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Assessing virtual water trade and inequalities in household water footprints across California's counties

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ABSTRACT

The concept of virtual water trade suggests water flows from water-rich to water-scarce regions, but local disparities are often overlooked. This study uses a multi-regional input-output (MRIO) model to assess virtual water transfers among California's 58 counties and the rest of the conterminous U.S. in 2017. Results show the Central Valley exported large volumes of virtual water via water-intensive crops (e.g., fruits and vegetables) but imported water embodied in industrial, mining, and thermoelectric processes. These imports eased water stress in the Central and South Coast but left Central Valley scarcity unresolved. Linking household consumption with MRIO reveals the highest-income group (over US$200k) had per capita water footprints 1.8 times larger than the lowest-income group (below US$15k). Although household size and consumption patterns mitigated this gap, Central Valley's high water intensity fueled excessive footprints. The study underscores the need for targeted, equitable water management policies, promoting more effective water conservation strategies.

1. Introduction

The U.S., with its vast and diverse landscape, faces significant challenges due to recurring droughts, particularly affecting regions with an uneven distribution of freshwater (Devineni et al., 2015; Martin et al., 2020). These geographical and climatic variations impact local economies and water consumption patterns (Marston et al., 2018). Despite a national decline in overall water use due to productivity improvements (Debaere & Kurzendoerfer, 2017), regional discrepancies persist (Sankarasubramanian et al., 2017). For example, surface water use has decreased, but fresh groundwater withdrawals, mainly for irrigation, have increased (Dieter et al., 2018), especially in the Southwest where climate heavily influences water usage (Worland et al., 2018). This emphasizes the pressing need for water sustainability, especially in arid regions.

California is responsible for nearly 9% of the U.S. total and freshwater withdrawals, primarily for irrigation (Dieter et al., 2018), which supports its extensive agricultural production. The state's unique climate, characterized by high temperatures and limited winter precipitation, frequently leads to droughts, making irrigation vital (NOAA, 2016). Despite occupying only 1.2% of the nation's farmland, California is a major producer of the country's vegetables, fruits, and nuts (Pathak et al., 2018), underpinning its role as the most diverse and productive agricultural state.

The California Central Valley, a key agricultural powerhouse, is confronted with major obstacles from persistent droughts and climate change, threatening its agricultural output. These issues, coupled with population growth, exert further pressure on the region's already strained water resources (Fuller & Harhay, 2010; Elias et al., 2015). During droughts, groundwater supplies roughly 60% of California's total water supply, compared to 40% during non-drought years (California Water Boards, 2022). This reliance has led to significant over-pumping of the region's aquifers, particularly during drought periods, leading to detrimental effects, such as soil subsidence (Galloway & Riley, 1999; Liu et al., 2019), lowered drinking water quality (Jurgens et al., 2010; Smith et al., 2018), and diminished water availability (Faunt, 2009) for both agricultural or non-agricultural uses. Being one of the most arid areas in the state, such high water demand places the

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Central Valley at the epicenter of California's water scarcity and groundwater depletion crisis. This stark imbalance underscores the acute challenges faced by the region, as well as highlighting the urgent need for sustainable water management strategies tailored to address the unique conditions of the Central Valley.

Water is a critical resource, consumed directly and indirectly in the production of goods and services, to fulfill both local and external demands (Yu et al., 2010). From consumers' perspective, direct water use is more tangible and easily recognized (Attari, 2014; Gómez-Llanos et al., 2020). However, indirect freshwater consumption for producing goods or services could be 20 times that of direct water use (Chini et al., 2017). For better water conservation plans, the virtual water concept has been conceived as a strategy for water-deficient regions to import water-intensive commodities from water-rich areas, thereby supplementing their local water shortages. Many studies have explored water footprint accounting in the U.S. (Marston et al., 2018; Konar & Marston, 2020) and specifically in California (Mubako et al., 2013; Fulton et al., 2014; Fulton & Cooley, 2015; Marston & Konar, 2017; Fulton et al., 2019) using methods of commodity or product flows. This method provides a detailed description of the movement of particular commodities or products and their direct resource implications, but it is limited by the inability to account for and accurately estimate pass-through flows within the country (Rushforth & Ruddell, 2018), and may therefore neglect sub-national supply chain effects. Although multi-regional input-output (MRIO) model provides a comprehensive approach for analyzing virtual water consumption and savings across interconnected sectors and regions, most MRIO studies have concentrated on specific areas and industries at the regional or state levels (Mubako et al., 2013; Mayer et al., 2016). This leaves a gap in understanding the distinct role of the inter-county virtual water transfers across California's 58 counties within the comprehensive national trade network, as well as the disparities in detailed water use across different region-sector combinations. Furthermore, as a major contributor to the total water footprint, the influence of household consumption on indirect water use has received less attention. This gap is notable in discussions about the composition of household water footprint and its inequality across various income groups. Understanding these socioeconomic factors is crucial for developing effective demand-side mitigation strategies and formulating targeted policy recommendations.

To fill these two important gaps, this study first built county-level water accounts for eight major water use sectors (crop farming sector consists of ten crop sectors), then constructed a county-level MRIO model to concurrently assess the inter-county virtual water flows, considering both production and consumption perspectives. Such detailed sub-state analyses provide a more granular understanding of water distribution and scarcity within a state, highlighting disparities at the county level that are often overlooked in national or state-level analyses. This capability to identify disparities in water usage across various sectors and regions aids in recognizing and quantifying complex interdependencies in water management. We then linked household consumption survey data with the MRIO model and analyzed water footprints across different income groups, representing a methodological advancement that can be applied to other regions and resources, enhancing the tools available for environmental and economic analysis. Furthermore, we decomposed and quantified the contribution of income, household size and household consumption patterns to the per capita water footprint gaps between income groups, supporting an understanding of how socioeconomic factors influence effective water demand. The detailed analysis not only provides the ability to better characterize California's water use dynamics, but also stresses the need for equitable water conservation strategies and facilitates more informed and balanced policy decisions.

2. Methods and data

2.1. Multi-regional input-output modeling (MRIO)

We employed an environmentally extended Multi-Regional Input-Output (MRIO) model to study virtual water flows in inter-county networks. MRIO models facilitate a comprehensive evaluation of environmental impacts caused by final consumption along the entire supply chain.

