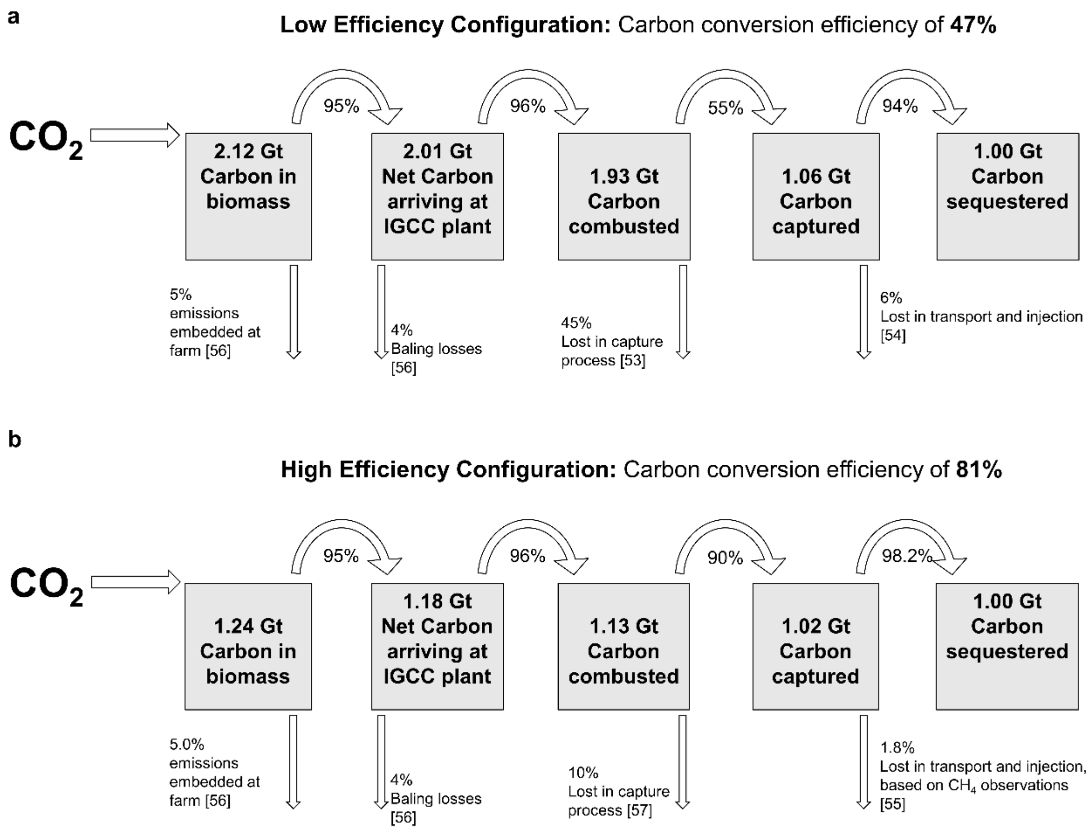

Figure 1. BECCS carbon supply chain in low and high efficiency configurations. The percentage values are carbon losses from literature.

Table 1. Inventory of previous studies used to collect data of water use efficiencies of biomass feedstock for BECCS.

BIOMASS FEEDSTOCK	SOURCE
POPLAR	[58-62]
MISCANTHUS	[63-68]
CROP RESIDUES	[66,70,71]
EUCALYPTUS	[72-76]
SWITCHGRASS	[66,68,71,77-79]
WILLOW	[80-82]
PERENNIAL GRASSES	[67-69]

5.3.4 Calculation of projected water consumption

We assessed projected water consumption from CO_2 sequestration in the 21st century multiplying technology-specific CO_2 sequestration from CCS processes (tonne CO_2) times their water footprints (m^3 per tonne CO_2). Carbon dioxide removal scenarios were taken from Realmonte et al., 2019 [33] and assessed using two well-established integrated assessment models – WITCH [83] and TIAM-Grantham [84]. With integrated assessment models, it is possible to evaluate the role of different carbon removal technologies in 1.5 and 2°C mitigation scenarios through a least-cost optimization, under a range of techno-economic assumptions (technology costs, energy requirements, and technical learning and growth rates). These scenarios were obtained imposing a carbon budget over the 2016-2100 period equal to 810 and 220 billion tonne CO_2, consistent with 1.5ºC and 2ºC warming respectively [85]. We chose the study of Realmonte et al., 2019 [33] for its detailed representation of a broad portfolio of carbon capture technologies, considering also DACCS and BECCS in addition to traditional CCS processes. Moreover, the inter-model study design ensures that our results are robust across model uncertainties, as the integrated assessment models adopted have complementary characteristics.

5.4 Results

5.4.1 Water footprint of low carbon electricity generation

Water use is becoming an increasingly important issue for low-carbon electricity generation [86]. Given the committed trillion-dollar investments in existing fossil fueled energy and industrial infrastructure [87], post-combustion CCS is the preferred economically viable technology to curtail CO_2 emissions because it can potentially be added to existing energy and industrial infrastructure without having to decommission them [88,89]. Using the IECM model, we estimate that a coal-fired power plant retrofitted with post-combustion CCS has a water footprint of 1.71 [0.50; 2.33] m^3/tonne CO_2 (*median* [*low percentile; upper percentile*] *across the ensemble*) (Figure 2). Receiving increasing attention is also the opportunity to retrofit natural gas power plants with post-combustion CCS [90]. We estimate that a natural gas combined cycle power plant retrofitted with post-combustion CCS has a water footprint of 2.59 [2.37; 3.16] m^3/tonne CO_2.

Figure 3 shows technology-specific water intensities of different post-combustion CCS technologies. We find that water intensity strongly varies with cooling technology and CCS technology (Figure 3). Once-through is the cooling technology with the highest water withdrawal intensity, while wet cooling is the technology with highest water consumption intensity. Amine absorption and temperature swing adsorption are the CCS technologies with the highest water intensity. Pressure swing adsorption and membranes systems are the least water intensive CCS technologies.

Pre-combustion CCS is another promising technology to decarbonize energy and industrial systems (Box 2). We considered two pre-combustion CCS processes: Oxy-combustion and integrated gasification combined cycle. We find that oxy-combustion has a similar water footprint to post-combustion CCS, equal to 2.22 [1.93; 2.69] m^3/tonne CO_2. But integrated gasification combined cycle has a smaller one, equal to 0.74 [0.65; 0.80] m^3/tonne CO_2 (Figure 2).

5.4.2 Water footprint of removing CO_2 from the atmosphere

Preventing global temperature from rising more than 1.5º C is likely to require the removal of CO_2 from the atmosphere with negative emission technologies such as BECCS and DACCS [91,92]. BECCS is the CCS technology with the highest water footprint. Under a low efficiency configuration (Figure 1a), BECCS has a water footprint equal to 575 [382; 766] m^3/tonne CO_2 captured, while under a high efficiency configuration (Figure 1b), it has a lower water footprint equal to 333 [221; 444] m^3/tonne CO_2 captured (Figure 3). The water footprint of BECCS is mainly from green water to grow biomass feedstock. Figure 4 shows the water footprint of

BECCS considering different dedicated biomass feedstock. The water footprints show large variations depending on feedstock type and phenology. Producing bioenergy and capturing CO_2 from eucalyptus plantations has the highest water footprint (Figure 4), while miscanthus and willow are the biomass feedstock with the lowest water footprint. In addition to BECCS, DACCS is emerging as a potentially important process to remove CO_2 from the atmosphere [34]. Despite DACCS is currently more expensive than BECCS, we find that DACCS is the most water-efficient way to remove CO_2 directly from the atmosphere, with a blue water footprint of 4.01 [2.00; 6.83] m^3/tonne CO_2.

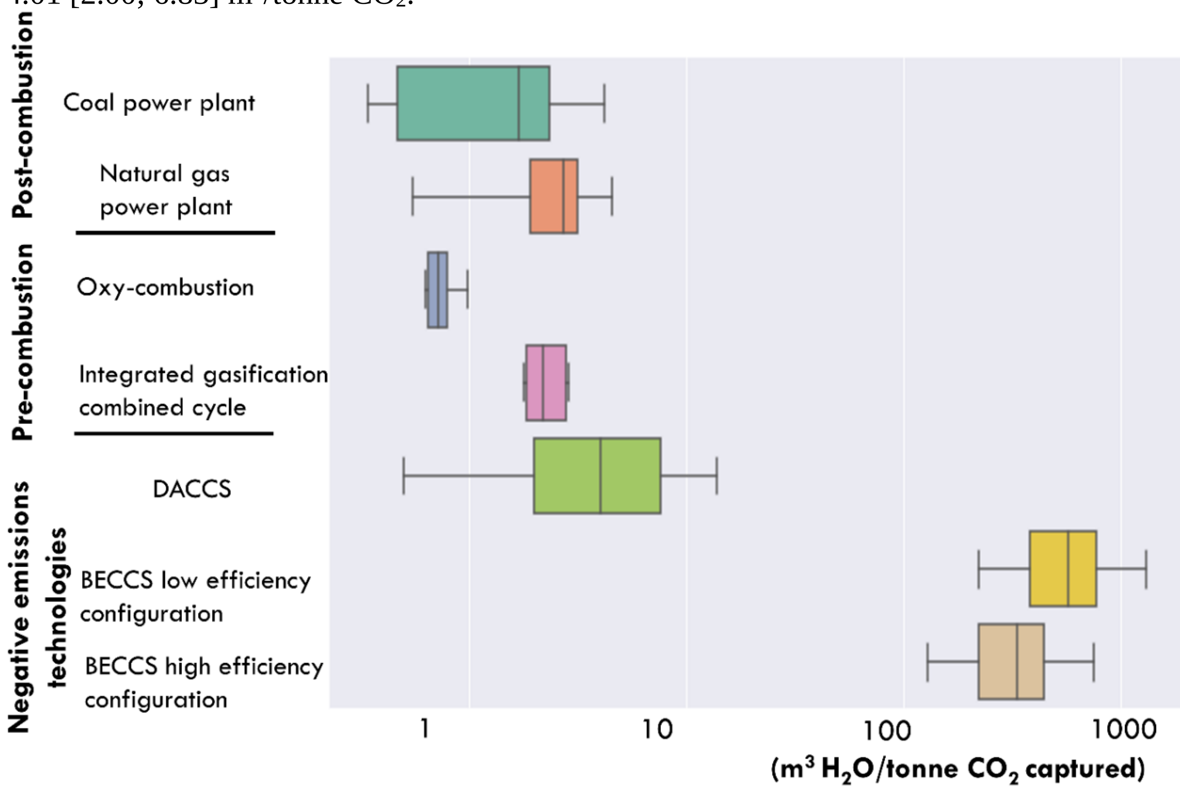

Figure 2. The water footprint of carbon capture and storage technologies. The boxplots reports a range of water footprints of post-combustion CCS, pre-combustion CCS, and negative emission technologies. The water footprint of BECCS is shown for the low and high efficiency configurations (Figure 1). The boxplots represent median, 25th and 75th percentile, and maximum and minimum values of water footprint among the ensemble, outliers are not shown in the figure. Note one cubic meter of water is equal to one tonne of water.

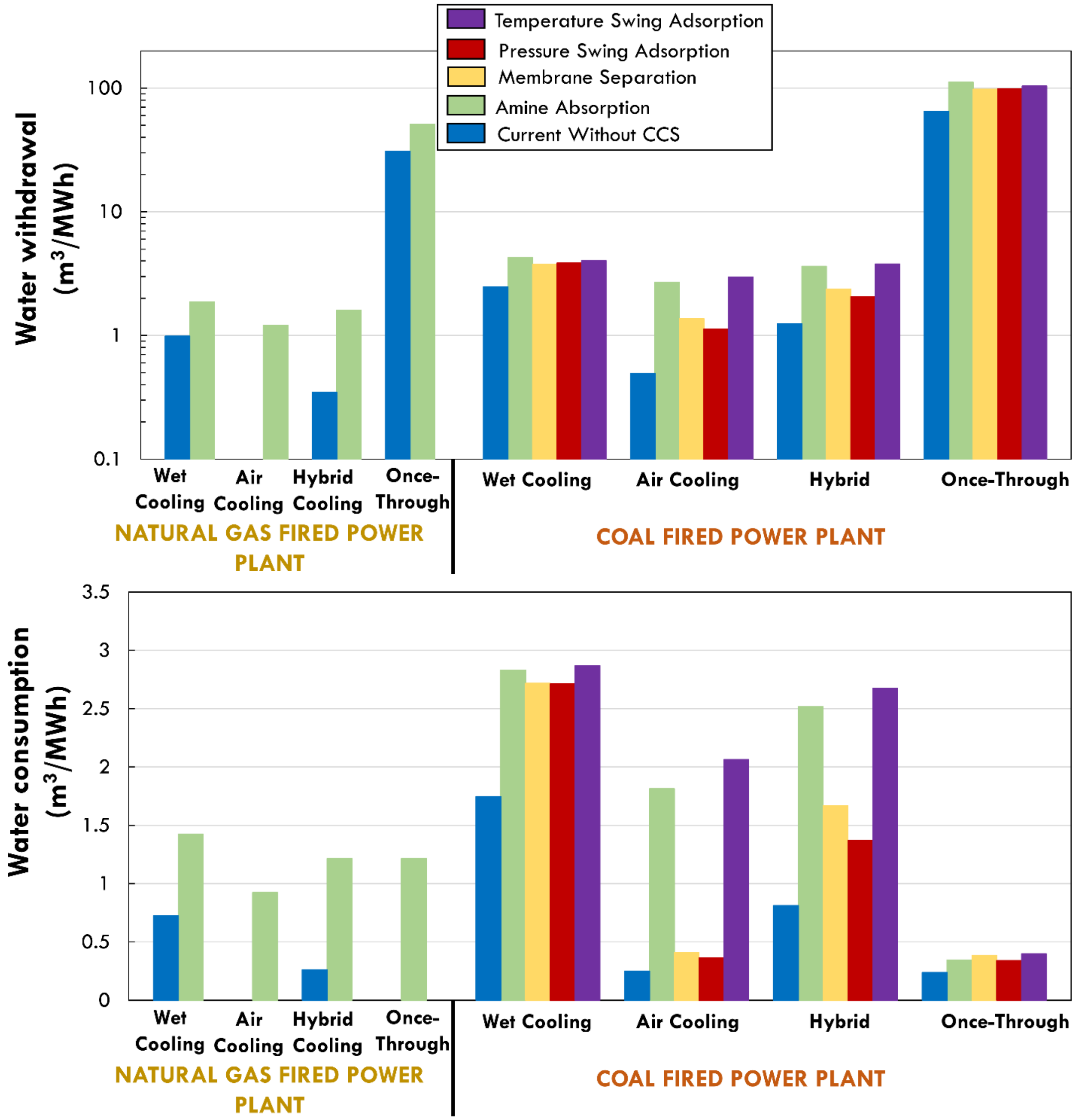

Figure 3. Water consumption and withdrawal intensities of coal-fired and natural gas-fired plants with and without post-combustion CCS. There are four prominent post-combustion CCS technologies: amine absorption, pressure swing adsorption, pressure swing adsorption, and membrane separation. Despite amine absorption is proven and commercially available, membrane separation and adsorption post-combustion CCS systems are still at lower stages of development [3]. The figure was generated running the Integrated Environmental Control Model (IECM Version 11.2) [43] and considering a different range of air temperatures, relative humidity, and gross power inputs. Note that water withdrawal intensity is shown using a logarithmic scale.

