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Salinity impacts on irrigation water-scarcity in food bowl regions of the US and Australia

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Abstract

Irrigation water use and crop production may be severely limited by both water shortages and increased salinity levels. However, impacts of crop-specific salinity limitations on irrigation water scarcity are largely unknown. We develop a salinity-inclusive water scarcity framework for the irrigation sector, accounting for crop-specific irrigation water demands and salinity tolerance levels and apply it to 29 sub-basins within two food bowl regions; the Central Valley (CV) (California) and the Murray–Darling basin (MDB) (Australia). Our results show that severe water scarcity (levels >0.4) occurs in 23% and 66% of all instances (from >17 000 monthly crop-specific estimates) for the CV and MDB, respectively. The highest water scarcity levels for both regions occurred during their summer seasons. Including salinity and crop-specific salinity tolerance levels further increased water scarcity levels, compared to estimations based on water quantity only, particularly at local sub-basin scales. We further investigate the potential of alleviating water scarcity through diluting surface water with lower saline groundwater resources, at instances where crop salinity tolerance levels are exceeded (conjunctive water use). Results from the CV highlights that conjunctive water use can reduce severe water scarcity levels by up to 67% (from 946 monthly instances where surface water salinity tolerance levels were exceeded). However, groundwater dilution requirements frequently exceed renewable groundwater rates, posing additional risks for groundwater depletion in several sub-basins. By capturing the dynamics of both crops, salinity and conjunctive water use, our framework can support local-regional agricultural and water management impacts, on water scarcity levels.

1. Introduction

One of the United Nations Sustainable Development Goals aims to substantially reduce the number of people suffering from water scarcity (UN 2015). Estimates suggest around 30% of the world's population are affected by water scarcity during parts of the year, with increasing risks in the future, due to increasing water demands with increases in population, higher living standard and changing consumption patterns (Wada *et al* 2016, Boretti and Rosa 2019). Traditionally, water scarcity is estimated through water quantity considerations only, as the ratio between the total volume of withdrawn freshwater to the total available volume of renewable freshwater (Alcamo *et al* 2003, Cui *et al* 2018). However, research has shown that poor water quality may severely limit its usability for different sectors, urging for both quantity and quality aspects to be considered within water scarcity frameworks (Zeng *et al* 2013, Liu *et al* 2016, 2017, van Vliet *et al* 2021). Water quality deterioration is also increasing in many regions (Damania *et al* 2019, Strokal *et al* 2021, Beusen *et al* 2022), posing additional limitations on the usability of water resources for different water use sectors.

Irrigation is the largest water use sector, sustaining over 40% of global food production (Wisser *et al* 2008). Water scarcity is a common problem for the irrigation sector, which may be driven by both water shortages and water quality issues (Pereira *et al* 2002). Salinity is one of the most critical water quality parameters for irrigation, because of the potential crop yield reductions that can result from the use of saline water and soil-salinity build-up (Russ *et al* 2020). Elevated salinity levels could thus severely limit the usability of water resources for irrigation and could increase water scarcity for this sector (Assouline *et al* 2015, Velmurugan *et al* 2020). Both water scarcity impacts for crop production (Caparas *et al* 2021), as well as salinity impacts on water scarcity (Jones and van Vliet 2018, Vliet *et al* 2021) have been studied separately. However, joint impacts of both crop- and salinity aspects on irrigation water scarcity are largely unknown. Addressing knowledge gaps of irrigation water scarcity from both the quantity and salinity perspective is a main scientific challenge with importance for sustainable crop production.

Further, irrigated agriculture is in most regions depends on both surface—and groundwater resources and its conjunctive use (Liu *et al* 2020, Lopez *et al* 2022). Conjunctive water use refers to the utilisation of both surface- and groundwater resources in an integrated way, in order to deal with water shortages or increasing water demands. Using groundwater resources to temporally buffer limited surface-water supplies (e.g. during or after droughts), reduces irrigation water scarcity and crop water stress (Zhang 2015). Conjunctive water use may also be utilised for dilution of saline irrigation water, to avoid exceeding crop-specific salinity tolerance levels and risks of yield reductions. Although groundwater quantity has been included in traditional water scarcity approaches, its consideration in a quality (salinity)-inclusive approach, as well as its potential role in alleviating water scarcity through dilution, has also been lacking.

This study aims to address the knowledge gaps outlined above, by developing a salinity-inclusive and crop-specific water scarcity framework accounting for conjunctive groundwater—surface water use for the irrigation sector. Our water scarcity framework can be applied at multiple scales (local to global), depending on the extent of available input datasets. For this study, we use a data-driven approach and focus our assessments on two important food-bowl, irrigated regions of the US and Australia with sufficient availability of monitoring data; the Central Valley (CV) in California and the larger Murray–Darling basin (MDB) in Australia. Specifically, irrigation water scarcity is estimated by accounting for crop-specific irrigation water demands and salinity tolerance levels on a monthly, sub-basin scale, including both surface—and groundwater resources. Secondly, we investigate the potential of conjunctive water use for alleviating irrigation water scarcity and discuss its suitability as a management option.

2. Methods

2.1. Salinity-inclusive water scarcity framework

Our salinity-inclusive water scarcity framework builds on earlier approaches including both quantity and quality dimensions (van Vliet *et al* 2017, 2021, Ma *et al* 2020). Our approach is based on sub-basin spatial scales and monthly temporal scales, and uses observed data from the two study regions as far as possible (further described in sections 2.3–2.5 and in the supplementary methods and supplementary tables 1 and 2). In previous approaches, water scarcity is estimated by comparing the ratio of sectoral water withdrawals of acceptable quality to water availability. Further, although some water scarcity approaches consider both the surface- and groundwater system (Mekonnen and Hoekstra 2016, Graham *et al* 2020), these approaches focus only on water quantity dimensions. We extend previous approaches by explicitly considering both surface and renewable groundwater resources, from both quantity and quality aspects. Our approach focusses on the irrigation water use sector, and on salinity as a key water quality parameter. Specifically, the water scarcity framework is developed to account for crop-specific salinity tolerances and crop-specific surface- and groundwater irrigation water withdrawal rates (figure 1).

We also include conjunctive water use and investigate its potential for alleviating water scarcity. Groundwater is used to dilute surface water volumes to meet crop-specific salinity tolerance levels when tolerance levels are exceeded for surface water, but not for groundwater. This concept of sectoral-specific dilution to reach acceptable water quality levels has previously been introduced (Ma *et al* 2020, van Vliet *et al* 2021). However, we here consider existing groundwater resources of both suitable salinity and availability to meet the dilution need, rather than considering external desalinated resources (van Vliet *et al* 2021), due to its minor contribution to the irrigation water use sector (Jones *et al* 2021).

