# RETOUCH NEXUS

REsilienT water gOvernance Under climate CHange
within the WEFE NEXUS

## Deliverable D3.2

# Case studies dashboard on baseline, model development, economic instruments and business model building, and performance evaluation

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RETOUCH NEXUS

# DELIVERABLE 3.2

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| DEM | Demonstrator, pilot, prototype, plan designs | |
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# ***Executive summary***

This report presents the application of the analytical framework developed in Tasks T3.1 and T3.2 to the project case studies through the implementation of integrated mathematical models grounded in a WEFE (Water–Energy–Food–Ecosystems) perspective. The analysis relies on a portfolio of complementary methods, including hydro-economic modeling, an environmentally extended input–output model, cost–benefit analysis, and a choice experiment, selected according to the specific analytical gaps identified in each case study. These modelling tools are used to estimate user-dependent water values and to design and assess economic instruments that are consistent with the physical, economic, social, institutional, environmental, and governance characteristics of each case study. The report synthesizes the key results and insights derived from the modelling exercises and discusses their implications for policy design, with the aim of supporting more efficient and context-specific water management strategies under a WEFE nexus approach.

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# Abbreviations

| | |
|---|---|
| SSP | Shared Socioeconomic Pathways |
| RCP | Representative Concentration Pathways |
| WEFE | Water, Energy, Food, Ecosystems |
| TETIS | A Catchment Hydrological Distributed Conceptual Model |
| CMIP6 | Coupled Model Intercomparison Project Phase 6 |
| CAPRI | Common Agricultural Policy Regionalized Impact |
| AQUACROP | Crop-Water Productivity Model |
| PRIMES | Price-Induced Market Equilibrium System |
| DWP | Dynamic Water Pricing |
| UWP | Uniform Water Pricing |
| MROC | Marginal Resource Opportunity Costs |
| SW | Surface water |
| GW | Groundwater |
| IO model | Input Output model |
| EEIO | Environmentally Extended Input-Output model |
| AC | Avoided Costs |
| BAU | Business As Usual |
| BASE | Baseline scenario |
| CAPEX | Capital Expenditures |
| CBA | Cost-Benefit Analysis |
| GSVH | Gewestelijke Stedenbouwkundige Verordening Hemelwater |
| INNO | Innovative scenario |
| NPV | Net Present Value |
| OPEX | Operational Expenditures |

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# 1. Introduction

Europe faces increasing water management challenges driven by climate change, population growth, and competing sectoral demands. These pressures affect not only the availability and quality of freshwater resources but also the sustainability of ecosystems and the resilience of socio-economic systems. Addressing these challenges requires approaches that go beyond individual sectors, recognizing the interconnected nature of water, energy, food, and ecosystems. Water is a critical resource whose sustainable management is essential for ensuring the security of the Water-Energy-Food-Ecosystem (WEFE) nexus. Addressing the interdependencies among these sectors requires integrated approaches that combine scientific understanding with practical economic tools. Economic instruments, tailored to the local context, can play a key role in promoting efficient water allocation while considering social, environmental, and governance dimensions.

To translate the WEFE perspective and the implications of economic instruments developed in Deliverable D3.1 into actionable insights, this document employs quantitative modeling approaches applied to representative European case studies. The case studies span diverse hydro-climatic conditions, economic structures, agricultural systems, and governance settings, reflecting the heterogeneity of water-related challenges across Europe. They include river basins characterized by strong surface–groundwater interactions and competing sectoral demands, economies where water use is embedded in complex supply chains, agricultural regions facing increasing drought risk and quality pressures, and island systems with high dependence on energy-intensive non-conventional water sources.

Ensuring efficient and sustainable water allocation is becoming increasingly central to water policy in Europe, as growing scarcity, degradation of water quality, and climate variability intensify competition among users and ecosystems. A consistent nexus-oriented framework is applied to evaluate how alternative management strategies and economic instruments can improve water allocation outcomes under diverse physical, economic, and institutional conditions. By linking system responses to policy interventions, the case studies identify economically efficient and environmentally sustainable instruments, generating insights that support robust, evidence-based decision-making.

This document presents the modelling approaches and results[1] for six European case studies. Additionally, the main results of the case studies are supported by an interactive web-based dashboard developed within the project: https://retouch-nexus.eu/dashboard/. The dashboard provides structured access to key model outputs and scenario results, allowing users to explore and compare outcomes across case studies and policy options. This complementary tool enhances transparency and usability of the modelling results, facilitates stakeholder engagement, and supports informed decision-making by enabling users to interactively assess the implications of alternative water management strategies within a WEFE nexus perspective.

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[1] ***The results presented for each case study are preliminary and subject to ongoing updates as the project progresses.***

# 2. Description of models and scenarios

## 2.1. The Jucar River Basin

### 2.1.1. Model description

The Júcar case study employs an integrated hydro-economic modeling framework developed by Macian-Sorribes (2017), which links biophysical, economic, and ecosystem components within the WEFE Nexus perspective.

The system boundaries of the basin include Jucar's major surface reservoirs, groundwater bodies, irrigated agricultural areas, hydropower facilities, and key ecological sites. It is represented by 27 nodes, 8 surface reservoirs, 5 groundwater bodies (modeled using the Embedded Multireservoir Model, Pulido-Velazquez et al., 2005), 7 subbasins, 18 consumptive demands, 9 hydropower plants, and 6 minimum environmental flows (five based on habitat suitability curves for native fish species and one corresponding to the minimum discharge to the sea) (*Figure 1*).

*Figure 2* illustrates the model's structure, which connects global climate and economic models with basin-scale hydrological and simulation models to assess the impacts of climate change and policy scenarios on water, energy, food, and ecosystems. The modeling chain integrates local hydrological and agronomic projections, derived from the eco-hydrological TETIS model and the AQUACROP model (for the Mancha Oriental area), with large-scale projections from CAPRI (including crop water requirements and production for citrus crops) and PRIMES. These inputs feed into the hydro-economic model STIG-HEM and the fish habitat models available for the Júcar basin, providing quantitative evidence on cross-sectoral trade-offs and synergies within the nexus. The hydroeconomic model is a multi-objective model that accounts for stream-aquifer interactions, improving the operation of the Jucar river system and maximizing both water allocation and the net total benefits. It operates on a monthly timescale for the period 1979-2050 and relies on a simulation approach under deterministic conditions.

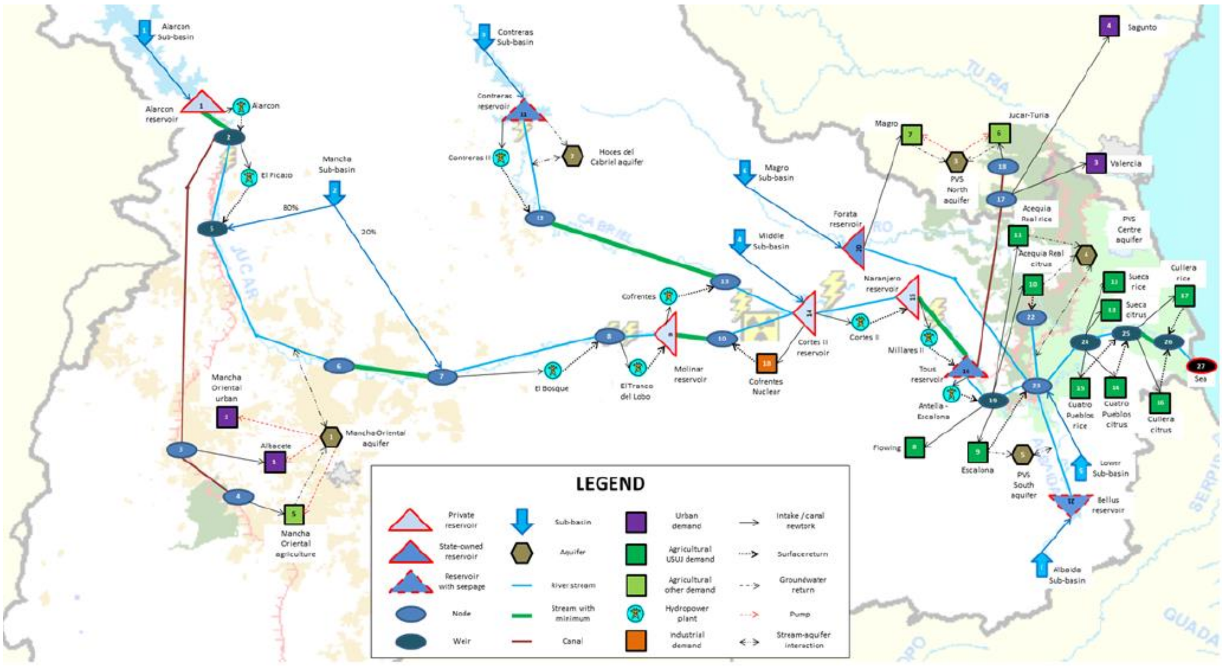

*Figure 1 Hydroeconomic model schematic for the Jucar River*

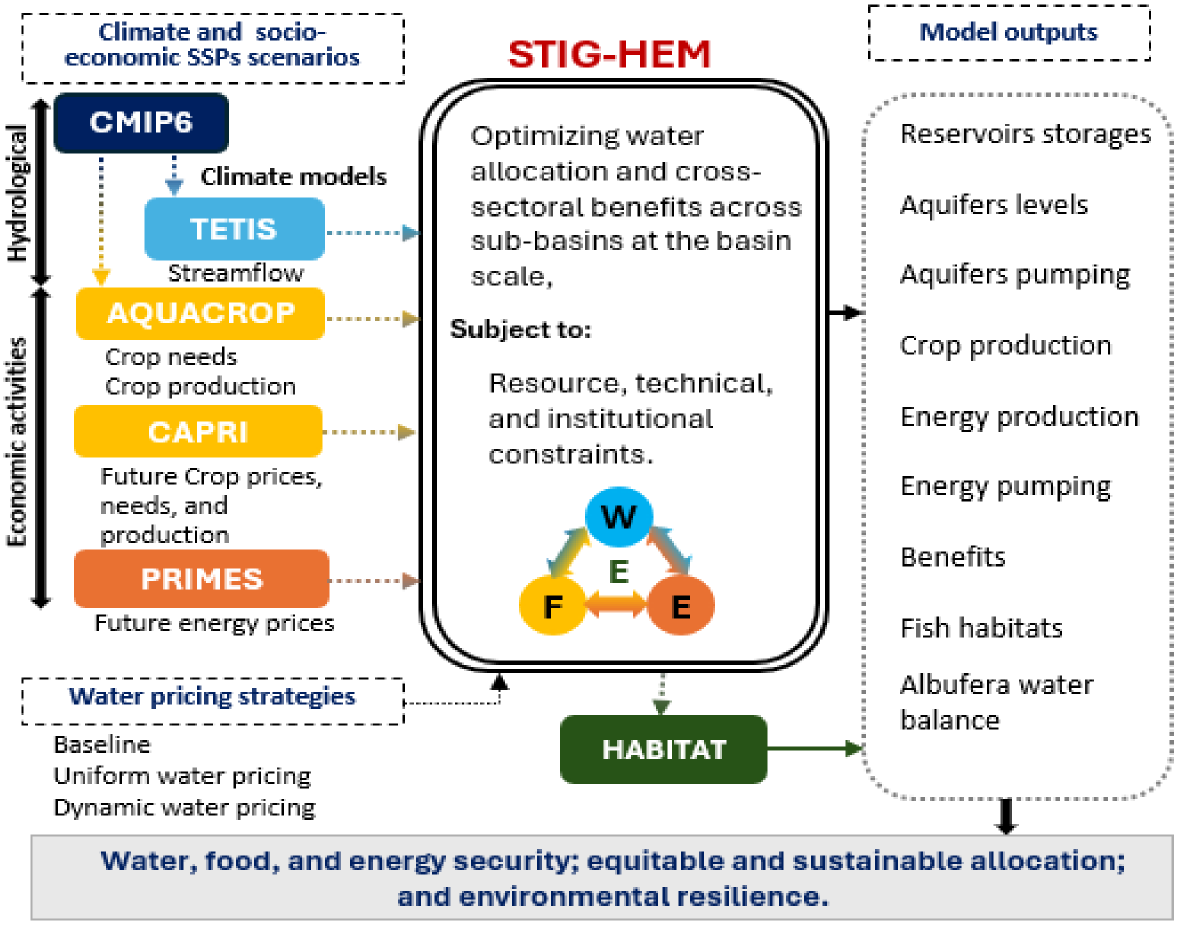

*Figure 2 Modelling chain used for generating the Jucar WEFE evidence*

### 2.1.2. Observation data and climate projections

To perform the robustness analysis for the Júcar basin case study, projected climate data were required as the basis for subsequent analyses. The climate change scenarios analyzed have been obtained from the Coupled Model Intercomparison Project (CMIP6), which is the most established information source for climate change projections in response to the scenarios developed by the IPCC. *Figure 3*a, from O'Neill et al. (2016), illustrates the link between RCPs and SSPs established by CMIP6, and how CMIP5 and CMIP6 scenarios can be compared.

CMIP6 scenarios encompass two activities: Scenario MIP (standard resolution, ranging from ~100 to 250km) and ISIMIP3b (Inter-Sectoral Impact Model Intercomparison Project, high-resolution data, ~50km). From the combinations between RCPs and SSPs, we have selected the ones referring to RCP 2.6 (related to achieving the goal of not surpassing 2 degrees of global warming as indicated in the Paris Agreement), RCP 7.0 (business as usual scenario considering the ongoing energy transition), and RCP 8.5 (worst case scenario).