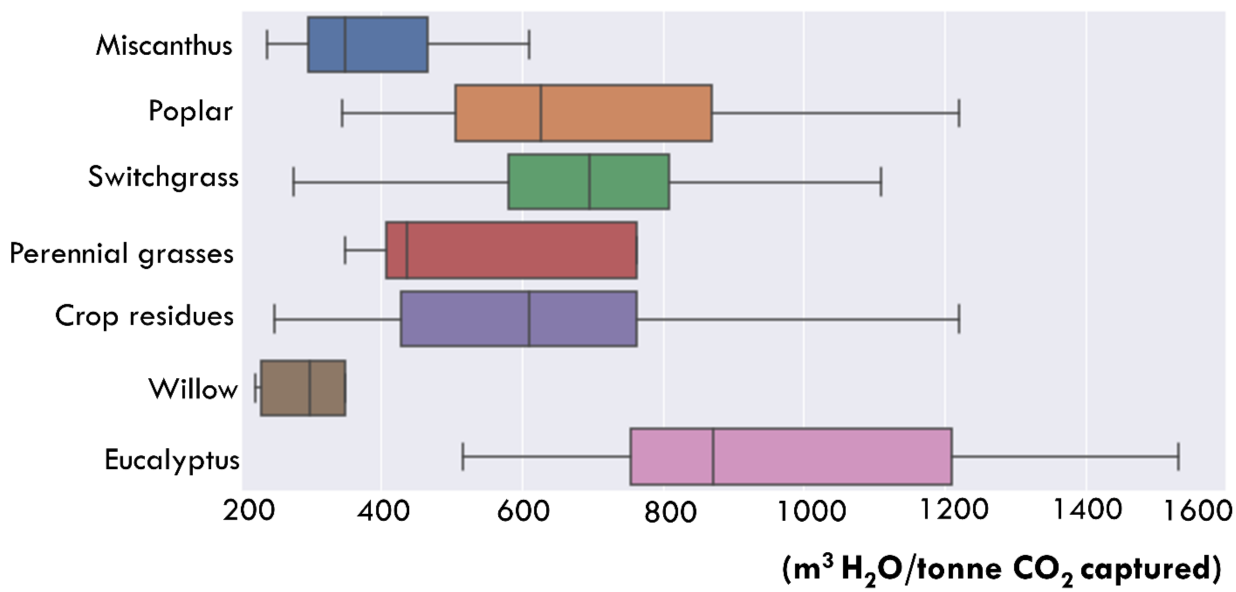

Figure 4. The water footprint of dedicated BECCS feedstock. The figure was generated considering feedstock-specific water use efficiencies from previous studies (Table 1) and a BECCS carbon conversion efficiency equal to 47% (Figure 1a). The figure shows green and blue water footprint. Because we do not assume that dedicated feedstock are irrigated, here, blue water for BECCS comes solely from the integrated gasification combined cycle process and it is equal to 0.74 m^3/tonne CO_2 (Figure 2). The boxplots represent median, 25th and 75th percentile, and maximum and minimum values of water footprint among the ensemble of data collected, outliers are not shown in the figure.

5.4.3 Projected water use to meet climate targets

In order to assess the water consumption that would result from the adoption of CCS to meet 1.5ºC and 2ºC climate change targets in the 21st century, we multiplied the projected amount of CO_2 sequestered by different technologies [33] by the water footprint values specific to each CCS process. Under a more conservative 2ºC climate change scenario, CCS would have a water footprint of 3,900-5,850 km^3 to sequester 15-47 billion tonne CO_2 yr^{-1} in year 2100 (Figure 5). We also find that meeting 1.5ºC mitigation targets will require substantially more water than the 2ºC climate scenario, with an estimated 5,085-8,564 km^3 of water necessary to sequester 21-47 billion tonne CO_2 yr^{-1} in year 2100. The 1.5ºC climate scenario will require more water because more CO_2 will need to be sequestered from the atmosphere along the century to limit warming. In all the scenarios, more than 97% of global water consumption will come from BECCS and therefore would mainly be from green water. Indeed, Figure 5 shows that the scenarios with multiple adoption of CCS technologies exhibit lower water consumption, while BECCS intensive scenarios require more water than the others do.

While our results show that large volumes of water will be required, future technological development could lower the water footprint of CCS processes. For example, to assess the water footprint of BECCS we considered a carbon conversion efficiency – the amount carbon from the harvested dedicated feedstock can be removed from the carbon cycle and sequestered – equal to 47% [52] (Figure 1). In case the carbon conversion efficiency of BECCS increased to 81% (Figure 1), the water footprint of BECCS would decrease from 575 m^3/tonne CO_2 to 333

m^3/tonne CO_2. This in turn would reduce global CCS water consumption from 5,085-8,564 km^3 to 3,000-4900 km^3 under a 1.5ºC climate scenario by 2100.

5.5 Discussion

5.5.1 Trade-offs between water resources and climate mitigation

Building on previous efforts that assessed the water footprint of anthropogenic activities [9,93], this study quantifies the water footprint of four prominent CCS technologies in the context of stringent climate change mitigation. The need to decarbonize the global economy has led to an increasing interest in CCS as a climate mitigation strategy from a policy-making perspective [8,24,94,95]. At the same time, concerns have been arisen about their sustainability and the impacts on water and land use, energy needs and ecosystems [8]. In particular, the adoption of CCS technologies will likely increase demand for water. We analyze the water footprint of future CCS deployment both for low-carbon energy generation and direct carbon dioxide removal from the atmosphere, which both play a large role in stringent climate change mitigation.

We show that the water footprint of CCS varies with technology and that some technologies remove CO_2 in a more water-efficient way than others. While, BECCS has the highest water footprint, DACCS is the most water efficient technology to directly remove CO_2 from the atmosphere (Figure 2). However, BECCS mostly uses green water while DACCS uses exclusively blue water and therefore may compete with municipal and industrial uses as well as irrigation. Conversely, green water uses for BECCS compete with agro-ecosystems for the use of land and associated rainwater needed for biomass production. Among the CCS technologies suitable for low carbon electricity production, oxycombustion is the process with the lowest water footprint. We also illustrate the projected water requirements of the widespread adoption of CCS that is required to meet climate targets, considering a combination of CCS adoption scenarios (Figure 5) and find that a diversified portfolio of CCS technologies is likely to have lower impacts on water resources than a scenario relying mainly on one technology, such as BECCS. Our results enable a more comprehensive understanding of water uses by the most prominent CCS technologies and can better inform management and policy decisions to identify the most effective use of water resources in meeting climate goals.

a

Fraction of carbon dioxide removal per CCS technology in year 2100

	TIAM Model				WITCH Model			
	1.5 Multiple CCS	1.5 BECCS Intensive	2 Multiple CCS	2 BECCS Intensive	1.5 Multiple CCS	1.5 BECCS Intensive	2 Multiple CCS	2 BECCS Intensive
BECCS	22%	50%	22%	67%	19%	58%	15%	67%
DACCS	64%	0%	64%	0%	66%	0%	68%	0%
Industrial	10%	46%	10%	29%	11%	39%	12%	29%
Electricity	4%	4%	4%	4%	4%	3%	5%	4%

b

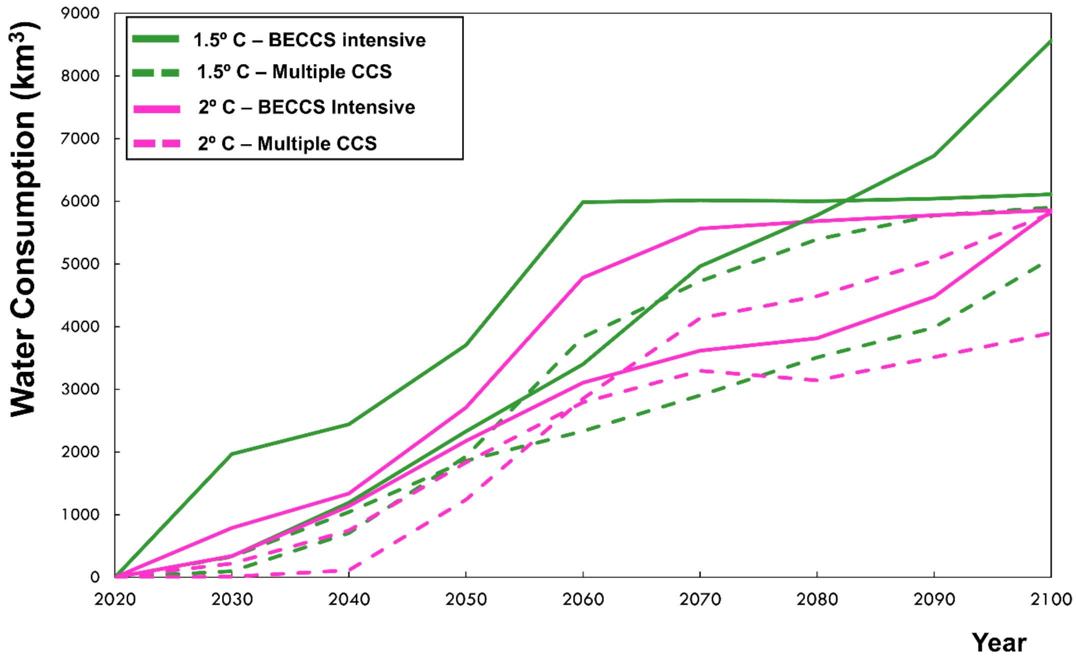

Figure 5. Global water consumption from CCS in the 21st century in a 1.5ºC and 2ºC consistent scenarios. The figure shows the water consumption required to achieve climate targets across different mitigation pathways. All pathways require carbon dioxide removal through CCS technologies, but the amount varies across climate scenarios, as do the relative contribution of post-combustion CCS, pre-combustion CCS, BECCS, and DACCS. This has implications for projected water consumption from CCS adoption. Projected carbon dioxide removal scenarios come from TIAM and WITCH integrated assessment models [33]. Panel a shows the share of carbon dioxide removal per technology in year 2100 [33]. Water consumption estimates were generated considering a BECCS carbon conversion efficiency equal to 47% (Figure 1a).

5.5.2 Biomass plantations and water resources

BECCS has the highest water footprint among CCS technologies and it is by far the process that will have greater impacts on global water consumption, accounting for more than 97% of the total water footprint from CCS technologies by 2100 (Figure 5). We here assume that BECCS feedstock consume solely green water resources. However, under irrigated condition or in the case of phreatophyte vegetation, blue water can also be used by biomass plantation. In fact, cheap blue water from the Columbia River in Oregon has been used to irrigate biomass plantations [96,97]. Irrigation will likely be deployed to increase yields in biomass plantations [33] and therefore reduce the large land footprint that would be needed to meet climate targets through BECCS [98]. In addition, feedstock plantations could also have impacts on downstream blue water resources [99] when tree plantations act as phreatophytes and tap blue water from shallow aquifers to sustain their high evapotranspiration rates. For example, eucalyptus trees have shown the ability to take up blue water from the underneath aquifers and deplete blue water availability for downstream users [100-102]. Of great concern is also the planting of large swaths of non-native tree species, many of which perish because their water needs are too great for local climate conditions [13]. Moreover, high CO_2 concentration in the atmosphere [103] and future technological development [104] will likely increase the efficiency and productivity of photosynthesis in crop plants, potentially reducing the water footprint of biomass plantations. C4 plants (corn, sorghum) will have higher water use efficiency than C3 crops [105]. Importantly, biomass plantations are likely to have other environmental liabilities in addition to impacts on water resources, such as nitrogen leakage, soil carbon and phosphorus loss, land use, albedo, and local climate change [8,50,106].

5.5.3 CCS and water planetary boundary

In some regions of the world, CCS adoption will likely put under additional stress freshwater resources that are already depleted, challenging water systems, rising concerns about water scarcity [107] and the Earth's ability to meet the water needs of humanity with its limited freshwater resources [22]. While, globally, the water footprint of humanity has not surpassed the planetary boundary of freshwater [108], societal water consumption is locally unsustainable in many regions worldwide. In fact, it has been estimated that 50% of blue water consumption [109] and 18% of green water consumption [110] overshoots maximum sustainable level for local green and blue water resources. An increase in water demand due to CCS deployment would draw humanity closer to the planetary boundary for both blue water [111] and green water [110], which are estimated to be 2,800 $km^3\ yr^{-1}$ and 18,000 $km^3\ yr^{-1}$, respectively (Figure 6). We find that CCS adoption would increase by 84 (±56) $km^3\ yr^{-1}$ the current blue water consumption of humanity, which is estimated to be 1,700 $km^3\ yr^{-1}$ [112]. CCS adoption – through BECCS – would require an additional 6,757 (±1,803) $km^3\ yr^{-1}$ of green water from the current green water consumption estimated to be 8,720 $km^3\ yr^{-1}$ [110], or approximately 10% of global total evapotranspiration [113]. Therefore, CCS may increase competition for freshwater resources with other human activities such as the agricultural, industrial, and domestic sectors [109,114,115] and generate unsustainable conditions for freshwater ecosystems [116]. Green water appears to be the primary concern, as BECCS plantations will likely draw humanity closer to the planetary boundary for green water and generate widespread green water scarcity.