Specifically, salinity-inclusive irrigation water scarcity is estimated for each monthly timestep and sub-basin according to equation (1) below

$$\mathrm{WS_{irr}} = \frac{\sum_{c}^{n} \left(D_{\mathrm{sw,irr,c}} + D_{\mathrm{gw,irr,c}} + dq_{\mathrm{sw,irr,c}}\right)}{(\mathrm{Q} - \mathrm{EFR})} \quad (1)$$

where $\mathrm{WS_{irr}}$ = salinity-inclusive irrigation water scarcity (−), $D_{\mathrm{sw,irr,c}}$ = irrigation water withdrawal from surface water resources for crop type c ($\mathrm{m^3\,s^{-1}}$); $D_{gw,irr,c}$ = irrigation water withdrawal from groundwater resources for crop type c ($\mathrm{m^3\,s^{-1}}$); Q = water availability, using observed discharge ($\mathrm{m^3\,s^{-1}}$); EFR = environmental flow requirements ($\mathrm{m^3\,s^{-1}}$);

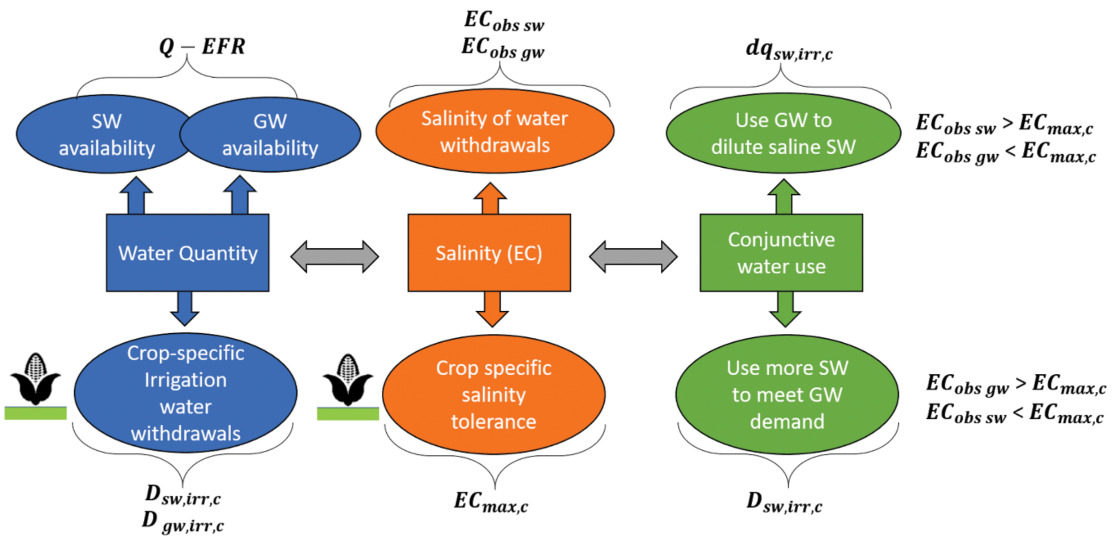

Figure 1. Conceptual illustration of the here developed water scarcity framework, tailored specifically to irrigation water use as the main sector and salinity as a key water quality constituent, using crop-specific salinity tolerance levels ($\mathrm{EC}_{\mathrm{max,c}}$) and including alleviation methods through conjunctive water use ($dq_{\mathrm{sw,\,irr,\,c}}$). Each of the illustrated components (water availability; irrigation water use; salinity and conjunctive use) are estimated per month, crop and sub-basin, using local-regional datasets (supplementary tables 1 and 2). All included variables are further described in equations (1)–(3).

$dq_{\mathrm{sw,\,irr,\,c}}$ = conjunctive groundwater use for dilution to obtain acceptable surface water salinity levels for each crop type, c ($\mathrm{m}^3\ \mathrm{s}^{-1}$).

As mentioned above, at each monthly timestep where surface water salinity (measured as electrical conductivity; EC) exceeds tolerance levels for each crop ($\mathrm{EC}_{\mathrm{obs,\,sw}} > \mathrm{EC}_{\mathrm{max\,c}}$), the increasing water demand for diluting the surface water below the tolerance levels is met by conjunctive water use, under the condition that the groundwater is of enough quality ($\mathrm{EC}_{\mathrm{obs\,gw}} < \mathrm{EC}_{\mathrm{max\,c}}$). On the other hand, if groundwater exceeds tolerance levels for a crop ($\mathrm{EC}_{\mathrm{obs,\,gw}} > \mathrm{EC}_{\mathrm{max\,c}}$), but surface water does not ($\mathrm{EC}_{\mathrm{obs,\,sw}} < \mathrm{EC}_{\mathrm{max\,c}}$), we instead allocate additional surface water to meet the total irrigation water demand for that crop ($D_{\mathrm{sw,irr,c}} + D_{\mathrm{gw,irr,c}}$), since dilution of groundwater with additional surface water is not a common practise (Singh 2014). Specifically, additional groundwater volumes needed for dilution of surface water ($dq_{\mathrm{sw,\,irr,\,c}}$) are estimated through a mass balance approach (equations (2) and (3)) in line with previous work (Ma *et al* 2020, van Vliet *et al* 2021), but further developed to account for crop specific salinity tolerance values and conjunctive water use. According to our approach, salinity loadings of the surface water irrigation demand, plus the groundwater salinity loadings of the water needed for dilution, must be equal or less to the maximum crop tolerance level, times the sum of water volumes for irrigation and dilution (equation (2))

$$\left(\mathrm{EC}_{\mathrm{obs,sw}} \cdot D_{\mathrm{sw,irr,c}}\right) + \mathrm{EC}_{\mathrm{obs,gw}} \cdot dq_{\mathrm{sw,irr,c}} = \mathrm{EC}_{\mathrm{max\,c}} \cdot \left(D_{\mathrm{sw,irr,c}} + dq_{\mathrm{sw,irr,c}}\right). \quad (2)$$

We then derive the equation used for estimating conjunctive use (equation (3)), by solving for the dilution need ($dq_{\mathrm{sw,irr,c}}$);

$$\Rightarrow \mathrm{EC}_{\mathrm{obs,sw}} \cdot D_{\mathrm{sw,irr,c}} + \mathrm{EC}_{\mathrm{obs,gw}} \cdot dq_{\mathrm{sw,irr,c}} = \mathrm{EC}_{\mathrm{max\,c}} \cdot D_{\mathrm{sw,irr,c}} + \mathrm{EC}_{\mathrm{max\,c}} \cdot dq_{\mathrm{sw,irr,c}}$$

$$\Rightarrow \mathrm{EC}_{\mathrm{obs,sw}} \cdot D_{\mathrm{sw,irr,c}} - \mathrm{EC}_{\mathrm{max\,c}} \cdot D_{\mathrm{sw,irr,c}} = \mathrm{EC}_{\mathrm{max\,c}} \cdot dq_{\mathrm{sw,irr,c}} - \mathrm{EC}_{\mathrm{obs,gw}} \cdot dq_{\mathrm{sw,irr,c}}$$

$$\Rightarrow D_{\mathrm{sw,irr,c}} \cdot \left(\mathrm{EC}_{\mathrm{obs,sw}} - \mathrm{EC}_{\mathrm{max\,c}}\right) = dq_{\mathrm{sw,irr,c}} \cdot \left(\mathrm{EC}_{\mathrm{max\,c}} - \mathrm{EC}_{\mathrm{obs,gw}}\right)$$

$$\begin{array}{ll} dq_{\mathrm{sw,irr,c}} = \dfrac{D_{\mathrm{sw,irr,\,c}} \cdot \left(\mathrm{EC}_{\mathrm{obs,sw}} - \mathrm{EC}_{\mathrm{max\,c}}\right)}{\mathrm{EC}_{\mathrm{max\,c}} - \mathrm{EC}_{\mathrm{obs,gw}}}, & \text{if } \mathrm{EC}_{\mathrm{obs,\,sw}} > \mathrm{EC}_{\mathrm{max\,c}} \\ dq_{\mathrm{sw,irr,c}} = 0, & \text{if } \mathrm{EC}_{\mathrm{obs,\,sw}} \leqslant \mathrm{EC}_{\mathrm{max\,c}} \end{array} \quad (3)$$

where $EC_{obs,sw}$ = observed EC (μS cm^{-1}) of surface water; $EC_{obs,gw}$ = observed EC (μS cm^{-1}) of the groundwater and $EC_{max\,c}$ = maximum salinity tolerance for crop type c (μS cm^{-1}).

