TITLE: Quantifying the link between crop production and mined groundwater irrigation in China

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ABSTRACT

In response to increasing demand for food, Chinese agriculture has both expanded and intensified over the past several decades. Irrigation has played a key role in increasing crop production, and groundwater is now an important source of irrigation water. Groundwater abstraction in excess of recharge (which we use here to estimate groundwater mining) has resulted in declining groundwater levels and could eventually restrict groundwater availability. In this study we used a hydrological model, WBMplus, in conjunction with a process based crop growth model, DNDC, to evaluate Chinese agriculture's recent dependence upon mined groundwater, and to quantify mined groundwater-dependent crop production across a domain that includes variation in climate, crop choice, and management practices. This methodology allowed for the direct attribution of crop production to irrigation water from rivers and reservoirs, shallow (renewable) groundwater, and mined groundwater. Simulating 20 years of weather variability and circa year 2000 crop areas, we found that mined groundwater fulfilled 20% - 49% of gross irrigation water demand, assuming all demand was met. Mined groundwater accounted for 15% - 27% of national total crop production. There was high spatial variability across China in irrigation water demand and crop production derived from mined groundwater. We find that climate variability and mined groundwater demand do not operate independently; rather, years in which irrigation water demand is high due to the relatively hot and dry climate also experience limited surface water supplies and therefore have less surface water with which to meet that high irrigation water demand.

KEYWORDS: groundwater, sustainability, hydrology, agriculture, food security, China

1. INTRODUCTION

Increasing global demand for food over the past several decades has forced agriculture to expand into water-scarce regions and increase irrigation water use substantially [Molden *et al.*, 2007]. With little additional land available for agricultural expansion except in tropical rainforests, future increases in crop production will likely rely on increases in irrigation and intensification, both in China and globally [Molden *et al.*, 2007]. Historically, China's agriculture was concentrated in the wetter southern half of the country, but significant expansion over the past 50 years has led to over 50% of current national crop production occurring in the dry northern regions [Ma, 2006]. Irrigated agriculture has expanded significantly in China in the past 75 years, increasing by more than 35 million hectares since 1939 to 51 Mha of planted land and 79 Mha of harvested land in 2000 [Calow *et al.*, 2009; Portmann *et al.*, 2010]. Groundwater exploitation has underpinned the agricultural intensification of northern China since the 1970's [Calow *et al.*, 2009; Currell *et al.*, 2012], where groundwater accounts for up to 40% of irrigation water [Wada *et al.* 2012]. Declining groundwater levels are threatening to limit the irrigation water supply for China's crop production [Kang *et al.*, 2009; Aeschbach-Hertig and Gleeson, 2012; Syed *et al.*, 2008]. There has been a 15 m drop in groundwater levels in the North China Plain since 1960 [Calow *et al.*, 2009], and the current rate of groundwater depletion is estimated to reach up to 1m per year regionally based on model studies [Aeschbach-Hertig and Gleeson, 2012], and up to 1.5 m per year and 3 m per year in unconfined aquifer systems and deep confined aquifer systems, respectively, based on monitoring data [Currell *et al.*, 2012]. This heavy reliance on groundwater for irrigation is driven largely by lack of sufficient surface water supplies [Wisser *et al.*, 2008; Wada *et al.*, 2012], and Northern China is now considered to be a region of physical water scarcity, i.e., more than 75% of river discharge is abstracted

[Molden *et al.*, 2007]. Global multi-model projections of irrigation water availability show significant reduction in Northern China for irrigation potential from renewable surface water by 2100 due to climate change (using a scenario of high greenhouse gas emissions (RCP8.5) [Elliott *et al.*, 2013]. Aquifer depletion could also significantly decrease irrigation water availability in the future. Despite the importance of groundwater and groundwater depletion for the future of Chinese agriculture, it is currently unknown how much food is produced as a direct result of irrigation with non-renewable groundwater mining.

Konikow and Leake (2014) show that groundwater depletion is a combination of decreasing aquifer storage and capture of water from subsurface storage. Over time, the relative proportion of storage reduction to water capture changes, and increases in subsurface water capture can reduce baseflow in rivers [Konikow and Leake, 2014]. Previous research on groundwater in China has shown that groundwater pumping has indeed lead to capture of baseflow, with implications not only for water quantity but also water quality. In the Hai River Basin in northeastern China, groundwater depletion has been identified as the cause of 40% of waterways drying up, and the disappearance of 194 natural lakes and depressions [Jiang, 2009]. A multi-model study of the Baiyangdian Lake catchment in northern China shows that increased exploitation of water resources over the past 50 years has lead to depletion of both groundwater and surface water levels [Moiwo *et al.*, 2010]. In the Shiyang River Basin in northwest China, localized regions have experienced groundwater level decreases of up to 14 meters, and seasonal drying up of river beds [Kang *et al.*, 2009]. Downstream sections of this basin have become significantly salinized over the past 50 years, and measured total dissolved solids in well water has increased three-fold [Kang *et al.*, 2009]. Salinization due to diminished groundwater

recharge has lead to degradation of riparian vegetation in the Tarim River Basin in western China [*Pang et al.*, 2010]. Further examples can be found in Liu and Zheng (2004), Tang et al. (2004), and Nakayama and Shankman (2013).

Large-scale surface water balance models can simulate the use of both surface water and groundwater for irrigation. Several model-based estimates of irrigation water demand in China have been made, ranging from 220 to 850 km³ yr⁻¹, circa 2000 [*Wisser et al.*, 2008], with most estimates in the range of 350 – 500 km³ yr⁻¹ [*Döll and Siebert*, 2002; *Siebert and Döll*, 2007; *Liu and Yang*, 2010; *Wada et al.*, 2012]. These estimates are from a range of hydrology model studies, which used the Water Balance Model [*Wisser et al.*, 2008], the Water-Global Assessment and Prognosis (WaterGAP) model [*Döll and Siebert*, 2002], the GID-based Environmental Policy Integrated Climate (GEPIC) model [*Liu and Yang*, 2010], the Global Crop Water Model (GCWM) [*Siebert and Döll*, 2010], and the PCR-Global Water Balance (PCR-GLOBWB) model [*Wada et al.*, 2012]. The proportion of irrigation water demand fulfilled by mined groundwater pumping is less well constrained. Groundwater (both renewable and mined in excess of recharge) provides up to 40% of China's irrigation water, and model results from Wada et al. [2012] show that 20 km³ yr⁻¹ (5% of irrigation demand) is drawn from non-renewable groundwater.

Water supply alone does not determine food production; cropped areas, crop choice, soil quality, and management practices all contribute [*Tilman et al.*, 2002; *Foley et al.*, 2012]. Agriculture's vulnerability to changes in water supply will necessarily also depend on these factors, which all vary spatially across China. Crop water productivity, the crop yield gained from one unit of

water, also varies spatially, even within individual watersheds [Cai *et al.*, 2011]. Global-scale studies of unsustainable water supplies and food production have used an empirical method to determine the relationship between water use and crop yields [Siebert and Döll, 2010], but this method is not suited for higher-resolution analysis [Siebert and Döll, 2010]. An alternative approach is to employ a process-based crop growth model that can capture these important spatially-variable production factors [Müller *et al.*, 2013].