Climate projections encompass five CMIP6 models: GFDL-ESM4, IPSL-CM6A-LR, MPI-ESM1-2-HR, MRI-ESM2-0, and UKESM1-0-LL, for both historical and future scenarios within the ISIMIP framework. The different models represent a combination of the warmest and freshest, as well as the driest and humid (*Figure 3*b). These five models have been selected as primary models in ISIMIP due to their data availability, process representation, structural independence, climate sensitivity, and performance in the historical period. The raw climate model outputs were subjected to a bias correction and downscaling procedure using the quantile mapping method, with ERA5 reanalysis data (Hersbach et al., 2020) serving as local reference observations.

For all scenarios, the historical period will be 1979-2014, and future assessments will be based on 2050 (deadline foreseen by the EU Green Deal).

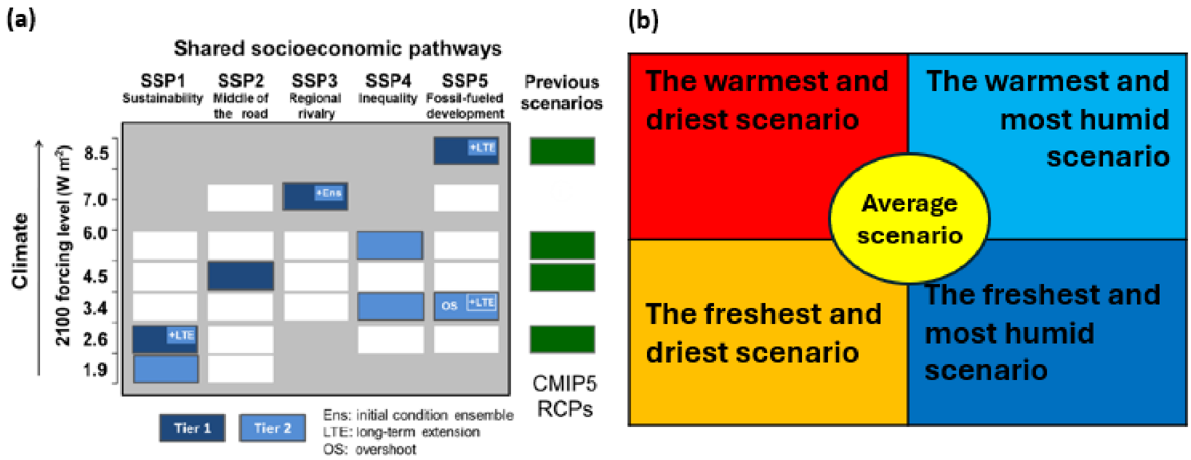

*Figure 3 Projections from Climate Models across RCP–SSP Pathways*

(a) Linkages between RCPs and SSPs implemented by CMIP6 (O'Neill et al. (2016). (b) The selected five climate models from the ISIMIP project.

### 2.1.3. Water pricing strategies

Water pricing strategies were developed and evaluated using the hydro-economic model to explore how alternative price signals can influence water use, economic performance, and resource sustainability across the system. In this framework, pricing instruments were grouped into two main categories: uniform water pricing and Dynamic water pricing, and systematically compared against a baseline scenario that reflects current allocation rules and tariff structures.

The uniform pricing approach tests the implications of applying a constant volumetric tariff across users and time, thereby assessing the effect of simplified and transparent pricing signals. In contrast, the dynamic pricing strategy introduces temporal and hydrological variability into the tariff structure, aligning prices with water availability, scarcity conditions, and marginal values generated within the system. These strategies enable us to quantify trade-offs among economic efficiency, distributional impacts, and resource conservation, providing insights into the potential of pricing reforms to support more resilient water management.

✓ **Uniform water pricing**

In this project, fixed volumetric water pricing is allocated separately for surface water (SW) and groundwater (GW), without differentiation by season, scarcity level, or spatial location, based on the information provided by the Jucar River Basin Water Authority (CHJ, 2022d).

The uniform tariff levels were defined as follows:

Surface water (SW): 0.10 €/m³ for all users.

Groundwater (GW): Urban sector: 0.18 €/m³

Agricultural sector: 0.27 €/m³

These tariff values were incorporated into the hydro-economic model as fixed price parameters applied to all water withdrawals from each source. By using a constant volumetric fee, the strategy aims to isolate the behavioral and economic effects of implementing a simplified pricing rule, allowing comparison with both the baseline scenario and the dynamic pricing alternative.

- ✓ **Dynamic water pricing**

For dynamic water pricing, we applied the methodology proposed by Pulido-Velazquez et al. (2006), using marginal resource opportunity cost (MROC) at a specific location and time to refer to the system-wide cost or forgone net benefit of having one additional unit of the resource available at that location and time. This shadow value varies dynamically in time and space, representing the marginal economic value of natural (raw) water at the source, considering the intersectoral competition for water allocation in space and over time. Its assessment requires simultaneously considering the value of water for all alternative water uses in the basin, as well as the system's variable operating costs.

The marginal economic value of water (MROC) is determined by assessing the impact on water use of a small change in streamflow and then applying economic value estimates to the resulting changes in water use (e.g., Brown 1990; Pulido-Velazquez et al. 2006). The methodology requires a calibrated simulation model of basin water management that has been developed, representing all relevant components, including surface and groundwater systems, infrastructure, demands, and existing allocation rules, so that it reproduces current operating conditions. Therefore, each simulated allocation is economically valued using unit benefit and cost functions for deliveries, flows, storage, pumping, and other operations, producing the total economic benefit of the system for a given hydrologic scenario (the base case). Finally, the model is run iteratively under controlled perturbations: a small additional water volume is introduced at a specific node and time, the system reallocates water following the same operating rules, and the resulting change in total economic benefit is computed. The ratio between the benefit change and the perturbation volume provides an estimate of the aggregated MROC, which represents the economic opportunity cost of water scarcity at that location and time and forms the basis for dynamic water pricing.

After estimating MROC values through sequential perturbation of the water system, each MROC estimate is paired with the reservoir storage level prevailing at the corresponding time step, resulting in a dataset that links scarcity values to hydrologic conditions. These paired observations are then grouped into storage classes to characterize how marginal water value increases as storage declines across the full range of simulated conditions. Based on this empirical storage-MROC relationship, a simplified and operational pricing schedule is derived by selecting a limited number of storage thresholds and assigning a representative scarcity price to each interval that reflects the distribution of MROC within that range. Finally, the resulting step-based tariff is validated against model outputs and refined to ensure that it captures the essential scarcity signal while remaining stable, transparent, and feasible for implementation (*Figure 4*).

In this analysis, there are two sets of prices for surface water:

- **Price set 1**: applicable for Mancha Oriental users, based on water values at the Alarcon reservoir, depending on its storage.
- **Price set 2**: applicable for the rest of the users, depending on the joint storage of Alarcon, Contreras, and Tous.

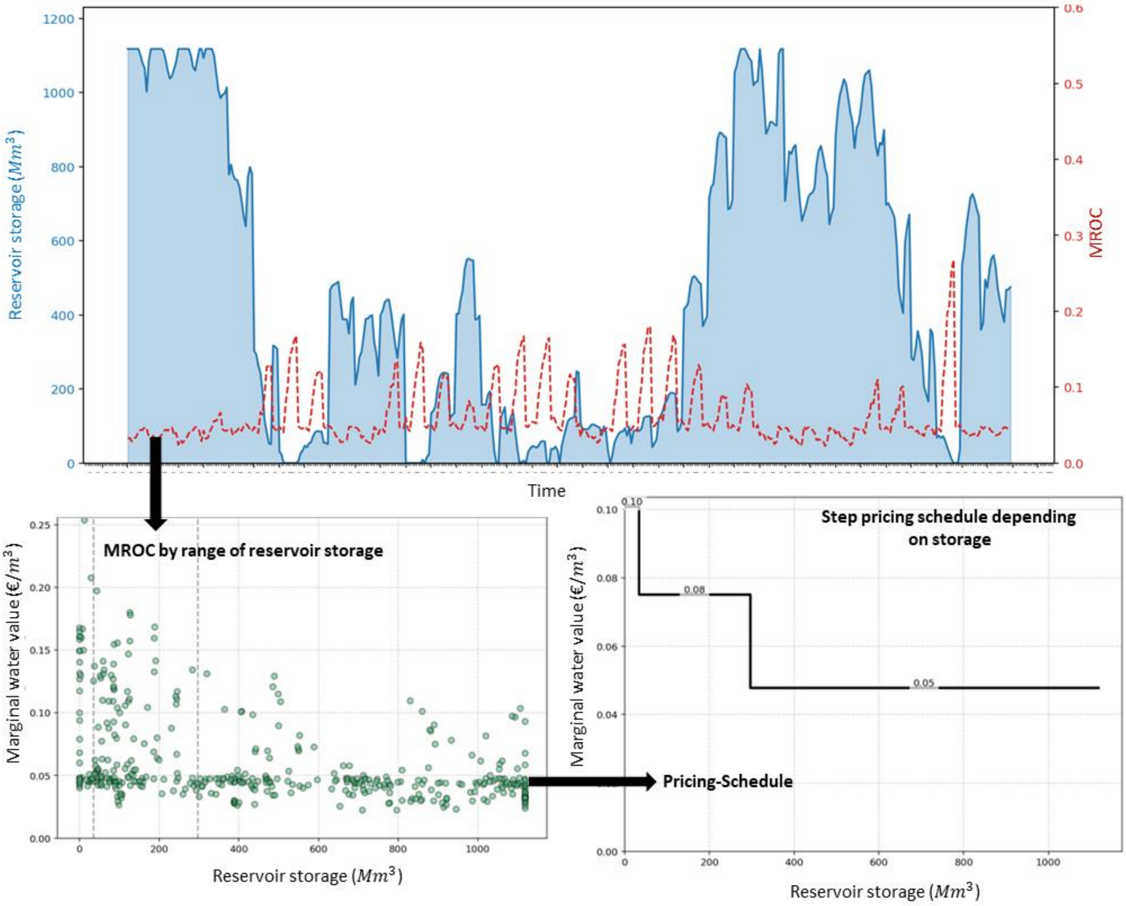

*Figure 4 Procedure for deriving the step pricing schedule from the MROC and storage time series*

So, the pricing policies derived are:

- For downstream users:
  - 0.10 €/m$^3$ if joint storage is equal to or below 300 Mm$^3$
  - 0.06 €/m$^3$ if joint storage is between 300 and 1100 Mm$^3$ (≈ the capacity of Alarcon reservoir)
  - 0.04 if joint storage is above 1100 Mm$^3$
- For Mancha users:
  - 0.10 €/m$^3$ if Alarcon storage is equal to or below 35 Mm$^3$
  - 0.08 €/m$^3$ if Alarcon storage is between 35 and 295 Mm$^3$
  - 0.05 €/m$^3$ if Alarcon storage is above 295 Mm$^3$

The same process as above has been repeated for the pricing policies of groundwater pumping. In this case, the one referring to users pumping from the Mancha Oriental aquifer refers to the storage in Alarcon. The combination of surface and groundwater pricing policies promotes conjunctive use in aquifers, as the decreasing block tariff for surface water and the increasing block tariff for groundwater imply that pumping is discouraged when surface water is abundant and, conversely, is promoted during periods of surface water scarcity.

The resulting pricing policies are shown in *Figure 5* and can be summarized as:

  - 0.019 €/m$^3$ if Alarcon storage is equal to or below 180 Mm$^3$
  - 0.031 €/m$^3$ if Alarcon storage is between 180 and 400 Mm$^3$
  - 0.047 €/m$^3$ if Alarcon storage is between 400 and 836 Mm$^3$
  - 0.085 €/m$^3$ if Alarcon storage is above 836 Mm$^3$

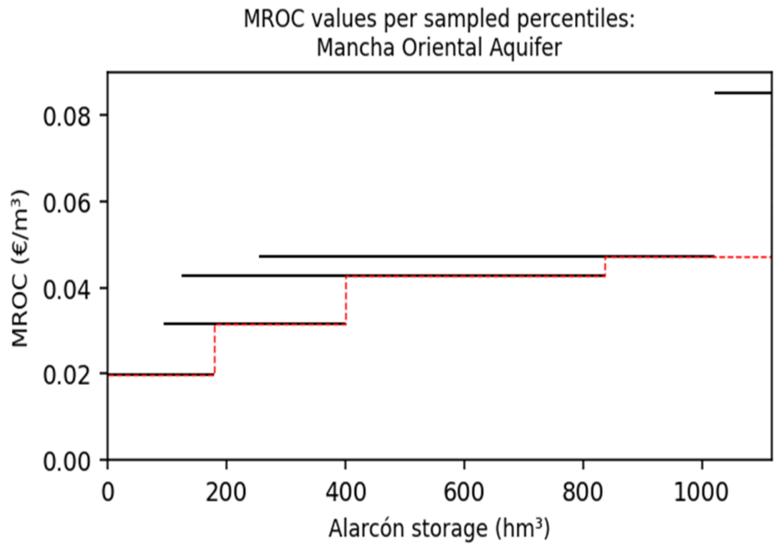

*Figure 5 Groundwater proposed pricing policy*

### 2.1.4. WEFE performance indicators

To estimate the WEFE index and assess the synergies and trade-offs among the different components of the WEFE Nexus, a set of performance indicators was employed to quantify and capture the impacts of water pricing strategies on the agricultural, hydropower, and ecosystem sectors. The modelling chain simulates key hydrological and operational processes, including river discharges, streamflows, reservoir storage dynamics, release decisions, groundwater storage and pumping rates, stream–aquifer exchanges, water deliveries, crop production, energy consumption and revenues in consumptive uses, hydropower generation and associated benefits, inflows to the Albufera lake, as well as habitat availability for selected fish species and stream reaches.