2.2. Definitions of water scarcity levels and instances

Water scarcity levels of our salinity-inclusive irrigation water scarcity index (WS_{irr}) are classified according to thresholds used by previous studies, with 'severe water scarcity' levels identified as equal or higher than 0.4 (Hanasaki *et al* 2013, van Vliet *et al* 2021). We also introduce two new water scarcity classes, which apply to estimates where the irrigation water demand cannot be met (for a specific crop, month and sub-basin). Firstly, the 'fully impaired (salinity) water scarcity' level occurs when both surface and groundwater resources exceed crop-specific salinity tolerance levels (i.e. conjunctive water use cannot be applied) and the demand cannot be met. Secondly, the 'fully impaired (quantity) water scarcity' level occurs when water availability is zero (i.e. Q − EFR = 0) and the irrigation demand cannot be met.

As outlined above, each water scarcity level is estimated per crop, month over the study period and sub-basin (figure 1 and supplementary figure 1), by accounting for each crop's specific salinity tolerance level (supplementary tables 3 and 4) and irrigation water demand. We hereafter refer to each such monthly, crop-specific and sub-basin water scarcity level estimate as an 'instance'. For presentation of our results, the amount (%) of instances per water scarcity level are compared to the total number of instances across all water scarcity levels. For the Central Valley, the full dataset includes 10 sub-basins, 15 crop-classes and 120 months over the period 2008–2018, leading to a total number of water scarcity instances of 17 186. For the MDB, the full dataset includes 19 sub-basins, 12 crop-classes and 120 months over the period 2008–2018, leading to a total number of water scarcity instances of 17 128.

In addition to the salinity-inclusive irrigation water scarcity index (WS_{irr}) developed here, we also compare our results to estimates based only on water quantity, through the standard water withdrawal to availability ratio (e.g. Vanham *et al* 2018, Oki and Kanae 2006 and references therein). Further, to investigate the impact of crop-specific salinity tolerances on water scarcity levels, we also quantify water scarcity both with and without using crop-specific salinity tolerances. For quantifications excluding crop-specific salinity tolerance levels, we instead used the global irrigation average threshold value of 700 μS cm^{-1} (Ayers and Westcot 1985).

2.3. Selection and data processing for sub-basins within main irrigated regions of the US and Australia

We focus our assessments on two main agricultural irrigated regions in the world; the CV in California, United States, and the MDB in Australia. These two regions were selected based on their importance for regional food production, being considered as the 'food bowls' of both the US and Australia. The CV region accounts for 75% of all irrigated area of California and produces the majority of nuts, vegetables and fruits across the US (Gebremichael *et al* 2021). The MDB is also an agricultural centre of Australia, with crop production accounting for around 40% of the country's total agricultural revenue (Wei *et al* 2011) and consisting of over 9000 irrigated agriculture businesses (MDBA 2021). Both these regions are also known for experiencing issues related to water shortages and salinisation (Wichelns and Qadir 2015, Hart *et al* 2020), making them relevant case study regions.

Due to the use of a predominantly data-driven approach, we selected specific sub-basins within these regions based on the availability of observed salinity data. EC data was synthesised using our global salinity dataset (Thorslund *et al* 2020), in combination with local dataset (supplementary section 1 and supplementary tables 1, 2). Initially, monitoring locations with available monthly (surface water; river; $EC_{obs,\,sw}$) and annual (groundwater; $EC_{obs,\,gw}$) observed salinity data within the period 2008–2018 were selected for each region. Due to the long-term focus of our assessments and the lack of complete salinity timeseries (a common issue for water quality data), all surface water stations with less than 15% data gaps (i.e. up to 18 months) were included. Due to the low availability of monthly groundwater salinity data, as well as due to its generally low intra-annual variability (supplementary figure 2), we included all groundwater locations with at least one measurement per year, over the full 2008–2018 period.

From this selection, we further included only salinity monitoring locations in irrigated areas, which were identified using model outputs from PCR-GLOBWB 2 (Sutanudjaja *et al* 2018; supplementary section 1). Finally, we estimated sub-basin average monthly surface water and annual groundwater salinity timeseries for each sub-basin, by spatial averaging of the data from all identified monitoring locations within irrigated parts of each sub-basin (supplementary figure 1).

2.4. Crop classification, salinity tolerance and irrigation water use

Crop-specific salinity tolerance levels were based on reported guideline values for irrigation water use, using local data sources as far as possible (supplementary tables 3 and 4). We consider the lowest salinity threshold guideline values, at which yield potential start to decrease if these crop-specific salinity levels in the applied irrigation water are exceeded. For crops missing salinity tolerance levels, we assumed the global average irrigation salinity threshold of 700 μS cm^{-1} from the Food and Agriculture Organization (FAO) (Ayers and Westcot 1985).

Crop-specific irrigation water use was estimated from reported local-regional datasets (for the MDB region), as well as from crop-specific land cover datasets, combined with crop-specific irrigation application rates (for the CV region; for more details see supplementary section 2). Due to the large numbers of different crops (e.g. the CV has >100 types), reclassification was made by grouping several crop types into one, based on the following criteria: similar crop types, similar maximum salinity tolerance and similar economic value. Crop-specific salinity tolerance levels were gathered from literature (supplementary tables 3 and 4 for CV and MDB, respectively).

To estimate the proportion of irrigation volumes taken from surface- and groundwater sources respectively, we used reported data on irrigation withdrawals (not crop-specific) from surface—and groundwater, at the scale closest to the sub-basin level as possible (county scale for CV sub-basins and sub-basin scale for the MDB). These ratios were then multiplied with the reported total irrigation withdrawal volumes, to get surface- and groundwater specific irrigation withdrawals per year and sub-basin. Annual crop-specific irrigation quantifications were then scaled to monthly values (supplementary section 3).

2.5. Water availability and environmental flows

Water availability (Q) was estimated from observed discharge data (mean monthly and annual) for each sub-basin, by synthesizing data from the most downstream monitoring station within each sub-basin (supplementary tables 6 and 7). Observed discharge was assumed representative of both surface water and groundwater availability, since renewable groundwater resources are part of observed streamflow, through the baseflow component (e.g. Winter *et al* 1998). Similarly, as for salinity, discharge stations with less than 15% monthly timeseries data gaps were included. Environmental flow requirements (EFRs) of the integrated ground- and surface water system (i.e. water availability Q) were quantified using the monthly variable flow method (Pastor *et al* 2014). To investigate the sustainability of conjunctive water use as a potential measure for water scarcity alleviation, we also quantified groundwater availability separately, using modelled data of groundwater renewable rates (recharge + irrigation return flows) and compared these to the calculated dilution rates. Full details on water availability processing is given in supplementary section 4.

3. Results

3.1. Salinity-inclusive water scarcity: impact of crop types

Crop distributions, used to estimate water scarcity for each sub-basin are shown in figure 2. Dominating crops in the CV sub-basins are forage crops (pasture for grazing) and nuts, which on average account for 43% (range of 25%–58% across sub-basins) and 14% (range: 2%–28%) of total irrigated areas respectively (figure 2(a)). Crops are relatively evenly distributed across the CV region, with some higher dominance of forage crops in the eastern sub-basins compared to the rest. Rice is also grown to a large extent in the three most northern sub-basins (Sacramento Stone-Corral, Honcut Headwaters and Upper Coon) while alfalfa is commonly grown in the west and southern sub-basins. Crops in the MDB are more heterogeneously distributed, with most of the sub-basins having multiple crops. For instance, both grapes, cereals and forage crops, on average, account for 13% of total irrigated areas throughout the region. Cotton is also a common crop, mainly grown in the central and northern sub-basins, where it reaches up to 80% of total irrigated area and (figure 2(b)). In contrast, grapes dominate irrigated areas in the southern sub-basins (35%–62% of total irrigated area across these sub-basins).