The MRIO framework has several key components. Considering there are k counties, then A, the technical coefficient matrix represents the inter-sectoral monetary flow from sector m in county p to sector n in county q per unit of total output of sector n in county q. A^{mn} is calculated by $a_{mn}^{pq} = z_{mn}^{pq}/x_n^q$, where z denotes intermediate consumption, x_n^q represents the total output of sector n in county q. Also, the final demand matrix Y, comprising y^{pq}, identifies the vector of output from each sector in county p used by final consumers in county q.

$$A = \begin{bmatrix} A^{11} & A^{12} & \dots & A^{1k} \\ A^{21} & A^{22} & \dots & A^{2k} \\ \vdots & \vdots & \ddots & \vdots \\ A^{k1} & A^{k2} & \dots & A^{kk} \end{bmatrix} \tag{1}$$

$$Y = \begin{bmatrix} y^{11} & y^{12} & \dots & y^{1k} \\ y^{21} & y^{22} & \dots & y^{2k} \\ \vdots & \vdots & \ddots & \vdots \\ y^{k1} & y^{k2} & \dots & y^{kk} \end{bmatrix} \tag{2}$$

$$X = \begin{bmatrix} x^1 \\ x^2 \\ \vdots \\ x^k \end{bmatrix} \tag{3}$$

Accordingly, the MRIO methodology can be outlined in the following way using matrix algebra (Miller & Blair, 2022):

$$X = AX + Y \tag{4}$$

Leading to:

$$X = (I - A)^{-1} Y \tag{5}$$

Where I denotes the identity matrix. The Leontief inverse matrix $(I - A)^{-1}$ represents the direct and indirect inputs required to meet each quantity of final demand in monetary terms.

To include water coefficients f, the following equation describes embodied water use in goods and services:

$$W = f(I - A)^{-1} Y \tag{6}$$

Here, W stands for the total amount of water needed to produce products and services to satisfy final demand. The f factor indicates the direct water usage per unit of total output for each sector within each county.

Additionally, to get each income group's water footprint, the following equation can be used:

$$W_{incomegroup} = f(I - A)^{-1} Y_{incomegroup} \tag{7}$$

Here, $W_{incomegroup}$ and $Y_{incomegroup}$ represents the water footprint and household final demand of each income group in every county, respectively.

Consumption in one area might indirectly affect water use in other regions through regional trade. Thus, we studied the water embodied in inter-county trade, including water embodied in inter-county exports (*WEE*), imports (*WEI*) and net outflows (*WEN*).

$$WEE^p = f^p (I - A)^{-1} Y^{-p} \tag{8}$$

$$WEI^p = f^{-p} (I - A)^{-1} Y^p \tag{9}$$

$$WEN^p = WEE^p - WEI^p \tag{10}$$

Total exports from county p to other regions are denoted by WEE^p. Direct resource usage coefficients for county p and final demand for all other regions are represented by f^p and Y^{-p}. Total imports into county p from other regions are calculated as WEI^p, where the direct resource usage for all regions aside from county p, along with the final demand of county p are denoted by f^{-p} and Y^p.

2.2. Index decomposition analysis

We utilized an index decomposition analysis (IDA) to quantify how various key elements contribute to the disparity in per capita water footprint (WF) among different income groups. This method helps us break down and measure the impact of three specific factors: the size of the households, household consumption volume, and water intensity, which reflects the water consumption relative to each dollar expended (Feng et al., 2015, 2021).

The per capita WF can be determined by the product of these three key factors:

$$WF_{percapita} = \frac{1}{s} \cdot c \cdot w \tag{11}$$

Here, $WF_{percapita}$ represents the per capita WF, s represents the household size, c denotes consumption volume, w represents the water intensity.

To analyze variations in per capita WF across two income categories, we consider:

$$\Delta WF_{\text{percapita}} = \left(\Delta \frac{1}{s}\right)(c \cdot w) + (\Delta c)\left(\frac{1}{s} \cdot w\right) + (\Delta w)\left(\frac{1}{s} \cdot c\right) \tag{12}$$

Here, Δ denotes the change, transforming the multiplicative factors in the initial equation into additive terms, reflecting the contributions of each factor to changes in per capita WF when holding other factors constant. Importantly, this decomposition does not have a unique solution. In our study, we adopted the widely-used polar decomposition method (Wang et al., 2020; Miller & Blair, 2022).

2.3. Water stress index

To identify areas that are vulnerable to water scarcity, we used the Water Stress Index (WSI) concept by Pfister et al. (2009). The WSI operates on a scale from 0 to 1, where 0 signifies the absence of water stress and 1 signifies the maximum possible level of water stress. A threshold is established at WSI = 0.5, with values exceeding 0.5 indicative of extremely severe water stress and a WSI of 0.5 representing moderate water stress.

$$WSI = \frac{1}{1 + e^{-6.4 \cdot WTA}\left(\frac{1}{0.01} - 1\right)} \tag{13}$$

To calculate the WSI index, we adopted WTA (yearly freshwater withdrawals to availability) from Hofste et al. (2019). Notably, while the scaling factor in the WSI calculation influences the absolute figures, our analysis primarily concerns the relative differences in water stress levels.

2.4. Data

To construct the county-level MRIO table, we utilized trade information and sector-specific economic input-output data drawn from IMPLAN (IMPLAN Group LLC, 2021). IMPLAN compiles and consolidates data from a wide variety of credible sources, and then uses them to construct comprehensive county-level MRIO tables. These tables facilitate the estimation of inter-county trade flows, and ensure consistency in economic data; further details are available in Appendix 1. We applied the RAS approach (Miller & Blair, 2022) to balance the MRIO table, iteratively adjusting the estimated interregional trade flows to meet constraints. The table includes California's 58 counties as regions 1-58, and the rest of the U.S. as region 59. We aggregated the original 526-sector MRIO table into 17 water use sectors, with 10 sectors dedicated to crop farming (detailed in Table A1 in the Appendix). For better visualization, we discussed 8 major water sectors (crop farming, livestock, aquaculture, domestic, industrial, mining, public supply, thermoelectric) in the main text. In the IMPLAN dataset, foreign import and export data were sourced from the U.S. Department of Commerce's Foreign Trade Statistics series (IMPLAN Group LLC., 2017), ensuring our model captures a comprehensive trade network.

Data on irrigation withdrawals were obtained from the 2018 Irrigation and Water Management Survey conducted by the U.S. Department of Agriculture (USDA, 2019), which closely corresponds to the year of the IMPLAN data. Methods to generate crop-level water use accounts were outlined in our prior study (Liu et al., 2024). Water data for other primary categories were provided by the U.S. Geological Survey (USGS) National Water Use Science Project (Dieter et al., 2018). Notably, for thermoelectric power water use, we considered the consumptive use (i.e. water isn't returned to the source after cooling) rather than withdrawals (Dieter et al., 2018). For the direct water use for households based on expenditure, water utility data by county and income group were collected from the IMPLAN database (IMPLAN Group LLC, 2019). Residential water rates data were collected from the U.S. Environmental Protection Agency (U.S. EPA, 2023).