5.5.4 CCS and local water scarcity

Water is a local resource and the planetary boundaries for water need to be calculated starting from a local water balance assessment. Differently, carbon budgets are defined on a global scale, as the impact of carbon emissions on climate change does not depend on their specific location, but on the global CO_2 concentrations. In the case of CCS technologies, the exact location where these systems will likely be deployed remains unknown. Our study does not investigate the impacts of CCS technologies on local water availability and water scarcity. We, here, calculate the global amount of water resources that will be claimed by CCS technologies to meet stringent climate targets. Therefore, planning for CCS mitigation strategies for climate change should account for local water availability and the patterns of blue and green water scarcity [107].

We posit that the additional water consumption from CCS could strongly affect the local and global water resources exacerbating and creating widespread green and blue water scarcity conditions worldwide. For example, Rosa et al., 2020 [26] estimated that 23% of global coal plant capacity would face longer periods of blue water scarcity if retrofitted with post-combustion CCS. It is therefore fundamental to deploy CCS at those facilities not to be impacted by blue water scarcity. Appropriate plantations for biomass production to be used as feedstock in BECCS systems should be as well planned only in areas not to be affected by green water scarcity so as not to require irrigation.

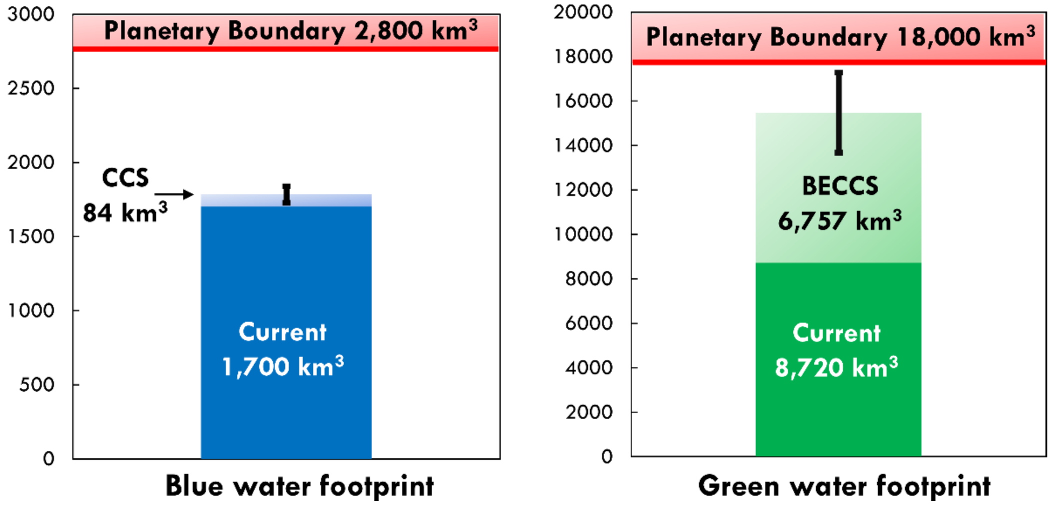

Figure 6. Estimates of green and blue water footprints relative to proposed planetary boundaries. Bars show current and CCS green and blue water footprints. Blue water footprint from CCS is from expected adoption of pre-combustion, post-combustion, and DACCS in year 2100 under 1.5ºC climate scenarios. Green water footprint from CCS is from BECCS in year 2100 under 1.5ºC climate scenarios. The error bar ranges represent the uncertainty range of consumption use of blue water and green water from different carbon dioxide removal scenarios [33]. The figure was generated considering a BECCS carbon conversion efficiency equal to 47% (Figure 1a).

This study quantified the water footprint of carbon capture and storage technologies. We showed that CCS adoption necessarily entails large water requirements, and that different CCS processes have different water requirements to capture carbon dioxide. There are already reasons of profound concern about whether the future food, energy, and fiber needs can be met using the limited freshwater resources of the Planet. The projected water requirements from CCS should be of paramount concern when designing future climate policies. The results of this study can thus form an important basis for further assessments of how climate mitigation policies will contribute to the water footprint of humanity in the coming decades. Future research is required to reduce the water footprint of CCS processes and minimize the competition for the already scarce freshwater resources of the Planet.

5.6 Conclusions

Water scarcity is progressively perceived as a socio-environmental threat that could constrain anthropogenic activities and impair ecosystems [117]. Water is also becoming an increasingly vexing factor in managing climate mitigation technologies such as carbon capture and storage. What is the global water footprint of carbon capture and storage under stringent climate change mitigation policy? We provide an answer to this question in the context of the four prominent CCS technologies. We estimate that to meet the 1.5ºC climate target, CCS would almost double the water footprint of humanity. Our results show that the water footprint of CCS strongly varies with technology. Some CCS technologies, however, consume much less water than others, suggesting that with appropriate decision it is possible to capture CO_2 in the most water-efficient way. Green water appears to be the primary concern, as BECCS plantations will likely draw humanity closer to the planetary boundary for green water and generate widespread green water scarcity. Our results show that a diversified portfolio with different CCS technologies and balanced strategies of mitigation and carbon removal will likely have lower water requirements than a portfolio relying mainly on one technology. The results of this study underscore the importance of integrating water footprints of CCS in future climate and energy policies. Our analysis provides important insights into the hydrological consequences of widespread CCS adoption.

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CHAPTER 6

The water-energy nexus of hydraulic fracturing: a global hydrologic analysis for shale oil and gas extraction

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6.1 Abstract

Shale deposits are globally abundant and widespread. Extraction of shale oil and shale gas is generally performed through water-intensive hydraulic fracturing. Despite recent work on its environmental impacts, it remains unclear where and to what extent shale resource extraction could compete with other water needs. Here we consider the global distribution of known shale deposits suitable for oil and gas extraction and develop a water balance model to quantify their impacts on local water availability for other human uses and ecosystem functions. We find that 31-44% of the world's shale deposits are located areas where water-stress would either emerge or be exacerbated as a result of shale oil or gas extraction; 20% of shale deposits are in areas affected by groundwater depletion and 30% in irrigated land. In these regions shale oil and shale gas production would likely compete for local water resources with agriculture, environmental flows, and other water needs. By adopting a hydrologic perspective that considers water availability and demand together, decision makers and local communities can better understand the water and food security implications of shale resource development.

6.2 Introduction

Shale oil and shale gas have recently emerged as new important energy sources expected to play a fundamental role in meeting energy demand in the near future [International Energy Agency, 2017]. Shales are low permeability sedimentary rocks that might contain high quantities of hydrocarbons [Holditch et al., 2007]. Various recent studies have shown how hydraulic fracturing, the technology generally used for shale hydrocarbon extraction, is associated with substantial amounts of water withdrawal and consumption [Nicot and Scanlon, 2012; Scanlon et al., 2014; Chen et al., 2016; Horner et al., 2016] as well as declines in regional water quality [Osborn et al., 2011; Rozell et al., 2012; Vidic et al., 2013, Jackson et al., 2013; U.S. Environmental Protection Agency, 2016]. Other possible environmental consequences of unconventional oil and gas extraction from shale are methane migration and groundwater contamination from faulty seals around well casings [Warner et al., 2012; Vidic et al., 2013, U.S. Environmental Protection Agency, 2016; Brantley et al., 2018], impacts on regional air quality [Vidic et al., 2013], low weights at birth in babies born near wells [Currie et al., 2017], seismic triggering associated with the choice to use deep wells as a disposal method for returned fracturing fluids and the so-called 'produced water' (the water resulting from oil and gas extraction) [Rutqvist et al., 2013; Kharak et al., 2013]. The development of shale deposits may also entail land use change [Jordaan et al., 2018], forest removal, habitat fragmentation, and biodiversity loss [Kiviat et al., 2013].

Shale oil and gas extraction also has important social, political, and economic implications. In the last decade, the North American fracking 'boom' has changed the world hydrocarbon industry and energy economy. Shale gas has provided an abundance of natural gas, a bridge fuel towards a low carbon future [Moniz et al., 2011]. For example, power generation in the U.S. is shifting from coal to the lower emitting natural gas [Obama, 2017]. In addition, the shale revolution has created new jobs and economic benefits in North America [Peplow, 2017]. Thus, shale extraction has the potential to enhance the economic growth and energy security of some regions and nations. Unconventional oil and gas from shale rocks are an opportunity for some countries to increase their energy security, while reducing costs of fossil fuel imports and potentially changing their import-export balance [Vidic et al., 2013]. However, leakages from natural gas infrastructure can offset the benefit in reduction of greenhouse gas emissions from a combustion process that is more efficient and cleaner than that of coal [Alvarez et al., 2012; Jenner and Lamadrid, 2013; Brandt et al., 2014; Caulton et al., 2014; Howarth, 2014]. Despite growing interest in shale resources, there is only a limited understanding of the pressure that their extraction could place on local water resources worldwide [Reig et al., 2014]. Globally, it remains unclear to what extent the water consumption of shale gas and shale oil production would compete with other human and environmental water needs and induce or exacerbate local water scarcity. Such a potential trade-off among water allocations is especially worrisome for regions already prone to water stress, where additional water may also be needed to support growing populations and the expansion of irrigation [Davis et al., 2017].

This limited understanding of the potential impacts of shale development on the local water balance thus prevents the implementation of a sustainable water management plan in places where shale extraction is possible. There is therefore a pressing need for a quantitative assessment and mapping of where shale resource mining could lead to an inadequate management of local water resources as well as intensify the competition for water between food and energy production [Rulli et al., 2016; Cook and Webber, 2016; Rosa et al., 2017; D'Odorico et al., 2017; Chiarelli et al., 2018; Habib et al., 2018].

Previous efforts [Clark et al., 2013; Jiang et al., 2014; Kondash and Vengosh, 2015] have assessed the water footprint of unconventional oil and gas extraction from shale from the life cycle assessment (LCA) perspective, focusing on a comprehensive accounting of all water costs associated with production and processing, but without examining the availability or source of the required water. Here we assess the impacts of global shale extraction on the local water balance using a hydrologic approach that links shale fuel extraction with hydrologic and environmental impacts. We examine the global distribution of known shale deposits suitable for oil and gas production [Kuuskraa et al., 2013] and identify the regions in which water consumption for hydraulic fracturing could compete with agriculture and other human activities. We analyze the average annual water stress [Mekonnen and Hoekstra., 2016] at 0.5° resolution (~50 km at the Equator) for the world's shale deposits and highlight those deposits in which shale hydrocarbon extraction would induce or enhance water stress. While water quality concerns by local population may be a limiting factor for the development of world shale deposits [e.g., Goho, 2012; Williams, 2017], here we focus on physical and environmental constraints resulting from water limitations.

Previous studies have quantified water stress resulting from water withdrawals for hydraulic fracturing in some shale deposits in the United States, Argentina, China, and Mexico [e.g., Scanlon et al., 2014; Freyman et al., 2014; Mauter et al., 2014; Guo et al., 2016; Galdeano et al., 2017]. In response to the need for a global-scale analysis of the hydrologic impacts of shale extraction, the World Resources Institute estimated that 39% of global shale deposits lie within surface water-stressed regions [Reig et al., 2014]. However, a global-scale quantitative analysis of the extent to which water consumption for shale gas and shale oil production would compete with agriculture and induce or exacerbate local water stress is still missing.

Here we also quantify and analyze the possible impacts of global oil and gas extraction from shale on groundwater resources, environmental flows, agricultural, industrial, and domestic water consumption. We use an updated global shale deposit dataset that includes all known deposits where the most profitable opportunities for oil and natural gas extraction exist [Kuuskraa et al., 2013]. We adopt a water balance approach [Mekonnen and Hoekstra, 2016] to quantify the impact of shale extraction on the local water resources, while accounting for the water required for other human needs (e.g., irrigation) and environmental flows. We conclude by comparing local volumes of water consumption by shale extraction to the amount of current irrigation water consumption [Hoekstra and Mekonnen, 2012].

6.3 Methods

6.3.1 World shale deposits

Global maps of shale deposits were acquired from Advanced Resources International, Inc., who have developed an up-to-date internationally recognized geo-referenced dataset of the spatial extent of shale areas [Kuuskraa et al., 2013]. In the case of the United States, the map of shale areas came from the U.S. National Energy Technology Laboratory [U.S. National Energy Technology Laboratory, 2016]. In this study we focus on shale areas (or "shale plays") that offer the most profitable opportunities for oil and natural gas extraction in the near future, while lower quality and less explored deposits, which likely hold additional shale resources, are not included in this assessment [Kuuskraa et al., 2013].

6.3.2 Generation of water-stress maps

Water stress (WS) is defined as the ratio of the local water consumption of human activities (WC) (i.e., municipal, agriculture, mining, and other industries) and the renewable blue water availability in a grid cell [Mekonnen and Hoekstra, 2016]. In water stressed areas, water is consumed at greater rates than local renewable water availability. This means that there is an unsustainable use of water resources typically associated with the use of environmental flows and/or groundwater depletion. Blue water stocks include freshwater resources in surface water bodies and aquifers, but do not include soil water storage in the unsaturated zone [Falkenmark and Rockstrom, 2004]. Renewable blue water availability was calculated following the methods by Mekonnen and Hoekstra, [2016].

6.3.3 Assessment of local renewable blue water availability

The global distribution of annual renewable blue water availability (WA) (at 0.5° resolution) was calculated following the methods by Mekonnen and Hoekstra, [2016], whereby the value of WA in a grid cell was expressed as the sum of the local renewable blue water availability in that cell (WA_{loc}) and the net blue water flow from the upstream grid cells, defined as the local surface renewable water availability in the upstream cells (WA_{up}) minus the blue water consumption by human activities in the upstream cells (WC_{up}). The net surface blue water flows were calculated using the upstream-downstream routing "flow accumulation" function in ArcGIS®, where the subscript i denotes the cells upstream from the cell j under consideration:

$$\mathrm{WA}_{\mathrm{j}} = \mathrm{WA}_{\mathrm{loc,j}} + \sum_{\mathrm{i}=1}^{\mathrm{n}} \left(\mathrm{WA}_{\mathrm{up,i}} - \mathrm{WC}_{\mathrm{up,i}}\right)$$

Local blue water availability was calculated as the local blue water flows generated in that grid cell minus the environmental flow requirement. We assumed that a fraction (y) of runoff is allocated to maintain environmental flows and the remaining fraction (1-y) is considered blue water locally available for human needs, WA_{loc} [Pastor et al., 2014; Steffen et al., 2015]. Environmental flow is defined as the minimum surface runoff that is required to sustain ecosystem functions; for irrigation to be sustainable, these minimum flow requirements need to be met even during dry season and low flow conditions [Pastor et al., 2014; Richter et al., 2012]. Three flow regimes were considered: low, intermediate, and high corresponding to less than the 25^{th} percentile, between the 25^{th} and the 75^{th} percentile, and greater than the 75^{th} percentile of annual runoff, respectively. Following Steffen et al. [2015], a different environmental flow requirement (i.e., value of y) was used for each flow regime [Pastor et al., 2014].