Regionally, our salinity-inclusive water scarcity estimations show that severe water scarcity (WS > 0.4) occurs in 23% (3952 out of 17 186) and 66% (11 304 out of 17 128) of all instances (i.e. monthly, crop-specific and sub-basin water scarcity level estimates; see section 2.2), for the CV and MDB region, respectively (figures 3(a), (b) and supplementary tables 8, 9). Locally, in the Central Valley, severe water scarcity due to ‘fully impaired (salinity)’ levels (i.e. when salinity levels of both surface and groundwater withdrawals exceed crop salinity tolerance levels), is predominantly occurring in the three near-coastal San Joaquin sub-basins (delta; lower and middle San Joaquin; figure 3(a) and supplementary figure 5). The Middle San Joaquin is also the sub-basin affected to the highest degree by severe water scarcity levels (55% of its total instances). These water scarcity patterns are related to the crop distributions; the majority being forage crops (pasture for grazing class) and nuts. Forage crops have the highest irrigation application rates (supplementary table 5) while nuts have low salinity tolerance (supplementary table 3). Both these factors, alone or combined, may drive instances of severe water scarcity to increase.

The Upper Coon also stands out, but rather driven by water quantity issues, with 66% severe water scarcity instances, due to ‘fully impaired (quantity)’ levels. This sub-basin has low natural water availability (i.e. discharge rates close to zero), resolved by importing water for irrigation to cope with natural

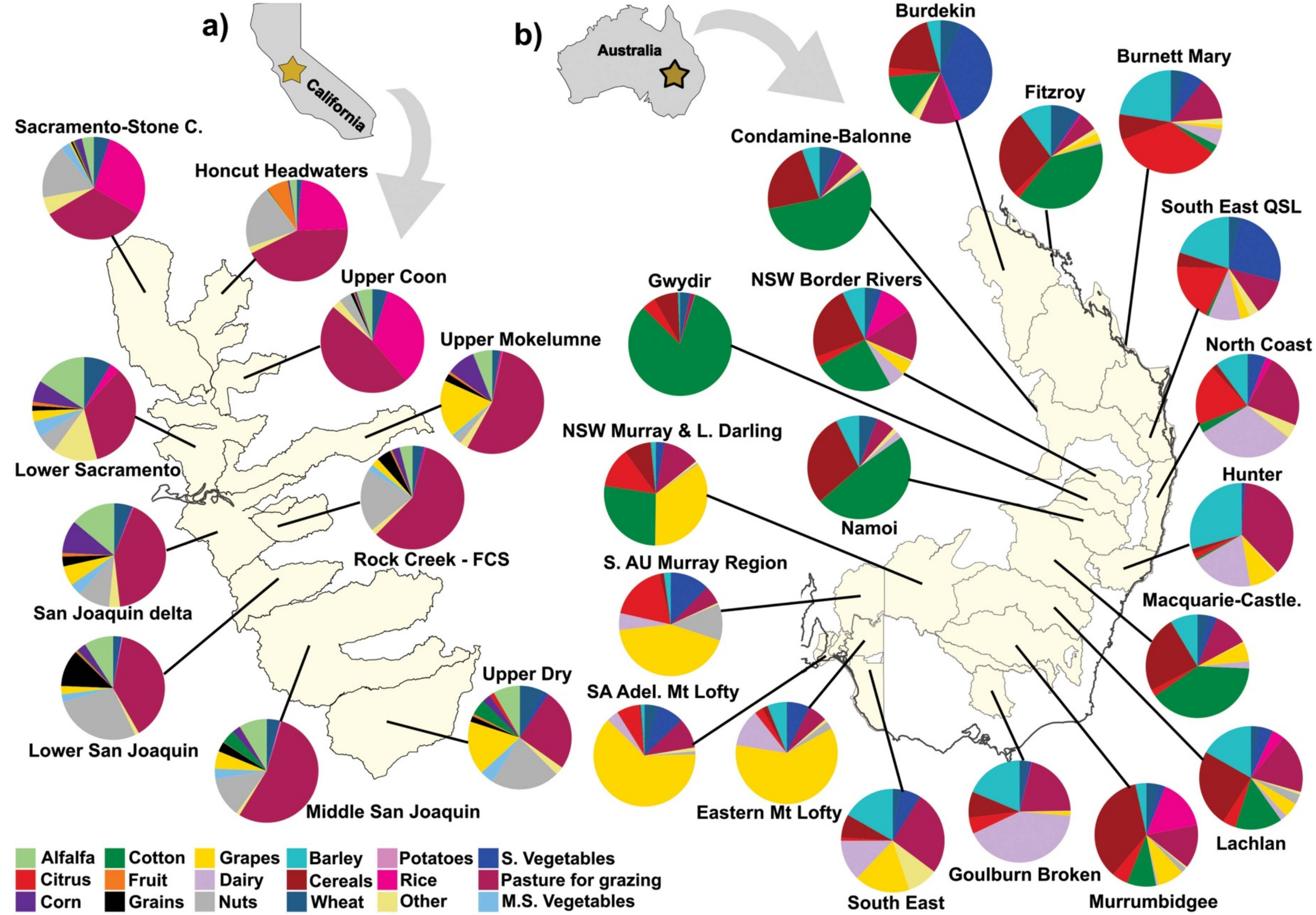

Figure 2. Crop distribution patterns for each sub-basin within (a) the Central Valley (California, US) and (b) Murray–Darling basin (Australia). Each pie diagrams shows the long-term (2008–2018) average distribution of irrigated crop areas within each sub-basin.