Income level household distribution data were collected from the American Community Survey (ACS) conducted by U.S. Census Bureau (USCB, 2017). These income levels were matched with the IMPLAN income groups under the assumption that households are evenly distributed across each income group. Following this, the household numbers from the ACS were reaggregated according to the income levels in the IMPLAN dataset. Similarly, data on household size by income level were collected from the Current Population Survey (CPS) conducted by the Bureau of Labor Statistics and the Census Bureau (USCB, 2018). Household expenditure data were provided by IMPLAN, which calculated these expenditures by merging comprehensive household commodity purchase benchmarks with the yearly Personal Consumption Expenditures (PCE) data from the National Income and Product Accounts (NIPA) and the Consumer Expenditure Survey (CES) conducted by the Bureau of Economic Analysis (BEA) (IMPLAN Group LLC, 2023).

2.5. Limitations

Several limitations should be acknowledged. The limitations inherent to input-output modeling have been extensively discussed in previous research (Wiedmann, 2009; Lenzen et al., 2010; Daniels et al., 2011; Wiedmann et al., 2011). Our MRIO model presents a comprehensive trade network that distinguishes between domestic and international trade (IMPLAN Group LLC, 2024), ensuring our analysis is not confined to an insular economy. However, our analysis primarily focuses on domestic inter-regional water flows, which means that the impacts of foreign exports and imports on local water use patterns are not explored in detail. This presents a potential avenue for future research to investigate these international dimensions more comprehensively. This study examines exclusively blue water due to its alignment with consumptive use across sectors. The completeness of our virtual water source accounting may not be absolute, possibly overlooking minor goods, services, or industrial processes. The estimations of virtual water transfers are based on assumptions about water intensity, which can vary due to differences in production techniques, technology, and efficiency across sectors and regions. Moreover, our findings are susceptible to the quality and availability of county-level data on the production, consumption,

and trade of goods and services. For example, our use of a constant average residential water rate across all counties may lead to an overestimation of direct water use for high-income households, as it does not account for the tiered pricing structure typically applied to higher water consumption levels. Water expenditure can include charges for services other than actual water consumption, like sewage and sanitation services. Additionally, in our analysis, the highest-income category in the IMPLAN dataset is defined as US$200,000+, which may not fully represent the water usage patterns of California's wealthiest individuals, as this group encompasses a broad range of high-income earners. While the data on household size per income-group is national, it is important to note that variations at the county level may exist. Additionally, IDA often faces challenges with data aggregation and model consistency (Xu & Ang, 2014).

Our study utilizes environmental, economic, and demographic data centered around 2017. Although this approach remains sufficient for our primary focus on identifying structural relationships and spatial patterns in virtual water transfers and household water footprint inequalities, it may not capture the temporal fluctuations that might arise from changing consumption patterns, climate conditions, and trade dynamics. To address this limitation and capture these dynamics, future studies should consider employing dynamic analyses to better understand how virtual water transfers evolve over time. Despite our county-level focus, significant intra-county variations in water use patterns could exist, which our study may not capture. Also, the study does not consider potential future changes such as advancements in water-saving technologies or changes in water availability due to climate change. Lastly, the impact of policy, socioeconomic, and demographic factors on virtual water flows and spatial inequalities might not be comprehensively incorporated. Despite these potential limitations, our research offers a foundational understanding upon which future work can build, aiming for ever more detailed insight into California's virtual water landscape.

3. Results

3.1. The virtual water flows across regions within California and the rest of the U.S.

Consistently with California's geography and agricultural production, we divided the 58 counties into 7 regions (Klonsky & Tourte, 1996; Kuminoff et al., 2000). In this section, the virtual water flows across these 7 regions were presented to show the spatial patterns of water use and identify potential areas of concern for water management and sustainability in California.

Fig. 1 offers a detailed view of the interregional trade of virtual water flows across the 7 regions of California and the conterminous U.S., highlighting the scale and direction of these flows. In Fig. 1(a), the WSI level underscores the severity of water stress in southern California compared to the north. Notably, the South Coast (WSI = 1.00) and the San Joaquin Valley (WSI = 0.84) faced the most acute water scarcity. Virtual water imports from other U.S. regions (1.4 billion m^3) and from within California, such as the San Joaquin Valley (0.9 billion m^3), partially mitigated the local water stress in the South Coast. However, despite its own severe stress, the San Joaquin Valley exported 8.0 billion m^3 of virtual water to the rest of the U.S. Fig. 1(b) shows that California's virtual water exports to the rest of the U.S. were three times higher than imports. Specifically, in the San Joaquin Valley, exports to other regions were about eleven times higher than imports. This imbalance exacerbates water stress in the San Joaquin Valley and highlights the significant impact of interregional trade on regional water scarcity in California.

Fig. 2 further addresses these dynamics at the sector level. In Fig. 2 (a), California's San Joaquin Valley, Sacramento Valley and Southeast Interior exported virtual water predominantly embodied in crop farming, reflecting their more intense agricultural activities compared to the South or Central Coast regions. For instance, the San Joaquin Valley alone exported approximately 10.0 billion m^3 of virtual water through crop farming. Conversely, the rest of the U.S. primarily exported virtual water linked to industrial processes, mining, and thermoelectric power generation, with 1.4 billion m^3 of virtual water originating from these sectors. At the county level, the largest virtual water exporters in California are Fresno, Tulare, and Kern, all located in the San Joaquin Valley. Meanwhile, Los Angeles and San Diego on the South Coast, along with Alameda on the Central Coast, were major importers of virtual water. For a detailed county-level visualization, please refer to Figure A1.

Fig. 2(b) presents both production and consumption-based water use patterns across California regions. The San Joaquin Valley and Sacramento Valley recorded the highest production-based water use at 18.8 billion m^3 and 7.1 billion m^3 respectively, primarily driven by crop farming, which contributed over 90% to their total production-based