Beyond these variables, the modelling chain enables the computation of WEFE indicators derived from the RETOUCH framework (Deliverable D1.4). For each WEFE component, indicators are classified into two dimensions: “Availability” and “Access”, which together capture both the physical and socio-economic performance of the system Table 1.

*Availability* refers to the physical presence or supply of a resource, including water volumes, energy generation, food production, and ecosystem services. It reflects the biophysical capacity of the system and its infrastructure to sustain production and ecological functions. High availability indicates that sufficient resources are available to meet human and environmental demands, whereas low availability can signal exposure to climate stress, overexploitation, or infrastructure limitations. *Access*, in contrast, captures the ability of users and sectors to benefit from and utilize these resources. It incorporates distributional, operational, and institutional aspects. High access ensures equitable and consistent service delivery, whereas limited access may reflect competition among users, economic barriers, and allocation constraints.

*Table 1 Computed WEFE indicators.*

| WEFE sectors | Indicators |
|---|---|
| **Water** | **Availability**<br>- Reservoir storage as % of capacity.<br>- Groundwater availability: how close current groundwater conditions are to sustainable levels.<br>- Economic efficiency of water use (€/m³).<br>**Access**<br>- % of irrigation demand satisfied.<br>- % of urban demand satisfied.<br>- Number of periods with water shortage (IEE < 0.3). |
| **Energy** | **Availability**<br>- Hydropower generation (GWh).<br>- Energy produced per unit of water (GWh/m³).<br>- Energy productivity (€/m³).<br>**Access**<br>- % of pumped water energy demand satisfied.<br>- Frequency of energy shortages. |
| **Food** | **Availability**<br>- Total production (MT).<br>- Crop water productivity (kg/m³).<br>- Agricultural water productivity (€/m³).<br>**Access**<br>- % of irrigation demand satisfied.<br>- Number of months with stress due to water shortage. |
| **Ecosystem** | **Availability**<br>- Albufera monthly deficits.<br>**Access**<br>- Number of periods below ecological thresholds (%HPU of habitats <0.5). |

Water pricing directly influences both Availability and Access in complex ways. High water prices can restrict access, particularly for vulnerable users. Conversely, low prices can encourage overuse, leading to scarcity that undermines both food security and ecosystem integrity. Carefully designed pricing structures that reflect resource scarcity can promote efficiency, but they must be balanced to ensure economic viability, social equity, and environmental protection. Together, Availability and Access provide a comprehensive understanding of system performance within the WEFE nexus. Their combined assessment enables the identification of synergies, where improvements in one dimension reinforce those in others, and trade-offs, where resource constraints or allocation decisions may reduce the performance of certain sectors.

### 2.1.5. Data used

Table 2 summarizes the main datasets used in the Júcar basin case study, including their sources, spatial and temporal resolutions. The data encompasses hydrological, agronomic, economic, and ecological information required to implement the integrated WEFE Nexus modeling framework.

*Table 2 Data sources for parameterization of the hydroeconomic model*

| PARAMETERS | DESCRIPTION | DATA SOURCES | SPATIAL RESOLUTION | TEMPORAL RESOLUTION |
|---|---|---|---|---|
| **HYDROLOGICAL** | | | | |
| WATER AVAILABILITY | Runoff, river discharge, groundwater recharge, environmental flow, and reservoir evaporation | CHJ (1979-2015) and CIMP6 and TETIS model simulations (Francés, F., 2018) | Asset / basin-level | Monthly (1979-2050) |
| WATER DEMAND | Monthly domestic water demand, Irrigation water demand, irrigation efficiency, cropping pattern, Irrigation needs | CHJ (2022a), (Pulido-Velazquez et al., 2006)<br>AQUACROP and CAPRI model simulation (FAO. (2021; Britz, (2004) | Asset-level<br>Asset-level | Monthly<br>Monthly |
| WATER INFRASTRUCTURE | Reservoir capacities, pumping capacities, hydraulic connection with rivers, diversion and conveyance, reservoir area-capacity function slope, evaporation and infiltration losses | CHJ (2022b) | Reservoir, and network level | Monthly |
| **ECONOMIC, AGRONOMIC, AND ECOLOGICAL** | | | | |
| AGRICULTURE | Input costs, and irrigated areas<br>Crop yields, crop prices | MAPA, (2010–2022)<br>CAPRI model simulation (Britz, 2004) | Municipal<br>Regional | Annual<br>Annual |
| URBAN | Urban demand curve | Pulido-Velazquez et al., (2006) | Municipal | Monthly |
| ENERGY | Electricity prices and pumping energy consumption | Iberdrola and PRIMES model simulation (Capros, Y., 1999) | National level | Monthly |
| ECOSYSTEMS | Habitat suitability, environmental flow requirements | (CHJ, 2022c) | Basin/reach level | Monthly |

## 2.2. The Upper Main River Basin

The Upper Main River Basin (UMRB) in Germany overlaps largely with upper Franconia, and represents a significant agricultural landscape characterized by extensive cereal cultivation and increasing hydrological stress (Figure 6). Here, nearly 200,000 hectares (43.2% of the total) are dedicated to agriculture, of which around 134,000 hectares (28.8%) specifically to cereal production with wheat, barley, and maize being the primary crops (Regierung von Oberfranken, 2024). In this traditionally water abundant region, the interaction between recent climatic trends and human activity has heightened the uncertainty for water availability since 2000, including declining groundwater recharge (Bayerischer Landtag, 2024). Specifically, seasonality causes rain to be abundant in the winter months but scarce during the cereal production season, and increases in temperature drive up the water requirements of the crops (Thaler et al., 2012). Consequently, more water is extracted from groundwater aquifers, rivers and lakes (blue water) to meet the demand. In parallel, the quality of the available water resources has also become a concern due to the run-off and leaching of nitrate compounds from overfertilization. These dynamics have direct implications for agricultural output, water security, and environmental quality, and are forecasted to increase in the future. Therefore, there is a pressing need to assess climate change-related pressures on the agricultural activity in the region from a water economics perspective, which could provide insights to improve water security through cross-sectoral governance.

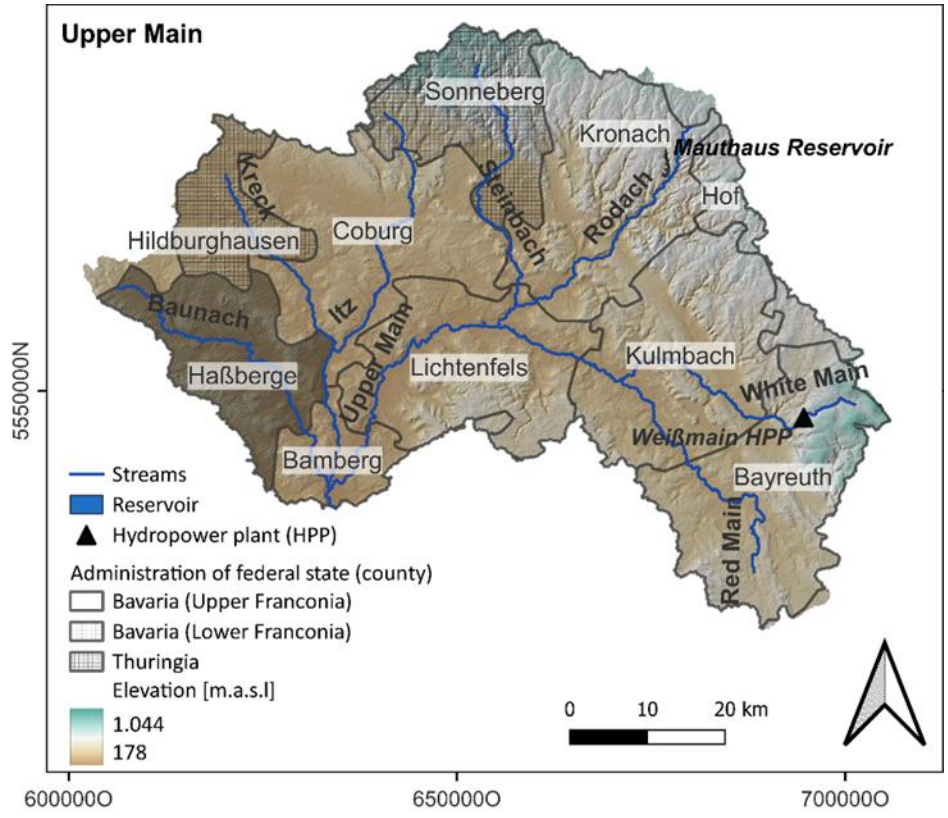

*Figure 6 Map of the Upper Main River Basin (Scherer, 2024).*

### 2.2.1. Model description

SWAT+ is the completely restructured successor to the Soil and Water Assessment Tool (SWAT), developed to address limitations of the original model and to meet emerging challenges in watershed modeling and management. The redesign introduces a more flexible, modular spatial framework that allows users to explicitly connect landscape elements—such as HRUs, landscape units, channels, ponds, aquifers, and reservoirs—and thereby improve the simulation of land–water interactions, flow routing, and management practices. Although the underlying process algorithms remain similar to SWAT, the new architecture enhances model maintainability, supports more realistic watershed configurations, and expands the range of environmental questions that can be assessed, including hydrology, water quality, land management impacts, and conservation planning. (Bieger et. Al., 2017)

In this project, the Chair of Hydrology and River Basin Management at the Technical University of Munich created a SWAT+ model of the UMRB. This serves as the foundational component of the hydrological subsystem, providing quantitative estimates of water availability under present and future climate scenarios. Model outputs feed directly into a system dynamics framework, enabling temporal assessment of groundwater and surface water interactions at the basin scale.

$$P(t) = EFR(t) + PCEvap(t) + SEvap(t) + R_{SW}(t) + R_{GW}(t) + G(t) \qquad (1)$$
$$t \in [2000, 2050]$$

$$Q_i(t) = Q_0 + \int_{t_0}^{t} \Big( R_i(t) - q_{ii}(t) - q_{ij}(t) - q_{ik}(t) \Big)\, dt \qquad (2)$$
$$i \in \{S(River\ and\ Streams\ Surface\ Water), G(Groundwater)\}$$
$$j \in \{H(Household), I(Industrial), AG(Agricultural)\}$$
$$k \in \{OF(Outflow), REvap(Recharge\ Evotranspiration),$$
$$BS(Baseflow\ and\ Seepage), Spr(Pring\ Water)\}$$

$$Q_i(t) \le \bar{Q}_t(t) \qquad (3)$$

$$\sum_i R_i(t) < \sum_i \sum_j q_{ij}(t) + \sum_i \sum_k q_{ik}(t) \qquad (4)$$

$$q_{iAG}(t) = \begin{cases} MCWR(t) - G(t)\ if\ MCWR(t) > G(t) \\ G(t) \qquad\qquad\qquad if\ MCWR(t) \le G(t) \end{cases} \qquad (5)$$

$$WR(t) = \frac{\sum_j q_{GWj}(t)}{R_{GW}(t)} \qquad (6)$$

$$Nitr(t) = \beta_0 + \beta_1 \cdot F(t) + \varepsilon \qquad (7)$$
$$\alpha = \beta_1$$

$$R(t) = NORM(\mu = 0.35, \sigma = 0.065) \qquad (8)$$
$$NO(t) = N(t) \cdot R(t)$$

$$N(t+1) = N(t) + \frac{\alpha \cdot F(t)}{Q_{SW}(t) + Q_{GW}(t)} - NO(t) \qquad (9)$$

$$MCWR(t) = \sum_{i=1}^{n} WR_i(t) \cdot L_i \qquad (10)$$

$$f(t) = a \cdot e^{b \cdot t} \qquad (11)$$

$$CR(t) = Y(t) \cdot Crop\ Price\ (t) \qquad (12)$$
$$SP(t) = \frac{\Delta CR(t)}{\Delta W(t)}$$

$$WQI(t) = 1 - \frac{WQ(t) - WQ_{initial}}{WQ_{max} - WQ_{initial}} \tag{13}$$

After establishing a basic SWAT+ model for the 425,000 ha Upper Main River Basin, an agricultural cropping system was incorporated to simulate crop production and nitrogen dynamics. Within the model, 164,984.3 ha are classified as agricultural land. This area was allocated among the dominant regional crops identified in the Bavarian crop statistics (Bayerisches Landesamt für Statistik, 2021), from which five major crops were selected: winter wheat, silage maize, winter barley, summer barley, and winter rape. Each crop was represented as a single annual crop with a fixed planting date (Table 3). The remaining agricultural area from crops not explicitly represented in the model was proportionally redistributed among the selected major crops to maintain the total cultivated area.

Information on crop management and fertilization practices was obtained from AELF Bayreuth–Münchberg (2024), ensuring that the model reflects region-specific agricultural conditions. Harvest dates were also fixed to a single day in cases where the crop's plant heat units (phu_plant) did not reach the required maturity threshold, thereby guaranteeing harvest operations under cooler or shortened growing seasons. Phosphorus stress was disabled in the model to focus the analysis on nitrogen processes. Nitrogen fertilization was simulated using a predefined schedule in which fixed amounts of elemental nitrogen were applied on specific dates throughout the growing season ( Table 4).