In this study, we used a hydrological model in conjunction with a process based crop growth model to evaluate Chinese agriculture's dependence upon mined groundwater, and quantify mined groundwater-dependent crop production across a domain that includes variation in climate, crop choice, and management practices. We computed irrigation water demand, as well as sources of water supply, across 20 years of climate variability. Model years 1981 – 2000 were chosen based on the validation period of the crop growth model. Using these two models allowed us to quantify the amount of food produced as a direct result of irrigation with groundwater mined in excess of recharge. We also defined an index of vulnerability to loss of mined groundwater resources that is a function of the amount of mined groundwater required for irrigation and the productivity of a crop irrigated from that water source. The main objective of this study is to quantify how much of China's food production is directly dependent upon mined groundwater irrigation.

2. METHODS

We used two models to simulate irrigation water demand, irrigated and rainfed crop yields, and crop production due to mined groundwater. A grid-based water balance model [WBMplus,

Wisser *et al.*, 2010] calculated daily fluxes and storage of water between and within different water storage pools (Figure 1). WBMplus was used to estimate the irrigation water demand of different crop types based on weather variables, soil properties, and crop parameters, and tracked the sources of irrigation water available to meet that demand [e.g., Wisser *et al.*, 2008].

WBMplus provided an estimate of the irrigated crop area depended upon mined groundwater by first calculating the total irrigation water demand, then assessing the availability of water from multiple sources. Surface water sources are used first to fully irrigate as much irrigated cropland as possible; the remaining irrigated cropland is assumed to be reliant on mined groundwater.

DNDC [Li *et al.*, 1992; 2007], a process-based crop growth and agroecosystem biogeochemistry model, was used to simulate fully-irrigated and rainfed crop yields for individual crops and multi-cropping systems for all counties in China. WBMplus provided an estimate of the irrigated crop area dependent upon mined groundwater. By computing DNDC's difference between irrigated crop yields and rainfed crop yields to these areas, we estimated the portion of total crop yields directly resulting from groundwater mining.

Model validation for WBMplus is discussed in Wisser *et al.* (2010), and for DNDC in Li *et al.* (1992a), Li *et al.* (1994), and Wang *et al.* (2008). In China, river discharge modeled by WBMplus is compared to station observation monthly mean river discharge from the Global Runoff Data Center [*Global Runoff Data Centre*, 2013] using the Nash-Sutcliffe model efficiency coefficient. WBMplus has a Nash-Sutcliffe coefficient of greater than zero (i.e., a better prediction than projection the observational mean) at 10 of the 14 observation stations, and better than 0.5 in 7 of the 14 stations. DNDC has also been validated for use in China. Wang *et al.* (2008) compared model soil organic carbon to six long-term (10-20 years) datasets from

Northeast, North, Northwest, and Central South China, and found a high level of agreement between all study sites and model results. Qiu et al. (2009) validated DNDC's representation of fertilizer and manure treatment against three study sites in Shanxi Province, China, and validated DNDC's representation of biogeochemical cycling in rice paddies against a study site in Jiangsu Province, China. Yu et al. (2014) validated DNDC maize yields against reported yield data in China and the U.S. under drought and non-drought weather conditions.

2.1 Water Balance Model

WBMplus computed a daily water balance for each 0.5 degree grid cell. Water was input through precipitation and irrigation, and outputs were evapotranspiration, runoff, and shallow rechargeable groundwater (Figure 1). Water was stored as soil moisture and in the shallow groundwater pool. Surface runoff and baseflow from shallow groundwater were transported downstream through the STN-30p river network [Vorosmarty *et al.*, 2000] taking into account the storage of water in large reservoirs. A detailed description of WBMplus's fundamental processes is given by Wisser et al. (2010). Here we describe WBMplus's method of irrigation and crop water use (Section 2.1.1), and WBMplus's updated method of simulating operation of large reservoirs (Section 2.1.2).

2.1.1 Irrigation and crop ET

Irrigation was simulated by abstracting water from rivers, reservoirs, and groundwater, then moving that water to the soil water pool. The amount of water abstracted was determined by an irrigation water demand that was based on root zone soil moisture, crop evapotranspiration, and irrigation efficiency. Evapotranspiration depended on each crop's planting date, growing season

length, growth stages, rooting depth, and a crop water use coefficient. Crop coefficients, growth stages, rooting depths, and depletion factors were from Siebert and Döll (2010), and the coefficient method used was based on FAO recommendations [Allen, 1998]. Each crop's evapotranspiration was calculated as:

$$E_c = K_c ET_0 \quad (1)$$

Where E_c [mm day⁻¹] is the crop's evapotranspiration, K_c [-] is a dimensionless crop- dependent and growth stage-dependent coefficient for crop c , and ET_0 [mm day⁻¹] is a reference-surface potential evapotranspiration. ET_0 was calculated using the Hamon method [Hamon, 1963; Vorosmarty *et al.*, 1998].

Crop evapotranspiration and percolation removed water from the soil moisture pool. When water inputs to the soil moisture pool brought the soil moisture stock above field capacity, the excess was diverted in equal proportions to surface runoff and percolation. Percolated water moved to the shallow groundwater storage pool (implemented as a linear reservoir); water from the groundwater storage pool flowed to the river network as baseflow. A daily soil moisture accounting was done in irrigated areas with inputs from precipitation, irrigation water, and snowmelt, and with evapotranspiration and percolation as outputs from the soil storage. If soil moisture fell below a crop-dependent threshold, C_t [mm], irrigation water was applied to bring the soil moisture up to field capacity. The crop-dependent threshold for soil moisture was calculated as:

$$C_t = C_s * R_d (F_{cap} - W_{pt}) \quad (2)$$

Where C_s [-] is a crop-specific scalar that represents a crop's inability to remove all water from the soil, R_d [mm] is the crop's rooting depth, F_{cap} [-] is the field capacity of the soil, and W_{pt} [-] is the wilting point of the soil. Crop-specific parameters C_s and R_d were from Siebert and Döll (2010), and F_{cap} and W_{pt} are from FAO/UNESCO [2003].

When soil moisture was below the crop-specific threshold, the crop had a positive irrigation demand (otherwise demand was zero). For each grid cell, a net irrigation demand, I_{net} [mm day⁻¹], was calculated daily as the area-weighted water demand of all the crops [26 crop types; taken from *Portmann et al.*, 2010] in the grid cell. Daily net irrigation demand was calculated as:

$$I_{net} = \sum_{c=1}^n A_i \times a_i \times I_c \quad (3)$$

where A_i [-] is the portion of the grid cell's area equipped for irrigation, a_c [-] is the portion of the irrigated area containing crop c (Table 1), I_c [mm day⁻¹] is the irrigation demand of crop c , and n is the number of crops.

Rice paddies require additional irrigation water due to inundation-induced percolation. We assumed that irrigation water was applied to rice paddies to maintain a 50 mm flooding depth throughout the growing season. To achieve this, irrigation water was applied on the first day of the paddy rice-growing season to fill the soil moisture pool 50 mm above field capacity, and each subsequent day water was applied to account for percolation plus evapotranspiration losses

minus precipitation gains. Rice paddy percolation was assumed to occur at a constant rate that is determined by the soil drainage class [e.g., *Wisser et al.*, 2008], estimated spatially from the FAO/UNESCO soil map of the world [*FAO/UNESCO*, 2002].