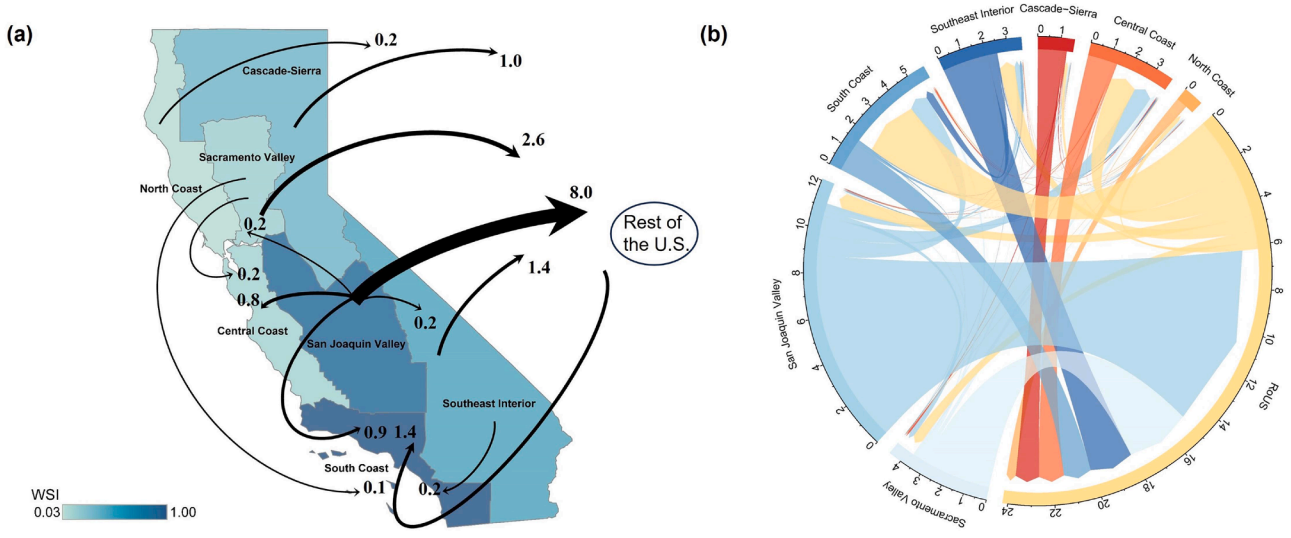

Fig. 1. Virtual water flows across California regions and the rest of the U.S. triggered by final demand of each region.
(a) The arrows indicate the largest net virtual water flows from exporting (producers) to importing (consumers) areas (unit: billion m^3), also showing each region's water scarcity on a scale from WSI = 0 (water abundant) to WSI = 1 (extreme stress), with the rest of the U.S. averaging a WSI of 0.03. (b) It illustrates the bilateral virtual water flows between regions (unit: billion m^3).

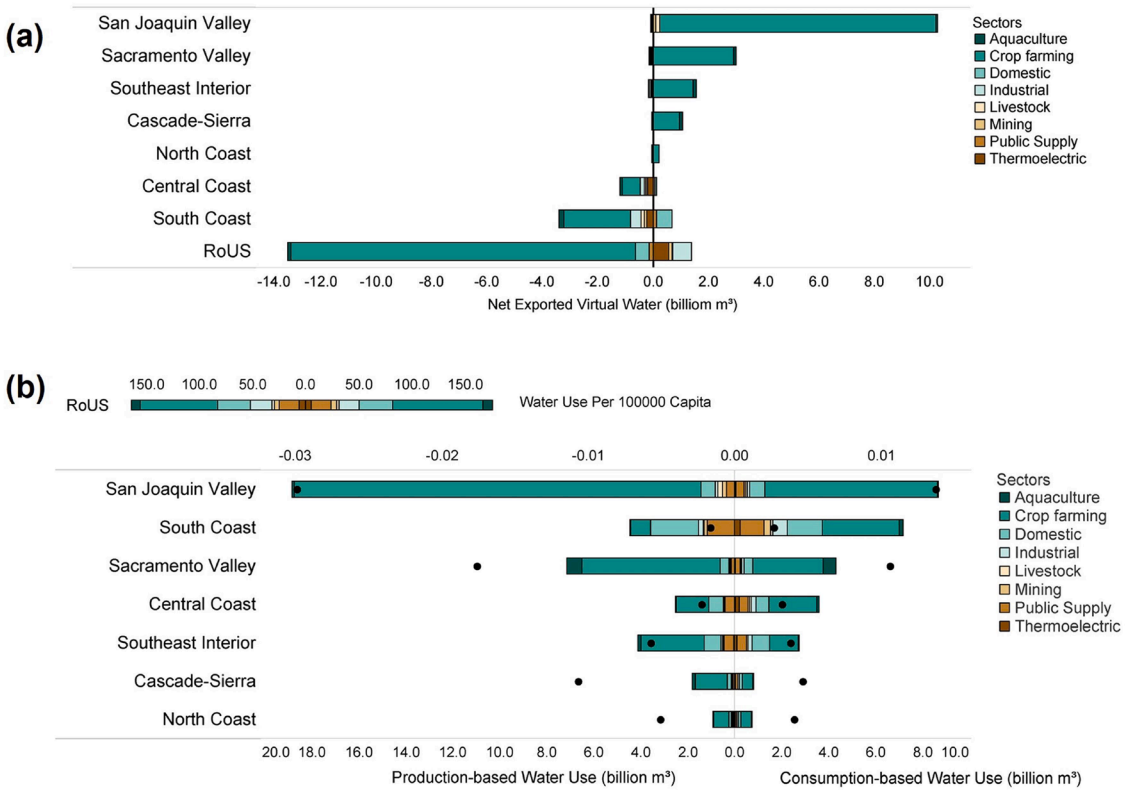

Fig. 2. Sectoral composition of (a) net exported water (b) production-based (direct water use) and consumption-based (water footprint) water use. Overlaying dots on the bars represent water use per 100,000 capita. Regions are in order of total consumption-based water. RoUS represents the rest of the conterminous U.S., with both per 100,000 capita of production and consumption-based water use standing at 0.06 billion m^3.

water use. These areas primarily cultivated fruits, tree nuts, and vegetables and melons, which were more profitable but also more water-intensive (See Figure A3 for a crop-level irrigation water breakdown). In consumption-based metrics, the San Joaquin Valley topped the list, consuming 8.6 billion m^3, while the South Coast followed closely behind with 7.2 billion m^3. Unlike these predominantly agricultural regions, the South Coast showed a diverse water usage profile, with non-agricultural sectors accounting for over 50% of both production and consumption-based water. In Los Angeles, non-agricultural use exceeded 95% of production-based water and contributed 1.9 billion m^3 to the total 3.9 billion m^3 in consumption-based water.

Within California, regions intensively engaged in agriculture demonstrated higher per capita water usage compared to other areas. Notably, the San Joaquin Valley had the state's highest per 100,000 capita water usage for both production-based and consumption-based metrics, at 0.03 billion m^3 and 0.01 billion m^3, respectively. Despite this, such figures were below the national average for the U.S., where per 100,000 capita water use was 0.06 billion m^3 for both metrics. By contrast, densely populated areas like the South Coast, especially Los Angeles, had lower figures, with consumption-based water use reaching only 0.003 billion m^3 per 100,000 capita. Detailed county-level results can be found in Figure A2.

Fig. 3 illustrates the virtual water flows associated with local consumption and inter-county trade, illustrating each county's share of total virtual water consumption and its contributions to exports. These results reveal extensive virtual water transfer across counties, highlighting the contrast between reliance on local versus imported water resources. In the San Joaquin and Sacramento Valleys, most of the final demand is met through local water sources, with a minor fraction of virtual water imported from other counties. These areas also exported substantial amounts of virtual water, intensifying local water scarcity issues. In contrast, counties in the South Coast and Central Coast relied more on imported virtual water, contributing less to virtual water exports. For example, Fresno used 1.6 billion m^3 of its water to meet local demand while importing only 0.4 billion m^3 and exporting 2.9 billion m^3 to other counties. Conversely, Los Angeles depended more on external sources, using 1.2 billion m^3 locally but importing 2.6 billion m^3, with a smaller export contribution of 0.8 billion m^3.