To calculate the upstream to downstream surface water availability we used the flow direction raster (at 0.5° resolution) from the World Water Development Report II [Vörösmarty et al., 2000 a-b]. Surface runoff estimates (at 0.5° resolution) were obtained from the Composite Runoff V1.0 database [Fekete et al., 2002].

6.3.4 Assessment of local water consumption

Water consumption (WC) is the volume of water that is withdrawn and not returned back to the environment as liquid water (i.e., consumptive use). Estimates of agricultural (crops and livestock), industrial, and domestic water consumption at 0.0833° resolution were from Hoekstra and Mekonnen, [2012] and were aggregated to 0.5° resolution to match with the water availability dataset. Crop water consumption was estimated using a crop-specific model of irrigation water requirements [Hoekstra and Mekonnen, 2012]. The rates of domestic and industrial water consumption were taken from Hoekstra and Mekonnen, [2012] using country-specific per capita values and population density maps.

6.3.5 Shale deposits and groundwater depletion

Water used for shale gas and oil extraction can be taken either from surface water bodies or from groundwater resources [Freyman et al., 2014]. Because the recharge and recovery of groundwater reserves occurs at much longer time scales, these resources can be more vulnerable to depletion under prolonged rates of withdrawals. With this in mind, we analyzed world shale deposits and their possible extraction impacts on freshwater aquifer stocks contained in global major groundwater basins [BGR/UNESCO, 2008]. In this study we do not consider brackish or saline aquifers.

If groundwater consumption occurs at higher rates than it is replenished by hydrologic processes, the aquifer is undergoing unsustainable use or ‘depletion’. In some cases freshwater stocks that were formed in the past centuries or millennia are depleted (‘mined’) in just a few decades [Gleeson et al., 2012]. To identify shale deposits located in areas affected by groundwater depletion, we overlaid a groundwater depletion map [Gleeson et al., 2012] with the global distribution of shale deposits.

6.3.6 Assessing water consumption for shale extraction

The water consumption of shale resource extraction (WC_{Frac}) was calculated as:

$$\mathrm{WC_{Frac}}\left(\frac{\mathrm{m}^3}{\mathrm{year}}\right) = (1 - \mathrm{F} \cdot \mathrm{R}) \cdot \mathrm{n} \cdot \mathrm{W}$$

where W is the water injected into one well using today’s hydraulic fracturing technology, n is the number of wells, and F and R are the fraction of the returning fracturing fluid and its recycled fraction, respectively.

The amount of water required to stimulate a horizontal well through hydraulic fracturing (W) depends greatly on local geology, deposit depth, technology used, and operational factors applied (e.g., average well lateral length) [Nicot and Scanlon, 2012; Scanlon et al., 2014; Gallegos et al., 2015]. Unfortunately, only limited data and scholarly work exist for shale deposits outside the

United States. Therefore, given the complexity and uncertainty of modelling water consumption for global deposits, our analysis requires simplifications and assumptions. We therefore considered eighteen water management scenarios (Table 1) based on the same parameters available for U.S. shale development and applied them to the other shale deposits outside North America. According to the literature, we assumed two values of water consumed per well (W) of 12,000 m^3 (low injection scenario) and 30,000 m^3 (high injection scenario) [Chen et al., 2016; Kondash and Vengosh, 2015]; and three water recycling (R) options ('no recycling', 50%, and 80% recycling). Depending on the geology, the returning hydraulic fracturing fluid (F) can be up to 70% of the injected water. To make a conservative analysis we assumed flow back water equal to 70% [Gregory et al., 2011].

The number of potential wells (n) that can be drilled in each shale deposit was assessed as the product of the area of each shale deposit (km^2) and the typical well spacing values (wells/km^2). Well spacing from developed shale oil and gas deposits ranges from 1.50 wells/km^2 (low), to 2.13 wells/km^2 (average), and 3.62 wells/km^2 (high) [Kuukstraa et al., 2011; McGlade et al., 2013; Rezaee, 2015]. In our analysis we used these three well spacing values. The rate at which wells are drilled and completed depends on numerous factors, including existing infrastructure availability (e.g., drilling rigs, trucks, pumps, water tanks, roads, and pipelines), economics (e.g., oil and gas prices and marginal costs of extraction), existing production within the shale basin, and technology adopted by shale companies [Kuuskraa et al., 2013]. Therefore the wells are not drilled and stimulated all at once but are drilled within a timeframe of a few decades, here assumed to be 30 years [U.S. Energy Information Administration, 2014]. In other words, we assume that the above values of well spacing is attained within a timeframe of 30 years, with n/30 wells added each year.

Results presented in the main text of this study consider an average scenario of water consumption, i.e. a well spacing equal to 2.13 wells/km^2, 80% recycling of the flow back water under the case 'low injection scenario' or 12,000 m^3 of water injected per well.

6.3.7 Assessing other related impacts

To identify shale deposits in which the extraction of oil and gas is expected to compete with food production in the near future, we examined areas in which the increase in agricultural production by closing the yield gap of major crops (i.e., wheat, maize and rice) – the difference between actual and attainable yields – to within 75% of attainable yield will require an increase in irrigation. To that end, we utilized data on the global assessment of irrigation-controlled yield gaps by Mueller et al., [2012].

The number of people living in areas underlain by shale deposits was estimated using population distribution data taken from CIESIN's Gridded Population of the World map (GPWv4) for the year 2010 [Center for International Earth Science Information Network, 2015]. Percentages in the results section are expressed as fractions of the total global shale area times 100.

6.4 Results

6.4.1 Regions in which hydraulic fracturing will intensify pressures on local water resources

We estimate that 31% of global extent of shale areas are located in water-stressed regions, defined as areas in which human consumptive water demand already exceeds local renewable blue water availability (i.e., surface + groundwater). Our global analysis of additional water stress potentially generated by shale deposit exploitation shows that, depending on future water consumption from hydraulic fracturing (Table 1), water-stressed areas over shale deposits could expand to as much as 44% of shale deposit areas. Deposits in currently stressed areas include those occurring in the south-central United States, Canada, Argentina, South Africa, northern Africa, China, India, and Australia (Figure 1).

Table 1. Annual water potentially consumed globally to extract oil and gas from shale resources under the 18 scenarios considered in this study. Results are represented using two values of water consumed per well (*W*) of 12,000 m^3 (low injection scenario) and 30,000 m^3 (high injection scenario); three well densities scenarios (1.50 wells/km^2, 2.13 wells/km^2, and 3.62 wells/km^2); and three water recycling (*R*) options ('no recycling', 50%, recycling, and 80% recycling).

	Low injection scenario [10^9 m^3 y^{-1}]	High injection scenario [10^9 m^3 y^{-1}]
No recycling		
1.50 wells/km^2	3.49	8.73
2.13 wells/km^2	4.96	12.39
3.62 wells/km^2	8.43	21.06
80% Recycling		
1.50 wells/km^2	1.54	3.84
2.13 wells/km^2	2.18	5.45
3.62 wells/km^2	3.71	9.27
50% recycling		
1.50 wells/km^2	2.27	5.67
2.13 wells/km^2	3.22	8.06
3.62 wells/km^2	5.48	13.69

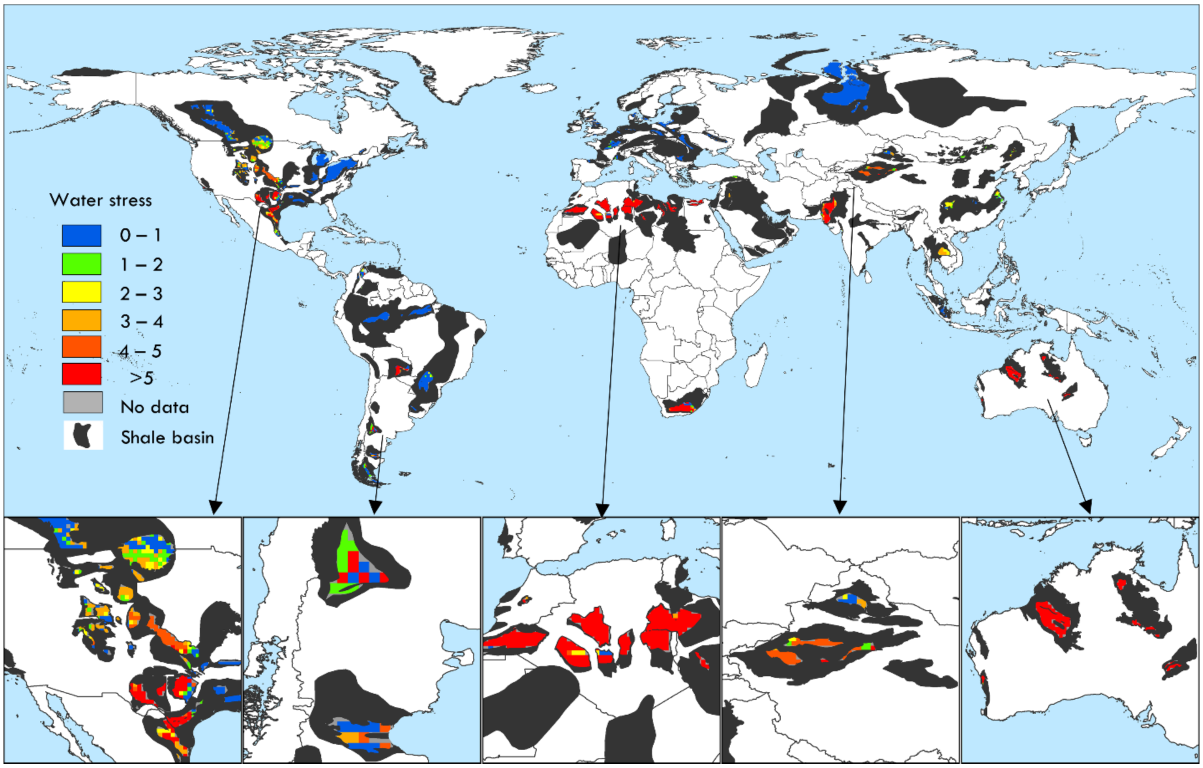

Figure 1. Map of water stress within shale deposits. Pixels with water stress indexes greater than one are subjected to unsustainable water consumptions (i.e., water consumption for human activities exceeds the limit imposed by environmental flow requirements).

Depending on the fraction of returning fracturing fluid that is recycled and well spacing adopted by shale companies, a total water demand ranging from 1.54 and 21.06 km^3 per year will be required to extract the global shale oil and shale gas reserves using current technology (Table 1). Even though the volume of water for shale oil and gas production is an order of magnitude smaller than that required for crop irrigation globally (899 km^3 annually [Hoekstra and Mekonnen, 2012]), we find that the effect of hydraulic fracturing on water resources could be substantial at the scale of individual shale deposits where the water demands of shale extraction can exceed local renewable blue water availability (Figure 2). Depending on future water consumption by hydraulic fracturing, the majority (51-74%) of global shale areas will require less than 1% of the locally available water availability for the extraction of natural gas or oil. However, certain arid regions (17-33% of world shale areas) will require more than 50% of regional water resources for complete shale extraction (Figure 2). Shale deposits in such arid regions also include the Cambay shale (India), Etel shale (Libya), Frasnian shale (Algeria and Tunisia), Gacheta shale (Colombia), Lower Silurian shale (Morocco), and Goodwood/Cherwell shale (Australia) (Figure 2).

Table 2. Overlap between shale deposits and irrigated croplands. Current water consumption from irrigation (*WC_{IRR}*), blue water availability (*WA*), fraction of local blue water availability needed for shale extraction (*WC_{Frac}/WA*), and current water stress (*WS*) over shale deposits. Values for blue water availability are reported after accounting for environmental flows. Note that only the top 15 shale areas with the highest demand for irrigation water are listed.

Shale deposits (country)	WC_{IRR} [10^9 m^3/y^{-1}]	WA [10^9 m^3/y^{-1}]	$\frac{WC_{Frac}}{WA}$ (%)	WS
Sembar (Pakistan)	33.624	4.069	0.16	8.68
Khatatba (Egypt)	6.264	13.682	0.05	1.16
Niobrara (US)	3.433	5.497	0.01	0.75
Permian-Triassic (India)	2.490	18.784	0.07	0.15
Mississippian Lime (US)	2.243	8.519	0.07	0.28
Nam Duk Fm (Thailand)	2.145	12.174	0.05	0.27
Wufeng/Gaobiajian (China)	1.672	35.520	0.02	0.18
Ketuer (China)	1.197	0.601	1.10	2.11
Collingham Whitehill Prince Albert (South Africa)	1.069	0.309	2.14	4.10
Colorado Group (Canada)	0.675	5.062	0.13	0.26
Cambay Shale (India)	0.643	0.000	100.00	>10
Longmaxi Permian Qiongzhusi (China)	0.497	10.279	0.08	0.05
Pimienta (Mexico)	0.407	5.773	0.04	0.13
Baxter (US)	0.406	0.604	1.09	0.71
Banff/Exshaw (Canada)	0.363	2.144	0.308	0.18
Other deposits	7.640	984.263	-	-
All deposits	64.768	1105.135	-	-

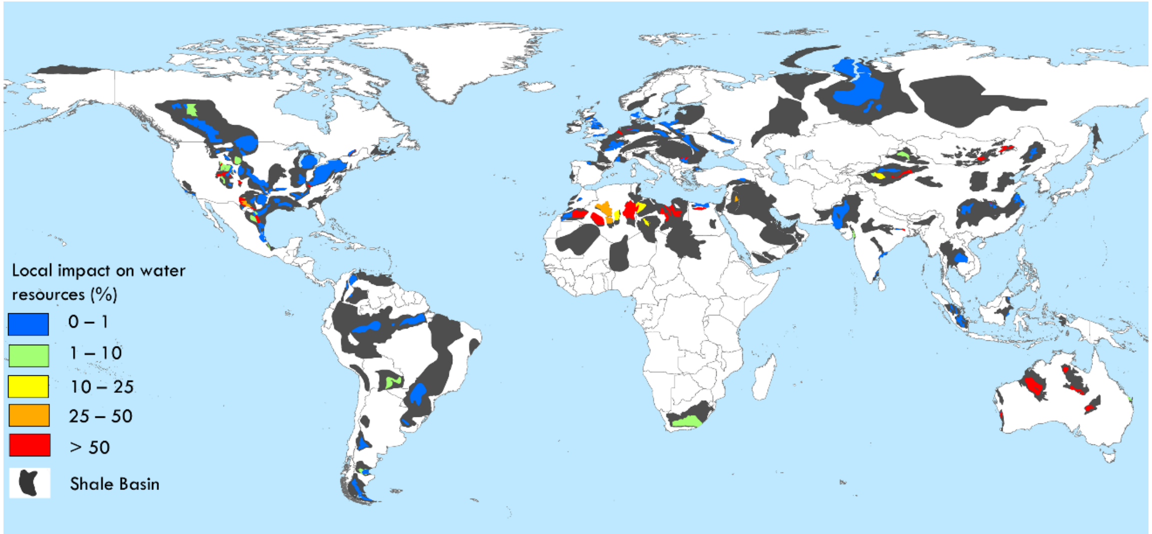

Figure 2. Fraction of local water availability needed for unconventional oil and gas extraction from shale rocks.

