

Approaching institutional matches of river basin systems

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Abstract

Increasing competition for water is leading to depletion of freshwater globally and calls for efficient institutions in water governance. Alignment of social-ecological system (SES) structures is an crucial approach to institutional matches, but understanding of its causal links and underlying processes are still weaknesses. To fill these knowledge gaps, the Yellow River Basin in China is a valuable case study, because it is one of the most anthropogenically altered large river basins. Under different institutions, its stream-flow was first overdrawn, then dried up, and finally has been successfully restored. We focused on two institutional shifts, the Water Allocation Scheme that began in 1987 (87-WAS) and the Unified Basinal Regulation that took over in 1998 (98-UBR), which re-framed different social-ecological system (SES) structures. By creating counterfactual against, our results suggested that during the decade following the introduction of the 87-WAS, observed water use of the YRB increased by **8.57%** more than expected, while 98-UBR ultimately decreased total water use. Furthermore, these heterogeneous effects aligned with our further theoretical marginal benefits analysis, supporting the hypothesis that SES structural changes played a vital role in YRB restoration. This quasi-natural experiment on the YRB offers profound insights into the links between SESs structures and outcomes, providing valuable guidelines for water depletion in basins worldwide.

Keywords: Yellow River, water use, water governance, social-ecological system, institutional fit

1 INTRODUCTION

Widespread freshwater scarcity and overuse challenge the sustainability of large river basins, resulting in systematic risks to economies, societies, and ecosystems globally [1–4]. In the context of future climate change, the gap between

supply and demand for water resources in large river basins is expected to become increasingly more prominent [5, 6]. Those river basin systems successfully supporting sustainable water resource use are structurally well-aligned with water provisioning and social-ecological demands, without inefficient competition or overuses [7]. However, balancing the water demands of ecosystems and development in heavily human-dominated river basins is a challenge because human activities and water are intertwined in their structures as complex social-ecological systems (SES) [8, 9].

For governing river basin systems, their SES structures can be reshaped by institutions, such as policies, laws, and norms [10, 11]. Representing all relative governance practices, institutions include interplays between social actors, ecological units, or between social and ecological system elements [12, 13]. Understanding how these complex interplays is crucial for developing strategies to effectively manage natural resources and enhance the resilience of social-ecological systems [14]. However, the best approach to designing effective institutions remains an open question [15–17].

Effective (“matched” or “fit”) institutions operate at appropriate spatial, temporal, and functional scales to manage and balance different relationships and interactions between human and water systems, supporting (but not guaranteeing) the sustainability of SES [7, 18]. Some institutional advances have had desirable water governance outcomes (e.g., the Ecological Water Diversion Project in Heihe River Basin, China [7], and collaborative water governance systems in Europe [19]). However, imposing institutional changes on a large, complex river basin may create or destroy hundreds of connections between social agents and ecological units, where matched social-ecological structures are not ubiquitous. Two particular weaknesses in existing knowledge of institutional matches include understanding: (i) the causal links between SES structures and outcomes; (ii) details of the underlying processes, and especially the coordination of the incentives of different participants, that result from an institutional lack of match. These weaknesses limit understanding of institutional design, and hinder approaches toward institutional matches for improving the sustainability of river basin systems.

To better understand how water governance institutions can be designed to match their social-ecological context, we take the Yellow River Basin (YRB), China, as an example *Study area* to dive into causal links between SES structures and outcomes. We used data on changes in official documents following two institutional shifts (the 87-WAS and the 98-UBR) to describe comparable changes in the SES structures associated with the YRB from 1979 to 2008 *Portraying structures*. We then used a method called ‘Differenced Synthetic Control (DSC)’ [20], which considers economic growth and natural background, to estimate theoretical water use scenarios without institutional shifts (*Differenced Synthetic Control* and *Appendix B: Robustness of DSC method*). This approach allowed us to create a counterfactual against which to explore the mechanisms linking SESs structure and outcomes for a deeper

understanding of the potential role of institutions in water governance worldwide. Finally, we further developed an approach for marginal benefits analysis, to interpret the underlying processes of match and mismatched institutions based on SESs structures (*Marginal benefits analysis*).

2 RESULTS

2.1 Institutional shifts and structures

The institutional shifts in the YRB in 1987 (87-WAS) and 1998 (98-UBR) were two widely recognized milestones in restricting water use among YRB's water governance practices (*Study area* and *Appendix A: Contexts of institutional shifts*). Until the 87-WAS, stakeholders (the provinces in the YRB) had free access to the YR water resources for development, but there were geographic and temporal differences between freshwater demand and availability. The YRCC had no links to the provinces regarding water use before 1987, and the provinces could link directly to the Yellow River reaches (Figure 1 C). To shrink water deficits, in 87-WAS, national authorities proposed in 87-WAS allocating specific water quotas between 10 provinces (or regions) along the YR basin (Table A1). Simultaneously, according to the extracted information from documents of the 87-WAS issued by national ministries, the YRCC started to report water use in each reach. As it was the first time the responsibility of the YRCC involved water use, this introduced new links between the YRCC and the ecological nodes (Figure 1 C). However, the controversial 87-WAS did not resolve water depletion. In 1998, another strategy (98-UBR) was developed to strengthen the responsibilities of the YRCC for integrated managing water use. Information from the 98-UBR documents demonstrated that the provinces had to apply their plan for an annual water use license to YRCC instead of direct access to the Yellow River water. Thus, the YRCC has been linked to the provinces since 1998 (Figure 1 C).

2.2 Institutional shifts impact on water use

Our estimation of theoretical water use suggests that the institutional shift in 1987 (87-WAS) stimulated the provinces to withdraw more water than would have been used without an institutional shift (Figure 2A). From 1988 to 1998, on average, while the estimation of annual water use only suggests 974.34 billion m^3 , the observed water use of the YRB provinces reached 1038.36 billion m^3 (an increase of 6.57%). However, after the institutional change in 1998 (98-UBR), trends of increasing water use appeared to be effectively suppressed. From 1998 to 2008, the total observed water use decreased by 0.49 billion m^3/yr per year, while the estimation of water use still suggests

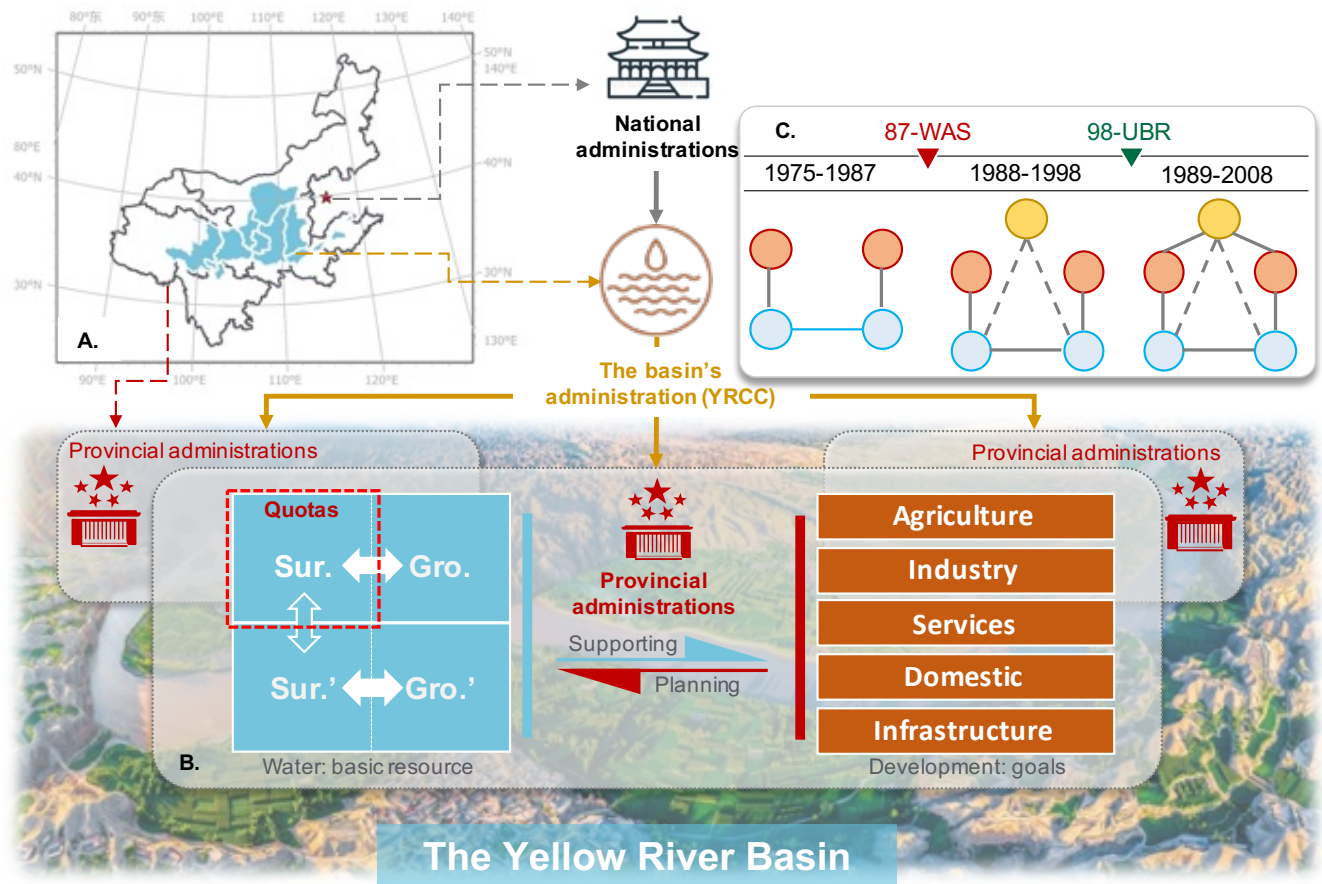


Fig. 1 Institutional shifts and related SES structures in the Yellow River Basin (YRB). **A.** The YRB crosses 10 provinces or the same-level administrative regions, 8 of which are highly relying on the water resources from the YRB (see [Appendix A: Contexts of institutional shifts](#) Table A1). The national administrations are the ultimate authority in issuing water governance policies, which are often implemented by basin-level agency (the Yellow River Conservancy Commission, YRCC) and each province-level agency. **B.** Provincial administrative agencies are the major stakeholders. Since the 87-WAS, with surface water withdrawal from the Yellow River restricted by specific quotas, each stakeholder plan and use water resources for development. However, the natural hydrological processes are connected. Although the institutions focus mainly on surface water (Sur.), it can also influence groundwater inside (Gro.) or water resources outside (Sur. and Gro.) through systematic socio-hydrological processes within the YRB. The YRCC only monitors water withdrawals at that time. **C.** Institutional shifts and following structures changes (details in [Appendix A: Contexts of institutional shifts](#)). (1) From 1979 to 1987, water resources were freely accessible to each stakeholder (denoted by red circles) from the connected ecological unit (the reach of Yellow River, denoted by the blue circles). (2) After 1987-WAS, the YRCC (the yellow circles) was monitoring (the dot-line links) river reaches with the water use quota. (3) Since the 98-UBR, stakeholders have to apply for water use licenses from the YRCC (the connections between the red and yellow circles).