3.2. *Water footprints of different household income groups*

In California, household final demand was the predominant trigger of virtual water consumption. Fig. 4 illustrates the direct water use by households and their total water footprint across various income groups. Collectively, household final demand from all income brackets accounted for 14.3 billion m^3 of virtual water consumption, which was 51.3% of the state's total virtual water consumption of 27.9 billion m^3, making households the largest contributors to virtual water consumption among all categories. Notably, in Fig. 4, the direct water use of households constituted only 5.9% to 7.8% of their total water footprint at the state level. Therefore, analyzing household consumption patterns and their water use impact, particularly across different income levels, is crucial for effective water conservation strategies.

Figure A4 reveals an unequal water footprint across various income groups within California. It demonstrates that higher-income households, particularly those earning above $100k, disproportionately contributed to the overall water footprint relative to their share of total households. In contrast, lower-income households, earning below $70k, had a water footprint that was smaller than their representation in the total household count. For example, in the South Coast region, households earning over $200k constituted 9.6% of total households but explained 17.4% of total water consumption. Meanwhile, households earning below $15k made up 10.0% of households yet only contributed 4.9% to total water consumption. This discrepancy primarily stems from

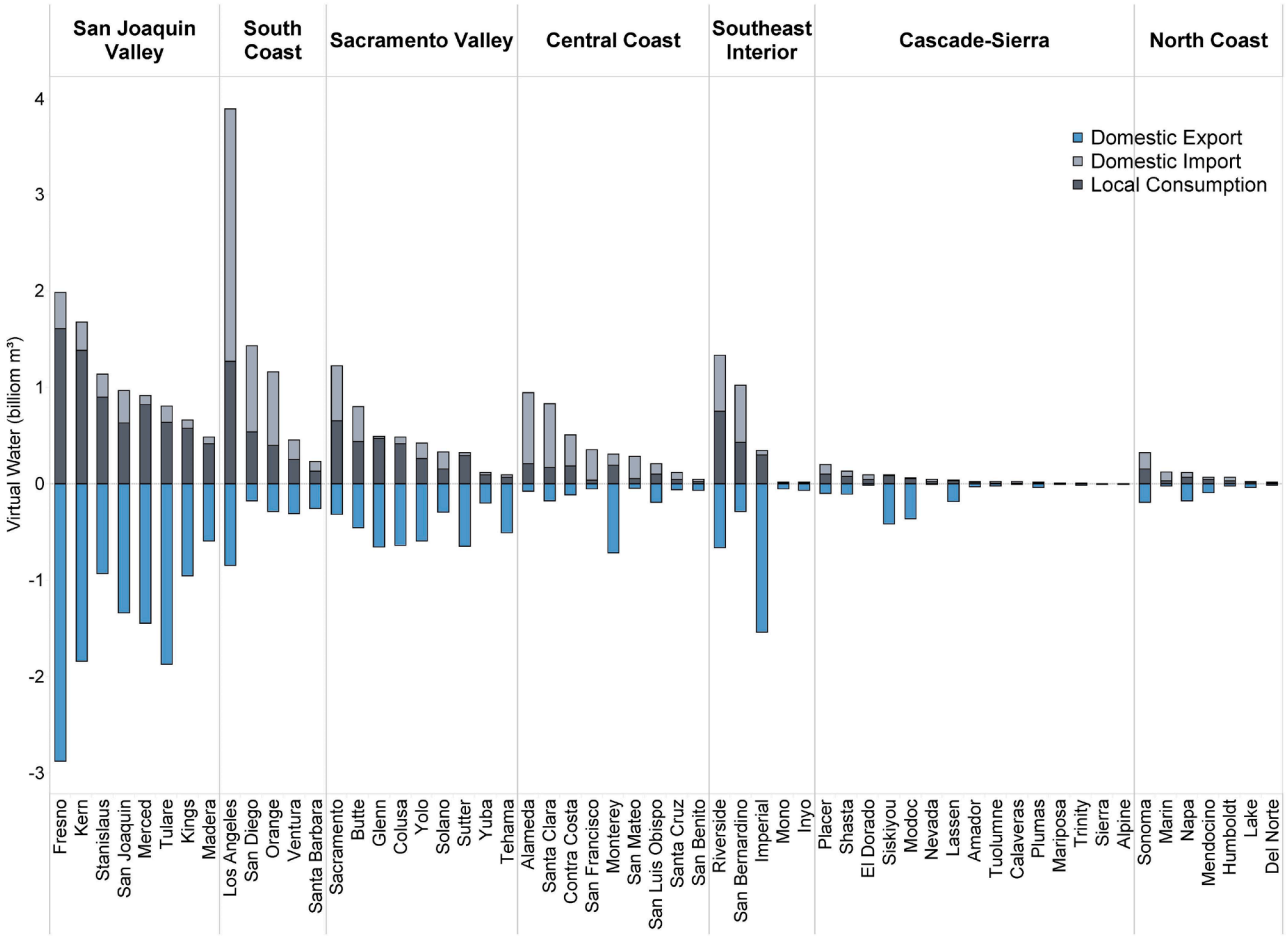

Fig. 3. Water embodied in local consumption, domestic import and export.
The positive y-values represent water embodied in domestic import and local consumption, negative y-values represent water embodied in domestic export. Counties (x-axis) are presented in order of their total virtual water consumption.

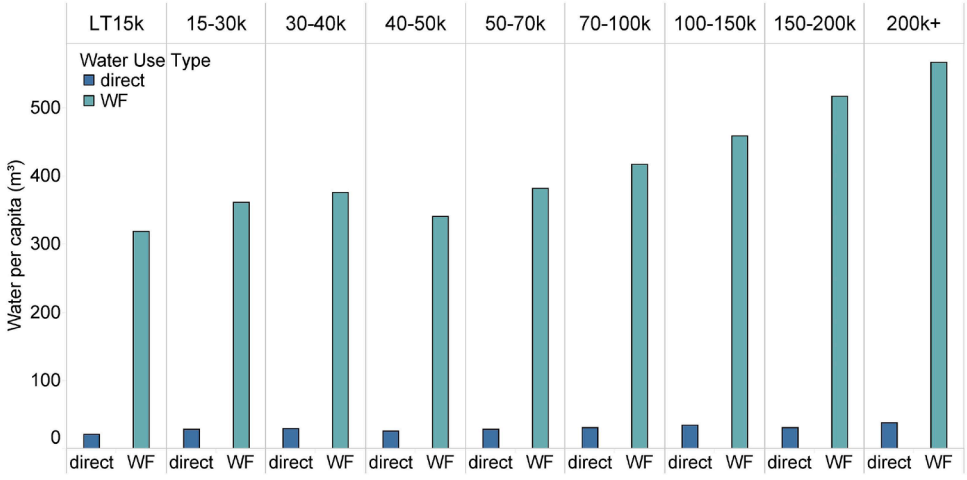

Fig. 4. Expenditure-based household water consumption by income categories: direct water use vs. water footprint.
Direct water use refers to residential water use. Water footprint refers to the total water consumption (direct and indirect). Incomes are ordered from lowest (less than $15k) to highest (more than $200k).

higher-income households having a larger water footprint per household compared to lower-income households, as detailed in Fig. 5.