6.4.2 Shale deposits and groundwater depletion

The extraction of shale deposits is expected to affect not only surface water resources but also more ubiquitous groundwater resources [Jasechko and Perrone, 2017]. In areas affected by water stress, the extraction of shale deposits could entail the reliance on unsustainable groundwater mining. Therefore, we investigated where the extraction of shale deposits could have an impact on freshwater aquifers around the world by analyzing the co-location of shale deposits and major groundwater basins [BGR/UNESCO, 2008]. Interestingly, we found that 59% of world's shale deposits are in the footprint of major freshwater aquifers (Figure 3). In addition, we find that 20% of shale deposits are located in regions affected by groundwater depletion (Figure 3). Some deposits in the south-central United States, northern India, and Pakistan are situated in groundwater basins that are experiencing substantial depletion (e.g., the U.S. High Plains and Indo-Gangetic Plain aquifers) because of groundwater pumping for irrigation [Rodell et al., 2009; Scanlon et al., 2012]. Further, 17% of the world's shale areas are affected by both water stress and groundwater depletion. These areas are found across the south-central United States, Mexico, Argentina, northern Africa, South Africa, South Asia, and China.

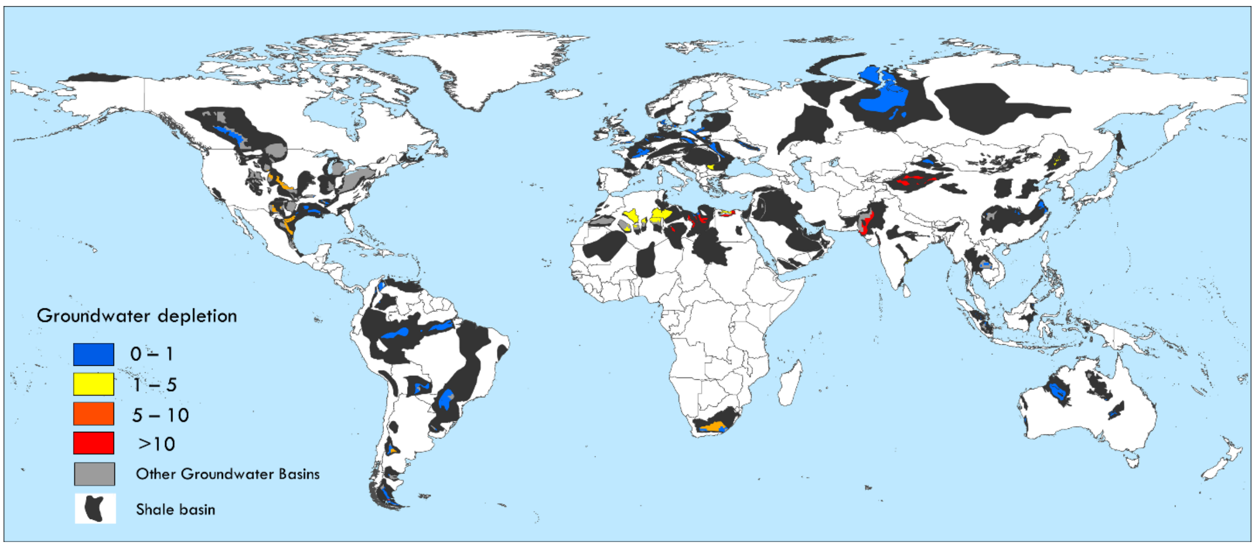

Figure 3. Groundwater depleted aquifers in the footprint of world shale deposits. Freshwater aquifers considered are major groundwater basins [*Gleeson et al., 2012*]. Pixels with groundwater depletion indexes greater than one indicate unsustainable water withdrawals (i.e., groundwater depletion).

6.4.3 Future shale development in irrigated areas

Globally, 7% (65 $km^3\ y^{-1}$) of total global annual irrigation water is consumed on croplands overlying shale deposits (Table 2). Some agricultural baskets over such shale deposits include the U.S. High Plains (Barnett, Niobrara, and Woodford shale), South and East Texas croplands (Eagle Ford and Haynesville shale), North Dakota's Great Plains (Bakken shale), Nile Delta (Khatatba shale), China's Sichuan Province (Sichuan shale basin), China's Xinjiang Province (Tarim shale basin), Indo-Gangetic Basin (Sembar and Cambay shale), and Thailand croplands (Nam Duk Fm shale) (Table 2, Figure 4).

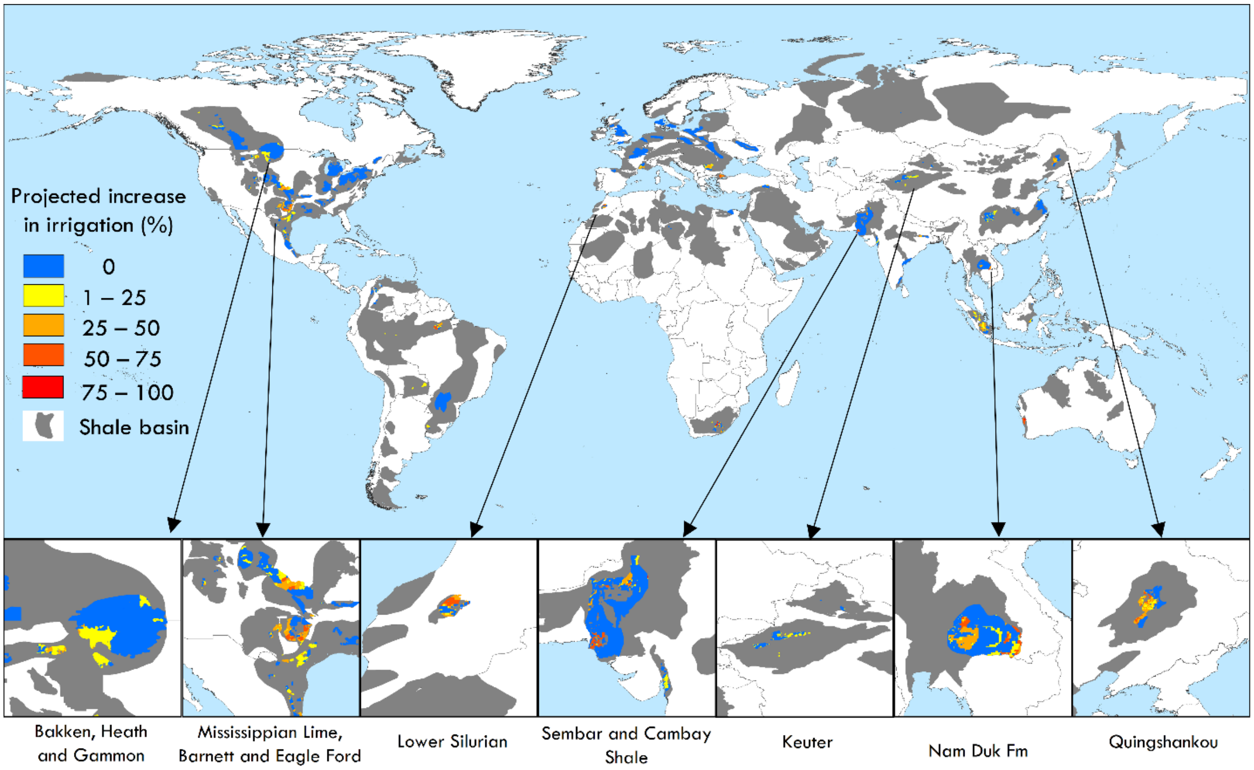

Figure 4. Irrigated areas overlying shale deposits. Projected increase in irrigated areas necessary to reduce the yield gaps of maize, rice and wheat to 75% of attainable yields [*Mueller et al., 2012*]. Bottom panels show the case of shale deposits in Canada, United States, Mexico, Morocco, Pakistan, India, China, and Thailand where we predict the occurrence of future competition between water for shale resource extraction and food production.

To better evaluate possible future competition for water resources between shale deposit extraction and agriculture, we examined the global distribution of areas in which irrigation is expected to increase to accommodate the growing demand for food products (Figure 4). We find that 30% of shale areas worldwide underlie irrigated agricultural areas. Some of these shale deposits in China, India, South Africa, Egypt, and Pakistan are located in water-stressed regions (Table 2). We estimate that 6% of the shale areas are located in regions where water consumption for irrigation has been projected to increase in order to reduce crop yield gaps by 75% – the difference between actual and attainable yields [Mueller et al., 2012]. Thus, pressure on water resources in these areas may not only increase due to potential shale energy production but may also be exacerbated by a greater need for irrigation water.

6.4.4 Domestic and industrial water consumption in areas underlain by shale deposits

Currently, 303 million people worldwide live over shale deposits. In these regions, water is also consumed in industrial production and for domestic water supply. We estimate that 43 km^3 y^{-1} of freshwater are consumed for domestic and industrial purposes over shale deposits – which is about 6% of the total global annual water consumption by these sectors [Hoekstra and Mekonnen, 2012] (Table 1). Those deposits are located in relatively highly populated regions of the United States, China, Ukraine, Pakistan, Egypt, and Thailand.

6.4.5 Use of water resources in shale plays where extraction has recently started

Beside the United States and Canada, shale oil and gas are commercially extracted in Argentina and China [U.S. Energy Information Administration, 2015]. In Argentina, oil and gas are extracted from shales in the Neuquén Basin. This basin is partly located in areas affected by water stress (Figure 1) and groundwater depletion (Figure 3). Here, water is stored in three artificial reservoirs along the Neuquén and Limay Rivers. We estimate that 8% of water availability (Table 3) is locally consumed to irrigate local crops, and the remaining fraction flows downstream where it is also consumed for irrigation. To overcome the additional water consumption from fracking activities and to prevent a further groundwater depletion, policy makers enacted a provincial decree that regulates water allocations associated with oil and gas extraction [Ministerio de Energia, 2012]. In particular, the decree prohibits groundwater withdrawal for hydraulic fracturing and requires the oil industry to report the amount of water consumed for fracking [Ministerio de Energia, 2012]. However, no limits are imposed on the rates of surface water withdrawal for hydraulic fracturing. Therefore, even though fracking activities account for only 1-2% of the local annual water availability (Table 3), in the event of prolonged extraction they are expected to enhance water stress, deplete freshwater storage in reservoirs, and reduce the amount of water available for irrigation [Mauter et al., 2014]. To address these concerns, a River Basin Management plan has been developed to resolve water demand conflicts in the Rio Negro, Neuquén, and Limay River Basins [Ministerio de Energia, 2012].

Table 3. Water resources in emerging shale plays outside North America. Current water consumption from irrigation (WC_{IRR}), blue water availability (WA), water consumption from the domestic and industrial sectors ($WC_{dom\&ind}$), and estimated water consumption from shale oil and gas extraction (WF_{Frac}). WF_{Frac} is reported for the high injection scenario and low injection scenario (30,000 m^3 and 12,000 m^3 of water injected per well, respectively) considering a well spacing equal to 2.13 wells/km^2 and 80% recycling of the flow back water.

Country	Shale Basin	Shale Deposit	WA [10^6 m^3/y^{-1}]	WC_{IRR} [10^6 m^3/y^{-1}]	$WC_{dom\&ind}$ [10^6 m^3/y^{-1}]	WC_{Frac} Low injection scenario [10^6 m^3/y^{-1}]
China	Sichuan Basin	Longmaxi	10278.80	496.82	3650.38	20.90
	Tarim Basin	L. Cambrian	14.24	236.14	36.10	6.23
		L. Ordovician	174.74	49.52	32.82	19.23
		M.-U. Ordovician	30.15	0.00	9.54	20.70
		Ketuer	600.75	1196.93	62.80	15.45
	Junggar Basin	Pingdiquan/Lucaogou	5.69	60.10	25.57	8.32
		Triassic	146.44	22.71	17.11	7.21
Argentina	Neuquen	Los Molles	1482.57	224.45	27.49	12.89
		Vaca Muerta	2864.14	124.75	26.19	11.33

In China, shale exploration and development is underway in the Sichuan, Tarim, and Junggar Basins. The Sichuan Basin is neither affected by groundwater depletion nor water stress. Hence, local water availability does not represent a significant constraint on production (Table 3). Chinese oil companies are procuring water using existing water withdrawal rights from the Wujiang River (the major tributary of the Yangtze River) [Guo et al., 2016]. Conversely, the Tarim Basin and Junggar Basin are located in intensively irrigated areas (Table 3) subjected to water stress (Figure 1) and groundwater depletion (Figure 3). Here, additional water consumption from hydraulic fracturing would likely require a significant fraction of locally available water resources, enhance water stress, and compete with irrigation in the region [Yang et al., 2013].