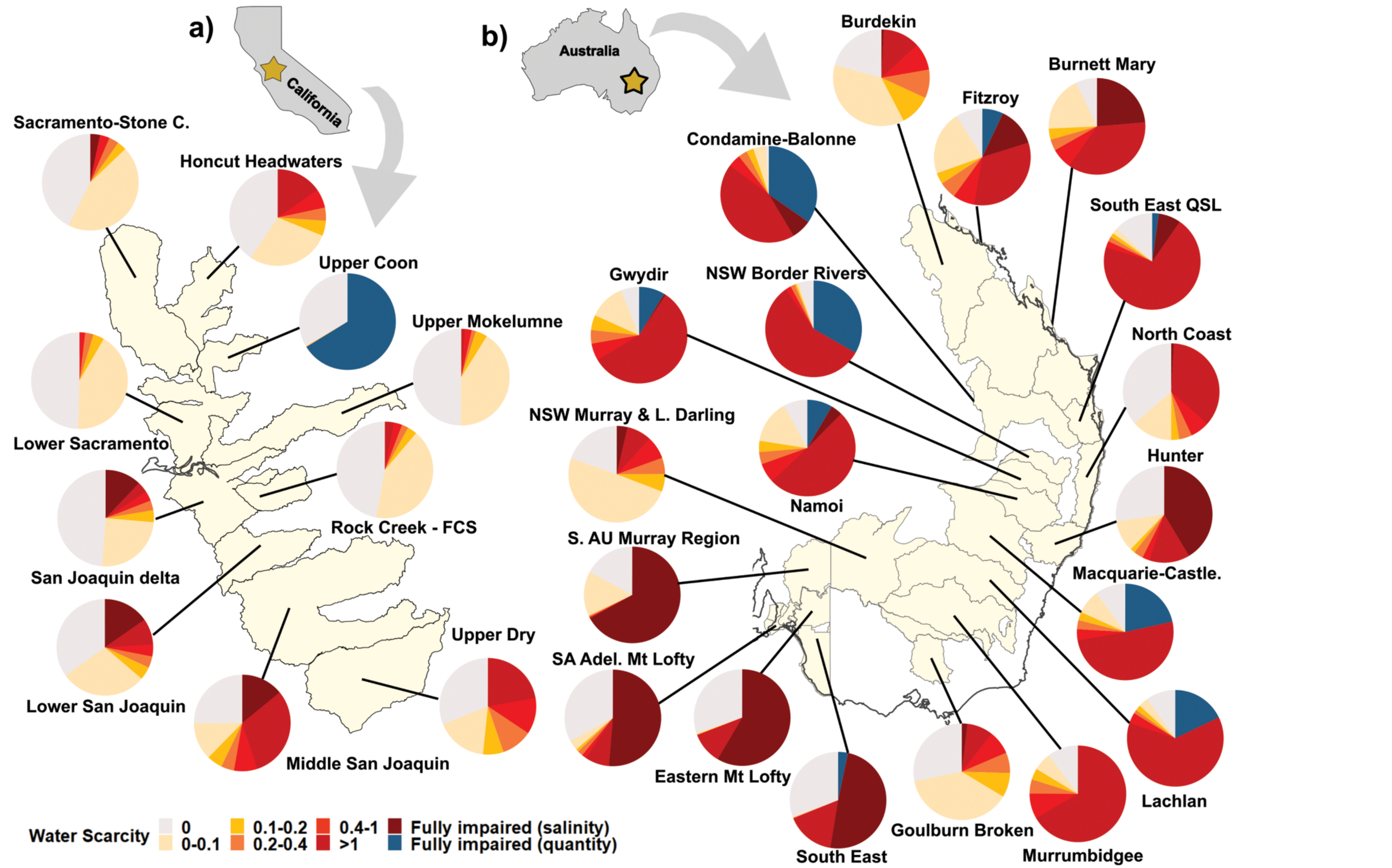

Figure 3. Sub-basin distributions of salinity-inclusive water scarcity levels for the irrigation water use sector. Each pie-diagram represents the distribution of water scarcity instances of each sub-basin (based on monthly, crop-specific estimates) over the 2008–2018 period, in (a) the Central Valley (total number of instances for all sub-basins in this region are 17 186) and (b) the Murray–Darling basin (total number of instances are 17 128) region. Severe water scarcity includes all levels above the 0.4 threshold (i.e. the levels: '0.4–1'; '>1'; 'Fully impaired (salinity)' and 'Fully impaired (quantity)').

water shortages (American River Basin IRWMP 2018). However, since our approach is based on using observed discharge data as an integrated measure of water availability (equation (1)), any irrigation water demand will contribute to severe water scarcity levels when this discharge is near or close to zero. Thus, the severity of water scarcity levels for this specific sub-basin are expected to be lower in reality than shown in our estimations.

Severe water scarcity in the MDB is not as localised as in the CV region, but affects most sub-basins. 'Fully impaired (salinity)' levels drive water scarcity levels in the most southern, near coastal areas (S. AU Murray; SA. Adelaide and Mt Lofty; Eastern Mt Lofty; South East), with severe water scarcity levels ranging between 66% and 89% of their total instances, whereas low water availability (i.e. 'fully impaired (quantity)' levels) has a larger impact on severe water scarcity levels in the most inland sub-basins (e.g. Condamine-Balonne; NSW Border Rivers; figure 3(b) and supplementary figure 6).

Including salinity levels on average increases severe water scarcity levels (WS > 0.4) from 21% (quantity only) to 23% (salinity-inclusive), out of all water scarcity instances in the CV and from 60% (quantity only) to 66% (salinity-inclusive) out of all instances in the MDB. Compared to these regional averages, effects were larger at individual sub-basin scales. Including salinity led to increasing severe water scarcity instances in all San Joaquin sub-basins in the CV region (Middle, lower and delta). For example, for all water scarcity instances in the Lower San Joaquin basin, severe water scarcity changed from 13% (quantity only) to 26% (salinity-inclusive). For sub-basins within the MDB, effects were seen across 10 out of the 19 sub-basins. For example, for all water scarcity instances in the Hunter sub-basin, severe water scarcity increased from 39% (quantity only) to 63% (salinity-inclusive) and from 1% (quantity only) to 66% (salinity-inclusive), of all instances in the SA Murray sub-basin (see supplementary tables 8 and 9 for all sub-basin results).

Excluding crop-specific tolerance levels also increased estimates of severe water scarcity, from 23% (incl. crop-specific tolerance levels) to 35% (excl. crop-specific tolerance levels), out of all instances in the CV region and from 66% (incl. crop-specific tolerance levels) to 72% (excl. crop-specific tolerance levels) out of all instances in the MDB (supplementary tables 8 and 9). Compared to these regional averages, differences were also here larger at individual sub-basin scales. Largest changes were seen across the San Joaquin sub-basins in the CV region (middle, lower and delta), with instances of severe water scarcity changing from 17% (incl. crop-specific tolerance levels) to 51% (excl. crop-specific tolerance levels) in the San Joaquin Delta. Locally within the MDB region, the SA Murray sub-basin showed the largest changes, with increased estimates of severe water scarcity from 66% (incl. crop-specific tolerance levels) to 97% (excl. crop-specific tolerance levels) out of all instances. All sub-basin changes are included in supplementary tables 8 and 9. Although some of these sub-basins grow relative saline sensitive crops (e.g. nuts and grapes), their salinity tolerance levels (1000 μS cm^{-1}) are still higher than the FAO global average irrigation guideline value (700 μS cm^{-1}). The San Joaquin sub-basins also to a large extent grow very saline tolerant crops (e.g. forage crops; 1800 μS cm^{-1}), which causes higher estimates of severe water scarcity when such crop-specifics are excluded. These results highlight the value of accounting for crop-specific salinity in water scarcity estimations, particularly at local (i.e. sub-basin) scales.

3.2. Salinity-inclusive water scarcity: temporal variation

For both studied regions, water scarcity levels are highest during their respective summer seasons, which correspond to the part of the year where irrigation water demand is highest and the climate is warmer and dryer. Specifically, in the CV region, severe water scarcity levels (WS > 0.4) occur in around 35% of total monthly instances during June, July and August (figure 4(a), left panel). For the MDB, severe water scarcity levels occur in around 75% of total monthly instances during November, December and January (figure 4(b) left panel). For individual sub-basins within the CV region, water scarcity is a persistent problem throughout most of the year for some sub-basins (e.g. the Middle San Joaquin; >30% of total water scarcity instances reach severe levels, for most months), whereas some sub-basins show low water scarcity levels throughout the year (e.g. the Sacramento-Stone Corral; <15% of total instances reach severe water scarcity levels; figure 4(a), right panels and supplementary figure 5). For sub-basins within the MDB, water scarcity levels vary substantially across the studied sub-basins. Comparing for instance the Codamine-Balonne and Southern Murray Region (figure 4(b) right panels), both show high levels of severe water scarcity (>50% of total instances for each month), but with different contributions; strongly impaired by water quantity (i.e. low water availability) in Codamine-Balonne and dominantly impaired by salinity in the Southern Murray Region. Water scarcity levels also show more heterogeneity in this region, with multiple sub-basins being mainly impaired by water quantity, salinity, or a combination of both (supplementary figure 6).