When adjusted for household size, as depicted in Fig. 5, this strong association regarding per capita household water footprints with income remains. Statewide, the highest-income group's per capita water footprint (WF) was 1.8 times greater than the lowest-income group. Meanwhile, in agricultural regions, households typically exhibited higher per capita WFs compared to those in non-agricultural areas. For example, households in the San Joaquin Valley earning over $200k had the highest per capita WF of 1117.6 m^3, which was 2.3 times the WF of

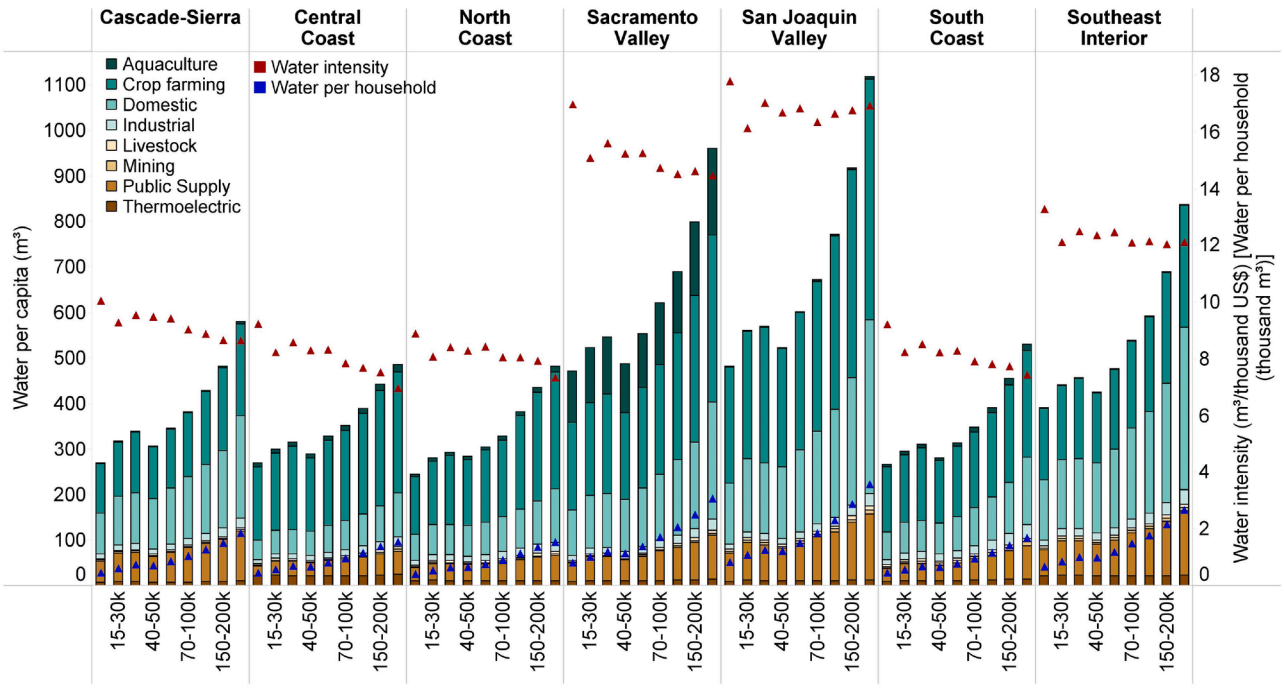

Fig. 5. Water footprints and water intensity of household consumption for 9 income groups.
The bars represent the composition of per capita household water footprint. The triangles represent water intensity measured as water footprint per thousand dollars of household expenditure (red) and water per household (blue, in thousand m^3).

the same income group in the Central Coast, primarily due to higher water usage in crop farming and domestic activities (water used for indoor and outdoor household purposes). For instance, the per capita WF in crop farming for households earning over $200k in San Joaquin Valley was 529.2 m^3, while in the South Coast, it was only 233.9 m^3. Thermoelectricity significantly impacted the water footprints in regions like the Central Coast and Southeast Interior. Here, high-income households in the Central Coast had a per capita WF from thermoelectricity of 24.4 m^3, nearly double that of those in the San Joaquin Valley at 13.0 m^3. Additionally, Fig. 5 illustrates that households earning $40-50k had a notably lower per capita WF relative to adjacent income brackets, a discrepancy accounted for by adjusting for household size and total expenditures per income group, as provided by IMPLAN. This variation is explained in Figure A6 by lower per capita expenditure due to a higher household count in this income bracket reported by the ACS during this period.

The water intensity, defined as water footprint per thousand dollars spent on household expenses, typically decreased when income increased. At the state level, the highest water intensity was observed in the lowest-income bracket (below $15k), remaining at about 10 m^3 per thousand dollars for households earning under $70k. Beyond this income level, water intensity starts to decline, though the reduction was relatively minor. For instance, the water intensity for the $200k+ income bracket was 27.5% lower than that of the below $15k group (11.2 m^3/thousand US$), primarily due to higher spending on services and transportation, which are less water-intensive compared to other expenses like food (refer to Figures A5 and A6 for expenditure breakdowns). Similarly to per capita WF, households in agricultural regions had higher water intensities compared to those in non-agricultural regions. For instance, households earning below $15k in the San Joaquin Valley had a water intensity of 17.8 m^3/ thousand US$, which was 2.1 times higher than those living in the North Coast region. Interestingly, in the San Joaquin Valley and Southeast Interior, water intensity for higher-income households actually increased from the $100-150k bracket to the $200k+ bracket, suggesting a more water-intensive lifestyle among these wealthier residents.

3.3. Drivers of water footprint gaps across income groups

Fig. 6 presents the IDA results, detailing how three factors influence per capita water footprint (WF) variations across five income brackets: lowest (below $15k), lower ($30-40k), middle ($50-70k), higher ($100-150k), and highest (above $200k). Despite the highest-income bracket earning over 10 times more than the lowest, the per capita WF only increased by 77.4%. The gaps in per capita WF were mainly attributed to differences in household expenditure between income groups. Assuming equal household sizes and water intensities, per capita WF rose by 167.1 m^3, or 52.3%, when comparing the lowest-income group to the lower-income group. For higher-income levels, the increase was even more pronounced, with a jump of 210.9 m^3 (46.0%) from the higher-income group to the highest-income group.