*Table 3 Crop Area, planting and harvesting schedule implemented in the Upper Main SWAT+ Model*

| Crop | Area [ha] | Share [%] | Planting Day [Day of Year] | Harvest Date [Day of Year] | Earlier Harvest if |
|---|---|---|---|---|---|
| Winter Wheat | 38,373.4 | 23.3 | 270 | 210 | phu_plant > 1.00 |
| Silage Maize | 32,812.9 | 19.9 | 105 | 280 | phu_plant > 1.05 |
| Winter Barley | 32,412.9 | 19.6 | 250 | 200 | phu_plant > 1.00 |
| Summer Barley | 36,330.5 | 22.0 | 74 | 280 | phu_plant > 1.00 |
| Winter Rape | 25,054.6 | 15.2 | 264 | 195 | phu_plant > 1.00 |

*Table 4 Fertilization schedule set in the Upper Main SWAT+ Model*

| Crop | | Fert. 1 | Fert. 2 | Fert. 3 |
|---|---|---|---|---|
| Winter Wheat | Day of Year | 60 | 90 | 120 |
| | N (kg/ha) | 70 | 60 | 60 |
| Maize Silage | Day of Year | 105 | 135 | - |
| | N (kg/ha) | 80 | 80 | |
| Winter Barley | Day of Year | 50 | 90 | 150 |
| | N (kg/ha) | 50 | 40 | 40 |
| Summer Barley | Day of Year | 74 | 95 | 115 |
| | N (kg/ha) | 60 | 30 | 20 |
| Winter Rape | Day of Year | 69 | 100 | 121 |
| | N (kg/ha) | 65 | 80 | 50 |

### 2.2.2. Observation data and climate scenarios

#### 2.2.2.1. Calibration of the Upper Main SWAT+ Model

To calibrate and validate the hydrological model, we used the Bayerischer Beobachtungsdatensatz (BayObs), a quality-checked gridded observation dataset provided by the Bayerisches Landesamt für Umwelt (LfU) (LfU, 2020). The Dataset offers daily values of precipitation, as well as minimum, maximum, and mean air temperature, for the entire territory of Bavaria from 1951 to 2019 at a spatial resolution of 5 km (LfU, 2020). BayObs is constructed by combining several high-resolution observation and reanalysis products. Precipitation is derived from the REGNIE dataset of the German Weather Service (DWD), originally at 1 km resolution, and aggregated to the 5 km target grid using area-weighted averaging (LfU, 2020; Rauthe et. Al., 2013). Temperature variables are based on the E-OBS dataset, complemented by HYRAS data in Alpine regions to ensure higher accuracy (LfU, 2020, Cornes et. Al., 2018). Both datasets are interpolated to the BayObs grid using a method based on elevation models and regional regression parameters (LfU, 2020). Due to the availability of the climate data input, the Hargreaves method was used for calculating potential Evapotranspiration.

he hydrological model was calibrated against the monthly mean discharge at the outlet of the Upper Main River Basin (gauge Kemmern). A single-objective calibration was performed, using the Nash-Sutcliffe-Efficiency (NSE) (Nash and Sutcliffe, 1970) as the primary optimization criterion. Figure 7 shows the discharge of the calibrated model compared to the observed discharge at the Kemmern outlet, as well as the goodness of fit criteria NSE, PBIAS, and RMSE for monthly mean discharges between 2000 and 2018. The NSE shows a good fit between simulation and observation, particularly in terms of the timing of the spring high-flow peaks. The model systematically underestimates the summer periods, where there is low discharge, which is a limitation when analyzing baseflow processes using this setup. Generally, the Upper Main SWAT Model performs well in terms of the NSE, with a reasonable model error (RMSE = 16.444 m³/s) and slightly underestimates the overall discharge (PBIAS = -7.87 %).

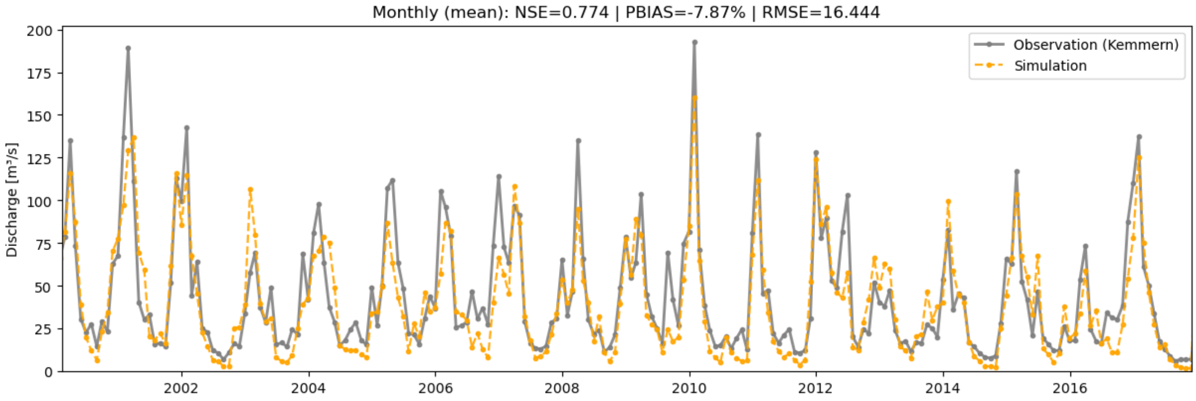

*Figure 7 Comparison of the calibrated SWAT+ Model against the observed discharge at the gauge Kemmern*

### 2.2.2.2. Climate Forcing Data

For the climate change impact assessment, the hydrological simulations were driven by regional climate projections from the Bayern-Ensemble, provided by (LfU, 2020). Specifically, the model uses the ICHEC EC-EARTH r12i1p1 global climate model dynamically downscaled with CLMcom-CCLM4-8-17 at a native resolution of 12.5 km, covering the period 1951–2100 for all three emission scenarios RCP2.6, RCP4.5, and RCP8.5 (LfU, 2020). The LfU applies a consistent statistical downscaling to a finer 5 km × 5 km grid using an elevation-based interpolation method (LfU, 2020; Marke, 2008). It then performs a bias adjustment of all variables using Quantile Delta Mapping, ensuring that the seasonal distributions of temperature and precipitation are aligned with the observed reference climatology (LfU, 2020). The dataset provides daily values for precipitation (pr), mean/min/max temperature (tas, tasmin, tasmax), relative humidity (hurs), global radiation (rsds), and wind speed (sfcWind) (LfU, 2020). For each scenario, a complete climate time series for the Upper Main River Basin was extracted and used to drive the calibrated SWAT model. Although the climate projection dataset provides additional meteorological variables compared to the BayObs observational dataset, the Hargreaves method was retained for calculating potential evapotranspiration to ensure methodological consistency with the calibration period. This setup was used to simulate annual hydrological outputs (runoff, evapotranspiration, baseflow, etc.), annual crop yields, as well as nitrogen cycle components (annual) for the period 2000–2050. While the chosen GCM–RCM combination is part of the plausibility-checked Bayern-Ensemble, relying on a single climate projection inherently limits the ability to represent the full spectrum of climate uncertainty. Consequently, the simulated hydrological response reflects only one possible climate trajectory, whereas alternative model chains could lead to wetter, drier, warmer, or more variable future conditions. The results should therefore be interpreted as scenario-specific outcomes, and the robustness of the conclusions could be strengthened by incorporating multiple climate models in future analyses.

### 2.2.3. Stochastic Frontier Analysis (SFA)

Stochastic Frontier Analysis (SFA) is applied to evaluate technical efficiency in cereal production with respect to key input variables, including water, labor, and fertilizer (Aigner et al., 1997). SFA decomposes deviations from optimal production into inefficiency effects and stochastic noise, thereby enabling estimation of productivity under varying environmental and economic constraints. Within the modelling framework, SFA quantifies the influence of water inputs on cereal output, offering insight into marginal productivity, shadow pricing, and resource optimization. Results highlight that water availability constitutes a principal determinant of agricultural technical efficiency within the basin.

$$y_i = f(x_i;\ \beta) \cdot e^{-u_i} \cdot e^{v_i} \tag{14}$$

$$u_i \sim |N(0, \sigma_u^2)| \tag{15}$$

$$v_i \sim N(0, \sigma_v^2) \tag{16}$$

We can take the logarithm of both sides, to turn the production function into an additive form that is linear in $\ln(x_i)$, further simplifying the model.

$$\ln(y_i) = \ln\big(f(x_i;\ \beta)\big) + v_i - u_i \tag{17}$$

$$\ln(y_i) = \alpha + \beta_1 \cdot \ln(W_i) + \beta_2 \cdot \ln(L_i) + \beta_3 \cdot \ln(K_i) + \beta_4 \cdot \ln(A_i) + \beta_5 \cdot \ln(N_i) + v_i - u_i \tag{18}$$

Empirical estimation of SFA parameters is conducted using maximum likelihood estimation (MLE), and the frontier package in R (Coelli & Henningsen, 2020).

### 2.2.4. System Dynamics (SD)

System dynamics modelling provides a quantitative methodology for representing causal interactions within complex socio-environmental systems (Sterman, 2018). The approach conceptualizes system behaviour as emergent from interconnected stocks, flows, and feedback loops, evolving over time. In this study, the model encapsulates six subsystems: hydrological supply (1), hydrological demand (2), agricultural water demand (3), cereal production (4), water quality (5), and economic output (6) bounded temporally from 2000 to 2050 and spatially by the Upper Main River Basin (Figure 8). Model construction enhances understanding of cross-sectoral dependencies within the WEFE nexus, enabling simulation of long-term outcomes under diverse climatic and management conditions.

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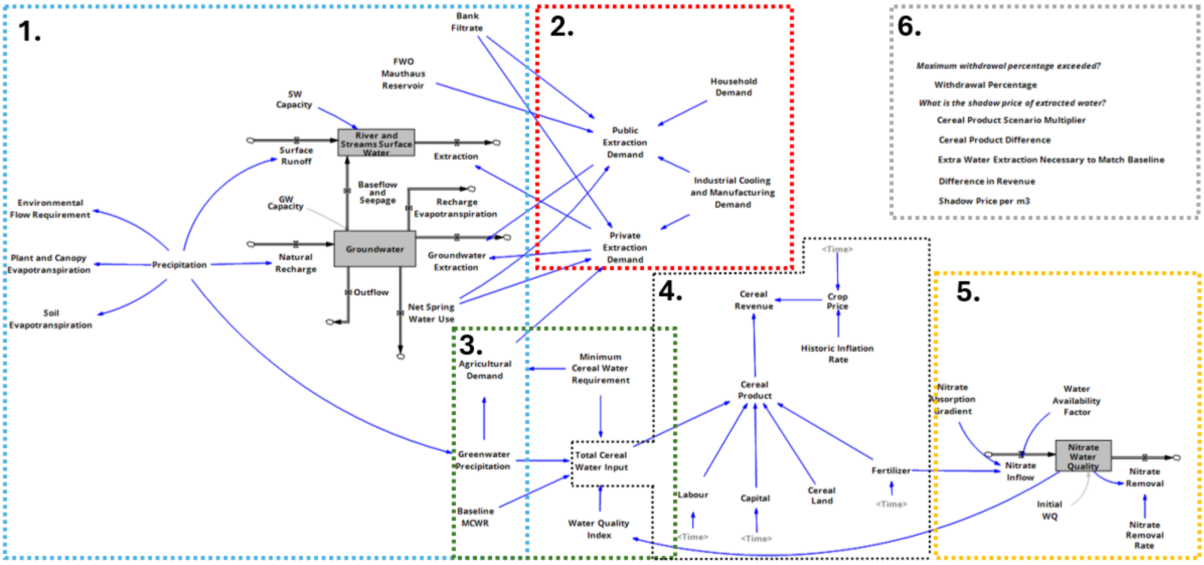

*Figure 8 Overview of the entire System Dynamic model. Source: Twohig (2023).*

### 2.2.5. WEFE Approach

The WEFE Nexus (Hoff, 2011; Carmona-Moreno et al., 2019) provides a valuable lens for analyzing the evolving challenges of the UMRB, where agricultural production, water availability, energy use, and ecosystem health are interlinked. Figure 9 illustrates the interdependencies within the UMRB, highlighting how water, energy, food, and ecosystems interact through multiple biophysical and socioeconomic pathways. The water pillar influences and is influenced by the other domains through processes such as blue and green water availability, water quality, irrigation efficiency, and recharge capacity. Energy interacts with water through cooling demands and fertilizer production, while also contributing to CO2 and other emissions that affect ecosystems. The Food pillar (mainly cereal production) depends heavily on precipitation or irrigation, while also generating nutrient pollution that affects ecosystem health. Finally, the ecosystem provides essential regulatory functions such as maintaining environmental flows and supporting the water cycle. Within our System Dynamic model for the Upper Main River Basin (Figure 8, above), the nexus framework is essential for understanding trade-offs between water availability, agricultural water use, agricultural production, other water demands, and ecosystem health, represented by subsystems 1, 3, 4, 2, and 5, respectively. The interdependence between agricultural extraction, water quality outcomes, and economic returns underscores the need for integrated governance mechanisms capable of balancing sectoral priorities while minimizing externalities.