Shale resource exploration efforts are underway in several countries, including Mexico [Castro-Alvarez et al., 2017], Algeria, Australia, Colombia, South Africa, and India [U.S. Energy Information Administration, 2015]. Their shale deposits are located in water stressed, groundwater depleted or arid areas (Figure 1 and Figure 3). In those regions, careful water resources management plans are required to avoid the enhancement of water stress or further depletion of freshwater aquifers. For example, regulations might require fracking companies to adopt water saving practices (e.g., re-use produced water, sourcing brackish groundwater, invest in low water and waterless technologies, or transport water from farther away), or prohibit oil companies from acquiring freshwater from the agriculture sector.

6.5 Discussion

Many shale deposits worldwide are located in water-scarce regions, where irrigation is critical for crop production and millions of people live. Although their extraction requires a small percentage of the annual local water resources available for human needs, in the long term the development of shale resources in these water-scarce areas could generate a depletion of water resources if water is consumed at rates exceeding those of replenishment by hydrological processes. Further, an increasing recycling volume of fracturing water could make an important contribution to alleviating the depletion of local freshwater resources.

While our analysis accounts for the total potential water consumption of shale development worldwide, many of the assessed shale areas are unlikely to be put under commercial production – for various economic, environmental, social, political, and technical reasons. Moreover, while our results show that large volumes of water will be required, future technological development and water management improvements offer promise for minimizing water appropriations for shale extraction [International Energy Agency, 2016]. For instance, industry is using brackish water – a globally abundant and underutilized resource – and is maximizing the reuse of returning hydraulic fracturing water [Nicot et al., 2014]. Research and development is also focusing on non-water alternatives for hydraulic fracturing fluid, including foams, which can reduce water usage but require more chemicals and extra safety precautions, while limiting the efficiency of hydrocarbon production [International Energy Agency, 2016].

The United States is the global leader in shale oil and gas production, and numerous studies show that water shortage is not a critical issue to the development of shale deposits (e.g., Marcellus, Barnett, Eagle Ford, and Bakken shale deposits) [Barth-Naftilan et al., 2015; Nicot and Scanlon, 2012; Nicot et al., 2014; Scanlon et al., 2014]. Our results are in overall agreement with these findings, in that global water use for shale deposit extraction is dwarfed by the local volumes used in agriculture and other activities. Nevertheless, water consumption by the shale industry would compete with other sectors (e.g., agriculture) in areas with limited water resources, such as Colorado, where recent reports show that shale oil and gas extraction has occurred at the expenses of water availability for irrigation [The New York Times, 2012; The Denver Post, 2015]. Indeed, oil and gas industry is willing to pay a premium price for the small amount of water (relative to agriculture) they use. For example, in Colorado, farmers trying to secure water for irrigation have been outbid by shale developers willing to pay US$0.81 or even

US$1.62 per cubic-meter for auctioned surplus water (vs. US$0.02 to US$0.08, the price farmers would typically pay) [The New York Times, 2012].

Our estimates, which are based on current North American technology and estimated size of extractable hydrocarbon deposits [Kuuskraa et al., 2013], are affected by the uncertainty associated with lack of detailed knowledge on the length of the wells (vertical and lateral), local geology, shale company, number of fracturing stages, type of water used, water recycling, technological, and economic factors [Nicot and Scanlon, 2012; Kondash and Vengosh, 2015; Gallegos et al., 2015]. Moreover, the resolution of the hydrological model used (~50 km at the Equator) and the annual scale of this analysis limit our ability to identify smaller scale impacts. However, the complexity of a global analysis lends itself to a scenario-based approach and to the use of suitable assumptions. These results will serve as a starting point for studies undertaking a finer scale, local analysis of the impacts of shale oil and gas extraction on water supplies.

Our global analysis does not account for regional site-specific factors that can be crucial to the feasibility of hydraulic fracturing in water-stressed areas, where water availability is critical for shale development. Indeed, in water-stressed regions of the United States, shale deposits are currently extracted using brackish water or withdrawing water from freshwater artificial reservoirs. Industry is using brackish groundwater resources in the Permian and Eagle Ford shale deposits (in West Texas and Texas-Mexico border regions, respectively) [Scanlon et al., 2014]. Shale companies in the Bakken shale deposit (in North Dakota) are withdrawing water from Lake Sakakawea, the third largest water reservoir in the United States [Horner et al., 2016]. Future research is required to investigate these site specific factors that could allow for shale oil and gas development even in water-stressed areas and minimize competition for freshwater resources with other human and environmental needs.

6.6 Conclusions

Economic, social, environmental, technical, and policy-related factors will combine to influence commercial-scale production from shale areas in the coming years. For water-scarce or water-stressed areas in particular, the development of shale deposits will need to overcome the additional challenge of regional water limitations and will likely enhance competition for water in many populated or agriculturally important areas. In some of these regions, oil and gas production from shale rocks could place unsustainable pressure on the water resources required to support other human needs. By adopting a hydrologic perspective that considers water availability and demand together, decision makers can better understand the water and food security implications of shale resource development.

6.7 References

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CHAPTER 7

The water-energy-food nexus of unconventional oil and gas extraction in the Vaca Muerta Play, Argentina

Reference: Rosa, L., & D'Odorico, P. (2019). The water-energy-food nexus of unconventional oil and gas extraction in the Vaca Muerta Play, Argentina. Journal of cleaner production, 207, 743-750.

7.1 Abstract

Vaca Muerta is the major region in South America where horizontal drilling and hydraulic fracturing techniques are used to extract unconventional shale oil and gas. Despite the growing interest in the Vaca Muerta resources, there is only a limited understanding of the impacts that their extraction could have on local water resources. This study uses a water balance model to investigate the hydrological implication of unconventional oil and gas extraction in this region. We find that, with current rates of extraction, water scarcity is observed for four months a year. We also find that water consumption per fractured well increased 2.5 times in the period 2012-2016 and produced water from unconventional shale formation sharply increased from roughly zero to 1.15×10^{6} m^{3} y^{-1} in the 2009-2017 period. We estimate that future projected water demand for unconventional oil and gas extraction will increase 2.2 times in the period 2017-2024 reaching 7.40×10^{6} m^{3} y^{-1}, while exhacerbating current water scarcity, likely competing with irrigated agriculture, the greatest water consumer in this semiard region. Produced water recycling, domestic wastewater reuse, irrigation water trade, brackish groundwater use, and waterless unconventional oil and gas extraction technologies are some of the strategies that could be adopted to meet future additional water demand. In the Vaca Muerta adequate water management plans are required to avoid water shortages.

7.2 Introduction

Many countries are transitioning from reliance on conventional to unconventional fossil fuels (Farrel and Brandt, 2006). This transition has been driven by increasing global energy demand (International Energy Agency, 2017), continuing reliance on fossil fuels (International Energy Agency, 2017), technological innovations in the oil industry that have reduced marginal production costs (Brandt et al., 2018), and depletion of conventional oil sources (Bentley, 2002; Sorrell et al., 2010; Höök and Tang, 2013). For example, Canada is producing unconventional oil mainly from oil sands (Rosa et al., 2017), the United States are producing record amount of oil and natural gas from their unconventional shale oil and gas deposits (U.S. Energy Information Administration, 2018), Estonia is retorting oil shale (Raukas and Punning, 2009), Mexico and China are increasing their efforts in unconventional shale oil and gas extraction (Castro-Alvarez et al., 2017; Masnadi et al.,2018), and Venezuela is mining heavy oil (Rosa et al., 2017).

Argentina is also undergoing a transition from conventional to unconventional fossil fuels extraction. Argentina was once the largest oil and natural gas exporter in South America (U.S. Energy Information Administration, 2018a), however, an increase in domestic energy demand and a decline in conventional oil and natural gas production made Argentina a net natural gas and oil importer since 2010 and 2013, respectively (BP, 2017).

In 2014, to prevent a further supply-demand energy imbalance, Argentina implemented a hydrocarbon reform to revive the energy sector and increase investments in mining its unconventional oil and gas (UOG) resources (U.S. Energy Information Administration, 2018a). Argentina has a world-class UOG shale endowment in the Vaca Muerta Play in the Neuquén Basin (Kuuskraa et al., 2013). The formation has 20 billion barrels of technically recoverable oil and 16 trillion cubic meters of technically recoverable natural gas (Kuuskraa et al., 2013). The presence of promising geological resources combined with existing natural gas infrastructures make Vaca Muerta an important region for the next shale boom (Mauter et al., 2014). Indeed, Vaca Muerta is at the preliminary stages of development and it is one of the major regions outside North America that is producing UOG using horizontal drilling and hydraulic fracturing (Suarez and Pichon, 2016).

As the use of hydraulic fracturing – the water demanding technology generally used to extract UOG from low permeability rocks – has become increasingly widespread, significant research has been conducted to determine its water consumption in the United States (Nicot and Scanlon, 2012; Clark et al., 2013; Kondash et al., 2015; Gallegos et al., 2015; Chen et al., 2016; Jiang et al., 2014; Jackson et al., 2014), Mexico (Galdeano et al.,2017), and China (Yu et al., 2016; Guo et al., 2016; Zou et al., 2018). Previous studies have shown that water shortages are not an obstacle to UOG extraction in the United States (Nicot and Scanlon, 2012; Jiang et al., 2014; Scanlon et al., 2014). Indeed, water consumption for UOG extraction is dwarfed by the volumes consumed in agriculture (Rosa et al., 2018). Moreover, fracking companies in the United States are willing to pay a premium price for the small amount of water (relative to agriculture) they use (Rosa et al., 2018). However, it has been estimated that large areas underlain by global UOG shale deposits are affected by water stress where irrigation is critical for crop production (Rosa et al., 2018). In these areas, including Vaca Muerta Play, it is not clear if physical water scarcity can be a constraint on hydraulic fracturing and/or create competition for water allocation in important irrigated agricultural areas between food and energy systems. This competition constitutes the core of the water-energy-food nexus debate (D'Odorico et al., 2018; Lant et al., 2018).

Despite the growing interest in the Vaca Muerta UOG resources, there is only a limited understanding of the pressure that their extraction through hydraulic fracturing could place on local water resources along the year. It remains unclear if water consumption of UOG extraction would compete with the irrigation sector and induce or exacerbate local monthly water scarcity, posing environmental, financial, reputational, and regulatory risks on both the hydrocarbon industry and local communities (Rosa et al., 2018). This limited understanding of the potential impacts of UOG development on the local water balance thus prevents the implementation of a sustainable water management plan in the Vaca Muerta. It is therefore necessary to assess how renewable water availability varies along the year and assess if UOG extraction could lead to an inadequate management of local water resources as well as intensify the competition for water between irrigated agriculture and hydraulic fracturing in the Vaca Muerta Play.

Here we first assess current and projected water consumption and production from hydraulic fracturing activities in the Vaca Muerta Play. We assess the impacts of UOG extraction on the local water balance using a hydrologic approach that links the water consumption of UOG with local monthly renewable blue water availability, blue water consumption from agriculture, municipal and industrial sectors. We then explore the potential impacts hydraulic fracturing might have on regional water users and how future water demand could be sustainably met in the region while avoiding water shortages. We conclude discussing current water management plans, opportunities, and challenges from the development of OUG in the Vaca Muerta.

7.3 Methods

7.3.1 Vaca Muerta Play

The Neuquén Basin includes the Vaca Muerta and the underlying Los Molles sedimentary geologic formations located in Argentina's Patagonia Region (Fig. 1). The Vaca Muerta formation is the primary source rock for hydrocarbons in the Neuquén Basin. It covers four different Argentinian provinces and it has an estimated prospective area of 30,000 km^2 and an average depth greater than 2400 m (Kuuskraa et al., 2013). The Basin is bordered on the west by the Andes Mountains and on the north and south by the Rio Colorado, Rio Neuquén, and Rio Limay. The Basin has a pluvio-nival regime, with most of the precipitation coming from the eastern Andes (Forni et al., 2018). The Rio Limay and Rio Neuquén have six hydropower plants with a total installed capacity of 5000 MW, representing 15% of Argentina's electricity supply (Forni et al., 2018). The Neuquén Basin has both freshwater and brackish or saline aquifers at depths ranging between 0 and 600 m (Magali et al., 2016).

The region is already producing oil and gas from conventional and tight sandstones and it is emerging as the premier UOG development in South America. The climate is semiarid with mean annual precipitation of 178 mm (Mauter et al., 2014). In the Neuquén province, most croplands are irrigated and in year 2010 its population accounted for 551,000 people (INDEC, 2010). The region is an important agricultural area with an intensive production of fruits with 10 million hectares of irrigated lands (Mauter et al., 2014).

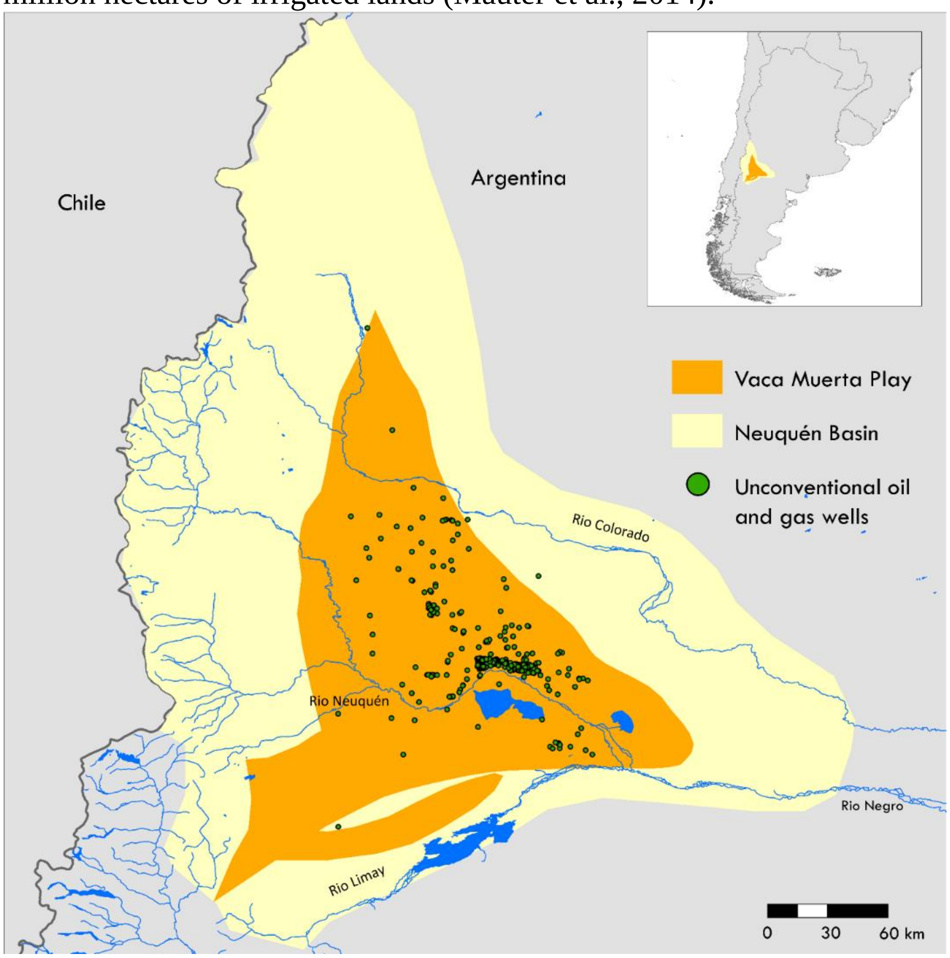

Figure 1. The Vaca Muerta Play in the Neuquén Basin in Argentina. The map shows major rivers, four artificial reservoirs in the footprint of the Vaca Muerta shale play, and the distribution of UOG wells. In the 2010-2017 period, 760 wells were drilled to extract UOG from the Vaca Muerta shale formation.