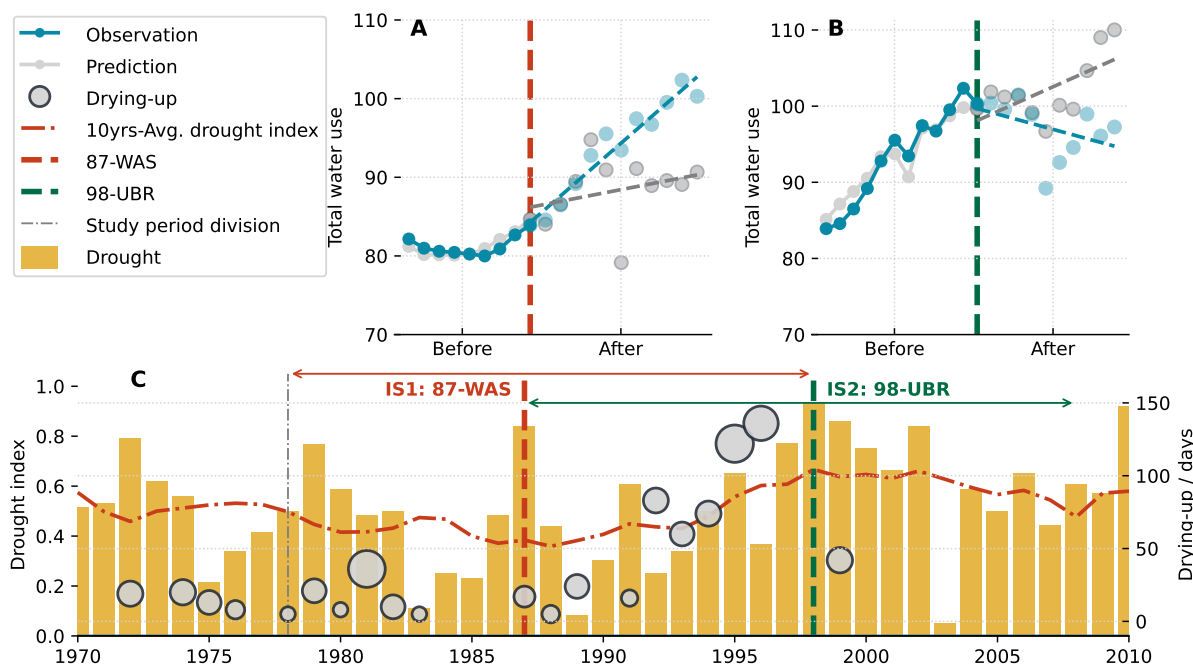


Fig. 2 Effects of two institutional shifts on water resources use and allocation in the Yellow River Basin (YRB). **A.** water uses of the YRB before and after the institutional shift in 1987 (87-WAS); **B.** water uses of the YRB before and after the institutional shift in 1998 (98-UBR). Blue lines are statistics derived from water use data; grey lines are estimates from the Differenced Synthetic Control method with economic and environmental background controlled; **C.** Drought intensity in the YRB and drying up events of the Yellow River. The size of the grey bubbles denotes the length of drying upstream.

0.82 billion m^3/yr increases (Figure 2 B). The increased water uses after 87-WAS aligns with the severe drying-down of the surface streamflow from 1987 to 1998, an obvious indicator of river degradation and environmental crisis (Figure 2C). On the other hand, the 98-UBR ended river depletion, despite subsequent increases in drought intensity (from 0.47 after 87-WAS to 0.62 after 98-UBR on average) (Figure 2C).

2.3 Heterogeneous institutional effects

Our results also suggest differences between patterns of provinces in their responses to the two institutional regulating. During the decade after the 87-WAS, the major water-using provinces (e.g., Inner Mongolia, Henan, Shandong) had apparent accelerations (Figure 3). The proportion of increased (or decreased) water use for each province (over the estimated water use by the model) correlated significantly (partial correlation coefficient is 0.77, $p < 0.05$) with actual water use from the Yellow River. On average, the major water users (Shandong, Inner Mongolia, Henan, and Ningxia) used 32.14% more water than predicted from 1987 to 1998. By contrast, after the 98-UBR (from 1998 to

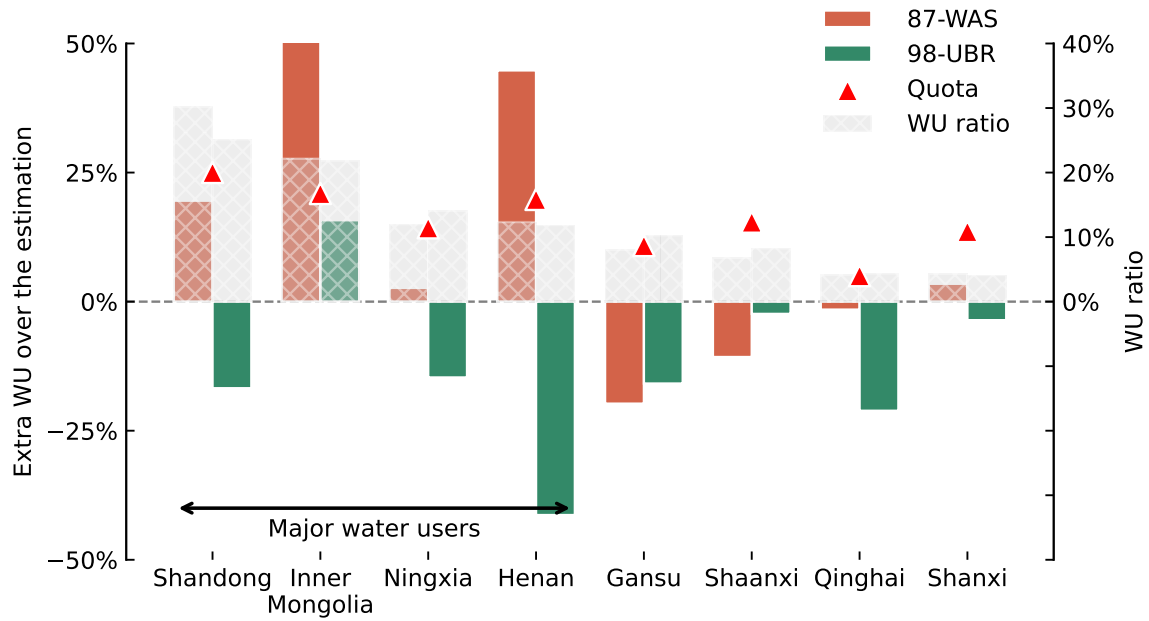


Fig. 3 Regulating differences for provinces in the YRB. Red (the 87-WAS) and green (the 98-UBR) bars denote an increased or decreased ratio for actual water use relative to the estimate from the model in the decade after the institutional shift. The grey bars indicate the proportions of actual water use for each province relative to their total water use in the decade after the institutional shift. The triangles mark the water quotas assigned under the institution, converted to ratios by dividing by their sum.

2008), almost all provinces have seen declines (-16.54% on average) in water use. Furthermore, the regulated water use of provinces was unrelated (partial correlation coefficient is 0.33, $p > 0.1$) to their proportional water use from the Yellow River.

2.4 Structure-based marginal benefit analysis

We compared the theoretical marginal returns and optimal water use under three different structural cases (analogy to Figure 1 C, see [Marginal benefits analysis](#) Figure 4, detailed derivation in [Appendix C: Optimization model for water use](#)). Assuming that water is the factor input with decreasing marginal output of each province, results show that varying incentives for water use in each province derive from the relationship between the benefits and costs of water use. Until water-use decisions are consolidated into unified management, each stakeholder's expectation that current water use helps bargain for a favorable water quota may intensify the incentive to use water, leading

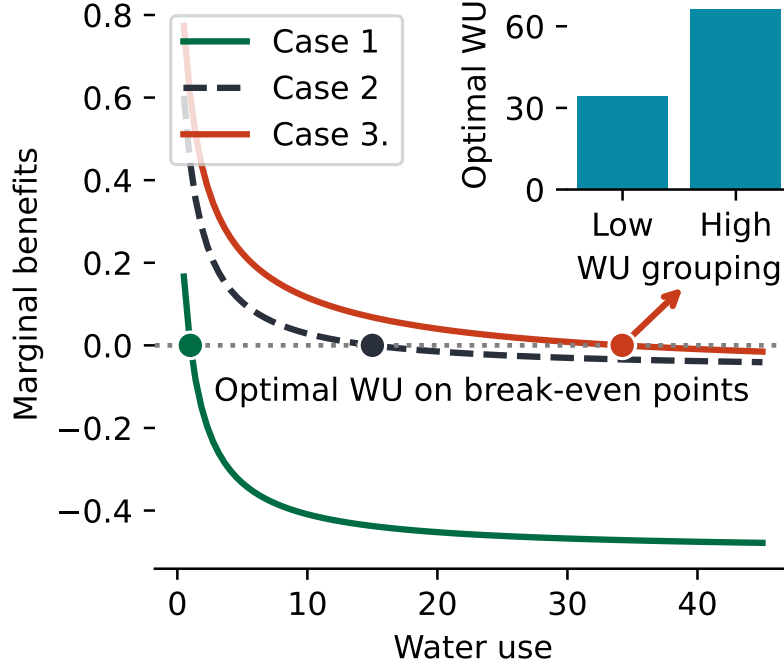


Fig. 4 The proposed relationship of marginal benefits and water use of individual province under varying cases (case 1 to case 3, corresponding to the different SES structures in Figure 1) Major water users' theoretically optimal water use is also larger (see [Marginal benefits analysis](#) and [Appendix C: Optimization model for water use](#)).

to higher water use. Furthermore, water users with higher capability are more stimulated by institutional shifts and away from the theoretically optimal water use under a unified allocation.