In addition to seasonality, we also quantified inter-annual variability in water scarcity levels during the warm-season (i.e. the three warmest months of each region and year) and evaluate its relation to regional drought (using the Standardised Precipitation-Evapotranspiration Index, SPEI Vicente-Serrano et al 2010; supplementary section 5).

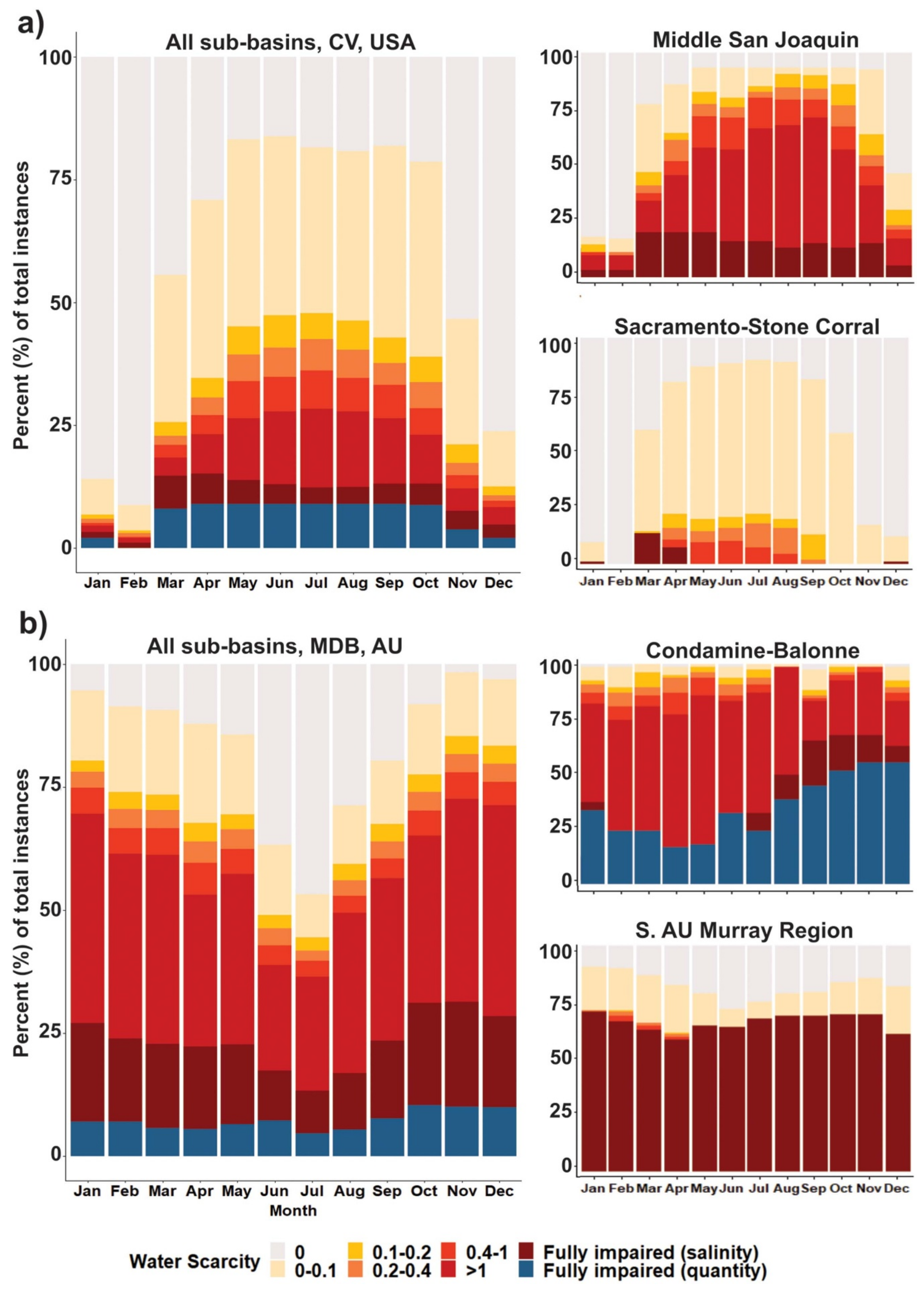

Figure 4. Distribution of water scarcity levels per month in (a) the Central Valley (CV; left panel; each month represents the total number of instances of that month during 2008–2018) and a sub-set of two individual sub-basin examples (right panel), and across (b) the Murray–Darling basin (MDB; left panel) and a sub-set of two individual sub-basin examples (right panel).

For the CV region (figure 5(a)), there is a relationship between warm-season water scarcity levels and drought, with overall decreasing water scarcity levels during relatively wet years (positive SPEI; 2008–2011) and increasing water scarcity levels when this region starts to experience more intense drought conditions from 2012–2015 (negative SPEI). For the MDB (figure 5(b)), there is a clear decline in water scarcity levels in 2010, which is the wettest year of the study period, as well as higher percent of instances of severe water scarcity during the later dryer years. However, inter-annual variations in water scarcity