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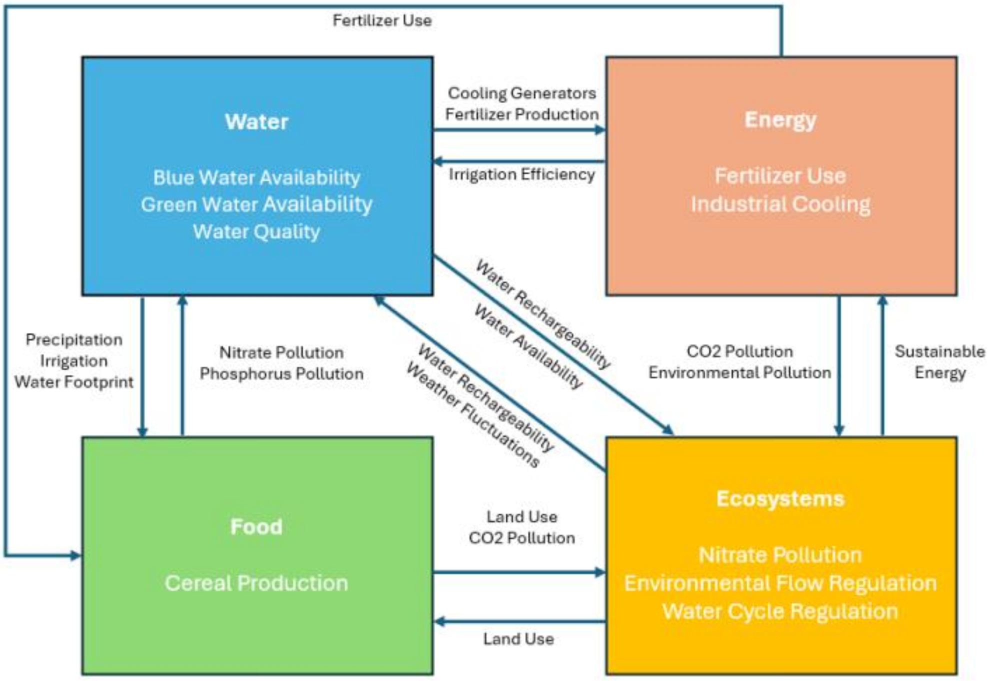

*Figure 9 Interdependencies within the Upper Main River Basin WEFE Nexus. Source: Twohig (2023).*

### 2.2.6. Scenario Analysis

The power of a System Dynamics model is that allows developing alternative scenarios by adjusting key variables throughout its components. We operationalize uncertainty by modelling our system's behavior under multiple climate and management assumptions. The study evaluates combinations of precipitation pathways for the region, derived from RCP 2.6, 4.5, and 8.5 (BayKIS, 2024, Figure 10), and minimum cereal water requirement trajectories (Drastig et al., 2016; Busschaert et al. 2022; FAO, 2024), categorized as low, medium, and high. Fertilizer use is modelled under stable (level recorded in 2000) and decreasing (i.e. following a country-wide trend) application regimes, resulting in nine total scenarios.

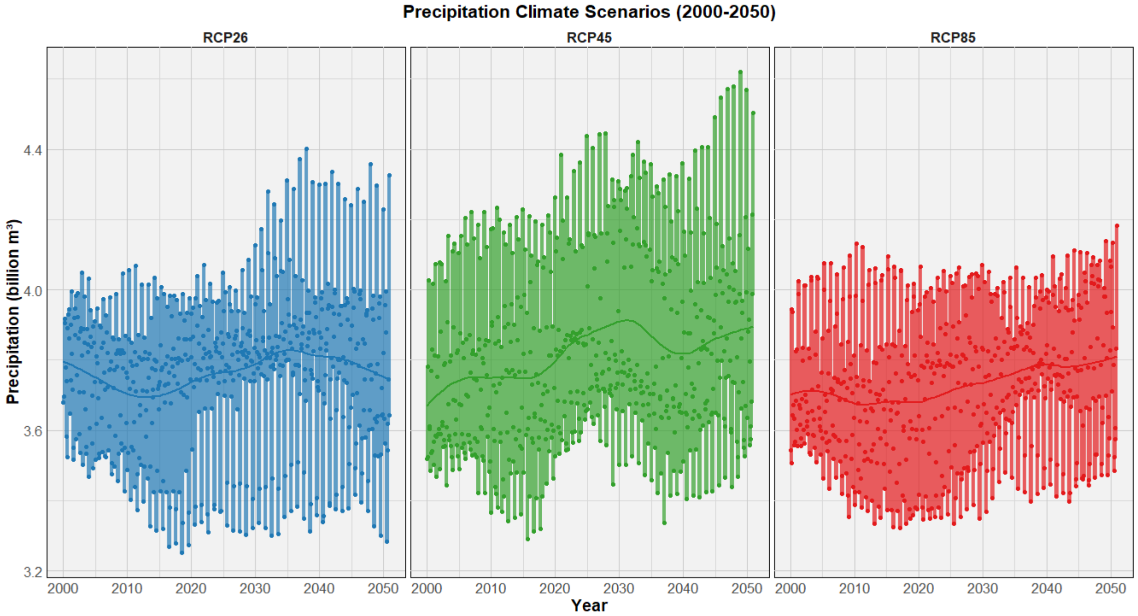

*Figure 10 RCP Precipitation Scenarios, 2000 – 2050 (BayKIS, 2024).*

### 2.2.7. Data Used

Table 5 provides an overview of all data sources use, categorized into hydrological, environmental, and economic domains. The hydrological section draws heavily on outputs from the SWAT model, including precipitation, evapotranspiration components, groundwater capacity, baseflow, seepage, and runoff, all processed using GIS tools to generate basin-specific datasets. Additional hydrological inputs such as surface water capacity, groundwater recharge, environmental flow requirements, and reservoir supply were sourced from Eurostat, regional governmental reports, and scientific literature. These datasets were preprocessed through scaling to the Upper Main River Basin (UMRB), GIS extraction, or direct integration when no further treatment was required.

The environmental and economic sections compile data from international databases, regional authorities, and scientific studies. Environmental variables include minimum cereal water requirements, water quality indicators, and nitrate removal rates, which were either incorporated directly or transformed using formulas and GIS procedures. The economic data encompass labor, cereal land, fertilizer, capital inputs, and crop prices, primarily sourced from FADN, Eurostat, and German federal statistical offices. These variables were scaled to the UMRB where necessary, ensuring consistency across spatial units. Together, the table presents a comprehensive and structured dataset supporting the integrated WEFE Nexus and system dynamics analysis.

*Table 5 Overview of data and sources.*

| ***Category*** | **Source** | **Variables** | **Preprocessing** |
|---|---|---|---|
| *Hydrological* | SWAT Model (Scherer & Holzer, 2024) (BayKIS, 2024) | Precipitation | GIS to Output Data |
| | | Plant & Canopy Evapotranspiration | GIS to Output Data |
| | | Soil Evapotranspiration | GIS to Output Data |
| | | Recharge Evapotranspiration | GIS to Output Data |
| | | Baseflow | GIS to Output Data |
| | | Seepage | GIS to Output Data |
| | | Groundwater Capacity | GIS to Output Data |
| | | Surface Water Runoff | GIS to Output Data |
| | Eurostat (2024a) | Surface Water Capacity | Scaling to UMRB |
| | Bayerischer Landtag (2024) | Groundwater Natural Recharge | - |
| | Liu et al. (2021) | Environmental Flow Requirement | GIS Masking & Extraction |
| | IWMI (2024) | Environmental Flow Requirement | Use of EFR Calculator |
| | FWO Oberfranken (2024) | FWO Mauthaus Reservoir Supply | - |
| | Lüers (2012) | Household & Industrial Water Demand | - |
| | | Bank Filtrate Supply | - |
| | | Spring Water Supply | - |
| *Environmental* | FAO (2024) | Minimum Cereal Water Requirement | See Formula 13 |
| | GEMStat (2024) | Water Quality | GIS to Panel Data |

| | | | |
|---|---|---|---|
| | Yan et al. (2024) | Nitrate Removal Rate | - |
| *Economic* | FADN (2024) | Labour | Scaling to UMRB |
| | | Cereal Land | Scaling to UMRB |
| | | Annual Capital Input | Scaling to UMRB |
| | | Total Assets | Scaling to UMRB |
| | Regierung von Oberfranken (2024) | Cereal Land | - |
| | Eurostat (2024a) | Fertilizer | Scaling to UMRB |
| | | Cereal Allocation | Scaling to UMRB |
| | Eurostat (2024b) | Crop Prices | - |
| | DEStatis (2024) | Cereal Product | Scaling to UMRB |
| | | Cereal Land Allocation | Scaling to UMRB |

## 2.3. Malta

All the islands in the Maltese archipelago are classified as a single river basin. With the high population density of 1670 km$^2$ and the semi-arid climate of the islands imply that Malta has the lowest per capita availability of freshwater in the EU. As a result, Malta is increasingly dependent on non-conventional water sources (i.e. recycled and desalinated water) to meet its water demand, including that of its agricultural sector. The processes involved are energy intensive and Malta's local production of renewable energy is limited to PV technology, the deployment of which is limited by space considerations and cost of land. Therefore, in this document we present the modelling approach and results for the Maltese case study.

### 2.3.1. Model description

The Maltese case study employs a dynamic sustainability model to describe the relationships between economic output and capital, and the use of energy and water resources from sustainable sources and through unsustainable extraction. The present model aims to determine sustainability of an economy based on inter-temporal dynamics of three state variables, namely:

I. Stock of energy resources;

II. Stock of water resources, and;

III. Stock of economic capital.

Key to the evolution of these three variables is a production function of economic output, which is itself specified in terms of use of energy and water resources from sustainable and unsustainable extraction and economic capital.

The model is developed in terms of dynamic and static behavioral equations which govern the development of the stock variables. It is solved on the basis of assumed but reasonable coefficients in order to assess plausibility of its behaviour and understand its characteristics. The model is excel based but it is extended through the use of Monte Carlo simulations to understand the effects of shocks to the model variables and coefficients on the dynamic results. The conceptual model is subsequently contextualized to the case of Malta, utilising suitable data including approximations and estimates where necessary, which will enable EWA to draw insight for policy design and give direction on future development of the model.

Conceptually, the stocks of energy, water and economic capital may be understood to be quantities of productive factors which can be depleted over time in to provide economic output and capital resources which would enable consumption. Conversely, the economy can invest part of its output back into these factors to enable future sustainability. These relationships are captured in Figure 11.

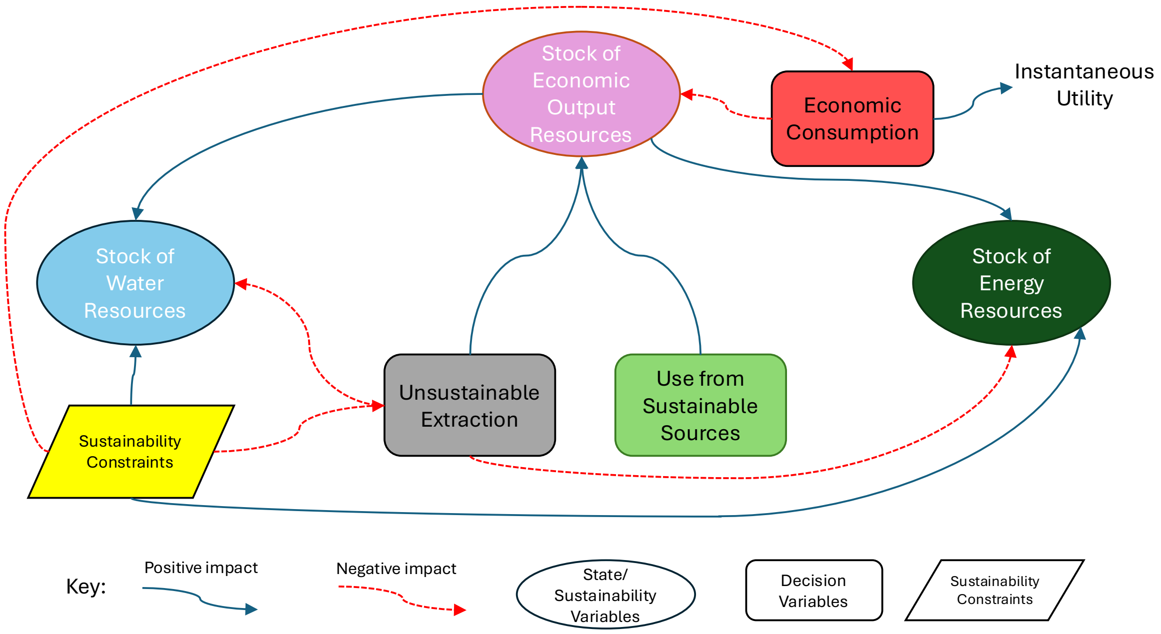

*Figure 11 Conceptual Flow Diagram*

Economic output and capital resources are derived through the use of energy and water resources obtained from unsustainable extraction, or alternatively, from sustainable sources. The latter has a positive impact on stock of water and energy resources. Sustainability constraints on extraction of these resources would limit unsustainable extraction hence positively impact these stocks. However, those sustainability constraints hinder economic consumption, thus also hinder instantaneous utility.