7.3.2 Assessing water consumption from UOG extraction

The water consumption of UOG extraction (WC_{UOG}) was calculated following Rosa et al. (2018), as:

$$WC_{UOG}\left(\frac{m^3}{month}\right) = (W - F \cdot R) \cdot n \tag{1}$$

where W is the water injected into one well using hydraulic fracturing technology; F is the produced water per well; R is the recycled fraction of the produced water (e.g., flowback and formation water); n is the average number of wells drilled and completed every month in the Vaca Muerta Play.

We used a value of water consumption per well (W) of 22,538 m^3, derived from wells data provided by Johannis and Triffiletti, 2017. This volume was assessed using data from 47 wells drilled in the Vaca Muerta formation between year 2012 and 2016. To our knowledge, this is the only publicly available dataset on water consumption for UOG operations in the Vaca Muerta. Indeed, state agency personnel confirmed that there are no publicly available datasets containing water injected and recycling volumes for oil and gas operations in the Vaca Muerta formation.

Past oil, gas, and water monthly production data for the 2010-2017 period for 760 wells were obtained from the Argentinian Ministry of Energy and Mining public database (Ministerio de Energia y Mineria, 2018). From this dataset we found that produced water (F) was 5,600 m^3 per well during the first year of production. Current and projected recycling fractions (R) were obtained through a personal communication with hydraulic fracturing operators in the Vaca Muerta, who reported a current recycling fraction up to 5% of produced water, and the goal to recycle up to 80% of produced water in the next few years.

To assess the number of wells (n), we considered three drilling scenarios: 1) current (year 2017), 2) recent future drilling forecast from recent well inventories (year 2024), and 3) a hypothetic energy boom scenario. The number of future wells is difficult to forecast since it depends on many parameters such as the rate at which wells are drilled and completed, economic drivers (e.g., oil and gas prices and marginal production costs), infrastructure availability (e.g., drilling rigs, trucks, pumps, water tanks, roads, and pipelines), existing production within the shale play, and technology adopted by shale companies (Rosa et al., 2018). In year 2017, 150 wells were horizontally drilled and hydraulically fractured in the Vaca Muerta Play and it has been forecasted that by year 2024 the number of wells will reach 400 (Wood Mackanzie, 2018). In the energy boom scenario, the projected number of future wells was estimated by evaluating the current drilling intensities in some of the most productive shale plays of the United States and assuming that this rate of drilling would apply to the Vaca Muerta. We consider that 2,400 wells per year will be drilled in the Vaca Muerta. This number has been assessed following the methods by Scanlon et al., (2014) and considering that a well density of 2 wells/km^2 is reached over the 25 years required to drill 60,000 wells at this rate. In the energy boom scenario, the projected number of future wells was estimated by evaluating the current highest established well density and assuming that this well density would expand throughout the play. These projections are not intended to be highly reliable but are simply used to provide a possible scenario regarding future water consumption and production.

7.3.3 Local blue water consumption for human activities

We considered human water appropriation of freshwater resources from the irrigation, municipal, and industrial sectors (Rosa et al., 2018). Previous studies provided an estimate of global monthly irrigation (Davis et al., 2018), municipal and industrial blue water consumptions (Hoekstra and Mekonnen, 2012) at 0.083° resolution (10 km resolution at the Equator). We aggregated these monthly water consumption values to a 0.5° resolution (50 km resolution at the Equator) to match the resolution of the renewable water availability dataset.

7.3.4 Local renewable blue water availability

Monthly renewable blue water availability (WA) was calculated following the methods by Rosa et al., (2018), whereby the value of WA in a grid cell was expressed as the sum of the local monthly blue water availability in that cell (WA_{loc}) and the net monthly blue water flow from the upstream grid cells defined as the local monthly renewable water availability in the upstream cells (WA_{up}) minus the monthly blue water consumption BWC of human activities (i.e., agriculture, municipal, and industrial uses) in the upstream cells (WC_{up}). The net monthly blue water flows were calculated using the upstream-downstream routing "flow accumulation" function in ArcGIS®.

Local renewable blue water availability (surface + groundwater) was calculated as the local blue water flows generated in that grid cell minus the environmental flow requirement (Rosa et al., 2018). Environmental flows define the quantities, qualities, and pattern of water flows to sustain freshwater ecosystems and the ecosystem services they provide (Acreman et al., 2014). Monthly local blue water flows are calculated in every grid cell as the difference between precipitation and evapotranspiration − using estimates by Fekete et al. (2002) − and therefore they account for surface and subsurface runoff (or "blue water flows") generated in that cell as well as for aquifer recharge. Thus, WA_{loc} accounts only for renewable blue water resources that can be sustainably used for human activities and excludes both environmental flows, artificial reservoir storage, and the (unsustainable) depletion of groundwater stocks. To calculate the upstream to downstream water availability we used the flow direction raster (at 0.5° resolution) from the World Water Development Report II (Vörösmarty et al., 2000a-b).

7.4 Results

In the Vaca Muerta, unconventional oil production totaled 8.05×10^{6} m^3 (51×10^{6} barrels), and unconventional natural gas totaled 7.35×10^{12} m^3; in the 2010−2017 period) (Fig. 2). In this period, 760 UOG wells were completed in the Vaca Muerta formation and producing 4.69×10^{6} m^3 of produced water (Fig. 2). Produced water volumes from UOG operations have increased sharply from roughly zero in 2009 to 1.15×10^{6} m^3 y^{-1} in year 2017. Figure 2 also shows that in year 2015 in the Vaca Muerta, as in other major North American shale plays (Scanlon et al., 2016), drilling rates dropped likely in response to a sharp reduction in oil prices.

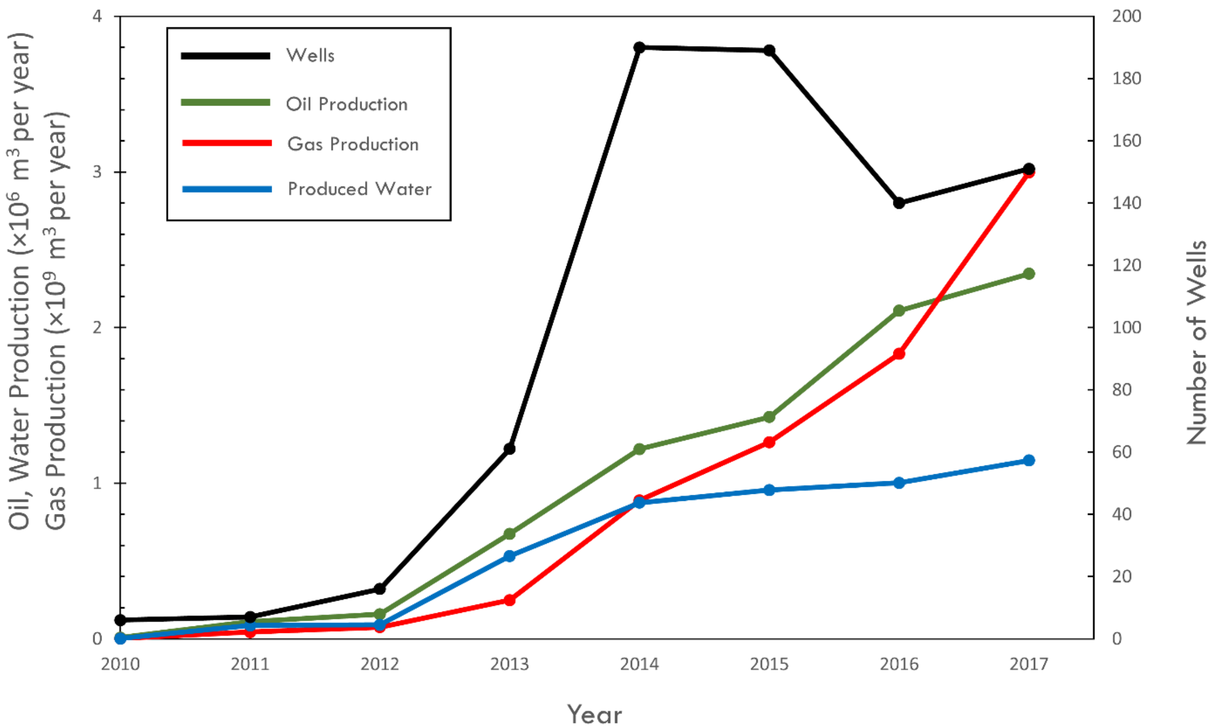

Figure 2. Total annual production volumes of oil, gas, and produced water from unconventional oil and gas wells and the number of wells drilled and completed in the Vaca Muerta shale formation.

7.4.1 Current water consumption

Current (year 2017) water consumption for hydraulic fracturing in the Vaca Muerta (3.41×10^6 m^3) is mostly consumptive because of limited recycling (operators recycle up to 5% of produced water). Mean total annual water consumption in the Vaca Muerta region (666×10^6 m^3 y^{-1}) is dominated by irrigation (84% of total yearly water consumption), followed by municipal (10%), industrial (5%), and hydraulic fracturing operations (1%).

It is important to notice that water consumption per fractured well in the Vaca Muerta Play is increasing due to the use of longer lateral wells, deeper wells, and increased number of fracturing stages (Table 1). Indeed, in year 2017, the average water consumption per well was 22,538 m^3, 2.5 times greater than in 2012 (Table 1).

Table 1. Wells data in year 2012 and 2016 in the Vaca Muerta Play (Johanis and Triffiletti, 2017).

Year	2012	2016
Stimulated Horizontal well length (m)	500	1250
Number of fracturing stages	5	15
Well Depth (m)	2600	3100
Water per 100 m horizontal stimulation (m^3)	1800	
Water consumption per well (m^3)	9,015	22,538

Currently, water scarcity occurs on four months a year (February, March, April, and December) (Table 2). In these months, renewable blue water availability is not sufficient to meet the demand for water for irrigation, industrial, municipal and UOG activities (Fig. 3). We estimate that current hydraulic fracturing activities exacerbate water scarcity by 1.14×10^6 m^3 y^{-1} (0.28×10^6 m^3 $month^{-1}$) in these four months, but do not create water scarcity in other months of the year. Water scarcity occurs in the austral summer when irrigation water consumption is at the highest rates (Table 2). Although there are four artificial reservoirs that can store a volume of about 30×10^9 m^3 of freshwater (Table 3), a capacity that is sufficient to meet annual irrigation needs, lack of distribution systems and the use the dams mainly for hydropower production, prevent to completely meet demand during the irrigation season. In these four water scarce months, water is pumped from groundwater stocks to meet irrigation demand. Indeed, Vaca Muerta is partly located in areas affected by groundwater depletion (Rosa et al., 2018).

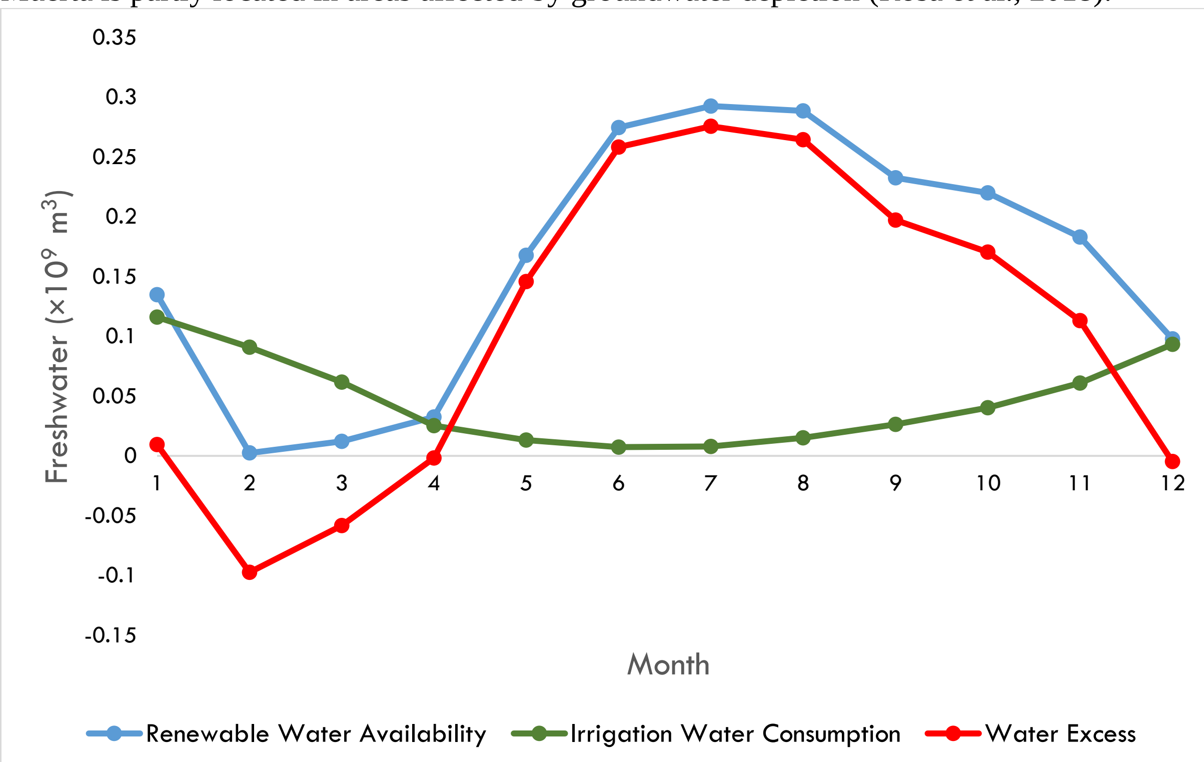

Figure 3. Current monthly water excess, renewable blue water availability, and irrigation water consumption in the Vaca Muerta. Water excess was assessed as renewable blue water availability minus human blue water consumption. Water scarcity is verified when water consumption exceeds renewable water availability.