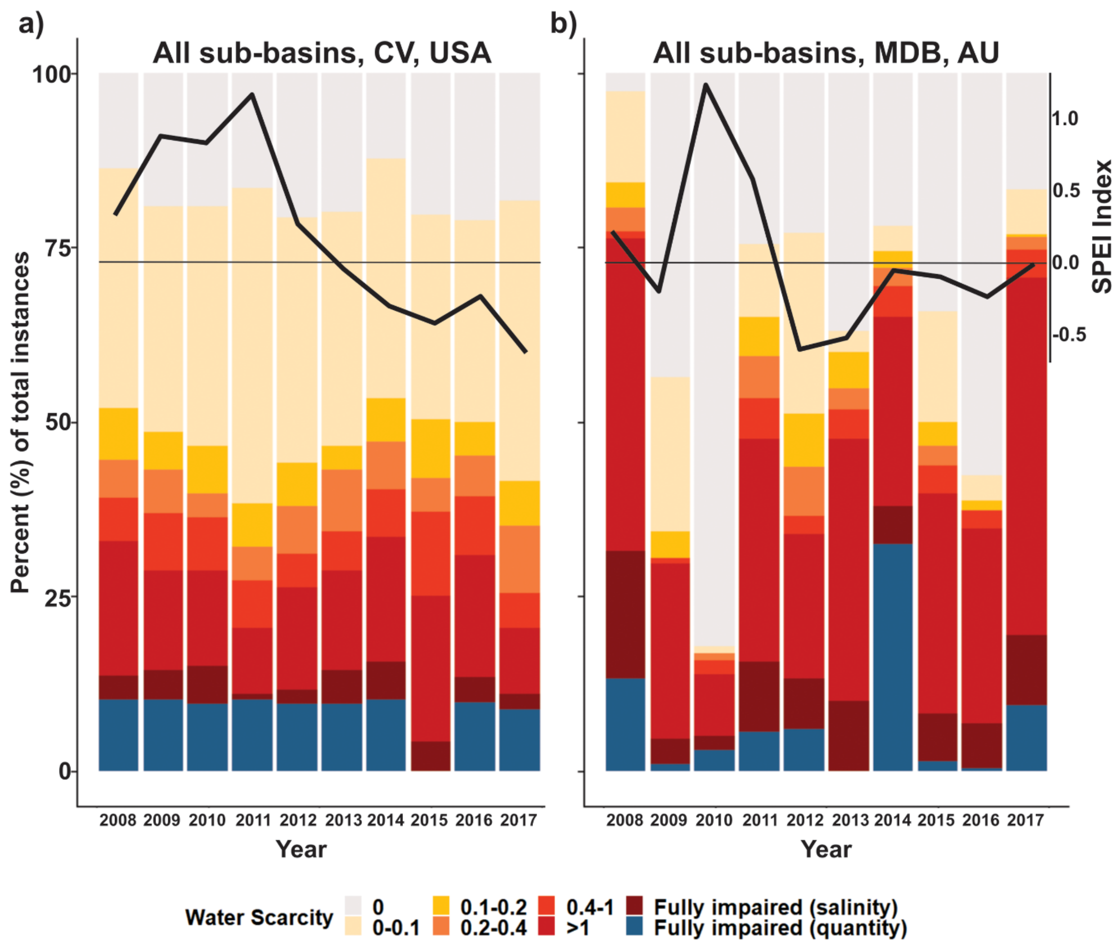

Figure 5. Warm-season inter-annual water scarcity levels, as well as drought conditions (represented by the Standardised Precipitation-Evapotranspiration Index; SPEI), for (a) the Central Valley (CV), and for (b) the Murray Darling (MDB) region. Warm-season annual average water scarcity is estimated from all instances within the three warmest months of each region and each year (June, July, August for CV and November, December, January for MDB, with N = 1443 and N = 1879 for CV, respectively MDB). SPEI index values (secondary *y*-axis on the right) are estimated from the same months as the WS index, with positive values representing non-drought and negative values representing drought conditions.

during the prolonged drought from 2012 to 2017 (e.g. decreasing water scarcity during 2015, 2016), suggests that drought alone does not fully explain the observed water scarcity dynamics.

3.3. Impacts of conjunctive water use on alleviating water scarcity

Our results show that conjunctive groundwater use may significantly alleviate severe water scarcity by diluting surface water at instances where salinity exceeds crop specific tolerance levels. Across the full CV region, crop-specific salinity tolerance levels were exceeded at 946 monthly instances over 2008–2018 (from the total of 17 388 monthly instances). By including conjunctive groundwater use to dilute the surface water in these specific instances, severe water scarcity on average decreased by 67%, with impacts ranging from 34% to 81% depending on the month (figure 6(a)). The Middle San Joaquin, a sub-basin strongly affected by salinity-driven water scarcity, is further illustrated as an example. In this sub-basin, conjunctive water use lowered instances of severe water scarcity by 23% on average (6%–49% monthly range; from a total of 1651 instances; figure 6(b)). The largest relative impact on alleviating severe water scarcity in the Middle San Joaquin occurs in the winter months (December–March), whereas alleviation impacts in the summer months (July–September) are comparatively low. This might be due to overall higher salinity and lower water availability in the summer, which means that the net effects of conjunctive use on alleviating severe water scarcity would be more limited. The main crops needing dilution in this sub-basin are nuts, grapes and forage crops (pasture for grazing crop class), which on average required over 280 million m^3 groundwater per month to dilute surface water for irrigation (figure 6(c)). Severe water scarcity occurs in this sub-basin, due to combinations of relatively low salinity tolerances and high irrigation water-use intensity of these crops (figure 6(d)).

Although conjunctive water use has a positive impact on decreasing severe water scarcity in this region, its suitability as a sustainable management

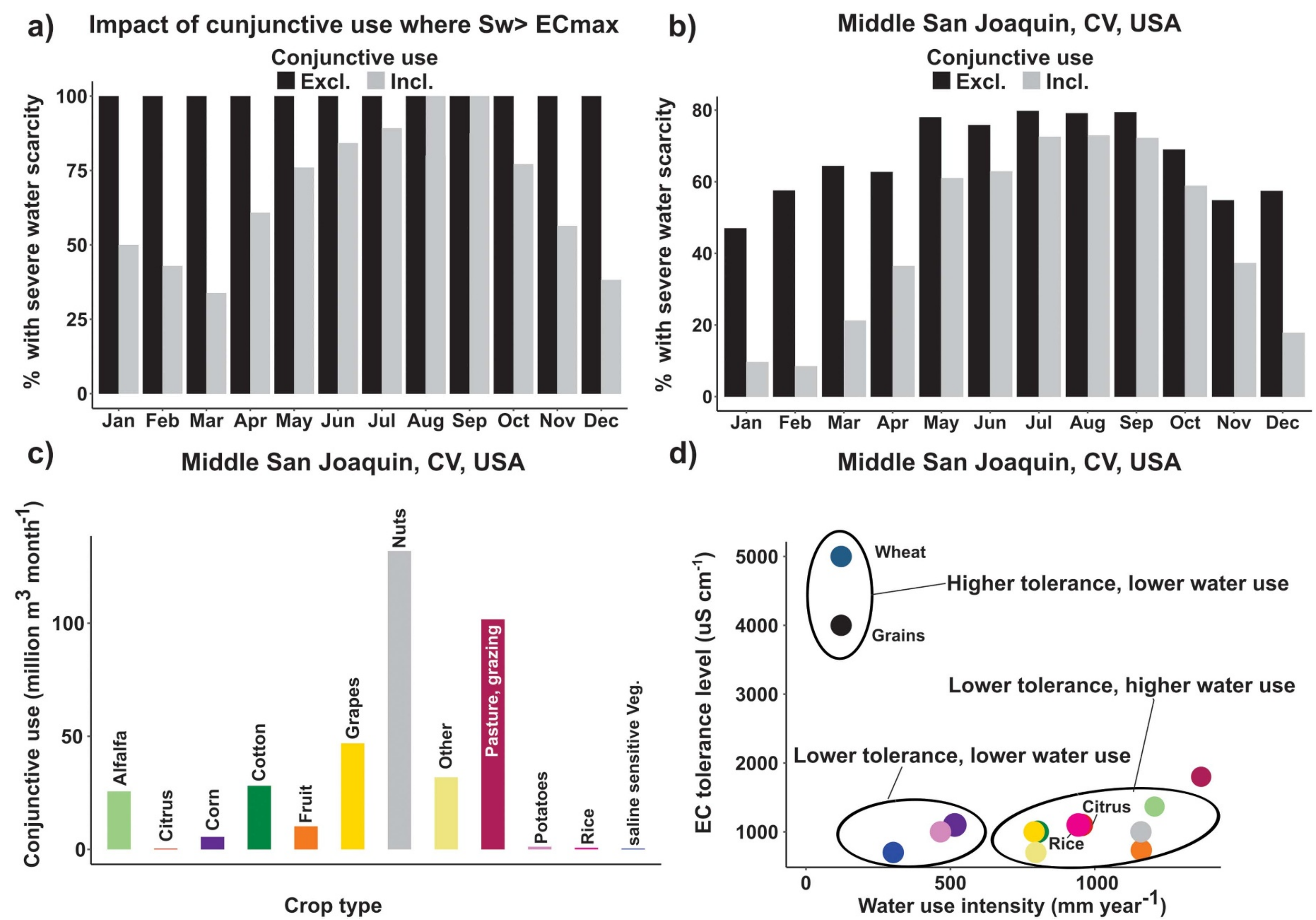

Figure 6. Impacts of conjunctive groundwater use to reduce severe water scarcity (>0.4) in the Central Valley (CV). Panel (a) shows the monthly levels of severe water scarcity for the CV region, when conjunctive use is included (grey bars), compare to when it is excluded (i.e. 100%; black bars), for all instances where surface water salinity levels exceed crop-specific tolerance values (n = 946 out of 17 186 total instances). Panel (b) shows the monthly levels of severe water scarcity for the Middle San Joaquin sub-basin (n = 1651), when conjunctive use is included (grey bars), compared to when it is excluded (black bars). Panel (c) shows the volumes of groundwater needed per month (long-term average 2008–2018) and per crop, to alleviate severe water scarcity through conjunctive use in this sub-basin and panel (d) shows the distribution of crops in terms of their salinity tolerance and irrigation water use intensity.