The model captures the following important behavioural elements:

- unsustainable resource extraction of both resource elements reduces the stocks of both elements over time, to possibly different extents depending on the relative coefficients;
- sustainable resource use can be modelled to detract to a lesser extent from the stock of resources, or to no extent at all if the relative coefficients are set to zero;
- resource use from both sustainable sources and unsustainable extraction contributes to the generation of economic output, but sustainable resource use involves higher costs and consequently, lower productivity thus use of resources from sustainable sources thus preserves the stock of resources, but at an economic cost;
- the stocks of resources are governed by exogenous targets with respect to the ratio of sustainable use to unsustainable extraction to be attained by the end of the model simulation period, as well as by the growth rates of the total use of each of water and energy resources, both of which serve as important policy inputs into the model.
- The lack of sustainability of a given scenario can be determined through a situation of continuous depletion of energy, water and economic capital.

The model uses a set of equations for modelling. Hereby, each equation is explained:

$$y_t = \left( \left( \frac{e_t^s}{R_1} + e_t^x \right)^{R_2} \times \left( \frac{w_t^s}{R_3} + w_t^x \right)^{R_4} \times K_{t-1}^k \right)^{\gamma} \tag{19}$$

$$k = 1 - R_2 - R_3 \tag{20}$$

Equation 1 above is used to model the production function. It is a Cobb-Douglas style production function where economic output depends on the use of energy and water flows, and economic capital. It is to be noted that the sustainable use of energy and water flows are adjusted by factors which denote their relative costs. The parameter k, which is the output elasticity of economic capital K, is set in equation (2) as one less the sum of the other elasticities for resources used, in line with the requirements for a production function with constant returns to scale.

$$E_t = E_{t-1} + e_E^s e_t^s + e_E^x e_t^x + w_E^s w_t^s + w_E^x w_t^x + y_E y_t \tag{21}$$

$$W_t = W_{t-1} + e_W^s e_t^s + e_W^x e_t^x + w_W^s w_t^s + w_W^x w_t^x + y_W y_t \tag{22}$$

Equation 3 and 4 are utilized to model the stocks of energy and water respectively. Equation (3) is a dynamic equation for the development of the energy stock. Its change from the previous period depends on sustainable and extractive use of energy and water resources multiplied by relative coefficients, and by a portion of output that is invested back into developing the energy stock. A similar approach applies to the development of the water stock, as indicated in equation (4).

$$c_t = c_I \times K_{t-1} + c_a \tag{23}$$

Equation (5) models consumption as autonomous consumption multiplied by economic capital, adding induced consumption.

$$K_t = K_{t-1} \times (1 - \delta) + y_t \times (1 - y_E y_W) - c_t \tag{24}$$

Equation (6) represents the development of economic capital. It consists of the economic capital of the previous time period after economic capital depreciation added to the economic output that is not dedicated to resupplying energy and water stocks, adjusted for consumption.

$$e_t^s = (e_T^s - e_0^s) \times \frac{1}{T} + e_{t-1}^s \tag{25}$$

$$w_t^s = (w_T^s - w_0^s) \times \frac{1}{T} + e_{t-1}^s \tag{26}$$

$$e_t^x = (e_T^x - e_0^x) \times \frac{1}{T} + e_{t-1}^x \tag{27}$$

$$w_t^x = (w_T^x - w_0^x) \times \frac{1}{T} + e_{t-1}^x \tag{28}$$

$$e_T^s = (e_0^s + e_0^x) \times (1 + g_e)^T \times S_1 \tag{29}$$

$$w_T^s = (w_0^s + w_0^x) \times (1 + g_w)^T \times S_2 \tag{30}$$

$$e_T^x = (e_0^s + e_0^x) \times (1 + g_e)^T \times (1 - S_1) \tag{31}$$

$$w_T^x = (w_0^s + w_0^x) \times (1 + g_w)^T \times (1 - S_2) \tag{32}$$

Equation (7 & 9) explains the unsustainable and unsustainable flow of energy, respectively while equations (8 & 10) represent the sustainable and unsustainable flow of water, respectively. Equation 7 is a dynamic equation for the development of the use of sustainably sourced energy. It consists of the difference between the initial and final target values of $e_t^s$ scaled down by $\frac{1}{T}$ added to the previous time period's use of sustainably sourced energy. In turn, the target final value of sustainably sourced energy is given in equation (11). This shows the final value of the variable at time T to be determined as an exogenously set ratio $S_1$ of the total final energy use, determined as the initial total energy use growing by an exogenously set growth rate $g_e$ over the time period T. Therefore, both $S_1$ and $g_e$ are critical parameters influenced by energy policy. Similar considerations apply for the other elements of resource use in equations (8) to (10) with their respective final values set in equations (12) to (14).

### 2.3.2. Observation data and scenarios

The model described above is applied to a stylised situation where the data inputs are calibrated to an extent possible to reflect the situation of the Maltese economy.

Table 6 shows the model inputs derived from a data calibration approach which are required to induce the model to approximate, to the extent possible, a situation which depicts the current and future state of the Maltese economy.

*Table 6 Calibrated Model Coefficient Values for Maltese Context*

| ***Model Coefficient*** | ***Hypothetical Value*** | ***Interpretation*** |
|---|---|---|
| $R_1$ | 0.075 | A 1% increase in the use of energy resources increase economic output by 0.075%.[2] |
| $R_2$ | 1.5 | Energy derived from sustainable sources costs 1.5 times that extracted from unsustainable ones. (H) |
| $R_3$ | 0.05 | A 1% increase in the use of energy resources increase economic output by 0.05% (H). |
| $R_4$ | 1.25 | Energy derived from sustainable sources costs 1.25 times that extracted from unsustainable ones (H). |
| $e_E^s$ | 0 | 1 unit of energy derived from sustainable sources does not detract from the stock of energy resources (H). |
| $e_E^x$ | $-1$ | 1 unit of energy derived from unsustainable extraction detracts 1 unit from the stock of energy resources |
| $w_E^s$ | 0 | 1 unit of water derived from sustainable sources does not detract from the stock of energy resources |
| $w_E^x$ | $-0.1$ | 1 unit of water derived from unstainable extraction detracts 0.1 units from the stock of energy resources |
| $e_W^s$ | 0 | 1 unit of energy derived from sustainable sources does not detract from the stock of water resources |
| $e_W^x$ | $-\frac{1}{2000}$ | 1 unit of water derived from unstainable extraction detracts 1/2000 units from the stock of energy resources |
| $w_W^S$ | 0 | 1 unit of water derived from sustainable sources does not detract from the stock of water resources |
| $w_W^x$ | $-1$ | 1 unit of water derived from unstainable extraction detracts 1 unit from the stock of water resources |
| $y_E$ | 0.00075 | 0.075% of economic output each period is invested into energy resources, based on an approximation using National Accounts data for 2023 |

---

[2] This is identical to the hypothetical model value, which is at this stage the most plausible estimate. In the case of this variable, and other variables which are similarly treated, inputs values would stand to be refined through further research. From here on, these variables are indicated by (H).

| | | |
|---|---|---|
| $y_W$ | 0.0006 | 0.06% of economic output per period is invested into water resources, based on an approximation using National Accounts data for 2023 |
| $K$ | 0.875 | A 1% increase in economic capital increases economic output by 0.875% (H) |
| $g_e$ | 4% | Demand for energy grows by 4% per period, in line with the average over the past 10 years. |
| $g_W$ | 3% | Demand for water grows by 3% per period, in line with the average over the past ten years. |
| $S_1$ | 40% | The ratio of energy use from sustainable sources at the end of the simulation is targeted at 40%, reflecting commitments under international agreements and national policy. |
| $S_2$ | 5% | The ratio of water use from sustainable sources at the end of the simulation is targeted at 5%, a relatively low value indicating the absence of specific targets in this respect |
| $\gamma$ | 0.125 | The scaling effect in the production function is 0.125 (H) |
| $\delta$ | 10% | Economic capital depreciates at 3.2% per period, in line with National Account Estimates. |

The initial values of the stock and flow variables for this illustration are also assumed arbitrarily as below:

$E_0 = 43{,}386$

$W_0 = 642$

$e_0^s = 964$

$e_0^x = 2{,}190$

$w_0^s = 0$

$w_0^x = 30$

$K_0 = 150$

The value for the initial stock of energy resources is 43,386MWh, representing the present value (with a time discounting rate of 2%) of the potential output from the following energy sources over their residual life of operation:

*Table 7 Energy Stock Assumptions*

| | *Capacity (MW)* | *Use* | *Annual TWH* | *Residual Life* |
|---|---|---|---|---|
| *Interconnector* | 200 | 60% | 1,051.2 | 20 |
| *D3* | 120 | 25% | 262.8 | 10 |
| *D4* | 200 | 80% | 1,401.6 | 15 |
| *PV (effective)* | 50 | 100% | 438.0 | 15 |
| ***Total*** | | | **3,153.6** | **16.25** |

The value for the initial stock of water resources, at 642 million cubic metres, represents the present value (with a time discounting rate of 2%), of the potential output from existing groundwater and reverse osmosis plants over their remaining residual life. The annual production capacity is circa 50 million cubic metres and a 15-year average useful life is here considered.

Energy use from sustainable sources at the initial period, at 964MWh, reflects the output from photovoltaic units and an estimate of one half of the energy sourced from the interconnector. The remaining energy use is assumed to be from non-sustainable sources.

All water use in the initial period is assumed to be from non-sustainable sources, and reflects the average demand for water, in millions of cubic metres, per annum as currently applicable.

The value for the initial stock of productive economic capital at €150 billion, is based on a reasonable estimate of the capital output ration in the economy.

## 2.4. EEIO Model of the Slovak Economy

Over the past few decades, global water resources have come under increasing pressure, driven primarily by rising water consumption, pollution from economic activities, and the accelerating impacts of climate change on water availability (IPCC, 2022). With severe water stress affecting many regions, it has become crucial to monitor water demand and supply at both local and global scales and to identify the economic sectors exerting the greatest pressure on water systems (Lenzen et al., 2013; White et al., 2015). Growing water scarcity underscores the need to treat water as a limited resource and to prioritize strategies for managing demand.

Consequently, emphasis is placed not only on identifying water-intensive sectors but also on understanding the water embedded in the products they generate. The concept of the water footprint was introduced into water management science to highlight the role of consumption patterns and the global nature of water governance (Galli et al., 2012; Hoekstra and Mekonnen, 2012). It captures both direct and indirect water use. Specifically, in the case of food products, the water footprint measures freshwater use throughout the entire life cycle of production, including water consumption (and, where relevant, pollution) across all stages of the supply chain, from primary production through to retail.

Building on this concepts, the Slovak case study employs a hydro-economic input–output (IO) modeling framework to assess water consumption across economic sectors, the water footprint embodied in produced goods, and the effects of water-related shocks on the economy, with particular attention to agriculture and the food industry. This approach makes it possible to capture both direct and indirect water use throughout interlinked parts of supply chains. Using a baseline model and four scenarios – reflecting shifts in dietary composition, improvements in irrigation efficiency, and restricted water availability for selected sectors or the entire economy, we evaluate percentage changes in key economic indicators, such as output, value added, employment, as well as changes in water consumption at both the sectoral and economy-wide levels

### 2.4.1. Model description

The Slovak case study employs a hydro-economic input–output (IO) modeling framework to evaluate water consumption across economic sectors, the water footprint embedded in produced goods, and the impact of water-related shocks on the economy, with a particular focus on agriculture and the food industry. This approach enables the capture of both direct and indirect water use across interlinked parts of supply chains.

The modeling approach based on IO analysis provides a systematic framework for examining the interdependencies among economic sectors and tracing how changes in one part of the economy propagate through others. Originally developed by Leontief, IO analysis is based on fixed technical coefficients that describe the input requirements needed to produce a unit of output in each sector. IO models reveal not only the resources used within a sector itself but also those embodied in upstream supply chains and delivered to downstream sectors. When combined with environmental satellite accounts (i.e., integrating economic and environmental dimensions), IO models can quantify both direct and indirect water use associated with economic activities and final consumption, providing insight into how water demand is distributed across sectors and products. (Miller and Blair, 2009). Figure 12 illustrates the analytical framework of an environmentally extended IO model.

An environmentally extended IO model combines the economic structure of a standard input-output model with environmental variables, such as water consumption, while allowing the inclusion of other indicators (e.g., material use and carbon emissions, etc.). This approach enables simulations of changes in economic performance, sectoral activity, and water consumption, as well as their distribution across the economy, in response to environmental or economic shocks. It also identifies sectors for which water may act as a limiting factor from an economic perspective. The main principles of an environmentally extended IO model can be schematically described as follows (based on Miller and Blair, 2009).