3 DISCUSSION

Our results show that while 98-UBR decreased water use in the YRB, 87-WAS increased it. The results challenged previous analyses (i.e., suggesting that 87-WAS "had little practical effect") because theoretically, there should be few gaps between actual and synthetic water use in the YRB if no effect is present [21, 22]. However, the significant net effect indicated by our analysis suggests 87-WAS was followed by more water use even after controlling for environmental and economic variables (see [Appendix B: Robustness of DSC method](#) Table B1). On the contrary, the 98-UBR reduced water competition, so many studies attributed the restoration mainly to the successful introduction of this institution [23–25].

Increased water use after 87-WAS aligns with concerns about frequently scrambling for water in some provinces during this period [26, 27]. Although reasons for the non-ideal effect of 87-WAS had been widely discussed [24] (such

119 as enforcement, feasibility, and equity), structural change has received limited attention. Our results show that the
120 correlation between current water use and changed (increased or decreased) water use was significant after 87-WAS
121 (Figure 3). This “major users use more” pattern supports the hypothesis that separated stakeholders (individual
122 provinces) will response to structure by maximizing utility (interpreted in our structure-based model, see Figure 4).

123 The validity of our theoretical analysis is supported two facts: (1) The water quotas of 87-WAS (or the initial
124 water rights) went through a stage of “bargaining” among stakeholders (from 1982 to 1987) [7, 28], where each
125 province attempted to demonstrate its development potential related to water use. The bargaining was also a process
126 for matching water shares to economic volume, because the major water users (like Shandong and Henan) needed
127 more water than their original quota (if only considering economic potentials when designing the institution) [29].
128 (2) Provinces with higher current water use might have greater bargaining power in water use allocation because
129 of information asymmetry between decision-makers and stakeholders. Therefore, stakeholders had considerable
130 incentives to prevent water quotas from hindering their economic potential, which aligned with their appeals to the
131 higher central government for larger shares [7, 28].

132 After 98-UBR, the YRCC could adjust water use quotas to match river conditions for the whole YRB. When
133 the YRCC began to coordinate among stakeholders, the external appeals of provinces for larger quotas turned into
134 internal innovation to improve water efficiency (e.g., drastically increased water-conserving equipment) [30, 31].
135 During this period, proportional decreased water use of provinces indicated a positive result of regulation (see
136 [Heterogeneous institutional effects](#)). The 98-UBR thus led to expected institutional outcomes at a basin scale,
137 indicating that successful governance of SES emerged by indirectly (or vertically) creating links between different
138 stakeholders. Our model demonstrates that in this case, a unified scale-matched institution was indispensable for
139 sustainable water use.

140 We explored causal linkages between SES structures and sustainability-related outcomes by quasi-natural exper-
141 iments (institutional shifts imposed by central government) in the YRB. The YRB provides an informative case for
142 two main reasons: (1) The top-down institutional shifts induced sharp changes in SES structures, enabling us to
143 estimate their net effects quantitatively. (2) Since few large river basins have experienced such radical institutional
144 shifts more than once, this case study provides a valuable natural experiment for understanding the impacts of
145 structural changes in SESs on natural resources.

146 The structural building blocks we depicted here (Figure 1) have also been reported in other SESs worldwide
147 [14, 32, 33]. Before 98-UBR, SES structure (i.e., fragment ecological units linked to separate social actors) was more
148 likely to be mismatched because isolated actors generally struggle to maintain interconnected ecosystems holistically
149 [34–37]. Institutional re-alignments since 98-UBR improved the authority of the YRCC and helped it match the

scale of resource provisioning in the YRB, leading to enhanced social-ecological fit and better outcomes [7, 38]. The comparison demonstrates again the challenge of finding win-win situations in coupled human-nature systems [39], and the need more deeply understand the role of social-ecological structures [35, 37].

3.1 LIMITATION, INSIGHTS AND IMPLICATIONS

Our approach has some inevitable limitations. First, the contributions of economic growth and institutional shifts are difficult to distinguish because of intertwined causality (institutional changes can also influence the relative economic variables); and second, when applying the DSC method, it is difficult to rule out the effects of other policies over the same time breakpoints (1987 and 1998). Our quasi-experiment approach nonetheless provides evidence supporting the view that there was a change in water use trajectory following the YRB’s unique institutional shifts, and offers insights into water governance (and particularly the importance of having a scale-matched, basin-wide authority for water allocation solutions [13, 40, 41]) Moreover, the ultimate success of the 98-UBR institutional shift theoretically and practically proved the importance of social-ecological fit. For sustainability in the future, therefore, it is necessary to emphasize the necessity of strengthening connections between stakeholders by agents consistent with the scale of the ecological system. From these perspectives, two scenarios based on the marginal benefit analysis (see *Appendix D: Model extensions*) can inspire institutional design on how to reduce mismatches. For example, water rights transfers may be another way to build horizontal links between stakeholders that also have the potential to result in better water governance. In addition, policymakers can propose more dynamic and flexible institutions to increase the adaptation of stakeholders to a changing SES context [41].

The structural building blocks that led to different outcomes are recurring motifs in global SESs, so our proposed mechanism is crucial to governing such coupled systems. Calls for a redesign of water allocation institutions in the YRB in recent years also illustrate the importance of institutional solutions for sustainability (see *Appendix A: Contexts of institutional shifts*) [42]. Given the changing environmental context, outdated and inflexible water quotas can no longer meet the demands of sustainable development [28]. Thus, the Chinese government has embarked on a plan to redesign its decades-old water allocation institution (see *Appendix A: Contexts of institutional shifts*). Our analysis suggests that these initiatives can benefit by actively incorporating social-ecological matched building blocks when developing new institutions [13]. Moreover, our research provides a cautionary tale of how institutions can create perverse incentives [39], while insights from the YRB contribute to improving guidelines for SESs management worldwide [43, 44].

178 4 CONCLUSION

179 Intense water use in one of the most anthropogenically altered large river basins, the Yellow River Basin (YRB),
180 once led to drying up. Alterations of institutions eventually successfully restored water governance practices on a
181 decadal time scale. We propose that the institutional shifts in the YRB (87-WAS and 98-UBR) framed two different
182 SES structures and depicted them as widespread building blocks. We quantitatively estimate the net effects of these
183 changes in the YRB and analyze the reasons from SES structural perspectives. Our results show that the historical
184 records, the responses from stakeholders to structural changes, and the theoretical analysis from the marginal
185 benefits analysis all support that fragmented ecological units linked to separate social actors frames a mismatched
186 SES structure. Through the quasi-natural experiments of the YRB, we demonstrate that social-ecological fits can
187 lead to successful SESs management worldwide with better sustainability outcomes.

188 5 MATERIALS AND METHODS

189 We first abstract the SES structures of water used in the YRB from 1979 to 2008, where two institutional shifts split
190 the period into three pieces. To process the data, we use the Principal Components Analysis (PCA) method to reduce
191 the dimensionality of variables affecting the total water use. We then estimated the net effects of two institutional
192 shifts on total water use, changing trends, and differences of the YRB's provinces, by Differenced Synthetic Control
193 (DSC) method [20]. Finally, for theoretical discussion, we developed a marginal benefit analysis based on identified
194 SES structures to provide the observed pattern of water use changes with a theoretical interpretation.

195 5.1 Study area

196 The Yellow River Basin (YRB), the fifth-largest river basin worldwide, is known for its vital role in the socio-
197 economic development of China. It supports 35.63% of China's irrigation and 30% of its population while containing
198 only 2.66% of its water resources (data from <http://www.yrcc.gov.cn>, last access: July 1, 2022). In the 1980s, intense
199 water use, accounting for about 80% of Yellow River surface runoff, combined with other forms of human interference
200 (e.g., soil conservation and water conservancy projects), caused consecutive drying events and substantial ecological,
201 economic, and social crises (e.g., wetland shrinkage, agriculture reduction, and a scramble for water). In response,
202 Chinese authorities implemented several ambitious water management practices in the YRB to relieve water stress,
203 such as reservoir regulation, the South-to-north Water Diversion Project (WDP), the 1987 Water Allocation Scheme
204 (87-WAS), and the 1998 Unified Basinal Regulation (98-UBR) [7, 45]. Those efforts led to ecological restoration
205 of wetlands and the estuarine delta. Drying up has been avoided for over 20 years, which is widely considered

a substantial management achievement. Instead of relying on engineering to increase water supply, institutional strategies like the 87-WAS (which assigned water quotas for provinces in the YRB) and the 98-UBR (under which provinces had to obtain permits from the Yellow River Conservancy Commission, YRCC, authority at a basin-level) focused mainly on limiting demand for water [27, 46]. While researchers have carefully evaluated and quantified the effects of engineering solutions on water supply[45], there have been few attempts to assess institutional contributions to successful water governance in the YRB.

5.2 Portraying structures

We apply the network [13] approach to portray SES structures by abstracting relationships between ecological units (river reaches), stakeholders (provinces), and the administrative unit (the YRCC) into general building blocks (or motifs) (see Figure A1), from the official documents. Empirical studies have suggested that such widespread building blocks in SES are the key to the functioning of structures. The network-based approach is to abstract connections between entities into links and nodes [14, 32, 47]. In this study, we examined the official documents of the two institutional shifts of concern (87-WAS and 98-UBR, see *Appendix Appendix A: Contexts of institutional shifts* for details). Besides the ecologically connected river reaches, the agents (provinces and the YRCC) are abstracted as nodes, and their required interactions regarding water use are summarized as links. The 1987-WAS requires the YRCC to monitor each river’s reach, while the 1998-UBR requires direct interactions (through water use licenses) between the YRCC and the provinces. Therefore, we linked the YRCC unit to each ecological unit after 87-WAS and each province unit after the 98-UBR. We tested whether focusing on SES structures rather than institutional details could reasonably explain the differences caused by institutional shifts in the YRB.