In each grid cell, water for irrigation could be withdrawn from large reservoir storage (see §2.1.2 below), if present, rivers flowing through the grid cell, shallow renewable groundwater, and mined groundwater (modeled as a distinct pool of water, Figure1). These water sources defined the water volume available for irrigation. In this model, the shallow groundwater pool captures all water that is available for groundwater recharge. While this pool is connected to the river network, it is not connected to the deep groundwater storage pool. This simplified representation of groundwater is used to separate potential groundwater recharge from groundwater that is abstracted in excess of recharge. We recognize that in real groundwater systems, complex interactions between recharge, groundwater abstraction, and capture of surface water due to groundwater pumping exist [e.g., *Konikow and Leake*, 2014], and that this simple representation does not capture those interactions. Rather, we aim to estimate the large-scale demand at a province- to national level for groundwater that exceeds the sum of surface water supplies. Since this water remains in the river, it is counted towards the surface water pool. All water abstracted from the deep groundwater pool is considered here to be potentially non-renewable, since no recharge can refill this supply. WBMplus applied sufficient irrigation water to each grid cell to fulfill the irrigation demand, bringing the soil in the grid cell's irrigated area up to field capacity. This water was first removed from the (renewable) shallow groundwater pool, then from the river discharge and reservoirs. If these water sources were not sufficient to fulfill demand, then mined groundwater was added from a limitless pool to make up the difference. While the

shallow groundwater pool was recharged through infiltration, the mined groundwater pool received no recharge. The mined groundwater pool represented groundwater abstractions in excess of recharge. We assumed that all irrigation demand was met, though in practice this may not always be the case.

Delivery of water from an irrigation water source to the irrigated field is inefficient. An efficiency factor, E_{eff} [0.34 for China [Döll and Siebert, 2002]], was applied to account for these losses. Gross irrigation demand, I_{gross} [mm day⁻¹] was:

$$I_{gross} = \frac{I_{net}}{E_{eff}} \quad (4)$$

Only the efficient portion of irrigation water (i.e., the water equal to irrigation demand) was added to the soil moisture pool. The daily “inefficient” water ($I_{gross} - I_{net}$) was split three ways and returned to other pools or fluxes in each grid cell: evaporation, groundwater water recharge, and surface runoff. The amount of inefficient water that evaporated, I_{evap} , is:

$$I_{evap} = \begin{cases} PET - AET & \text{if } (I_{gross} - I_{net}) \geq (PET - AET) \\ I_{gross} - I_{net} & \text{if } (I_{gross} - I_{net}) < (PET - AET) \end{cases} \quad (5)$$

If $(I_{gross} - I_{net}) > PET - AET$, then the remaining inefficient water was divided evenly between groundwater recharge and river discharge.

The sources of irrigation water, both the efficient and inefficient portions, were recorded by the model. We assumed that the contribution of each water source to the total irrigation water amount was the same for both the efficient and the inefficient portions.

2.1.2 Large reservoirs

Large reservoirs were represented in the model as river grid cells with large river water storage capacity.. Unlike other river grid cells, the rules for outgoing discharge were governed by reservoir operation rules. The model applied the same set of rules to all reservoirs. If the reservoir storage was below 80% full, then water was released as the log of the reservoir level, and if the reservoir storage was above 80% full, then water was released as the exponent of the reservoir level; release at 80% capacity was equal to annual mean river discharge in that grid cell. Minimum allowed reservoir release was 20% of the 5-year average annual discharge [Prusevich *et al.*, 2013].

2.2 Crop Yield Model

The DeNitrification-DeComposition (DNDC) model, a process-based model of carbon and nitrogen biogeochemistry in agroecosystems, simulated irrigated and rainfed crop yields (kg C/ha/yr). The model simulates soil temperature and moisture regimes, soil carbon and nitrogen dynamics, crop growth and yield, nitrogen leaching, and emissions of trace gases for both individual crops and multi-cropping systems, and requires as inputs soil properties, daily weather, and crop management details (e.g., crop type and rotation, fertilization, irrigation, tillage, planting and harvest dates) [Li, 2007a]. DNDC has been used extensively for studies of both Chinese and global agriculture [e.g., Zhang *et al.*, 2002; Wang *et al.*, 2008; Qiu *et al.*, 2009; Li *et al.*, 2010; Deng *et al.*, 2011; Han *et al.*, 2014].

Unlike the gridded spatial structure of WBMplus, DNDC modeled crop growth on a county-based (polygon) system. The model was run from 1981 to 2000 to capture 20 years of weather-

driven crop yield and irrigation water demand variability for each county in China. NASA's MERRA climate reanalysis product [*Rienecker et al.*, 2011] was used for the temperature and precipitation inputs to DNDC, matching each county to the MERRA grid cell closest to the county polygon center. The average county size in China is similar to the size of a MERRA grid cell [$\sim 2,500 \text{ km}^2$]. Data on crops grown in single- and multiple-crop rotations for each of China's ~ 2400 counties came from Qiu et al. (2003). County soil properties used by DNDC (texture, bulk density, pH and carbon content) were from digitization of the Chinese Third National Soil Survey maps [*Shi et al.*, 2004; *Tang et al.*, 2006]. All crops in all counties were simulated with both full irrigation (no water stress) and no irrigation (rainfed) for all 20 years, to capture weather-driven interannual variability in rainfed and irrigated crop yields. For the simulations presented here, to represent general changes in fertilization across China, we applied a constant increase ($\sim 2\% \text{ yr}^{-1}$) in crop-specific fertilizer application rates across China from 1981 to 2000, and a step decrease (36%) in manure application rate in 1990 [*NBS*, 2008]. The crop distribution represented circa 2000 conditions, and simulated crop yields were de-trended with respect to increases in fertilizer and decrease in manure application rates to achieve c.2000 crop yield rates for all 20 years of the simulation, so yield interannual variability resulted from weather effects only. DNDC planting and harvest dates are from Cui et al. (1994), and crop-specific growth parameters are reported in Li (2007b).

2.3 Data

Both WBMplus and DNDC used daily climate drivers of temperature and precipitation from the MERRA NASA reanalysis product, years 1981 - 2000. Soil and non-crop vegetation data were from the FAO/UNESCO soil map of the world [*FAO/UNESCO*, 2003], and the GRanD database

of global dams and reservoirs [Lehner *et al.*, 2011] was used for the location and storage capacity of ~7000 large reservoirs globally. DNDC soil parameters are from the Third National Soil Survey [Shi *et al.*, 2004]. We chose this soil input for the purpose of making this study's results comparable to published DNDC-model based studies by the Chinese Academy of Agricultural Sciences [Teng *et al.* 2006; Qiu *et al.*, 2009; Li *et al.*, 2008]. We use the Soil Map of the World [FAO/UNESCO, 2002] to determine soil drainage classes for WBM, which is the only soil parameter needed for this model. While more recent gridded soil maps have been produced, we find that model sensitivity to soil drainage classes is low (see Section 4.3 below).

The MIRCA2000 gridded crop maps and crop calendars were used to determine crop areas, planting month, and harvest months in WBMplus for each of 26 crop classes, and up to 4 subcrop types [Portmann *et al.*, 2010]. The MIRCA2000 crop maps and calendars include multi-cropping systems. In China, multi-cropping systems include summer/winter combinations of maize/wheat, rice/rice, rice/wheat, rice/vegetables, wheat/wheat, and rice/rice/rice (triple-season cropping) [Portmann *et al.*, 2010]. DNDC's treatment of multi-cropping systems has been validated against field study data [Li *et al.*, 2008; Qiu *et al.*, 2009]. Crop class coefficients (K_c), rooting depths (R_d), and proportional lengths of growing stages were from Siebert and Döll (2010). Crop calendars (for planting and harvesting dates) were downscaled from monthly to daily values by assuming mid-month planting and harvesting dates.