Table 2. Current (year 2017) monthly water availability and water consumptions in the Vaca Muerta Play. Values for renewable blue water availability are reported after accounting for environmental flows.

Month	Water availability ($\times10^6$ m^3)	Irrigation ($\times10^6$ m^3)	Domestic ($\times10^6$ m^3)	Industrial ($\times10^6$ m^3)	Hydraulic Fracturing ($\times10^6$ m^3)	Water Excess ($\times10^6$ m^3)
January	134.69	116.04	5.81	2.98	0.28	9.58
February	2.39	90.77	5.81	2.98	0.28	-97.45
March	12.27	61.60	5.81	2.98	0.28	-58.40

April	32.54	25.23	5.81	2.98	0.28	-1.76
May	167.88	13.10	5.81	2.98	0.28	145.71
June	274.63	7.32	5.81	2.98	0.28	258.24
July	292.61	7.90	5.81	2.98	0.28	275.64
August	288.64	15.14	5.81	2.98	0.28	264.42
September	232.47	26.25	5.81	2.98	0.28	197.15
October	219.92	40.39	5.81	2.98	0.28	170.46
November	182.92	60.84	5.81	2.98	0.28	113.01
December	97.80	93.43	5.81	2.98	0.28	-4.70

Table 3. Artificial reservoirs in the Neuquén Basin (Messager et al., 2016).

Name	Volume of water stored ($\times10^9$ m^3)
Embalse Los Barreales	2.78
Embalse Mari Menuco	1.38
El Chocón Dam	22.3
Embalse de Casa de Piedra	4

7.4.2 Future trends in water consumption

By 2024, water consumption for hydraulic fracturing is projected to increase 2.2 times in the Vaca Muerta (Tab. 4). We also analyze an energy boom scenario where the projected future number of wells was estimated by evaluating current drilling activities in some of the most productive UOG plays in the United States (Bakken, Eagle Ford, and Permian plays). This scenario is not intended to be highly accurate but is used to provide a possible scenario regarding future water consumption in this region. The projected water consumption is based on 60,000 wells drilled using 2016 parameters (Tab. 1), and a drilling rate of 2,400 wells per year (200 per month). In this energy boom scenario, we estimate that the annual hydraulic fracturing water consumption would reach 44.42×10^6 m^3 y^{-1}, totaling $1{,}110\times10^6$ m^3 over the 25 years required to hydraulic fracture the additional 60,000 wells.

Table 4. Current and future wells and consumption and production water from UOG activities in the Vaca Muerta Play. WC_{UOG} is annual water consumption from hydraulic fracturing operations.

Year	Number of wells per year	WC_{UOG} ($\times10^6$ m^3 y^{-1})	Produced water ($\times10^6$ m^3 y^{-1})	Recycling Fraction of Produced Water
2017	150	3.41	1.15	5%
Scenario 2024	400	7.40	2.24	80%
Energy Boom Scenario	2400	44.42	13.44	80%

We estimate that in this energy boom scenario, the increased drilling rates would not increase the number of months experiencing water scarcity conditions. However, in the four months of water stress (February, March, April, and December) water scarcity will be exacerbated. We estimate that hydraulic fracturing would require an additional 3.42×10^6 m^3 of freshwater per month (41.01×10^6 m^3 y^{-1}). Additional domestic water will also be required to support the needs of the temporary oil field crews and other people that will move to work in the area for economic reasons (Horner et al., 2016). Interestingly, the total water excess in the spring, winter and fall seasons would still exceed the total water deficit in the four months of water deficit even in the most water demanding hydraulic fracturing scenario. Therefore adequate reservoir management and water distribution systems could be sufficient to meet the water needs in this region.

7.4.3 Meeting the increasing water demand

In the United States, water management plans were changed to meet the increasing demand from hydraulic fracturing (Scanlon et al., 2016). We think that in the Vaca Muerta seven strategies could be implemented to meet future additional water demand while avoiding water shortages. First, the future plans to recycle 80% of produced water could save up to 1.68×10^{6} m^3 y^{-1} by year 2024 and 10.06×10^{6} m^3 y^{-1} in the energy boom scenario (Tab. 4). Second, domestic wastewater could also be recycled and reused in either hydraulic fracturing or local irrigation operations. For example, in the nearby city of Mendoza, treated wastewater is already used for irrigation purposes (Vélez et al., 2002). Third, trading irrigation water during the four water scarce months (e.g., through a local water market could be a valuable strategy to meet the energy boom scenario water demand (Debaere et al., 2014; Endo et al., 2018). We estimate that hydraulic fracturing would require the acquisition of 3% of annual regional irrigation water consumption (14.81×10^{6} m^3 y^{-1}). However, this water trading would reduce local irrigated food production, making some croplands water stranded. Forth, as for Lake Sakakawea in the Bakken shale Play in the United States (Scanlon et al., 2016), easements could be provided to access freshwater from the four artificial reservoirs in the region and expanding pipeline infrastructure. A development of water supply pipelines would also reduce truck traffic (approximately 500-1,000 trucks are typically used per well) (Kurz et al., 2016), and consequently reduce greenhouse gases emissions, noise pollution, and improve local air quality (Josifovic et al., 2016; Carpenter, 2016). Fifth, brackish groundwater in the eastern Vaca Muerta could also be used as an alternate water source. For example, brackish groundwater is successfully deployed in the arid Permian Basin in Texas (Scanlon et al., 2014). Sixth, waterless drilling technologies could also be developed and implemented to save freshwater. Research and development is evaluating the feasibility of using foams (International Energy Agency, 2016), liquid nitrogen (Wang et al., 2016), and the development of new techniques that do not require the use of water as alternatives to hydraulic fracturing (e.g., fishbone drilling) (Jorgensen et al., 2014; Torvund et al., 2016). Alternatively, in the absence of appropriate regulations, water withdrawals for hydraulic fracturing could deplete environmental flows in the Rio Limay, Rio Negro, and Rio Neuquén to meet this additional water demand. Dams along these rivers have already modified the natural flow regimes reducing biodiversity and promoting the invasion of non-native species (Poff et al., 1997). A further modification of the flow regimes of these rivers would exacerbate native biodiversity loss (Sabo et al., 2018).

7.4.4 Water regulations

Groundwater has been already depleted in the Vaca Muerta region for agriculture irrigation, i.e. the abstraction rates are higher than the recharge rates (Rosa et al., 2018). To prevent further groundwater depletion policy makers enacted a provincial decree that prohibits groundwater withdrawal for UOG extraction (Ministerio de Energia, Ambiente y Servcios Publicos, 2012). Moreover, Vaca Muerta is located in four Argentinian provinces that have different water regulations. To address these concerns, a River Basin Management plan has been implemented to resolve water demand conflicts in the Rio Negro, Rio Neuquén, and Rio Limay River Basins

(Ministerio de Energia, Ambiente y Servcios Publicos, 2012). However, with the complexity of varying environmental/water regulations in the four different provinces, progress is relatively slow.

7.5 Opportunities and challenges

While water allocation was the primary concern in the early stages of shale development in arid and semiarid regions of the United States, there is an increasing awareness about the management of large volumes of formation water and flowback water (Warner et al., 2013; Kondash et al., 2017; Rosenblum et al., 2017). In the Vaca Muerta produced water reached 1.15×10^6 m^3 y^{-1}, and as drilling rates increase, produced water will also increase and appropriate management plans should be implemented. Produced water from UOG contains high levels of salt and heavy metals and, if disposed of inappropriately, would risk to contaminate both soil and water (DiGiulio et al., 2018). Produced water cannot be disposed into low permeability reservoirs such as shale formations, but is usually disposed into depleted conventional oil reservoirs or non-producing geological formations (Scanlon et al., 2017). Therefore, UOG operators in the Vaca Muerta need to carefully handle wastewater discharge developing adequate wastewater treatment capacity and identifying appropriate location for disposal wells. Alternatively, produced water can be reused for hydraulic fracturing after appropriate treatments (Scanlon et al., 2017).

While in our analysis we consider hydrological impacts of hydraulic fracturing, a transition from conventional to unconventional fossil fuels brings other environmental, economic and strategic impacts (Farrel and Brandt, 2006). Other possible environmental consequences of UOG operations are reduction of regional water quality (Osborn et al., 2011; Rozell et al., 2012; Vidic et al., 2013; Strinfellow et al., 2014), methane migration and groundwater contamination from faulty seals around well casings (Warner et al., 2012; Brantley et al., 2018), impacts on regional air quality (Josifovic et al., 2016; Carpenter, 2016), and seismic triggering associated with the choice to use deep wells as a disposal method for returned fracturing fluids and produced water (Murray, 2013; Davies et al., 2013; Weingarten et al., 2015; Walsh and Zoback, 2015).

UOG extraction has also important social, political, and economic implications. UOG is an opportunity for Argentina to increase its energy security, while reducing costs of fossil fuel imports and potentially changing its import-export imbalance. UOG extraction can create new jobs and enhance economic growth (Peplow, 2017; Neville et al., 2018). However, at local levels, UOG extraction – as for all natural resources extraction – can also be characterized by a short term economic boom during drilling and extraction followed by an economic bust when resources are fully extracted (Christopherson and Rightor, 2012), populations and jobs leave the region, and economic activities decrease altogether. UOG wells and fields exhibit steep production declines within a few years and require large amounts of industrial and financial capital to sustain production (Hughes, 2013). Further, relative to conventional hydrocarbon wells, UOG wells are more expensive and experience more unpredictable commercial flux of hydrocarbons, both of which entail financial risks for fracking companies. Moreover, the development of UOG may have infrastructure challenges even in areas with previous conventional oil and gas extraction. Indeed, in the Vaca Muerta logistic is one of the major constraints to transports proppant, water, workers, and drilling rigs (Gomes and Brandt, 2016).

This study is a first attempt at understanding the hydrological impacts and water demands associated with UOG extraction from shale rocks in the Vaca Muerta region. Our results provide the likely range of water demand for shale oil and gas extraction in the region under current and future conditions. These results could serve as a starting point for policymakers to develop water management plans and evaluate the competition for water that could emerge in dryland shale basins outside North America. Unfortunately, we found that there are no publicly available databases as detailed as the North American Frac Focus (fracfocus.org), which can be used in the study of the hydrological impacts of shale oil and gas extraction, in the Vaca Muerta region. The only repository that is publicly available contains information about oil, gas, and water production per well, but not the quantity of water injected for UOG operations.

7.6 Conclusions

The fracking boom that many expected in the Vaca Muerta has not yet happened, even though Argentinian operators continue to move up the learning curve with strong well performance and lower costs. Vaca Muerta is located in a populated semiarid region, where agricultural production relies on irrigation. Currently the region is affected by water scarcity for four months a year. The increasing volumes of water required to perform hydraulic fracturing will enhance local water scarcity in these four months but will not create additional water scarcity conditions for new months of the year. To avert future water shortages, the four provinces in the footprint of Vaca Muerta Play should coordinate common water management plans to ensure that the increasing water demand will be met with an adequate and sustainable supply, while avoiding further groundwater depletion, environmental flows disruption, and enhancement of water scarcity. By adopting a hydrologic perspective that considers renewable water availability and consumption together, we provide a useful framework for decision makers and local communities to better understand the water implications of OUG development in the Vaca Muerta Play.

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CONCLUSIONS

In this dissertation I developed a quantitative assessment of irrigated agriculture worldwide. I developed a modelling framework to investigate the extent to which agricultural production can be increased by the adoption of sustainable irrigation practices in areas that are presently rainfed. Using a global water balance model approach, I mapped areas in which irrigation can be sustainably expanded without depleting environmental flows or groundwater stocks. I found that 2.8 more billion people can be fed by sustainably expanding irrigation into rainfed areas. I also related existing patterns of unsustainable irrigation to trade flows and consumer decisions afar. I identified the trade flows that are responsible for unsustainable water use highlighting the need for an approach to watershed management that accounts for both physical and virtual water flows (i.e., water flows embodied in the trade of agricultural goods). I also used a process-based hydrological model to identify regions of the world affected by green water scarcity" (i.e., regions where crop production would be enhanced by irrigation), "blue water scarcity" (i.e., lack of water resources to meet the local irrigation water requirements), and "economic water scarcity" (i.e., croplands exposed to green water scarcity, where irrigation water is locally available but investments in irrigation infrastructure are lacking). This analysis identified agricultural areas where crop production is limited by institutional and/or investment capacity.

An important part of this dissertation focuses on energy production and the extent to which it may compete for water resources with agriculture. Part of the work concentrates on fossil fuel extraction, particularly unconventional deposits such as oil sands and shale formations. I estimated the amount of water needed for their extraction, developing a spatially-explicit water balance model at the global scale that accounts for local water uses for shale deposit extraction, agriculture, municipal and industrial needs. This analysis allowed the identification of the areas where competition for water between the energy and agricultural sectors are expected to emerge. I further addressed the issue of water sustainability in the energy sector using an inventory of almost 2000 coal-fired power plants. I investigated whether enough water resources would be locally available to sequester the carbon emitted by such power plants by retrofitting them with modern carbon capture and storage technologies. Likewise, I have assessed the water needs of other approaches to carbon capture and storage, highlighting the extent to which the adoption of such methods can be limited by water availability.

These findings shed light on the importance of freshwater in future decision making. The results of this dissertation have the potential to inform water, energy, and food security policies at global, regional, national, and local scales and to provide new insights to achieve global sustainability targets.