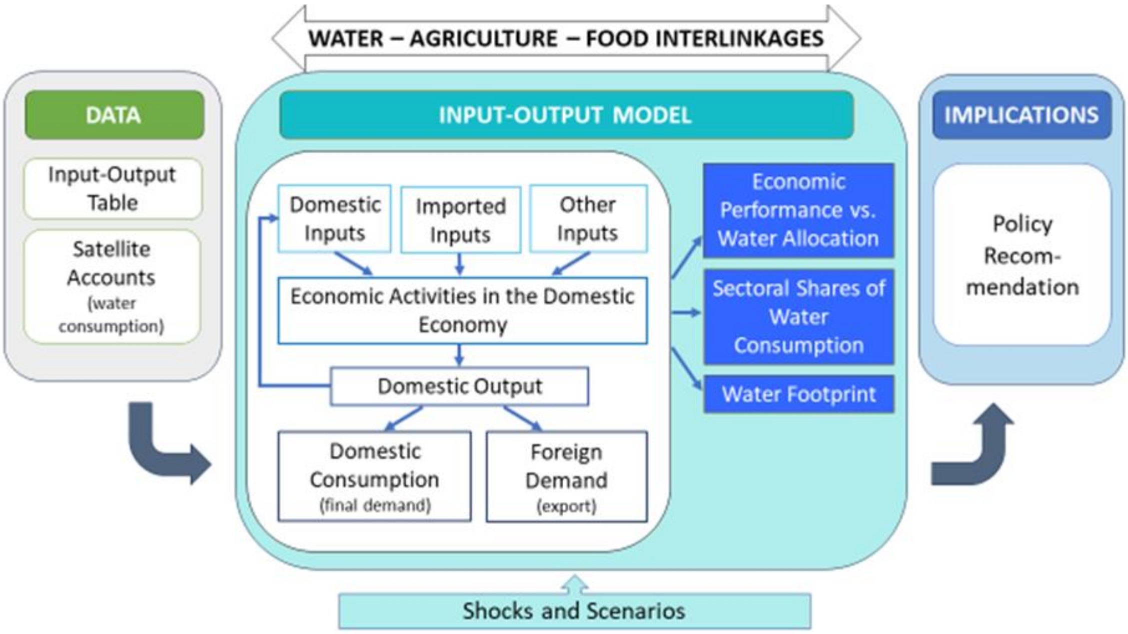

*Figure 12. Analytical Framework of Environmentally Extended Input-Output Model*

The model assumes Leontief production technology in each sector of the economy. A matrix A of technical coefficients $a_{ij}$ can be obtained from the economy's input-output table, with elements:

$$a_{ij} = {}^{x_{ij}}/_{x_j} \qquad (34)$$

where $a_{ij}$ denotes the input from sector i needed per one unit of production in sector j, $x_{ij}$ represents inputs from sector i to sector j, and $x_j$ is the total output of sector j.

Total output required to satisfy a given level of final demand in the economy is then calculated as:

$$x = (I - A)^{-1}\, y \qquad (35)$$

where $x$ represents the vector of sectoral total output, $(I-A)^{-1}$ is the Leontief inverse matrix, and $y$ is the vector of sectoral final demand.

An environmental coefficient matrix F with coefficients fij can be derived from satellite accounts, which complement the input-output table to include environmentally relevant variables. Its elements are:

$$f_{kj} = {}^{m_{kj}}/_{x_j} \tag{36}$$

where $f_{kj}$ is the environmental input k per unit of production in sector j, $m_{kj}$ represents the total environmental input k to sector j, and $x_j$ is the total output of sector j.

Environmental resources use in the economy can then be calculated as:

$$E = F\,(I-A)^{-1}\,y \tag{37}$$

where $E$ is the vector (or matrix, if multiple environmental variables are tracked) of environmental inputs, $F$ is the matrix of environmental coefficients, $(I-A)^{-1}$ represents the Leontief inverse matrix, and $y$ is the vector of sectorial final demand.

The model also allows scenario analysis. Changes in final demand, technical or environmental coefficients, or restrictions on sectoral output can be implemented, and their economic and environmental effects (by linking water use coefficients to sectoral outputs) can be computed using equations (2) and (4).

### 2.4.2. Baseline model and scenarios

To explore the economic and environmental implications of water use in Slovakia, a set of model scenarios was developed and implemented within the environmentally extended input-output framework. The IO model ensures that both direct and indirect effects of shocks are captured, providing a view of system-wide impacts. These scenarios developed reflect potential changes in final demand, production technologies, and water availability, allowing for an assessment of how sectoral water consumption, as well as key economic indicators, respond to different conditions. For the economic and environmental assessment, the following indicators are used:

- Output value,
- Value added,
- Employment,
- Water consumption.

The description of the baseline model and the developed scenarios is provided below:

- ✓ **Baseline**

The year 2021 serves as the baseline for the analysis, representing the current structure of the Slovak economy and its water consumption. The baseline model allows for the assessment of both economic performance and water use, providing a reference point against which alternative scenarios can be compared. Economic performance is evaluated in terms of key indicators, including sectoral and economy-wide output, value added, and employment. Water use is analyzed comprehensively, considering total water consumption in the economy, sectoral water use, and the water footprint of goods produced across different sectors. This baseline provides a detailed picture of how water resources are allocated within the Slovak economy and identifies the sectors and products that exert the greatest pressure on water resources.

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- ✓ **Scenario 1 – Changes in Food Demand Composition**

This scenario is motivated by evidence from the literature that different food products require substantially different amounts of water to produce, meaning that shifts in dietary habits can have important implications for water demand across the economy. Studies have shown that animal-based foods, particularly meat, generally require more water than plant-based foods, and national consumption patterns strongly influence overall water use (Wang et al., 2024; Liu and Savenije, 2008; Hoekstra and Chapagain, 2006). Systematic reviews further indicate that adopting healthier or more plant-based diets can reduce the green water footprint, although the blue water footprint may remain largely unchanged (Harris et al., 2020). Regional studies, including Vanham et al. (2013) for Europe and He et al. (2019) for China, highlight that dietary choices and eating habits, as well as population size and food demand, drive differences in water consumption, emphasizing the potential for water savings through shifts toward plant-based diets.

Based on these findings, the scenario is formulated as follows: final demand for plant-based foods (cereals, fruits, and vegetables) increases by 20%, while final demand for meat-based foods decreases by 5%. The primary research question addressed by this scenario is: What are the implications for water consumption and economic indicators if final demand for different types of food changes in this way?

- ✓ **Scenario 2 – Improved Water Efficiency in Agriculture**

This scenario is motivated by empirical research showing that modern irrigation techniques and improved on-farm water management can significantly increase water-use efficiency or reduce water consumption per unit of agricultural output. For example, a meta-analysis across China found that optimized water-saving irrigation in wheat production, especially via drip or micro-sprinkler systems, reduced irrigation water use by approximately 35.1% while maintaining yields, and increased water-use efficiency by roughly 18% under drip and 10% under micro-sprinkler irrigation (Ma et al., 2025). Jägermeyr et al. (2015) similarly estimate, in their global-scale study, that replacing surface (flood) irrigation with sprinkler or drip systems reduces non-beneficial water consumption (losses from evaporation, conveyance, and return flows) at the basin level by 54-76% and does not decrease crop yields. A field trial on maize demonstrated that drip irrigation achieved higher water-use efficiency and substantial water savings compared to conventional border or furrow irrigation (Liu et al., 2023). Water-saving irrigation methods have been demonstrated to enhance both water productivity and economic outcomes, making them a viable option for sustainable agriculture (Sajjad et al., 2025).

Given these findings, the scenario assumes an improvement in water efficiency for agricultural crop production, with a 5% reduction in the water coefficient for the respective sub-sectors, representing gains from more efficient irrigation, improved management, or the adoption of new technologies. Moreover, improved water efficiency and lower unitary water needs of crops can result from improved soil health. This scenario helps to answer what happens to water consumption in the economy and whether economic indicators change if the agricultural sector becomes more efficient in water use and reduces its direct water intensity.

- ✓ **Scenario 3 – Water-Use Restriction in Agriculture**

This scenario is motivated by studies showing that reductions in water availability due to drought, regulatory caps, or over-exploitation of freshwater resources can force agriculture to reduce its water consumption and irrigation withdrawals, with ripple effects throughout the economy. Water scarcity is increasingly threatening irrigated agriculture globally, leading to yield losses, reduced cultivated areas, and broader economic impacts when irrigation water is constrained (Biswas et al., 2025).

Coupled hydrological-economic models find that limiting water supply leads to pronounced drops in agricultural output (up to 60% by 2050 in some regions) and reductions in overall water withdrawals at the regional scale (Hejazi et al., 2023).

In this scenario, the total water availability for agriculture is reduced by 5%. Unlike efficiency improvements, this reduction limits the total water consumption of the sector, mimicking real-world conditions under drought or legal water-use restrictions. The production capacity of agriculture is therefore constrained, and supply limitations propagate both upstream (through inputs such as feed, fertilizers, and energy) and downstream (through processing, distribution, and services). Simulations under this scenario help to quantify what happens in the Slovak economy if agriculture is required to reduce its total water use by 5%. Specifically, how do supply chains adjust, and how are sectoral outputs, economy-wide water use, and key economic indicators such as output, value added, and employment affected?

- ✓ **Scenario 4 – Economy-Wide Water Constraint**

This scenario is motivated by growing evidence that water scarcity at the national or regional level can act as a binding constraint on economic activity across all sectors, not only agriculture. When total water availability is restricted (due to drought, overuse, regulatory caps, or climate change), sectors must adjust their outputs in response to a limited water input, resulting in economy-wide effects. If water supply constraints are integrated into hydrologic-economic models, the trade-offs between water use, economic output, and social welfare can be assessed. A study for a water-stressed Chinese region found that rationing water use in key sectors can lead to pronounced economic losses, job reductions, and environmental trade-offs (reduced grey water footprint), demonstrating how water limits propagate through entire supply chains (Zhao et al., 2021). Efficient water allocation policies under drought can significantly reduce economic losses compared to unmanaged rationing, highlighting the importance of considering inter-sectoral water trade-offs in policy design (Ortuzar et al., 2025).

By simulating this scenario, the IO model will reveal which sectors are most vulnerable to water scarcity. Total national water availability is reduced by 5%. The research question is: How must sectoral outputs adjust under this constraint? How does total economy-wide water consumption change, and what are the likely economic impacts? Water rationing is imposed across all sectors simultaneously, rather than targeting a specific sector; the policy does not specify how the rationing is allocated at the sectoral level. A binding water constraint for the entire economy translates into maximum output limits for sectors, and these sectors must scale back their production to align with the reduced water availability.

### 2.4.3. Data used

Input–output models rely on input-output tables, which are typically compiled by national statistical offices within the System of National Accounts. Environmental satellite accounts are then complemented by additional sector-specific data sources. In this study, however, the Slovak IO model used to analyze how water consumption relates to economic performance is based on the GLORIA (Global Resource Input–Output Assessment) database, a multiregional IO dataset covering 164 world regions and providing associated satellite accounts (Lenzen et al., 2017; Lenzen et al., 2022; IELab – Gloria). GLORIA offers IO tables disaggregated into 120 economic sectors and includes blue water consumption among its environmental extensions. The high level of sectoral detail was the primary reason for preferring GLORIA over the nationally available IO tables compiled by the Statistical Office of the Slovak Republic. From GLORIA, we extracted the IO table for Slovakia for the base year 2021.

The extracted table was reorganized into 39 economic subsectors, ensuring that agriculture and food processing retained a higher level of detail, while sectors not central to our case study were aggregated (with all service activities combined into a single sector). In addition to the sectoral structure and intermediate transactions, the dataset contains the standard IO components: final demand categories, value added, total output, and total input, all expressed in USD. The IO table was extended with the corresponding satellite account for blue water consumption (in $m^3$). Blue water refers to freshwater originating from surface water or groundwater, and blue water consumption captures the volume of water withdrawn that is either incorporated into products or lost through evapotranspiration or evaporation during crop growth and industrial processes.

Results for sectoral and the overall performance of the Slovak economy and its water consumption are presented for the following aggregated parts of the economy:

- Agriculture (including forestry and aquaculture),
- Food industry,
- Manufacturing,
- Services,
- Total economy.

## 2.5. The HHNK case study

This section sets out the analytical framework and scenario context for the HHNK case study. We first describe the random utility/discrete choice model that underpins the choice experiment and explains how farmers' preferences for different drought-management and water-storage options are represented. We then outline the relevant climate change and drought conditions in the HHNK service area, discuss why many farmers have not yet substantially adapted, and present the main policy instruments and behavioral responses explored in the choice scenarios. Finally, we position the case study within a WEFE (Water–Energy–Food–Ecosystems) perspective, highlighting the broader system trade-offs that motivate the design and interpretation of the model.

### 2.5.1. Conceptual model

The HHNK case study uses a choice experiment because the main objective is to understand how farmers trade off different elements of drought-management and water-storage policy packages, rather than to value a single, isolated change. So far, many farmers have been reluctant to invest in adaptation and tend to rely on the water authority to ensure sufficient supply, even though recent years and climate projections point to more frequent and severe water scarcity. By varying attributes such as abstraction bans, support for own storage, cooperation arrangements, and annual costs across alternatives, the CE allows us to recover marginal utilities and willingness to pay for each component of a policy package, and to see under which conditions farmers would be willing to take more responsibility for adaptation. The CE is embedded in a broader survey that documents farmers' experiences with drought and salinity, their current practices and their attitudes towards risk and cooperation, so that valuation results can be interpreted in context and linked to farmer heterogeneity. This combined CE–survey design also gives farmers a structured opportunity to express their preferences and concerns about future drought-management options, going beyond what can be inferred from existing market behavior or purely technical assessments.

The HHNK case study applies a discrete choice modelling framework, interpreted as a Random Utility Model (RUM), to analyze farmers' preferences for investments in on-farm water storage and related drought management measures. In the survey, farmers are repeatedly asked to choose between three options: two hypothetical policy/investment alternatives (A and B) and a status quo option in which

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the current situation on their farm remains unchanged. The hypothetical alternatives describe possible future situations along a fixed set of attributes, while the status quo reflects the respondent's existing practices and lack of new storage investment.

Formally, the utility that farmer i derives from alternative j in choice task t is written as U_ijt = V_ijt + ε_ijt, where V_ijt is the systematic (observable) component of utility and ε_ijt is an unobserved random term. The systematic component is specified as a function of the attribute levels of the alternatives and, in extended models, of interactions between these attributes and farmer characteristics. Farmers are assumed to choose, in each task, the alternative with the highest utility. Under standard assumptions on the distribution of ε_ijt, this yields a logit-type choice probability model.