option must also be balanced against the pressure it puts on renewable groundwater resources (i.e. in terms of aquifer over-exploitation). Our results show that total groundwater volumes required for dilution in the Middle San Joaquin sub-basin greatly exceeds renewable groundwater rates (supplementary table 10). Even though the salinity of used groundwater resources are below crop-specific tolerance levels, large dilution volumes arise when available groundwater is still relatively saline. This suggests that it is not economically realistic, nor sustainable to use conjunctive water use as an alleviation measure in this sub-basin. We also see similar results for the Upper Dry sub-basin, whereas dilution needs across other sub-basins in this region are below renewable groundwater rates (supplementary table 10). To limit irrigation water scarcity and aquifer over-exploitation in this region, other methods need to be considered. Such methods could include changing from nut production (relatively high water use and low salinity tolerance) to more salt tolerant crops with lower irrigation rates (e.g. wheat and grains, figure 6(d)), as well as salinity management.

4. Discussion and conclusion

Addressing the challenge of water and food security in irrigated regions will require multiple and cross-disciplinary efforts (McDermid *et al* 2021). Our salinity-inclusive water scarcity approach takes a step towards understanding the effects of both water quantity, salinity and crop patterns on water scarcity in irrigated regions. Although the framework can be applied at various scales, as a first basis, we assess salinity-inclusive water scarcity using a data-driven approach, at sub-basin scales. We show that, across two important food bowl regions of the US and Australia, many sub-basins face severe water scarcity. These results are in line with known issues of water scarcity within these regions, but where joint impacts of both water quantity and salinity have previously not been considered (Wichelns and Qadir 2015, Hart *et al* 2020). Our results further highlight an increase in water scarcity levels when crop-specific irrigation and salinity tolerance levels are accounted for, compared to estimations based on water quantity only, particularly at local (i.e. sub-basin) scales. Although salinisation is a well-known risk for yield reductions (Katerji *et al* 2003), limited water scarcity approaches have considered the impact of salinity on irrigation water use specifically (van Vliet *et al* 2021), none using observed-driven salinity data of both surface- and groundwater resources.

Our methodology can aid in evaluating the impacts of specific crops and agricultural practices (e.g. crop changes) on water scarcity (Cuevas *et al* 2019). Our framework captures the impact of both crop water usage and crop-specific salinity tolerances, both of which impact water scarcity levels. It would be beneficial for future studies to explore also the sensitivity of overall water scarcity levels to crop-specific salinity thresholds, to assess the sensitivity of crop-specifics on overall water scarcity levels. By doing so, impacts of crop changes can be evaluated more thoroughly. In terms of management, agricultural changes must also be feasible from a crop revenue perspective, to avoid negative trade-offs. For instance, our results show that nuts have relatively low salinity tolerance and are extensively irrigated in California (figure 6(d)), which contributes to observed high water scarcity levels, but they are also the highest crop-based commodity on market sales (Fulton *et al* 2019). Our framework can thus be used and further tailored to estimate impacts of crop changes on irrigation water scarcity, including economic analyses and trade-offs. It should also be recognized that our approach currently does not focus on the effect of engineering structures, such as water transfer. Reservoirs, desalination and other water transfer infrastructure may significantly aid in reducing water scarcity, also for the here studied sub-basin regions (USBR 2021). It is therefore important for future assessments to try to incorporate such impacts of water infrastructure, to further reduce uncertainties and improve water scarcity estimates (Ma *et al* 2020).

Water scarcity is also expected to become exaggerated with future hydro-climatic changes, particularly in arid and semi-arid regions, which is where most irrigated areas are located (Winter *et al* 2017, Leal Filho *et al* 2022). In the CV for example, earlier onset of snowmelt and less precipitation falling as snow may impact multiple sub-basins, which depend on snowmelt to refill reservoirs and support irrigation during dry seasons (Mehta *et al* 2013). Drought frequency and intensity is projected to become more common in these regions (Hosseinizadeh *et al* 2015, Balting *et al* 2021), with increasing risks of lower water availability and increasing salinity levels (Mosley 2015). Impacts of droughts on water scarcity were also here seen in our results, which is in line with earlier quantifications of drought-induced salinity and water scarcity increases in Texas (Jones and van Vliet 2018). Other studies across the US have also shown that combined effects of heat and drought can further intensify water scarcity, with impacts on crop production in irrigated agriculture (Luan *et al* 2021). All these changes increase the risks of water scarcity and yield reductions in global irrigated areas (Caparas *et al* 2021).

To sustain irrigated agriculture and to buffer when water demands temporally exceeds supply, for example during droughts, proper management of agricultural water is central (Singh 2014, Caparas *et al* 2021). We evaluate the impacts of conjunctive groundwater use as a management strategy for water scarcity alleviation, by diluting surface water at instances where salinity levels exceed crop specific tolerance levels. Conjunctive water use is common for irrigation, including for the here studied regions (Scanlon *et al* 2012). However, its impact on water scarcity from both a water quantity and quality perspective has not been assessed previously. Our results show that severe water scarcity can be substantially lowered through conjunctive groundwater use, suggesting it should be accounted for in water scarcity estimations. For the CV sub-basins, however, the estimated volumes needed for dilution frequently exceeded modelled renewable rates, suggesting that this is an unsustainable management option for several sub-basins in this region (supplementary table 10). Although modelled renewable recharge rates comes with inherited uncertainty, particularly at local sub-basin scales (Reinecke *et al* 2021), unsustainable groundwater use for irrigation is a known issue in the Central Valley, as well as across large parts of the US (Scanlon *et al* 2012, Lopez *et al* 2021). If further groundwater depletion continues, there are increasing risks of both land subsidence and further pollution issues (Faunt *et al* 2016, Smith *et al* 2018, Miro and Famiglietti 2019).

Exploring conjunctive use as a water scarcity alleviation strategy therefore needs to be evaluated against the pressure it puts on the groundwater system (i.e. renewable rates), particularly in groundwater-stressed regions. Future research also needs to include management aspects not considered here, such as wastewater recycling and reuse (Misra 2014), desalination solutions (van Vliet *et al* 2021) and technological advancements, including smart irrigation techniques (Dinar *et al* 2019).

Our results highlight that crop changes and reducing salinity levels in agricultural systems need to be a priority to limit irrigation water scarcity in the here studied regions. Beyond our studied regions, the sustainability of irrigated agriculture and crop production is also a global challenge. Our framework can provide a basis for future studies assessing the impact of salinity and crop distributions on global water scarcity levels. To do this, however, more knowledge on salinity in water-soil-crop systems are required and better spatial coverage of surface and groundwater salinity monitoring data across the world is needed. Although research on salinisation is growing, a main challenge is to better understand soil and water salinity interactions and its drivers. For instance, irrigation is known to be both a driver and a victim of freshwater salinisation (Russ *et al* 2020, Thorslund *et al* 2021), but large-scale understanding of current feedbacks between irrigation water use, climate and changing salinity levels in soil, ground- and surface waters is still lacking (Cunillera-Montcusí *et al* 2022). Integrated modelling efforts of water availability, salinity and crop changes under future conditions are also needed to better predict and sustainably manage irrigation and crop production under global change (Winter *et al* 2017).

Data availability statement

The data that support the findings of this study are openly available at the following URL/DOI: https://doi.org/10.5281/zenodo.6620860 (Thorslund *et al* 2022).

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