5.3 Dataset and preprocessing

We choose datasets and variables to compare on actual and estimated water use of the YRB. The actual water uses are accessible in China’s provincial annual water consumption dataset from the National Water Resources Utilization Survey, whose details are accessible from Zhou (2020) [48]. To estimate the water use of the YRB by assuming there were no effects from institutional shifts, we focused on variables from five categories (environmental, economic, domestic, and technological) water use factors. Their specific items and origins are listed in Table B1.

Among the total 31 data-accessible provinces (or regions) assigned quotas in the 87-WAS and the 98-UBR, we dropped Sichuan, Tianjin, and Beijing because of their trivial water use from the YRB (see *Appendix Table A1*). We then divided the dataset into a “target group” and a “control group”, treating provinces involved in water quota as the target group ($n = 8$) and other provinces as the control group ($n = 20$) for applying the DSC.

Using the normalized data of all variables, we performed the PCA reduction to capture 89.63% explained variance by 5 principal components *Appendix Appendix B: Robustness of DSC method*. Bayan had proved that combining PCA and DSC can raise the robustness of causal inference [49]. We first applied the Zero-Mean normalization (unit variance), as the variables' units are far different. Then, we apply PCA to the multi-year average of each province, using the Elbow method to decide the number of the principal components (*Appendix Appendix B: Robustness of DSC method Figure B5*). Finally, we transform the dataset and input the dimensions-reduced output into the DSC model.

5.4 Differenced Synthetic Control

Using the Differenced Synthetic Control (DSC) method, we estimate water use without the effect of the institutional shift. The DSC method is an effective identification strategy for estimating the net effect of historical events or policy interventions on aggregate units (such as cities, regions, and countries) by constructing a comparable control unit [21, 22, 50].

This method aims to evaluate the effects of policy change that are not random across units but focuses on some of them (i.e., institutional shifts in the YRB here). By re-weighting units to match the pre-trend for the treated and control units, the DSC method imputes post-treatment control outcomes for the treated unit(s) by constructing a synthetic version of the treated unit(s) equal to a convex combination of control units. Therefore, the synthetic and actual version difference can be estimated as a net effect for a treated unit.

In practice, all treated units (i.e., provinces) were affected by institutional shifts in 1987 and 1998, each taken as the “shifted” time t_0 within two individually analyzed periods T : 1979-1998; 1987-2008. We include each province in the YRB ($n = 8$, see *Dataset and preprocessing*) as the treated unit separately, as multiple treated units approach had been widely applied [51]. Then, we consider the $J + 1$ units observed in time periods $T = 1, 2 \dots, T$ with the remaining $J = 20$ units are untreated provinces from outside. We define T_0 to represent the number of pre-treatment periods $(1, \dots, t_0)$ and T_1 the number post-treatment periods (t_0, \dots, T) , such that $T = T_0 + T_1$. The treated unit is exposed to the institutional shift in every post-treatment period T_0 , unaffected by the institutional shift in all preceding periods T_1 . Then, any weighted average of the control units is a synthetic control and can be represented by a $(J * 1)$ vector of weights $\mathbf{W} = (w_1, \dots, w_J)$, with $w_j \in (0, 1)$. Among them, by introduce a $(k * k)$ diagonal, semidefinite matrix \mathbf{V} that signifies the relative importance of each covariate, the DSC method procedure for finding the optimal synthetic control (W) is expressed as follows:

$$\mathbf{W}^*(\mathbf{V}) = \underset{\mathbf{W} \in \mathcal{W}}{\text{minimize}} (\mathbf{X}_1 - \mathbf{X}_0 \mathbf{W})' \mathbf{V} (\mathbf{X}_1 - \mathbf{X}_0 \mathbf{W}) \quad (1)$$

where $\mathbf{W}^*(V)$ is the vector of weights \mathbf{W} that minimizes the difference between the pre-treatment characteristics of the treated unit and the synthetic control, given \mathbf{V} . That is, \mathbf{W}^* depends on the choice of \mathbf{V} –hence the notation $\mathbf{W}^*(\mathbf{V})$. Therefore, we choose \mathbf{V}^* to be the \mathbf{V} that results in $\mathbf{W}^*(\mathbf{V})$ that minimizes the following expression:

$$\mathbf{V}^* = \underset{\mathbf{V} \in \mathcal{V}}{\text{argmin}} (\mathbf{Z}_1 - \mathbf{Z}_0 \mathbf{W}^*(\mathbf{V}))' (\mathbf{Z}_1 - \mathbf{Z}_0 \mathbf{W}^*(\mathbf{V})) \quad (2)$$

That is the minimum difference between the outcome of the treated unit and the synthetic control in the pre-treatment period, where \mathbf{Z}_1 is a $(1 * T_0)$ matrix containing every observation of the outcome for the treated unit in the pre-treatment period. Similarly, let \mathbf{Z}_0 be a $(k * T_0)$ matrix containing the outcome for each control unit in the pre-treatment period, and k is the number of variables in the datasets. The DSC method generalizes the difference-in-differences estimator and allows for time-varying individual-specific unobserved heterogeneity, with double robustness properties [52, 53].

5.5 Marginal benefits analysis

To infer the mechanisms underlying the results, we developed an marginal benefits analysis based on marginal revenue to analyze how the institutional shift could have led to differences in water use.

Assumption 1. (*Water-dependent production*) Because of irreplaceably, water is assumed to be the only production function input with two production efficiency types.

Assumption 2. (*Ecological cost allocation*) Under the assumption that the ecology is a single entity for the whole basin, the water use cost is equally assigned to each province.

Assumption 3. (*Multi-period settings*) There are multiple settings periods with a constant discount factor for the expectation of future water use.

Under the above-simplified assumptions, we demonstrate three cases -corresponding to the abstracted SES structures (Figure 1), inference of how SES structure alters the expected marginal benefits and costs of provinces making decisions. As one of the possible interpretations for the causality between SES structure and institutional effects, the derivation of the model based on the above three assumptions can be found in *Appendix Appendix C*:

285 *Optimization model for water use*, and some simple model-based extensions are involved in *Appendix Appendix D:*
286 *Model extensions*.

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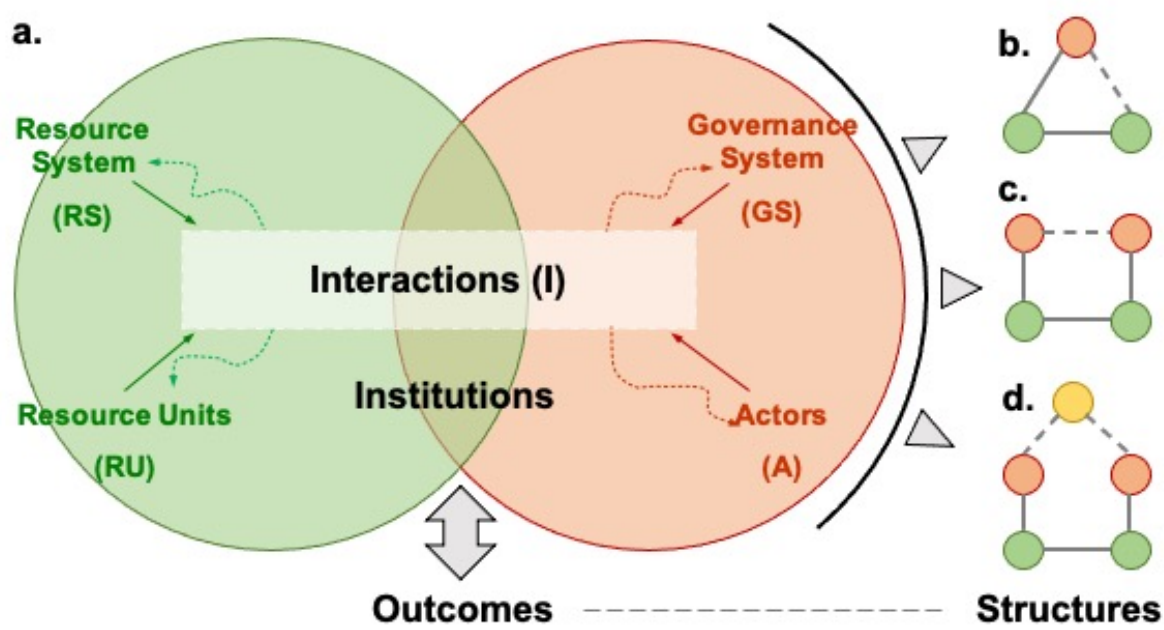
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409 **A Appendix A: Contexts of institutional shifts**

410 We aim to abstract the water allocating institutions from the description in official documents with necessary
411 context into SES building blocks (Figure A1) Widespread building blocks in SES are the key to the functioning of
412 structures, and a network-based description is a widely used way to depict them by abstracting links and nodes
413 [14, 32, 47].



Supplementary Figure A1 Framework for understanding linkages between SES structures and outcomes. **a.** The general framework for analyzing social-ecological systems (SESs) (adapted from Ostrom [40]). Institutions embedded in SESs may reshape structures by changing the interactions between core subsystems, resulting in different outcomes. Three typical types of abstracted SES structures are shown as **b.**, **c.** and **d.** (adapted from Bodin, 2017)[13]. Red circles indicate social actors, and green ones indicate ecological components. Connection (ties between two ecological components), collaboration (ties between two social actors), or management (ties between a social actor and an ecological component) exist when gray lines link two units. According to empirical evidence, the gray dashed lines show aligned SES structures that are more likely to achieve a desirable outcome.