Additionally, U.S. households with larger incomes tend to be larger. This trend implies that more individuals share the same resources, which in turn leads to a lower per capita WF. Our analysis indicates that the differences in household size alone significantly reduced the disparity in per capita WF among the lowest- (below $15k, 1.74 people) and the lower-income households ($30-40k, 2.23 people), with a 19.0% decrease in per capita WF observed for the lower-income group compared to the lowest when keeping the other two factors constant. For the higher-income households ($100-150k, 3.01 people) and the highest-income households ($200k+, 3.21 people), this number was 5.5%, due to the smaller differences in household size.

Besides, the impact of household consumption patterns on per capita WF differences was relatively minor for households earning under $50-70k compared to other factors. However, these patterns significantly impacted the higher-income households. For instance, changes alone in household consumption patterns would result in a decrease of 69.8 m^3 (15.2%) in per capita WF of the highest-income households ($200k+) compared to the higher-income households ($100-150k).

Regionally, the trends identified in household per capita WF were generally consistent across areas. In the San Joaquin Valley and Sacramento Valley, inequalities in per capita WF among different income groups were significantly influenced by household expenditure, which

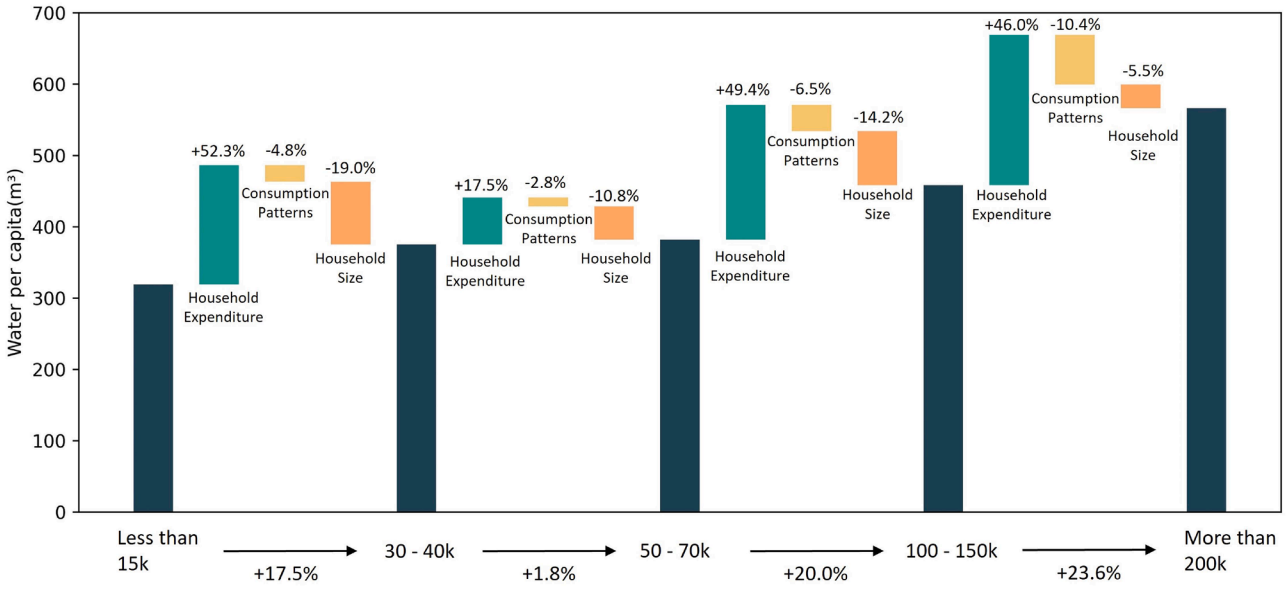

Fig. 6. Contributions of major factors in explaining the difference of per capita WF across income groups.
The dark bars represent per capita water footprint across income groups; the green, yellow, orange color bars show how major factors contribute to the disparity.

tends to widen the gap. The impact of household expenditure on the gap was more pronounced in these regions, while the influence of consumption patterns was considerably less impactful. In the San Joaquin Valley, the increase in per capita WF from the higher-income bracket ($100-150k) to the highest-income bracket ($200k+) was a unique example where changes in consumption patterns widen the gap. For a detailed regional visualization, please refer to Figure A7.

4. Discussion

Interregional free-trade of products typically transfers embodied (virtual) water from more water-productive regions to less productive ones, potentially leading to overconsumption beyond local carry capacities (Motoshita et al., 2023). For instance, while irrigation can significantly enhance agricultural productivity in arid regions, it can also cause local water scarcity (Fereres & Soriano, 2007). Thus, grasping the local water implications as well as the virtual water effects exchanged across regions and their spatial distribution is essential for mitigating the indirect environmental consequences of consumption and ensuring the sustainability of the entire economic system.

Our research provides an in-depth examination of California's water footprint across county-sectoral divisions, linking water usage to specific final consumption categories, and quantifying the virtual water flows between counties. Although many studies provide valuable insights on California's water footprint, they are often impeded by the lack of up-to-date, high-quality data (Sun et al., 2021). In contrast, our study leverages detailed and recent datasets to overcome this limitation. For example, while Fulton et al. (2014) and Fulton and Cooley (2015) examined the water footprint of California's agricultural, industrial, and energy sectors, their models were limited to state-level aggregations and selected sectors. This approach restricted their ability to assess inter-county or inter-sectoral virtual water transfers. Our research advances this by disaggregating data at the county level, offering a granular view of sectoral dynamics and virtual water flows.

Meanwhile, the existing studies using bottom-up, process-based approaches provide granular views, but often focus solely on the most significant processes, thus missing the full industrial supply chains (Wang et al., 2021) leading to gaps in understanding sector interconnections. In contrast, our top-down approach, utilizing the MRIO modeling approach, is grounded in comprehensive water data from USGS and USDA, enabling a holistic assessment of water use across the entire economic spectrum. Marston and Konar (2017) applied the commodity flow approach to investigate the water footprints and virtual water transfers of the Central Valley's agricultural products at the county level. However, their analysis did not extend to virtual water flows associated with secondary products, such as processed goods, livestock, or meat products, due to a lack of comprehensive supply chain data. Our study fills this gap by integrating the most complete MRIO data available, which enables us to include these secondary products and extend our analysis to other economic sectors.

Mubako et al. (2013) used an input-output model to analyze virtual water transfers in California and Illinois at the state-level, but only focused on water use differences and bilateral trade between these two states. Our study delves deeper by examining water footprints and virtual water transfers across all 58 counties within California, revealing previously unidentified interdependencies between regions and sectors. Our model also enables the study of final demand categories, particularly from the perspective of household water footprints, thus providing valuable insights into the socioeconomic dimensions of water use. This comprehensive approach not only advances the current understanding but also sets a foundation for future policy and management strategies.