2.4 Analysis

While the Qiu *et al.* (2003) and MIRCA2000 crop distributions for China had substantial overlap, there were enough differences that two data analysis steps were necessary to harmonize the DNDC model output with the WBMplus output and the MIRCA2000 crop maps. First,

county-based DNDC yield data were gridded to the resolution of the WBMplus model (0.5 degree) assigning a crop yield value to each grid cell equal to the area-weighted yields of all counties that overlapped with the grid cell. Second, for each grid cell, the individual crop yields were aggregated to produce one average annual irrigated crop yield value and one average rainfed crop yield value, both in units of $\text{kgCha}^{-1}\text{yr}^{-1}$ (note that multiple crop yields in a single year were aggregated to a single annual average crop yield). All crop production results are therefore based on the composition of irrigated and rainfed crops in DNDC's crop maps, as opposed to MIRCA2000 crop maps. This aggregation allowed for a comparison of land under irrigated versus rainfed cropping.

Both models were run for 20-year periods, 1981 - 2000, to capture impacts of variability in temperature and precipitation on irrigated and rainfed crop yields and irrigation water demand. The crop areas, irrigated areas, and growing season were static, representing year 2000 agriculture [*Portmann et al.*, 2008; *Qiu et al.*, 2003], and all crop yields were de-trended to year-2000 values with respect to time-varying DNDC model inputs (fertilizer and manure). All time series results therefore show variability in water demand and crop yields as a function of climate, and are not meant to be representative of historical trends in crop yields. From 1981 – 2000, the NASA-MERRA climate product reports an average of 640 mm of rainfall per year over agricultural land in China. The driest year for Chinese agriculture was 1992, in which precipitation averaged 596 mm over agricultural land; the wettest year was 1990 with an average of 695 mm of precipitation over agricultural land. The range in inter-annual variation in individual counties or grid cells was greater than the national aggregate of ~10%. From 1981 – 2000, the NASA-MERRA climate product shows a slight increase in annual precipitation over croplands, with a linear trend of 0.4mm/year. Other data products show similar trends; the

APHRODITE Monsoon Asia climate dataset [Yatagai *et al.*, 2012] shows an increasing precipitation trend of 4.5mm/year over cropland, and the Global Meteorological Forcing Dataset for land surface modeling [Sheffield *et al.*, 2006] reports a 2.1mm/year increasing trend in cropland precipitation over the same time period.

Surface water and mined groundwater use for irrigation in each grid cell were assumed to be spread among all irrigated crops proportional to demand. Crop yields directly due to the application of mined groundwater were determined for each grid cell by applying the difference between irrigated and rainfed crop yields to the irrigated area supplied by the mined groundwater:

$$Y_{MinedGW_j} = f_{MinedGW_j} \times A_{I_j} \times (Y_{I_j} - Y_{RF_j}) \quad (6)$$

Where $f_{MinedGW_j}$ is mined groundwater as a fraction of total irrigation water demand for grid cell j , $A_{I,j}$ is irrigated area [ha] in grid cell j , and $Y_{I,j}$ and $Y_{RF,j}$ are irrigated and rainfed crop yields [$\text{kgCha}^{-1} \text{yr}^{-1}$] in grid cell j . Gridded results were aggregated to 31 provincial totals.

Crop yield per unit area due to mined groundwater is a function of both mined groundwater demand and the difference between irrigated and rainfed crop yields. Both of these variables are climate-dependent, but also vary in different ways based on other factors such as local surface water storage, soil properties, crop composition, and management practices. The combined effects of these variables determine how vulnerable each province's crop production is to groundwater depletion. To capture the combined effects of these two variables, we defined crop groundwater productivity, CGP [$\text{kgC ha}^{-1} \text{mm}^{-1}$] to describe the yield gains per unit of mined groundwater for each province p , based on province-level data:

$$CGP_p = \frac{Y_{MinedGW_p}}{MinedGW_p} \quad (7)$$

where CGP_p is the crop groundwater productivity for province p [$\text{kgC ha}^{-1} \text{mm}^{-1}$], $Y_{minedGW_p}$ is the 20-year average crop yield due to mined groundwater in province p [$\text{kgC ha}^{-1} \text{yr}^{-1}$], and $MinedGW_p$ is the 20-year average amount of mined groundwater demand in province p [mmyr^{-1}]. CGP provides a method for directly comparing provinces' combined reliance on both mined groundwater use and yield impact for crop production to each other.

3. RESULTS

3.1 Irrigation Water Demand and Supply

Over the 20 years simulated by WBMplus, Chinese agriculture required an average of $330 \text{ km}^3 \text{yr}^{-1}$ irrigation water withdrawals in order to fulfill gross irrigation demands (Table 2). Irrigation water demand standard deviation over the 20-year simulation was $33 \text{ km}^3 \text{year}^{-1}$, or 10% of demand. This deviation is significantly higher than the deviation in annual average rainfall over cropped land, which is only 5% of the mean. Irrigation water demand varies significantly across China, but generally follows patterns of irrigated area (Figure 2).

WBMplus tracks the sources of irrigation water, grouping them into three categories: 1) rivers and reservoirs, 2) renewable groundwater, and 3) mined/fossil groundwater mining (Figure 3). Note that plotted yields in Figure 3 are based on c.2000 cropping area and management, and only represent interannual variability in weather. Actual grain yields in China increased by about 2.5% - 3% per year during most of 1980 to 2000, due to improved management [FAO AQUASTAT, 2008; National Bureau of Statistics of China, 2008]. Over the 20-year simulation period, the nationally aggregated amount of irrigation water available from the first two sources

are very steady. Rivers and reservoirs supply $115 \text{ km}^3 \text{ yr}^{-1}$, with a standard deviation of only $7 \text{ km}^3 \text{ yr}^{-1}$ and a range of $105 - 134 \text{ km}^3 \text{ yr}^{-1}$. Similarly, the average groundwater recharge supplied to irrigation is $107 \text{ km}^3 \text{ yr}^{-1}$, with a standard deviation of $7 \text{ km}^3 \text{ yr}^{-1}$ and a range of 96 to $125 \text{ km}^3 \text{ yr}^{-1}$. Due to the extremely steady supply of river and reservoir water as well as groundwater recharge, most of the variability in irrigation water demand leads to widely ranging fossil groundwater mining demands. Average mined groundwater demand is $125 \text{ km}^3 \text{ yr}^{-1}$, only slightly over one third of the total average irrigation water demand. However, at its maximum demand of $209 \text{ km}^3 \text{ yr}^{-1}$, mined irrigation water makes up 49% of that year's total irrigation water demand, and at its minimum demand of $58 \text{ km}^3 \text{ yr}^{-1}$ it makes up only 20% of total irrigation water demand.

Demand for mined groundwater varies significantly across China, both in absolute volume and in its relative fraction of total irrigation water demand (Figure 4 and Table 3). These two measures of the importance of mined groundwater do not always vary together. Anhui, the province with the largest irrigated area in MIRCA2000, also has the greatest demand for mined groundwater both in terms of absolute volume ($32 \text{ km}^3 \text{ yr}^{-1}$) and fraction of the province's total irrigation water demand (58%). In contrast, Fujian province has a relatively low mined groundwater demand ($3 \text{ km}^3 \text{ yr}^{-1}$), yet this demand makes up 64% of its total irrigation water demand. Yet a different pattern appears in Shandong province, which has the greatest total irrigation water demand ($33 \text{ km}^3 \text{ yr}^{-1}$), but only 20% of this demand is from mined groundwater.