414 Water allocation institutions are widespread in large river basin management programs throughout the world
415 (see *Appendix* Figure A2) [46]. This was the first basin in China for which a water resource allocation institution
416 was created, and institutional shifts can be traced through several documents released by the Chinese government
417 (at the national level)[28]:

- **1982:** The provinces and the Yellow River Water Conservancy Commission (YRCC) are required to develop a water resource plan for the Yellow River [28, 54].
- **1987:** Implementation of the Allocation Plan. (http://www.gov.cn/zhengce/content/2011-03/30/content_3138.htm#, last access: July 1, 2022).
- **1998:** Implementation of unified regulation. (http://www.mwr.gov.cn/ztpd/2013ztbd/2013fxkh/fxkhswcbcs/cs/flfg/201304/t20130411_433489.html, last access: July 1, 2022).
- **2008:** Provinces are asked to draw up new water resources plans for the YRB to further refine water allocations [28, 54].
- **2021:** A call for redesigning the water allocation institution (http://www.ccg.gov.cn/cggg/zygg/gkzb/202107/t20210721_16591901.htm, last access: July 1, 2022).

Since 1982, administrations attempted to design a quota institution, and the 2008 document marked the maturity of the scheme (complete establishment of basin-level, provincial, and district water quotas). Between the period, two significant institutional shifts can be analyzed by using the 1987 (87-WAS) and 1998 (98-UBR) documents.

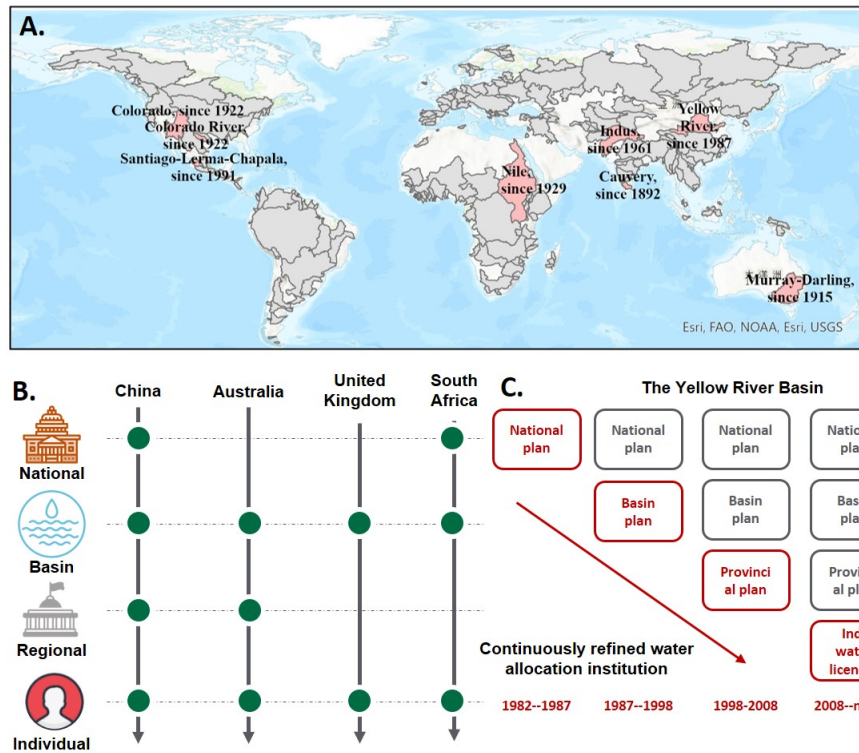
The official documents in 1987 (http://www.gov.cn/zhengce/content/2011-03/30/content_3138.htm#, last access: July 1, 2022) convey the following key points:

- The policy is aimed at related provinces (or regions at the same administrative level).
- Depletion of the river is identified as the first consideration of this institution.
- Provinces are encouraged to develop their water use plans based on a quota system.
- Water in short supply is a common phenomenon in relevant provinces (regions).

The official documents in 1998 (http://www.mwr.gov.cn/ztpd/2013ztbd/2013fxkh/fxkhswcbcs/cs/flfg/201304/t20130411_433489.html, last access: July 1, 2022) convey the following key points:

- The document points out that not only provinces and autonomous regions involved in water resources management (see *Article 3*), the provinces' and regions' water use shall be declared, organized, and supervised by the YRCC (*Article 11 and Chapter III to Chapter V, and Chapter VII*).
- Creating the overall plan of water use in the upper, middle, and lower reaches is identified as the first consideration of this institution (*Article 1*).
- With the same quota as used in the 1987 policy, provinces were encouraged to further distribute their quota into lower-level administrations (see *Article 6 and Article 41*).

- They emphasize that supply is determined by total quantity, and water use should not exceed the quota proposed in 1987 (see *Article 2*).



Supplementary Figure A2 Overview of water allocation institutions. **A.** Major river basins in the world with water resource allocation systems (shaded red); the YRB first proposed a resource allocation scheme in 1987 (designed since 1983) and then changed to a unified regulation scheme in 1998 (designed in 1997 but implemented in 1998) [46]. **B.** Different water resource allocation system design patterns; the YRB is typical of a top-down system. **C.** The four periods of institutional evolution of water allocation of the YRB.

Based on the above documents, we abstracted the structural changes of SES (see *Appendix S2*) after the two institutional changes, as shown in Figure 1 C.

B Appendix B: Robustness of DSC method

Explanatory variables are the key to constructing a robust synthetic control method. We used a total of 24 variables related to water consumption Table B1, which datasets have been used in previous studies to explain changes in

Table A1 Water quotas assigned in the 87-WAS

Items (water volume, billion m^3)	Qinghai	Sichuan	Gansu	Ningxia	Inner Mongolia	Shanxi	Shaanxi	Henan	Shandong	Jinji
Demands in water plan	35.7	0	73.5	60.5	148.9	115	60.8	111.8	84	6
Quota designed in 1983	14	0	30	40	62	43	52	58	75	0
Quota assigned in 1987	14.1	0.4	30.4	40.0	58.6	38.0	43.1	55.4	70.0	20
Average water consumption from the Yellow River from 1987-2008	12.03	0.25 ^a	25.80	36.58	61.97	21.16	11.97	34.30	77.87	5.85 ^a
Proportion of water from the Yellow River in total water consumption	48.12%	0.10 ^b %	30.79%	58.45%	47.82%	73.55%	44.39%	24.77%	34.41%	3.11% ^b

[a]Calculated by data from 2004 to 2017.

[b]The share is too small, thus the provinces (or region) Sichuan and Jinji not to be considered in this study.

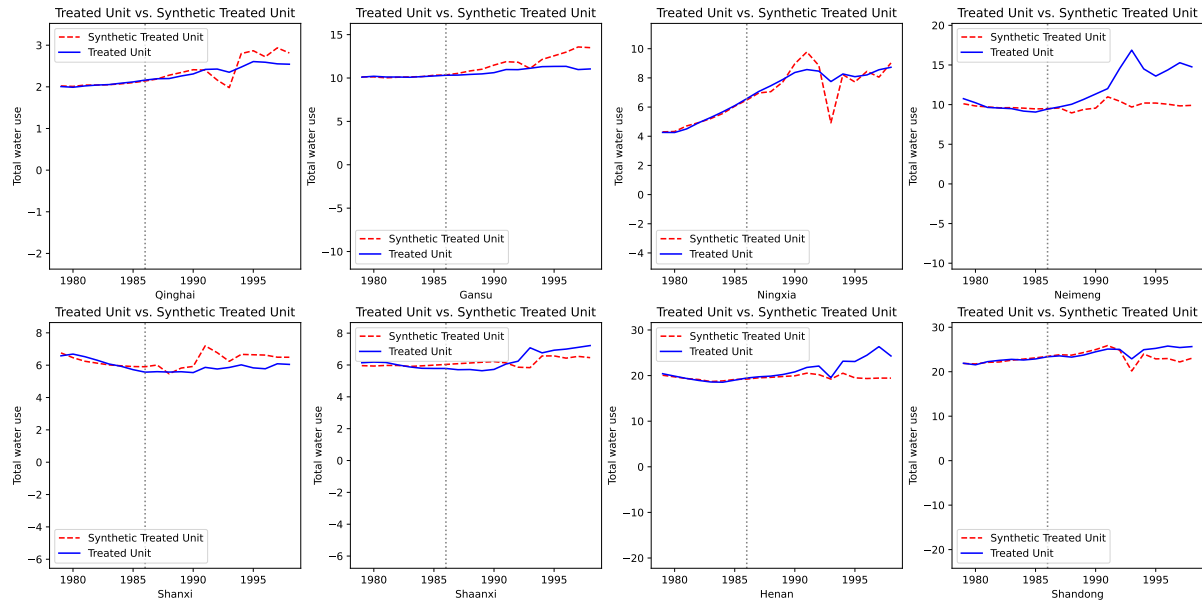
water use in China [48]. In addition, we selected 5 principal components as input by the elbow method because selection in autocorrelated variables reduces dimensions and then enhances the robustness of the DSC (Figure B5). There are two approaches to validity testing of the DSC: (1) comparing the post-treated and pre-treated reconstructions and (2) testing robustness through placebo analysis. For (1), differences between each province and their synthetic are significant in post-treated periods and small in pre-treated periods (Figure B1 and figure B2), which show good reconstructions of their water use changes' estimation. For (2), we applied the in-place placebo analysis described by [50]. In most provinces, ratios of post-MSPE to pre-MSPE are higher than the median of other placebo units, which suggests the institutional shifts in treated time (1987 and 1998 here) influenced them more than most of the other provinces (figure B3, figure B4, Table B2).