A sectoral breakdown reveals that California's virtual water exports were largely due to crop farming, while imports were mainly associated with industrial processes, mining, and thermoelectric power. Although imports potentially saved water in the South Coast, significant exports from the San Joaquin Valley intensified local water stress. Our study also differentiates production-based and consumption-based water footprints across counties. Central Valley counties, heavily involved in crop farming, showed high production-based water use, producing goods for both local and external demands. Conversely, populous counties like Los Angeles and San Diego showed high consumption-based water use, driven by their dependence on imported virtual water, stimulating production elsewhere. Recognizing these water use disparities between sectors and regions discloses the multifaced relationships in water management, enabling more equitable policy development.

Our findings reveal that household final demand is the primary driver of virtual water consumption in California. By linking household consumption survey data with the MRIO model, we analyzed water footprints across different income groups, revealing the socioeconomic dimensions of water use, understanding how income, household size, and consumption patterns impact effective water demand. Regarding direct water use, households earning below $15k had a significantly

lower per capita metric compared to other income groups, a disparity primarily due to affordability issues. These households often face financial constraints, such as having to allocate funds from other expenses to pay for water (Mack & Wrase, 2017), which limit their direct water usage. In terms of indirect water use, high-income households typically exhibited a higher water footprint, reflecting their increased consumption of goods and services and, consequently, greater virtual water use. This difference was further pronounced when considering the influence of household size, maintaining the inequality in per capita WF among income groups. In agriculturally intensive regions like the San Joaquin Valley, households' involvement in water-intensive farming significantly boosted their water intensity and per capita WF. In contrast, in non-agricultural regions, such as the highly urbanized and affluent Central Coast, increasing energy demands led to higher thermoelectric water use and an elevated overall water footprint.

Furthermore, lower-income households typically spend a larger proportion of their income to essentials like food (USDA, 2023), while higher-income households often allocate more on non-essential and luxury items, such as recreational activities (e.g. restaurants), education, services, and other leisure. These varying consumption patterns shape each income group's water footprint, resulting in lower water intensity for higher-income households. However, our index decomposition analysis reveals that differences in consumption patterns have minimal effect on reducing inequality. The significant difference in per capita WF across income categories is largely due to differences in total consumption levels. It's also important to note that our highest-income category does not represent the wealthiest households in California. Consequently, the water footprint associated with "luxury" consumption, such as pools and extensive landscaping, might not be fully represented in our analysis.

Our findings underscore the necessity for tailored water management policies that address both statewide efficiency and local scarcity. The emphasis on socioeconomic factors also highlights the need for equitable water conservation strategies, which can inform more inclusive and effective policymaking. To effectively address California's water management challenges, comprehensive and forward-thinking strategies are required to balance the needs for both agricultural and residential sectors. In this regard, it is essential to control both production and consumption sides of water use. Irrigation for crop farming constitutes the predominant sector in production-based water use, and thus warrants consideration in water resource management. Enhancing agricultural sustainability requires financial incentives that support less water-intensive farming methods, such as tax incentives for water-efficient devices such as drip irrigation (Dagnino & Ward, 2012) and subsidies for crop rotation and water conservation (Wang et al., 2015). Water usage can be further decreased by funding research and development into sustainable practices like drought-resistant crops and effective irrigation techniques. The government and business sector working together to enhance farming infrastructure is also essential (Eanes et al., 2019). Furthermore, the implementation of precision agriculture technologies can optimize resource utilization. Educational programs can encourage farmers to transition to organic and diverse farming practices (Gerdes et al., 2020), thus contributing to a wider degree of environmental sustainability. In the residential sector, adopting or enhancing tiered rate structures to target high water use households (Harmon et al., 2021) and implementing more incentive programs for water-efficient behaviors and technologies in high-income households (Novak et al., 2018) are recommended. Investing in infrastructure for rainwater harvesting (e.g. cistern) (Dallman et al., 2021) and wastewater recycling (Maier et al., 2022) can significantly contribute to reducing overall water demand. Campaigns to raise public awareness can influence consumers' preferences for more environmentally friendly goods (Sánchez-Bravo et al., 2021). Additionally, revising water management policies to reflect the scarcity of water through effective pricing and regulations (Molle, 2009) is crucial for both agricultural and residential sectors.

Building on this study's outcomes, future research should explore the complex relationship between income disparities and water usage patterns, investigating how different income levels respond to water conservation policies and their impact on direct and indirect water use. A thorough examination of additional sectors, such as manufacturing and the service industry, could provide a more comprehensive understanding of California's water usage dynamics. It is important to appraise the effectiveness of diverse policy interventions, including different water pricing models and technologies, in promoting sustainable and equitable water management. Analyses at finer income scales and consumer sectors would offer insights into the water consumption behaviors of the wealthiest groups, contributing to a more nuanced understanding of California's water use. Additionally, future studies could combine analyses of green water and gray water to provide a more holistic view of water management.

5. Conclusion

This study offers a detailed analysis of California's water distribution and scarcity by examining virtual water flows at the county level. The inter-county disparities revealed by this study, which are often overlooked in broader national or state-level analyses, provide a clearer picture of local water use dynamics. Our study identifies critical disparities in water usage across various sectors and sub-state regions, emphasizing the complex interdependencies inherent in water management. For instance, while virtual water imports alleviated water stress in the South Coast region, significant exports from the San Joaquin Valley exacerbated local water scarcity. These insights are essential for informing more balanced and effective policy decisions.

By linking household consumption data with the MRIO model, we were able to analyze water footprints across different income groups, revealing significant socioeconomic dimensions of water use. Our findings show that income, household size, and consumption patterns play crucial roles in determining effective water demand, with higher-income households typically exhibiting a larger water footprint due to increased consumption of goods and services.

While our findings underscore the necessity for tailored water management policies that address both statewide efficiency and local scarcity, we acknowledge several limitations in translating these insights into actionable policies. First, the use of aggregated data for certain sectors may obscure localized nuances, potentially limiting the precision of policy recommendations. Additionally, the reliance on virtual water transfer data assumes stable supply chain dynamics, which may not hold under changing environmental or economic conditions. These factors suggest that any policies derived from our findings must be implemented with flexibility and regularly updated based on new data and changing conditions.

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CRediT authorship contribution statement

Baobao Liu: Writing – original draft, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Kuishuang Feng:** Writing – review & editing, Supervision, Formal analysis, Conceptualization. **Laixiang Sun:** Writing – review & editing, Conceptualization. **Giovanni Baiocchi:** Writing – review & editing. **Daoping Wang:** Methodology, Data curation. **Fernando Miralles-Wilhelm:** Writing – review & editing.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.strueco.2025.02.008.

Data availability

Data will be made available on request.

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