3.2 Crop Yields

We estimate an average total national crop production of $495 \text{ mega-tonnes (Mt)C yr}^{-1}$ (Table 2)

if all irrigation requirements are fully met on irrigated cropland (as defined by MIRCA2000 irrigated crop maps). Irrigated land produces 315 Mt C, and rainfed land produces 180 Mt C. Grid-cell crop yields for irrigated and rainfed crops are area-weighted averages across all crops and rotations occurring in the grid cell (Figure 5). These yields are direct results from the DNDC model, and reflect the spatial variability of crop maps, soil quality, climate, and management practices, in particular single- vs. multi-cropping. The difference in yields follows a general north-south pattern (Figure 5), with the largest differences in the relatively dry north, and the smallest differences in the wetter south.

3.3 Crop Production from Mined Groundwater

When all irrigation water demands are fully met, mined groundwater directly contributes to a large portion of China's national crop production (Figure 6). We estimate an average of 102 Mt C over the 20 year period modeled, with a wide range varying from 79 to 130 Mt C per year due to weather variability (Table 2). The maximum contribution is seen in the driest weather year, and makes up 27% of the total national production; the minimum occurs in the wettest weather year and is 15% of the total national production. Without mined groundwater, the average national crop production is 393 Mt C/year (Table 2), which is 79% of the 20-year average crop production with mined groundwater used as needed.

There are significant regional differences in crop yields produced from irrigation water (Figure 5 & Table 4). Of all the major agricultural provinces, Sichuan has the highest crop irrigation water productivity, gaining 26 kgC/ha/mm (Table 4). There is a decline in crop irrigation water productivity with an increase in precipitation (Figure 7), which indicates that rainfed crop yields

in the southern high-precipitation regions of China are close to the maximum yield possible given soil quality, crop selection, and management choices.

We use the crop groundwater productivity, *CGP* (Eq. 7), to compare the vulnerability of provinces to the loss of unsustainably mined groundwater. Xinjiang Province had the maximum 20-year average mined groundwater demand, 596 mm/year, and Tianjin Province had the maximum 20-year average crop yield due to mined groundwater, 3,300 kgCha⁻¹. Tianjin also had the highest *CGP*, 11.6 kgCha⁻¹mm⁻¹. Beijing and Xinjiang have a *CGP* of 0 kgC ha⁻¹ mm⁻¹ because simulated use of mined groundwater is zero. Of the major agricultural producers (red in Table 5), Henan, Shandong, and Sichuan have the highest *CGPs* (Figure 8), 7.3, 5.3, and 3.9 kgC ha⁻¹ mm⁻¹, respectively. Demand for mined groundwater and crop irrigation water productivity in these provinces were also the most variable based on changes in climate over the 20-years of climate input modeled. The relative importance of mined groundwater versus crop yield gains due to mined groundwater vary between provinces; Sichuan Province has a high *CGP* mainly due to its high mined groundwater demand, while Henan Province has a high *CGP* because of its large crop yield gains due to mined groundwater. However, all provinces have a roughly linear relationship between low *CGP* and high *CGP* (Figure 8). This relationship is not unexpected, as both variables contributing to *CGP* are affected by climate; dry regions have a high demand for mined groundwater and a large crop irrigation water productivity, while wet regions have the opposite (Figure 8).

4. DISCUSSION

4.1 Comparison to other studies

4.1.1 Irrigation water demand and supply

Our results for China's irrigation water demand are similar to other model-based estimates; previous results range from 220 to 850 km³ yr⁻¹, with most studies estimating an average of 350 – 400 km³ yr⁻¹ circa 2000. These results assume that gross irrigation water demand was always fulfilled and that the infrastructure is in place to change the source of water depending on the availability. While global irrigation water demand estimates also vary, this study's estimate of 330 km³ yr⁻¹ is roughly 10% - 20% of world-wide demand [*Döll and Siebert, 2002; Wisser et al., 2008; Wada et al., 2012*]. FAO's AQUASTAT database reports China's irrigation water withdrawals to be 358 km³ yr⁻¹, and Döll and Siebert (2002) report an irrigation water demand of 364 km³ yr⁻¹ for all of East Asia. Wada et al. (2014) use the PCR-GLOBWB model, which is similar in structure to WBMplus, and report China's irrigation water demand to be 519 km³ yr⁻¹ when the model is driven with the MERRA climate product.

Previous studies also report that China's groundwater withdrawals for irrigation are ~100 km³ yr⁻¹ (Table 6), however the only study that distinguishes groundwater recharge from groundwater mining estimates a significantly smaller amount of mined groundwater, 17 - 27 km³ yr⁻¹ [*Wada et al., 2012*], than this study (58 – 209 km³ yr⁻¹). Wada et al. [2012] also estimate a “nonlocal water resource” demand of 11 km³ yr⁻¹, a water supply that is represented in this study's mined groundwater accounting. The remainder of the difference is likely due to the use of different methods for estimating renewable groundwater, and for partitioning of inefficient irrigation water, which has the potential to significantly alter the groundwater recharge. Wada et al. (2012) allow inefficient irrigation water to contribute to groundwater recharge, limited only by the hydraulic conductivity of the soils underlying irrigated areas. In this study, inefficient irrigation water first evaporates to fulfill the difference between AET and PET, then any remaining water

is divided equally between groundwater recharge and the river network (Eq. 5).

4.1.2 Crop production

This study estimates a similar total crop production as the National Bureau of Statistics of China, which reports year 2000 crop production was 500 Mt C. This estimate is significantly lower than the year-2000 total crop production of 715 Mt C reported by FAO (Table 6). The difference between the FAO's annual production and this study's production is mainly due to differences in rainfed crops, as the irrigated crop production estimates are comparable at 315 MtC (this study) versus 323 Mt C [FAO, 2003]. Our results were consistent with other crop productivity studies that show crop production increases by slightly less than two times due to irrigation [*Siebert and Döll*, 2010; *Molden et al.*, 2007], since irrigated and rainfed harvested areas in China are roughly equal [MIRCA2000; Table 2].

In a study by Ye et al. (2013), the CERES crop models (one for each crop simulated) are used to estimate baseline (years 1961 – 1990) crop production in China, as well as future (2030 – 2050) crop production under various climate change scenarios. Baseline crop yields match well with this study at 532 MtC, while future estimates show increasing yields but make no considerations of future water use or availability. Siebert and Döll (2010) use a gridded hydrology model that is similar in structure to WBMplus, and input the MERRA climate product to simulate global crop water demand. They then use an empirical method that relates the simulated AET/PET ratio to the relative ratio between irrigated and rainfed crop yields to calculate crop production due to consumptive irrigation water [*Siebert and Döll*, 2010]. While they do not report values for China, their results show that total crop production across all of East Asia is 545 MtC, and 78.2%

(or 457 Mt) is from irrigated agriculture.