C Appendix C: Optimization model for water use

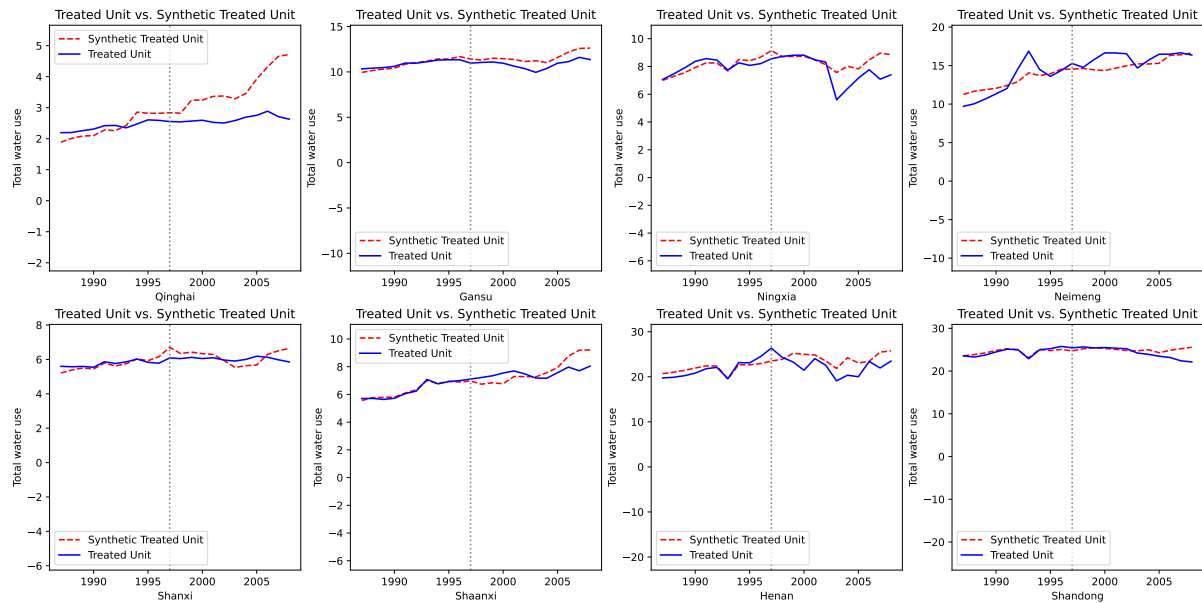
Setup

To understand the mechanisms through which the SES structure impacts provincial water use, we developed a dynamic marginal benefits analysis to analyze how institutional mismatch could have led to the changes in water use, especially among provinces with high incentives for excess water use. Specifically, we modeled individual provincial decision-making in water resources before quota execution.

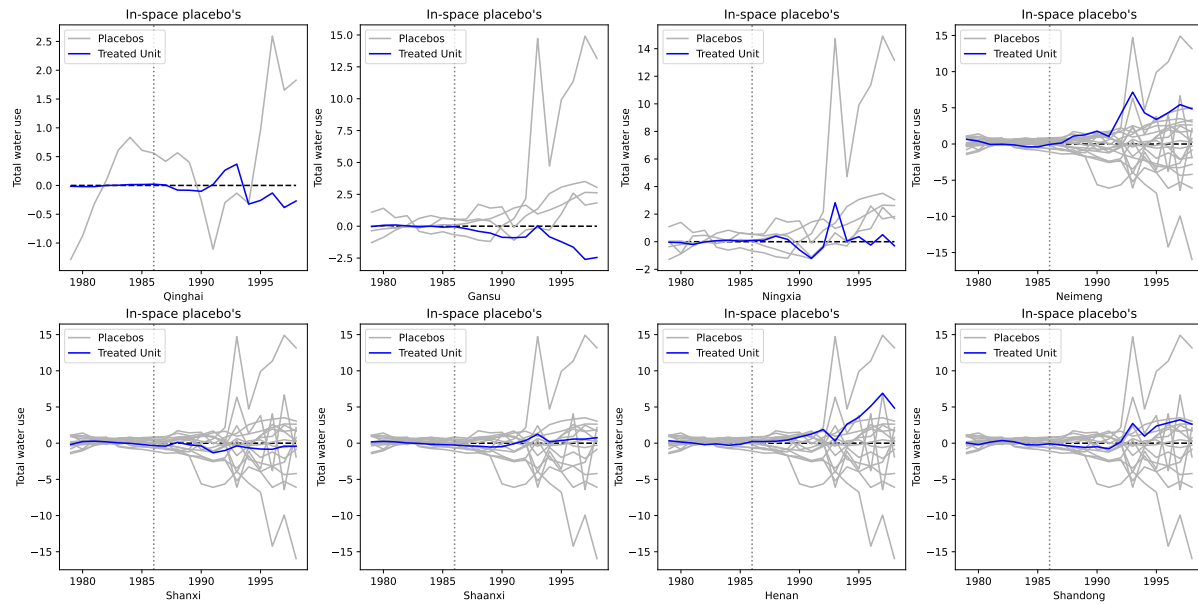
We proposed three intuitive and general assumptions:



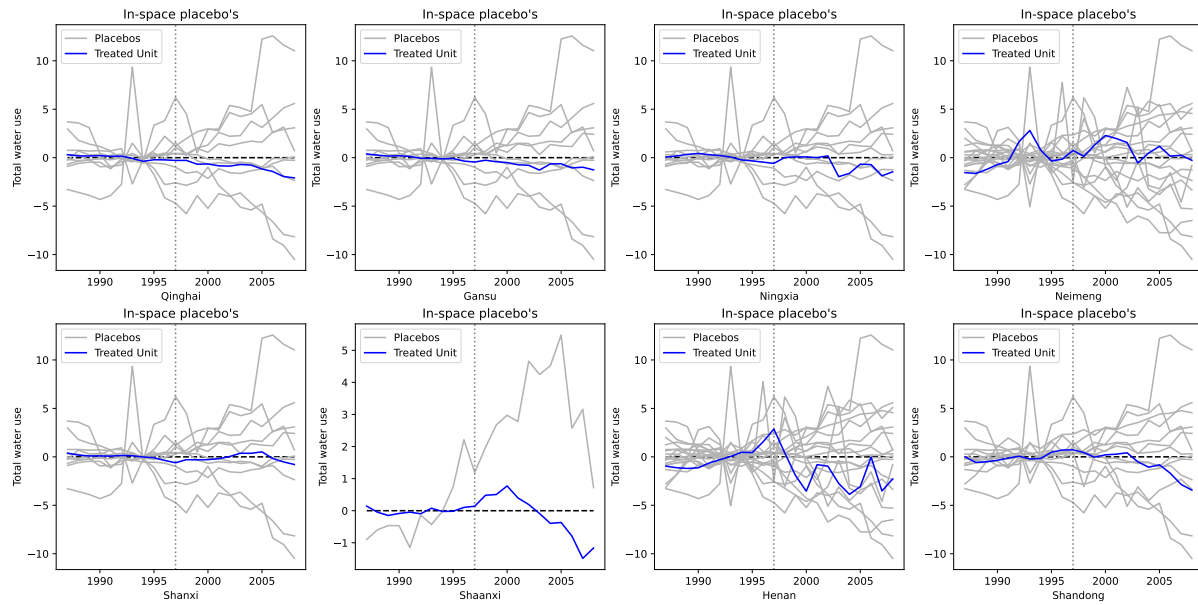
Supplementary Figure B1 Comparisons between YRB' provinces and their synthetic controls around the 87-WAS.



Supplementary Figure B2 Comparisons between YRB' provinces and their synthetic controls around the 98-UBR.



Supplementary Figure B3 Gaps in change in water use between provinces outside the YRB and their synthetic control, around the 87-WAS, excluding the provinces with high pre-treatment RMSPE (more than 3 times of treated units' RMSPE).



Supplementary Figure B4 Gaps in change in water use between provinces outside the YRB and their synthetic control, around the 98-UBR, excluding the provinces with high pre-treatment RMSPE (more than 3 times of treated units' RMSPE)

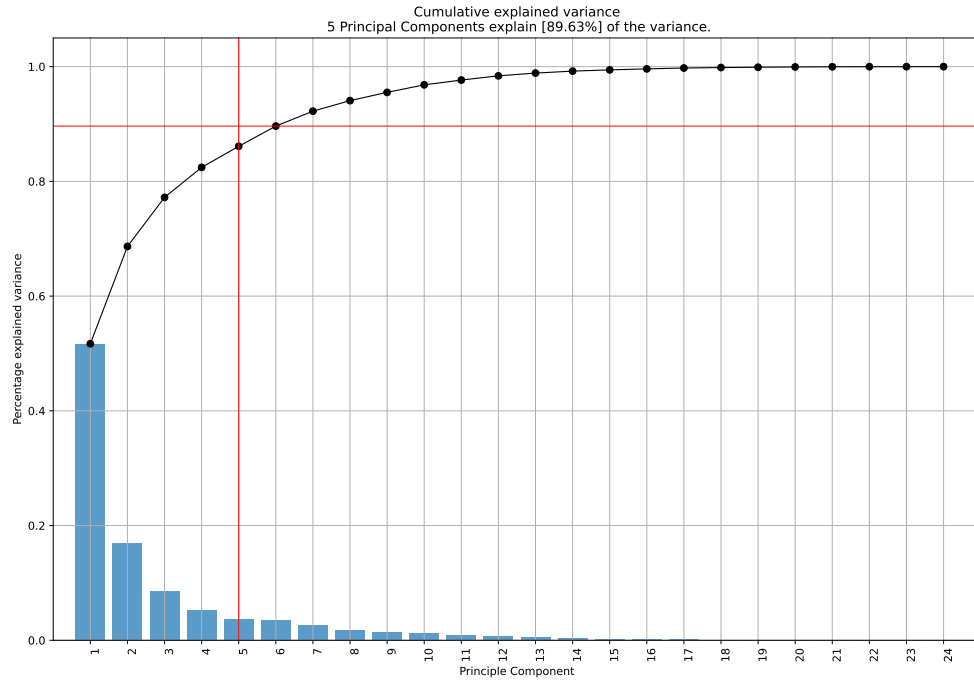
Table B1 Variables and their categories for water use predictions

Sector	Category	Unit	Description	Variables
Agriculture	Irrigation Area	thousand ha	Area equipped for irrigation by different crop:	Rice, Wheat, Maize, Fruits, Others.
				Textile, Papermaking, Petrochemicals, Metallurgy, Mining, Food, Cements, Machinery, Electronics, Thermal electricity, Others.
Industry	Industrial gross value added	Billion Yuan	Industrial GVA by industries	Ratio of industrial water recycling, Ratio of industrial water evaporated.
	Industrial water use efficiency	%	The ratio of recycled water and evaporated water to total industrial water use	
Services	Services gross value added	Billion Yuan	GVA of service activities	Services GVA
Domestic	Urban population	Million Capita	Population living in urban regions.	Urban pop
	Rural population	Million Capita	Population living in rural regions.	Rural pop
	Livestock population	Billion KJ	Livestock commodity calories summed from 7 types of animal.	Livestock
Environment	Temperature	K	Near surface air temperature	Temperature
	Precipitation	mm	Annual accumulated precipitation	Precipitation

Assumption 4. (Water-dependent production) Because of irreplaceability, water is assumed to be the only input of the production function with two types of production efficiency. The production function of a high-incentive province is $A_H F(x)$, and the production function of a low-incentive province is $A_L F(x)$ ($A_H > A_L$). $F(x)$ is continuous, $F'(0) = \infty$, $F'(\infty) = 0$, $F'(x) > 0$, and $F''(x) < 0$. The production output is under perfect competition, with a constant unit price of P .

Assumption 5. (Ecological cost allocation) Under the assumption that the ecology is a single entity for the whole basin involved in N provinces, the cost of water use is equally assigned to each province under any water use. The unit cost of water is a constant C .

Assumption 6. (Multi-period settings) There are infinite periods with a constant discount factor β lying in $(0,1)$. There is no cross-period smoothing in water use.



Supplementary Figure B5 Choose number of principal components by Elbow method, 5 principal components already capture 89.63% explained variance.

Table B2 Pre and post treatment root mean squared prediction error (RMSPE) for YRB's provinces

Provinces	1987-WAS				1998-UBR			
	Pre-RMSPE	Post-RMSPE	Ratio	Significant ^a	Pre-RMSPE	Post-RMSPE	Ratio	Significant ^a
Qinghai	0.016	0.231	14.606	True	0.230	1.170	5.096	True
Gansu	0.056	1.307	23.265	True	0.244	0.841	3.448	True
Ningxia	0.097	0.944	9.697	True	0.332	1.091	3.284	True
Neimeng	0.335	3.846	11.479	True	1.320	1.183	0.896	False
Shanxi	0.208	0.675	3.241	False	0.264	0.401	1.520	False
Shaanxi	0.181	0.572	3.164	False	0.096	0.724	7.579	True
Henan	0.210	3.207	15.292	True	1.222	2.479	2.029	False
Shandong	0.209	1.840	8.785	True	0.431	1.517	3.516	True

[a] Larger post/pre RMSPE than the median of the placebos.

480 Under the above assumptions, we can demonstrate three cases to simulate the water use decision-making and
 481 water use patterns in a whole basin.

482 Under the above assumptions, we can demonstrate three cases consisting of local governments in a whole basin
 483 to simulate their water use decision-making and water use patterns.

484 **Case 1.** *Decentralized decision: This case corresponds to a situation without any high-level water allocation institution.*

485 *When each province independently decides on its water use, the optimal water use x_i^* in province i satisfies:*

486
$$AF'(x) = \frac{C}{P},$$

487 *where A_H and A_L denote high-incentive and low-incentive provinces, respectively.*

488 *When the decisions in different periods are independent, for $t = 0, 1, 2 \dots$, then:*

489
$$x_{it}^* = x_i^*$$

490 **Case 2.** *Mismatched decision: This case corresponds to a mismatched institution.*

491 *The water quota is determined at $t=0$ and imposed in $t=1, 2, \dots$. Under the subjective expectation of each province
 492 that current water use may influence the future water allocation determined by high-level authorities, the total quota
 493 is a constant denoted as Q , and the quota for province i is determined in a proportional form:*

494
$$Q_i = Q \cdot \frac{x_i}{x_i + \sum x_{-i}}.$$

495 *Under a scenario with decentralized decision-making with a water quota, given other provinces' decisions on
 496 water use remain unchanged, the optimal water use of province i at $t=0$ satisfies:*

497
$$AF'(x_{i,0}) = \frac{C}{P \cdot N} - \frac{\beta}{1-\beta} \cdot A \cdot f\left(Q \cdot \frac{x_{i,0}}{x_{i,0} + \sum x_{-i,0}}\right) \cdot Q \cdot \frac{\sum x_{-i,0}}{(x_{i,0} + \sum x_{-i,0})^2},$$

498 *where A_H denotes a high-incentive province and A_L denotes a low-incentive province.*

499 **Case 3.** *Matched institution: This case corresponds to the institution under which water use in a basin is centrally
 500 managed.*

501 *When the N provinces decide on water use as a unified whole (e.g., the central government completely decides
 502 and controls the water use in each province), the optimal water use x_i^* of province i satisfies:*

503
$$F'(x) = \frac{C}{P}.$$

504 We propose Proposition 1 and Proposition 2:

505 Proposition 1: Compared with the decentralized institution, a matched institution with unified management
 506 decreases total water use.

507 The optimal water use under the three cases implies that mismatched institutions cause incentive distortions
 508 and lead to resource overuse.

509 Proposition 2: Water overuse is higher among provinces with high water use incentives than low- water use
 510 incentives under a mismatched institution.

511 The intuition for this proposition is straightforward in that all provinces would use up their allocated quota
 512 under a relatively small Q . As production efficiency increases, the marginal benefits of a unit quota increase, and
 513 the quota would provide higher future benefits for a pre-emptive water use strategy. Provinces with high production
 514 efficiency have higher optimal water use values under the decentralized decision. The divergence in water use would
 515 be exaggerated when the water quota is expected to be implemented with greater competition.

516 Extensions of the model are shown in Supplementary Material S3.

517 Appendix: Water Use Optimization

518 Case 1. Centralized decision

519 When the N provinces decide on water uses as a unity, the marginal cost is C , equal to its fixed unit cost. The
 520 water use of province i aims to maximize $P \cdot A \cdot F(x) - C$. Hence, x_i^* satisfies $P \cdot A \cdot F'(x) = C$, i.e., $AF'(x) = \frac{C}{P}$,
 521 where A denotes A_H for a high-incentive province and A_L for a low-incentive province.

522 Case 2. Decentralized decision

523 When each of the N provinces independently decides on its water use, the marginal cost of water use would be
 524 $\frac{C}{N}$ as a result of cost-sharing with others. Hence, the optimal water use in province i at period t , denoted as \hat{x}_i^* ,
 525 satisfies $P \cdot A \cdot F'(x_{it}) = \frac{C}{N}$, i.e., $A \cdot F'(x) = \frac{C}{P \cdot N}$. Since F' is monotonically decreasing, $\hat{x}_{it}^* > x_i^*$.

526 Case 3. Forward-looking decentralized decision under quota restrictions

527 When the water quota would constrain future water use, the dynamic optimization problem of province i is shown
 528 as follows. In $t = 1, 2, \dots$, there would be no relevant cost when the quota is bound that each province takes ongoing
 529 costs of $\frac{P \cdot Q}{N}$ regardless of the allocation. Therefore, it is sufficient to consider only the total water quota is less than
 530 total water use in Case 2 since a “too large” quota doesn’t make sense for ecological policies.

$$\begin{aligned}
 531 \quad & \max \quad P \cdot A \cdot F(x_{i,0}) - \frac{C \cdot \sum x_{i,0} + x_{-i,0}}{N} + \beta P \cdot A \cdot F(x_{i,1}) + \beta^2 P \cdot A \cdot F(x_{i,2}) + \dots \\
 532 \quad & = P \cdot A \cdot F(x_{i,0}) - C \cdot \frac{x_{i,0} + \sum x_{-i,0}}{N} + \frac{\beta}{1-\beta} P \cdot A \cdot F(Q \cdot \frac{x_{i,0}}{x_{i,0} + \sum x_{-i,0}})
 \end{aligned}$$

533 First-order condition: $P \cdot A \cdot F'(x_{i,0}) - \frac{C}{N} + \frac{\beta}{1-\beta} [P \cdot A \cdot f(Q \cdot \frac{x_{i,0}}{x_{i,0} + \sum x_{-i,0}}) \cdot Q \cdot \frac{\sum x_{-i,0}}{(x_{i,0} + \sum x_{-i,0})^2}] = 0$

534 where $f(\cdot)$ is the differential function of $F(\cdot)$.

535 The optimal water use in province i at $t=0$ $\tilde{x}_{i,0}^*$ satisfies $P \cdot A \cdot F'(x_{i,0}) = \frac{C}{N} - \frac{\beta}{1-\beta} \cdot P \cdot A \cdot f(Q \cdot \frac{x_{i,0}}{x_{i,0} + \sum x_{-i,0}}) \cdot$

536 $Q \cdot \frac{\sum x_{-i,0}}{(x_{i,0} + \sum x_{-i,0})^2}$, i.e., $A \cdot F'(x_{i,0}) = \frac{C}{P \cdot N} - \frac{\beta}{1-\beta} \cdot A \cdot f(Q \cdot \frac{x_{i,0}}{x_{i,0} + \sum x_{-i,0}}) \cdot Q \cdot \frac{\sum x_{-i,0}}{(x_{i,0} + \sum x_{-i,0})^2}$.

537 Since $F' > 0$ and $F'' < 0$, $\tilde{x}_i^* > \hat{x}_i^* > x_i^*$, taken others' water use $x_{-i,0}$ as given. Since the provincial water
538 use decisions are exactly symmetric, total water use would increase when each province has higher incentives for
539 current water use.

540 Proof of Proposition 1:

541 Because $F' > 0$ and $F''(x) < 0$ is monotonically decreasing, based on a comparison of costs and benefits for
542 stakeholders (provinces) in the three cases,

543 $\tilde{x}_i^* > \hat{x}_i^* > x_i^*$.

544 The result of $\hat{x}_i^* > x_i^*$ indicates that individual rationality would deviate from collective rationality under unclear
545 property rights where a water user is fully responsible for the relevant costs. The result of $\hat{x}_i^* > x_i^*$

546 The difference between x_i^* and \hat{x}_i^* stems from two parts: the effect of the marginal returns and the effect of the
547 marginal costs. First, the "shadow value" provides additional marginal returns of water use in $t = 0$, which increases
548 the incentives of water overuse by encouraging bargaining for a larger quota. Second, the future cost of water use
549 would be degraded from $\frac{P}{N}$ to an irrelevant cost.

550 Proof of Proposition 2:

551 Since $A_H > A_L$, $F'(x_H) < F'(x_L)$, Eq.(xxx) implies a positive relation between x_{i0} and A , when β, P, C, Q , and
552 other provinces' water use are taken as given.