4.2 Groundwater depletion and crop production

In an analysis with multiple global climate models, gridded crop models, and hydrological models of renewable surface water resources, Elliot et al. (2014) simulated future crop production (up to 2090) and found significant decreases due to a combination of climate change and reduced water supply. However, Elliot et al. (2014) included only surface water supplies in their analysis. We explicitly separated the fossil mined groundwater irrigation from renewable groundwater irrigation and surface water irrigation so that we can estimate irrigation water coming from a non-renewable source. Therefore, our estimates of crop yields due to irrigation from mined groundwater show that without this water source, China's total annual crop production would decrease by 15% - 27%, bringing it down to 352 – 448 MtC (Table 2). Annual grain production in China was ~ 350 MtC in the early 1980s, and ~450 MtC in the early to mid 1990s [NBS, 2008]. Our results assume that groundwater mining occurred at the level necessary to fulfill irrigation water demand after renewable water sources were used. While it is likely that not all irrigated areas are always provided with 100% of their irrigation water demand, there is abundant evidence, both observational and modeling studies, that groundwater depletion is occurring in China, particularly in the North China Plain [Wada *et al.*, 2012b; Aeschbach-Hertig and Gleeson, 2012; Syed et al., 2008; Tang et al., 2013]. Results from the Gravity Recovery and Climate Experiment (GRACE) satellite show contemporary groundwater depletion rates of 8.4 – 14 mm yr⁻¹ in the North China Plain, and further analysis by Tang et al. (2013) shows these rates may underestimate depletion by ~7.6 mm yr⁻¹. This study estimates a national average annual mined groundwater demand of 125 km³/year, which is equal to 1.58 m/yr of depletion averaged

over all irrigated areas, or 0.80 m/yr of depletion over all cropped area.

The North China Plain includes much of Henan, Hebei, Shandong, northern Jiangsu, and northern Anhui provinces, all of which are significant agricultural producers (Table 5). We estimate groundwater depletion levels of 22 mm yr⁻¹ (Sichuan) to as much as 400 mm yr⁻¹ (Shandong) in this region, with an average of 68 mm yr⁻¹. Compared to analysis of the GRACE satellite results [Tang *et al.*, 2013], 22 mm yr⁻¹ in the North China Plain is in agreement, but 68 mm yr⁻¹ is higher than satellite results have shown. This high estimate of regional groundwater depletion is not unexpected due to the model assumption that 100% of irrigation water demand is always fulfilled. However, Zhang *et al.* (2004) report that current irrigation schedules used in the North China Plain have lead to average annual groundwater mining of 200 mm yr⁻¹, which is twice the amount estimated by this study's model results. Yuan and Shen (2013) estimated that net groundwater consumption for irrigation in Hebei province was 15 – 20 km³ yr⁻¹ around the year 2000, which is in close agreement with this study's model results of 18 km³ yr⁻¹ in Hebei province. The five provinces in the North China Plain collectively produce 152 Mt Cyr⁻¹ (31% of total national production). Anhui and Jiangsu rely on mined groundwater for 42% and 49% of their total irrigation water supplies, and Henan, Hebei, and Shandong rely even more heavily on mined groundwater, requiring on average 73%, 77%, and 80% of their annual irrigation water to come from groundwater mining. Without the use of mined groundwater, crop production in the North China Plain would drop to 101 MtC yr⁻¹, a 10% loss in national production.

Groundwater depletion is also occurring in the northern-most and western-most parts of China [Aeschbach-Hertig and Gleeson, 2012; Syed *et al.*, 2008], affecting Heilongjiang, Jilin, Nei Mongol, and Xinjiang Provinces, which are all relatively small agricultural producers (Table 5).

Of these smaller agricultural producers, Jilin has the lowest relative reliance on mined groundwater, at 70%, and Xinjiang has the highest at 80%. Northern provinces also have extremely large differences between irrigated and rainfed crop yields (Figure 5), so that even a small loss of irrigation water causes a substantial decrease in their crop production. These potential losses in crop production may occur not only in the case of complete groundwater depletion, but also in the case of groundwater quality degradation. A national groundwater quality assessment in China revealed that over half of the ~5,000 groundwater samples collected were considered too contaminated for use [Curell, 2014]. Both industrial contamination [Curell, 2014] and wastewater treatment [Tang *et al.*, 2004] contribute to this problem.

We used the crop groundwater productivity, *CGP* (see Eq.7), to assess which provinces are vulnerable to the loss of unsustainably mined groundwater. Vulnerability can be due to either a high reliance on mined groundwater, significant crop yields dependent upon mined groundwater, or a combination of both these factors. Provinces that have a high *CGP* and are large agricultural producers fall along a band of precipitation with precipitation roughly between 1000 and 1500 mm yr^{-1} (Fig.8 inset). South of this band, provinces have low reliance on mined groundwater and small increases to crop yields due to use of mined groundwater. North of this band, provinces have high *CGPs*, but they are not large agricultural producers. Therefore we expect that if groundwater levels continue to drop, the provinces in this precipitation band will have decreased crop yields unless they can secure alternative water supplies or significantly increase crop water use efficiency [e.g., Liu *et al.*, 2010]. China's southern provinces, many of which are significant agricultural producers, are less at risk of aquifer depletion than northern provinces [Döll *et al.*, 2012], and their *CGPs* are low.

4.3 Limitations and uncertainties

Uncertainties, sensitivity analysis, and model validation for WBMplus are discussed in Wisser et al. (2010), and for DNDC in Li et al. (1992a), Li et al. (1994), and Wang et al. (2008). Of these previously-assessed uncertainties, the most important to this study is the uncertainty in irrigation water demand that arises from differences in potential input climate data sets. Wisser et al. [2008] found that differences in input climate can alter the irrigation water demand by up to 42% in China, circa year 2000. We compared the MERRA climate input reanalysis product [Reinecker et al., 2011] used in this study to two other, station data-based climate input products: the Global Meteorological Forcing Dataset for land surface modeling [Sheffield et al., 2006], and the APHRODITE Monsoon Asia data product [Yatagai et al., 2012]. The first of these two products is a coarser, 1.0-degree resolution data set, and the second is a finer-resolution, 0.25-degree data set that has been produced with the goal of achieving greater accuracy in representing monsoon season precipitation across Asia. Annually, MERRA precipitation is on average 19.8% higher than the Global Meteorological Forcing Dataset between 1981 and 2000, and 12.6% higher than the APHRODITE Monsoon Asia precipitation. We find that using these input data sets, the total irrigation water demand increases by 9.0% and 8.9%, respectively. A smaller amount of uncertainty in China's irrigation water demand – 17% to 20% - arises from differences in maps of irrigated crop area in China [Wisser et al., 2008]. Additional uncertainty introduced in this study arises from combining the two models' results, which are based on different crop maps. While the national total irrigated and rainfed areas are similar, MIRCA2000 (WBMplus input) and DNDC's crop maps disagree on which type of crops are grown on 63% of irrigated areas, and 31% of rainfed areas. DNDC also represents crop rotations at a more

detailed level than MIRCA2000 [Qiu *et al.*, 2003]. This uncertainty does not significantly impact the results of this study because both sets of model results have been validated or tested against other studies, and are both predicting results consistent with observational data and other modeling studies. Our conclusions and interpretations only address the total water demand and crop production of all irrigated and rainfed crops, and we do not make any interpretations about individual crop types; this type of analyses would require crop maps with better agreement.

Paddy rice is a major consumer of irrigation water and so representation of paddy water management and water balance has significant impact on WBMplus results for China. Percolation from paddy rice fields is estimated based on soil drainage classes [FAO/UNESCO, 2003], ranging from 2mm/day to 8mm/day in China. Uncertainty in soil mapping is therefore important to quantify. We tested WBMplus's sensitivity to soil drainage class parameters by setting all soil drainage rates to the minimum rate of 2mm/day, and the maximum rate of 8mm/day, and assessing the resulting change in annual mined groundwater demand. We find that increasing all rice paddy percolation to 8mm/day increases the national mined ground water demand by 6.3% (minimum over 20 years is 5.1%, maximum is 8.1%) over the results reported using the Soil Map of the World. Decreasing all rice paddy percolation to 2mm/day decreases the mined groundwater demand by 3.5% (maximum 4.6%, minimum 2.6%). Therefore, uncertainty in soil drainage classes leads to an uncertainty of 9.8% in average annual mined groundwater demand. Based on the FAO/UNESCO Soil Map of the World, China's average rice paddy percolation rate is 3mm/day, which is close to the lowest soil drainage class. Only small portions of northwest China are reported to have a high soil drainage class, which is consistent with other soil maps such as the Global Soil Dataset for use in Earth System Models

[Shangguan, *et al.*, 2014]. It is unlikely that the increase in mined groundwater demand that results from increasing all rice paddy areas to a drainage class of 8mm/day is realistic. There was a significant change in rice paddy water management in China during 1980-2000, with most farmers adopting a mid-season draining or drying management scheme [Li *et al.*, 2002]. This practice was adopted to save labor and energy, and may also conserve water (we are not aware of any studies that have quantified water savings impacts). Paddy mid-season draining/drying has not been implemented in WBMplus, so paddy irrigation water use may be overestimated in our analysis. This over-estimation may be offset by an under-estimation of the water amounts required to initially flood rice paddies. WBMplus's rice paddy flooding implementation applies 50mm of water above soil field capacity, which would be insufficient to inundate many field soils. Finally, macro-scale modeling that relates flooded paddy percolation losses to soil texture may overestimate losses in paddies that have been in managed use for centuries (e.g., the Sichuan Basin), where paddy percolation losses may be lower than would be estimated from regional soil properties.

Several simplifying assumptions about irrigation activity also lead to uncertainty in this study. WBMplus consistently draws irrigation water first from the shallow renewable groundwater pool, second from rivers and reservoirs, and third from mined groundwater. This ordering of abstraction does not consider water quality, nor individual preferences for groundwater versus surface water use. However, sensitivity analysis of this assumption shows that switching the preferred order of the first two water sources results in an average increase in total irrigation water demand of 1.2%. This small impact is due to the fact that the shallow renewable groundwater pool is connected to the river network, providing baseflow. Another simplifying

assumption is the disconnection between the shallow renewable groundwater pool and the mined groundwater pool. The disconnect does not represent the interactions of recharge and fossil water in real groundwater systems, but rather is a mechanism to quantify the water that could contribute to groundwater recharge separately from mined aquifer water. A source of uncertainty to the crop yield conclusions of this study is the assumption that there is a linear relationship between rainfed crop yields and fully irrigated crop yields, and that all irrigated crops are fully irrigated (as opposed to employing deficit irrigation). Deficit irrigation can produce crops with a higher yield than this assumption predicts, with spatial and temporal variation [Molden and Oweis, 2007]. China's annual crop yields are reported to be 500 Mt C [National Bureau of Statistics of China, 2008] to 715 Mt C [FAO AQUASTAT, 2008], while this study estimates 474 – 527 Mt C. This study's underestimation of the lower end of this range is likely due to the underestimation of deficit irrigation crop production.

5. Conclusions

This study is the first to combine a hydrologic model with a process-based crop growth model to simulate national-scale agricultural yield and irrigation water use. This methodology allows for the direct attribution of crop yields to irrigation water from rivers and reservoirs, groundwater, and fossil mined groundwater (Figure 2), as well as computation of the spatially varying crop water productivity from total irrigation water, and the crop groundwater productivity. We find that mined groundwater fulfills 20% - 49% of China's national irrigation water demand, which directly leads to 15% - 27% of national crop production. Crop water productivity and mined groundwater demand vary spatially across China, with the combination leading to a smaller percentage of crop yields dependent upon mined groundwater than the percent of total irrigation

water demand fulfilled by mined groundwater. I.e., while mined groundwater fulfills 20% - 49% of irrigation water demand, crops irrigated with mined groundwater only account for 15% - 27% of national crop production, which indicates that the regions with high crop yields are not all directly dependent upon mined groundwater. This study calculated a crop groundwater productivity to determine which provinces would be most vulnerable to the loss of access to mined groundwater. We found that provinces across central China are most vulnerable due to their combination of significant agricultural production, high demand for mined groundwater, and high crop yield gains from the use of mined groundwater.

China, like all major world agricultural producers, has increased irrigation to the point of over-exploitation of water resources in order to achieve greater levels of food production. While irrigation is typically employed to reduce agriculture's vulnerability to weather and climate variability, in the case of water over-exploitation irrigated agriculture may be vulnerable to changes in the water supply caused by both climate variability and diminishing water resources. We find that these two factors – climate variability and mined groundwater demand – do not operate independently, but rather magnify one another by increasing the demand for irrigation water in a hot and dry year while simultaneously reducing the water available for irrigation use. Due to this magnification, the amount of food production dependent upon irrigation from unsustainable water supplies varies significantly from year to year.

Understanding the sources of irrigation water supply and their relative importance to crop production across China will help provide context for water resource management in China, especially with regards to groundwater storage and the South-North Water Transfer Project.

Global depletion of groundwater aquifers, including the North China Plain, will require future groundwater use to achieve greater levels of sustainability than are seen today. Simultaneously, crop production will need to increase to feed the growing world population. Increases in crop water productivity are required to achieve these two goals. A major finding of this research is that crop irrigation water productivity varies significantly across China and that inter annual weather variability results in widely ranging responses in crop irrigation water productivity, as well as in relative reliance on mined groundwater. The wide range in crop water productivity can help identify the regions in China where climate, soil conditions, and management practices work together to achieve high levels of “crop per drop”. Increasing cropping efficiency will likely be a key component of feeding the world’s growing population in a sustainable manner [Brauman *et al.*, 2013]. Similarly, the distinction between crop yield losses due primarily to yield differences versus reliance on mined groundwater illustrates that different strategies (e.g., increasing rainfed yields versus reducing reliance on mined groundwater) will be best suited to different regions in order to achieve the future requisite increases in crop productivity within the constraints of future water availability.

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Fig. 1

Fig. 2

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Fig. 6

Fig. 7

Fig. 8

Table 1: Data sets input to WBMplus and DNDC.

Data Type	Variables	WBMplus Source	DNDC Source
Climate Drivers	Temperature Precipitation	MERRA	MERRA, (Reinecker et al. 2011)
Soil Properties	Field Capacity Wilting Point Non-crop rooting depth Soil drainage class	FAO Soil Map of the World	Third National Soil Survey (Shi et al. 2004)
Crop Distribution	Irrigated crops and areas Rainfed crops and areas	MIRCA 2000	Qiu et al. 2003
Crop Calendar	Plant Date Harvest Date	MIRCA 2000	Cui et al. 1984
Crop water use parameters	Rooting Depth Crop Coefficient Growth Stages	Siebert and Doll (2010)	Li 2007b

Table 2: China national 20-year mean and range (minimum to maximum) of irrigated and rainfed areas, annual irrigation water demand, crop yields, and the difference between sustainable and unsustainable yields.