553 The difference between \tilde{x}_i^* and \hat{x}_i^* (i.e., $\frac{\beta}{1-\beta} \cdot A \cdot f(Q \cdot \frac{x_{i,0}}{x_{i,0} + \sum x_{-i,0}}) \cdot Q \cdot \frac{\sum x_{-i,0}}{(x_{i,0} + \sum x_{-i,0})^2}$) represents the incentive of
554 water overuse derived from an expectation of water quota allocation. The incentive of water overuse increases by A .

555 D Appendix D: Model extensions

556 Using the marginal benefits analysis (see the Methods section in the main text), we also explored the response of
557 stakeholders to water quota policies. We considered two additional scenarios for stakeholders: technology growth and
558 one that felt different valuations through time (via the discount rate) of economic benefits and ecological costs. In

the following scenarios, the cost is assumed to be untransferable, which could be fully allocated to the one incurring the water use. Explaining plausible scenarios for these stakeholders will help us better understand the causes of water overuse and potential solutions. We argue that water overuse remains robust even if a complete and equitable system.

Case 4. *Forward-looking decentralized decision, taken ecology cost into considerations*

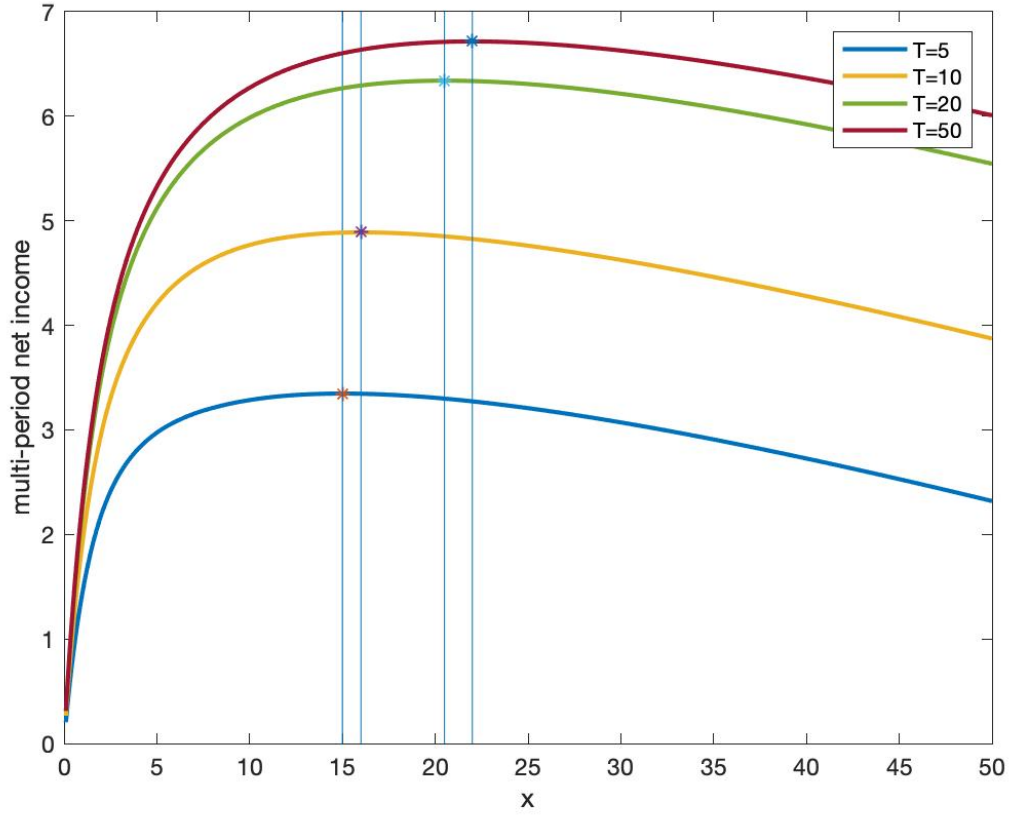
Even if the negative externality of water overuse is eliminated by “fair” ecology cost of $\frac{x_{i,0}}{x_{i,0} + \sum x_{-i,0}} \cdot Q \cdot C$, it is possible that the future growth opportunities and “remote” ecological costs provide enough incentive for the sprint. Water overuse has the value of future economic benefits by slacking the water use constraint in the future. The heterogeneous production efficiency is omitted in this section, and we set $A=1$.

(a) technology growth

Assume that there is an exogenous technology growth rate of g in the scenario of N provinces bargaining for water use under total quota Q , with unit price of output P , unit cost C , and discount factor β . For simplicity, consider a finite-period water use optimization:

$$\begin{aligned} \max \quad & P \cdot (1+g)^t \ln(1+x_{i,0}) - \frac{C}{N} + \beta^t \sum_{t=1}^T [P \cdot (1+g)^t \ln(x_{i,t}+1) - C \cdot x_{i,t}] \\ \text{s.t.} \quad & x_{i,t} \leq Q \cdot \frac{x_{i,0}}{x_{i,0} + \sum x_{-i,0}} \quad \text{for } \forall t \end{aligned}$$

We depict the relationship between multi-period profit and water use $x_{i,0}$ in different horizons in Figure D1, and thus find out the optimal water use pattern under technology growth. The higher marginal water output might create enough incentive to offset the untransferable cost since a higher allocated quota provides growth option value. On the other hand, as the provincial decision is under a longer horizon, there is a more significant sprint effect due to higher accumulated yield and relatively tighter water use constraints over time.



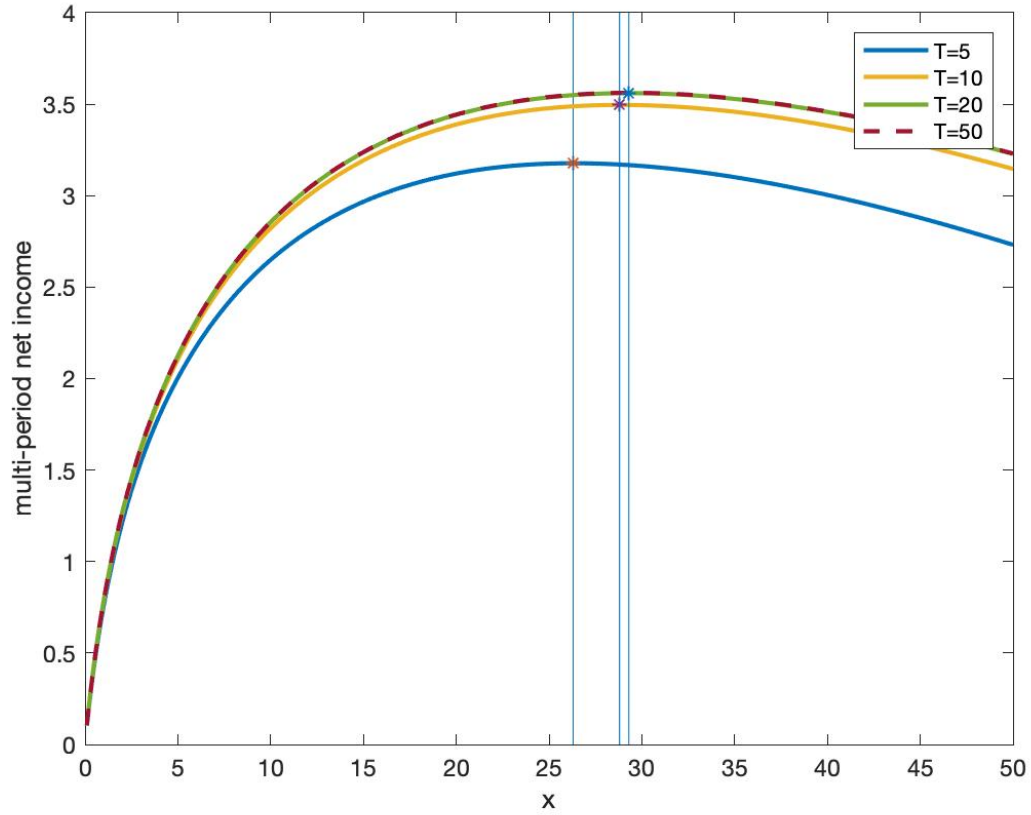
Supplementary Figure D1 Multi-period optimization of optimal water use under technology growth. The figure depicts the relationship of multi-period benefits of province i and water use under Case 3 with technology growth. Assume $F(x) = \ln(1 + x)$, $N = 8$, $P = 1$, $C = 0.5$, $\beta = 0.7$, $g = 0.2$, and $Q = 8$.

579 (b) Economic benefits and “remote” ecological costs with different discount factors

580 Assuming that there is a high discount rate for economic benefits and a low discount rate for ecological costs,
 581 in the scenario of N provinces bargaining for water use under total quota Q , with unit price of output P , unit cost
 582 C , discount factor β^{economy} and β^{ecology} ($\beta^{\text{economy}} > \beta^{\text{ecology}}$). For simplicity, consider the following finite-period
 583 water use optimization, noting the water use of province i at period t :

$$\begin{aligned}
 584 \quad & \max \quad P \cdot \ln(1 + x_{i,0}) - \frac{C}{N} + \beta_1^t \sum_{t=1}^T [P \cdot \ln(x_{i,t} + 1)] - \beta_2^t \sum_{t=1}^T [C \cdot x_{i,t}] \\
 585 \quad & \text{s.t.} \quad x_{i,t} \leq Q \cdot \frac{x_{i,0}}{x_{i,0} + \sum x_{-i,0}} \quad \text{for } \forall t
 \end{aligned}$$

586 We depict the relationship of multi-period net income and water use $x_{i,0}$ in different horizons in Figure D2 ,
587 and thus find out the optimal water use pattern under “remote” ecological costs. The higher discounted ecological
588 costs might create enough incentive to set off the untransferable cost. On the other hand, as the provincial decision
589 is under a longer horizon, a more significant sprint effect is due to a higher accumulated yield.



Supplementary Figure D2 Multi-period optimization of water use under “remote” ecological cost. The figure depicts the relationship of multi-period benefits of province i and water use under Case 3 with “remote” ecological cost. Assume $F(x) = \ln(1 + x)$, $N = 8$, $P = 1$, $C = 0.5$, $\beta_{economy} = 0.7$, $\beta_{ecology} = 0.3$, and $Q = 8$.