Variable	Mean	Range
Irrigated area (ha)	79,100,000	
Rainfed area (ha)	76,600,000	
Irrigation water demand (km ³ /yr)	330	270 - 420
Mined groundwater demand (km ³ /yr)	125	58 - 209
Fully irrigated yield (Mt C/yr)	315	308 - 321
Rainfed yield* (Mt C/yr)	180	164 - 207
Sum: Irrigated + Rainfed yields	495	474 - 527
Sustainable irrigated yield (Mt C/yr)	127	114 - 142
Sustainable rainfed yield* (Mt C/yr)	265	238 - 305
Sum: Irrigated + Rainfed yields	393	352 - 448
Unsustainable minus Sustainable Yields (Mt C/yr)	102 (21%)	79 – 130 (15% - 27%)

* Rainfed sustainable yields are higher than rainfed unsustainable yields because in the sustainable scenario, crops that were previously considered irrigated have been moved into the rainfed category.

Table 3: Province summary of irrigation water demand, and mined groundwater demand.

Province	Harvested Irrigated Area (Mha)	Irrigation Demand (km ³ /yr)	Mined Groundwater Demand (km ³ /yr)	Mined Groundwater Demand (fraction total)
Anhui	6.8	32.4	13.7	0.42
Jiangsu	5.2	25.6	12.5	0.49
Hunan	4.9	19.4	6.4	0.33
Jiangxi	3.6	15.6	5.1	0.33
Zhejiang	2.7	11.8	4.1	0.35
Hubei	3.4	13.1	5.1	0.39
Shandong	6.6	33.3	26.6	0.80
Henan	7.1	24.8	18.4	0.73
Guangdong	2.2	8.4	2.8	0.33
Guangxi	2.5	7.9	2.6	0.33
Hebei	4.6	23.7	18.2	0.77
Fujian	1.3	5.2	1.7	0.33
Guizhou	1.3	4.6	1.5	0.33
Yunnan	2.2	5.3	1.7	0.33
Sichuan	2.6	15.5	10.7	0.69
Chongqing	1.0	6.3	4.1	0.65
Jilin	2.2	5.8	4.1	0.70
Liaoning	2.2	5.6	3.9	0.69
Xinjiang Uygur	3.9	28.6	22.9	0.80
Heilongjiang	2.5	6.1	4.3	0.70
Shaanxi	2.1	6.8	5.1	0.75
Hainan	0.18	0.8	0.26	0.33
Shanghai	0.14	1.1	0.47	0.43
Shanxi	1.45	2.9	2.4	0.83
Nei Mongol	3.3	12.7	10.3	0.81
Tianjin	0.54	2.4	1.8	0.77
Gansu	1.3	3.9	3.1	0.80
Ningxia Hui	0.51	2.5	2.1	0.82
Qinghai	0.49	1.2	0.90	0.75
Beijing	0.15	0.3	0	
Xizang	0.16	0.4	0	

Table 4: Crop yield gains per mm of irrigation water for each of the provinces that contribute to the top 50% of cumulative national yields.

Province	Irrigation Water (mm/yr)	Irr Yield – RF Yield (kgC/ha)	Yield Gain per mm Irrigation Water (kgC/ha/mm)
Sichuan	30.7	810.2	26
Henan	135.0	2163.8	16
Shandong	253.1	1853.5	7
Anhui	219.2	1415.9	6
Hunan	91.7	534.8	6
Jiangsu	267.2	1412.7	5
Guangdong	48.7	182.1	4
Jiangxi	93.4	278.7	3

Table 5: China province 20-year average sustainable and unsustainable crop yields, with 20-year minimum and maximum yields in parentheses.

	Province	Total Crop Yield [Mt C]	Crop Yield without mined groundwater [Mt C]	Cumulative percent of national total crop yield	Cumulative % of total crop yield without mined groundwater
	Hunan	39 (37, 41)	38 (36, 40)	8	8
	Henan	37 (33, 42)	23 (14, 34)	16	13
	Anhui	36 (33, 39)	31 (23, 37)	24	20
	Shandong	29 (25, 32)	14 (5, 25)	30	23
	Jiangsu	28 (27, 30)	24 (19, 31)	36	28
	Jiangxi	27 (26, 29)	27 (25, 29)	42	33
	Sichuan	26 (23, 28)	22 (19, 25)	47	38
	Guangdong	25 (24, 26)	25 (24, 26)	53	43
	Guangxi	24 (23, 25)	23 (22, 24)	58	48
	Hubei	23 (21, 26)	21 (17, 24)	63	53
	Guizhou	22 (20, 23)	21 (20, 22)	67	57
	Hebei	22 (20, 24)	9 (3, 16)	72	59
	Zhejiang	16 (16, 17)	16 (15, 17)	75	63
	Chongqing	15 (13, 17)	13 (10, 15)	79	65
	Yunnan	14 (14, 15)	13 (13, 14)	81	68
	Fujian	14 (12, 14)	13 (12, 14)	84	71
	Shaanxi	13 (10, 13)	7 (4, 10)	87	72
	Jilin	10 (8, 12)	7 (5, 10)	89	74

	Liaoning	9 (7, 10)	6 (5, 8)	91	75
	Nei Mongol	9 (9, 10)	2 (1, 4)	93	76
	Xingjiang	9 (8,9)	2 (2, 4)	95	75
	Heilongjiang	7 (6, 9)	5 (4, 7)	96	77
	Shanxi	5 (4, 6)	3 (1, 4)	97	77
	Gansu	4 (4, 5)	1 (1, 3)	98	77
	Tianjin	3 (3, 3)	1 (0, 2)	99	78
	Ningxia Hui	2 (2, 2)	0 (0, 1)	99	78
	Beijing	1 (0, 1)	0 (0, 0)	99	78
	Hainan	1 (1, 2)	1 (1, 2)	100	78
	Quinghai	1 (1, 1)	0 (0, 1)	100	78
	Shanghai	1 (1, 1)	1 (0, 1)	100	78
	Xizang	0 (0, 0)	0 (0, 0)	100	78

^aProvinces are ordered from largest to smallest average annual yields. Color coding on the left indicates: red = top 50% of cumulative national yields; blue = top 75% of cumulative yields; yellow = top 95% of cumulative yields. See Figure S01 for a map of provinces.

Table 6: Estimates of irrigation water use, groundwater abstraction, and crop production from this study and others.

China Irrigation Water Demand (km³)	Groundwater Abstraction (km³)	Year	Source
270 – 420	58 – 209	2000	This study
403	97 (20 mined)	2000	Wada et al. 2012
537	95	2000	Wada et al. 2014
358	101	2005	FAO AQUASTAT
220 – 850	-nr-	2000	Wisser et al. 2008
358	-nr-	2005	Jiang 2009
364*	-nr-	1995	Doll and Siebert 2002
China annual crop production (Mt C)	China irrigated annual crop production (Mt C)	Year	Source
474 – 527	315	2000	This study
500	-nr-	2000	China Statistical Yearbook
532	-nr-	1961 - 1990	Ye et al. 2012
715	323	2000	FAO AQUASTAT
545*	457*	1998 - 2002	Siebert and Doll 2010
474**	-nr-	2009	Fan et al. 2012

-nr- results not reported.

*Estimates for all of East Asia

**Cereal production only