In our baseline specification, the systematic utility is linear in the attributes: V_ijt = β_1 Ban_jt + β_2 Bridge_jt + β_3 Coop_jt + β_4 Cost_jt + … , where the main policy attributes are:

- **Abstraction ban (Ban):** number of weeks per year that irrigation with surface water is not allowed (1, 3 or 6 weeks/year).
- **Bridging with own storage (Bridge):** duration for which the farmer can continue irrigation using on-farm storage if abstraction from surface water is not possible (1 or 3 weeks/year).
- **Cooperation (Coop):** Cooperation mode in water storage:
    1. no cooperation (only own investment),
    2. cooperation with HHNK (joint investment with other farmers, implemented by HHNK),
    3. cooperation without HHNK (joint investment with other farmers).
- **Cost (Cost):** net cost per hectare per year for water storage, including investment and maintenance (1500, 2000 or 2500 EUR/ha/year).

Given a negative cost coefficient β_4, marginal utilities for the non-monetary attributes can be converted into implicit willingness-to-pay (WTP) measures by taking the ratio of attribute coefficients to the cost coefficient. This yields, for example, the WTP for extending bridging capacity from one to three weeks, or for moving from individual to cooperative investment modes.

The panel structure of the data (multiple choices per farmer) allows us to capture preference heterogeneity by letting coefficients vary across individuals, either through interaction terms (e.g. with farm size, past drought or salinity experience, or attitudes such as risk perception and trust in HHNK) or through random-coefficients specifications. In all cases, the interpretation remains demand-side: we model how farmers trade off costs, restrictions, and cooperation arrangements when considering investments in water storage.

This stated preference RUM framework is complementary to physically based models. It does not simulate the hydrological system directly, but provides quantitative evidence on the behavioural and economic responses that underpin adoption of drought-adaptation measures in a setting where conventional supply-side expansion is severely constrained.

### 2.5.2. Climate change and drought in the HHNK service area

Climate projections for the Netherlands indicate warmer and drier summers, more frequent and intense drought events, and increasing pressure from salinity, particularly in low-lying coastal regions such as Noord-Holland. These changes are expected to result in:

- more frequent periods of low surface water levels and reduced possibilities for freshwater intake from main water bodies;
- increased risk of salt intrusion into surface waters and potentially into shallow groundwater, affecting both crop yields and the usability of water for irrigation;
- higher reliance on existing pumping infrastructure and storage capacity to maintain target water levels in polder systems.

In the HHNK service area, the combination of low-lying land, high drainage intensity and limited space for large-scale new reservoirs means that conventional supply-augmentation options are limited or costly. Historically, farmers have relied on the water authority to secure sufficient supply and have been reluctant to invest in on-farm adaptation, even as recent summers and climate projections point to more frequent and severe water scarcity.

Adaptation in this context requires a portfolio of measures that reduce demand during peak stress periods, increase local buffering and storage where feasible, and improve the timing and coordination of abstractions at both farm and network level. Against this background, the choice experiment introduced above represents stylised future policy and investment options for drought management and water retention in agriculture. Rather than modelling detailed climate scenarios, the HHNK case focuses on how farmers respond to different combinations of restrictions, support for storage and cooperation arrangements under a climate regime in which drought risks are expected to intensify.

### 2.5.3. Why farmers may not yet be adapting

Despite increasing awareness of climate risks, many farmers in Noord-Holland have not yet implemented substantial on-farm adaptation measures beyond incremental adjustments. Several strands of theory help explain this adaptation gap:

- **Risk and ambiguity aversion**: Investments in storage or other adaptation measures require upfront costs for benefits that are uncertain in timing and magnitude. Under ambiguity and learning about climate risks, farmers may postpone investment even when it is economically rational in expectation.
- **Credit and liquidity constraints**: Especially in sectors with thin margins, farmers may lack access to affordable finance or sufficient liquidity to undertake lumpy investments in storage infrastructure or new irrigation technology.
- **Behavioral barriers and status quo bias**: Farmers may underestimate future drought risk or over-weight recent wet years, leading to underinvestment in long-lived adaptation. Established routines and limited time can reinforce inertia.
- **Collective action and coordination problems**: Some measures (for example, shared storage, coordinated abstraction, or collective management) require cooperation among farmers or between farmers and HHNK. Without clear rules, monitoring, and enforcement, free-riding and strategic uncertainty can stall investment.
- **Perceived policy uncertainty**: If farmers are unsure about future policy on abstraction bans, compensation rules or cost-sharing arrangements, they may wait until the policy environment becomes clearer before committing to irreversible investments.

The choice experiment is explicitly designed to shed light on these mechanisms by varying, in a controlled way, the presence of cooperation arrangements, the probability and severity of droughts, and the costs of, and by linking choices to farmer characteristics, experiences, and perceptions.

The choice experiment scenarios explore different combinations of instruments that could induce farmers to invest in adaptation and water retention, and to move away from a passive reliance on the water authority towards more proactive on-farm measures. In particular, the CE directly varies three elements:

- **Support for own storage**: Changes in the duration of bridging with own storage represent different levels of effective support for maintaining irrigation during droughts, which can reduce perceived risks of investing in on-farm storage.
- **Conditional abstraction rules**: Temporary abstraction bans or restrictions, combined with different levels of bridging capacity, shift incentives towards investment while safeguarding minimum water levels and preventing over-abstraction during peak stress.
- **Cooperation and collective schemes**: Cooperative arrangements – for example, through investments implemented together with HHNK or with other farmers – may exploit economies of scale in storage, reduce local conflicts, and lower transaction costs of adaptation.

In the broader policy context, information and advisory services also play a role: providing targeted information on climate risks, technical options, and contractual terms can reduce uncertainty, address misconceptions, and lower non-monetary costs of participation, even though this aspect is not explicitly represented as a separate attribute in the CE.

The estimated WTP measures derived from the RUM provide quantitative guidance on how attractive different combinations of these elements are to farmers, and how strongly they respond to cost signals, restrictions, and cooperation opportunities. This allows us to identify policy packages that are not only technically promising but also behaviorally and economically acceptable to the farming community.

### 2.5.4. WEFE perspective for HHNK

The design and interpretation of the choice experiment explicitly consider WEFE trade-offs:

- **Water**: Measures that expand local storage and improve coordination can reduce peak abstractions and contribute to more stable water levels, thereby supporting overall water security in the polder system and reducing pressure during dry spells.
- **Energy**: Additional storage and pumping entail energy use for filling and managing basins, but may also reduce the need for emergency pumping and high-frequency interventions by HHNK. In addition, there is scope to time pumping and storage operations to periods when electricity is relatively cheap or low-carbon, and to coordinate with existing on-farm energy systems, which strengthens the water–energy link.
- **Food**: Improved drought resilience can stabilize yields and reduce crop losses in dry years, thereby supporting farm income stability and the continuity of local agricultural production. The primary motivation here is not national food security in a strict sense, but the economic viability of farms and associated supply chains in the region, which may indirectly contribute to a robust food system.
- **Ecosystems**: Reducing low-flow events and avoiding extreme water level fluctuations can benefit aquatic and wetland ecosystems, reduce salinity stress, and improve water quality, particularly in environmentally sensitive peat meadow (veenweide) areas. In these landscapes, water level management interacts with soil subsidence, greenhouse gas emissions, biodiversity (e.g., meadow birds), and recreational and landscape values. Abstraction patterns and drought-

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management policies therefore affect a broader bundle of ecosystem services than crop production alone.

The choice experiment thus provides evidence on the acceptability and perceived attractiveness of WEFE-relevant measures from the farmer's perspective. These preference-based results can later be linked to governance or hydrological assessments from other work packages to evaluate WEFE outcomes at the system level.

### 2.5.5. Data used

The empirical analysis is based on a dedicated survey and choice experiment among farmers in the HHNK service area in Noord-Holland. The questionnaire was developed jointly by HHNK and the Vrije Universiteit Amsterdam within the RETOUCH NEXUS project and focuses on how farmers currently cope with drought and water scarcity, what they expect for the future, and how they value different options for on-farm water storage.

The survey instrument consists of four main components:

1. **Background questions on the farm**: Respondents provide information on basic farm characteristics, such as farm type, land use, size, irrigation practices and the presence (if any) of existing storage or drought measures. These questions establish the context for interpreting the choice tasks and allow for segmentation by farm type or region.
2. **Experiences, expectations and measures related to drought**: This block documents how farmers have experienced recent dry years and salinity problems, the degree of hindrance from water shortages, measures already taken to deal with drought, and expectations about how drought and water availability will develop in the future. It also asks what farmers consider feasible to do themselves and what they expect from HHNK and other actors. These variables help explain heterogeneity in preferences and put willingness-to-pay (WTP) estimates in the context of actual exposure, perceived risk, and existing adaptation behavior.
3. **Choice experiment module**: Each respondent answers a sequence of choice tasks, each presenting two hypothetical policy/investment alternatives (A and B) and a status quo option in which the current situation on the farm remains unchanged. The alternatives differ along the attributes described in Section 2: the number of weeks per year with an abstraction ban, the duration of bridging with own storage, the form of cooperation (none, with HHNK, or with other farmers), and an annual cost per hectare. The design uses blocking and randomisation to distribute choice sets across respondents and to ensure sufficient variation and statistical efficiency of parameter estimates.
4. **Closing questions and socio-demographics:** In the final part of the questionnaire, respondents are asked how they interpreted the choice tasks, how realistic they found the scenarios and whether they experienced any difficulties when answering. This is followed by socio-demographic questions (e.g. age, education) and additional attitudinal items on risk, cooperation and trust in the water authority. These responses allow for robustness checks on the validity of the CE answers and for linking preferences to broader attitudes and personal characteristics.

The survey uses a mix of closed questions with fixed answer categories, open questions for qualitative comments and explanations, and the choice questions that form the core of the CE. This design provides both quantitative data for econometric analysis and qualitative information to interpret and contextualise the modelling results.

Survey responses are cleaned and pre-processed to construct:

- a respondent-level dataset with farm characteristics, drought experiences, expectations, attitudes, and socio-demographics;
- a long-format CE dataset, containing attribute levels, the chosen alternative and respondent- specific covariates to estimate willingness-to-pay levels.

At the time of writing, data collection is still ongoing. The results presented in this document and in the accompanying dashboard are therefore based on an interim sample and should be considered preliminary and illustrative only.

## 2.6. Belgian case study

Flanders faces increasing pressure on its water system due to climate change, growing water demand, and recurring droughts and flooding events. In response, the Flemish government has launched the Blue Deal, recognizing the urgency of strengthening water resilience across sectors. At the same time, the implementation of integrated approaches, such as the Water–Energy–Food–Ecosystems (WEFE) nexus, remains challenging due to fragmented responsibilities and the complex interactions between agricultural, energy, and environmental objectives.

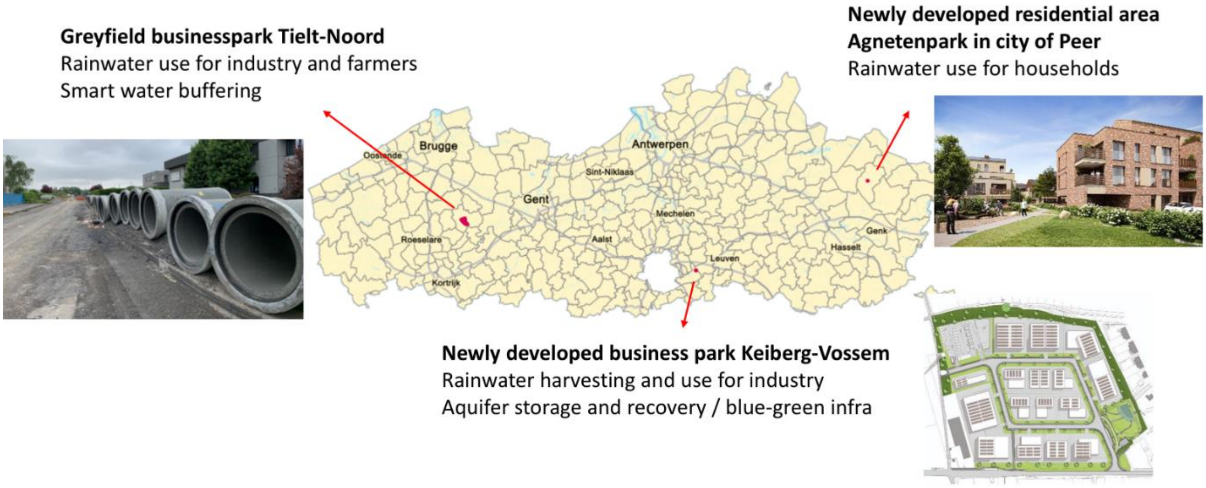
Ensuring long-term water availability in water-stressed regions such as Flanders requires complementing the centralized drinking water supply system with local and decentralized solutions. Circular rainwater management (e.g., buffering, rainwater use, and infiltration) can reduce pressure on drinking water production, lower wastewater treatment costs, and enhance resilience to both droughts and extreme rainfall events. Accordingly, Flemish policy increasingly emphasizes the circular use of rainwater and wastewater as a key element in securing a reliable 24/7 drinking water supply.

The Belgian case study examines the costs and benefits of innovative rainwater harvesting and management solutions across three demonstration sites. These sites represent different contexts of water use, including industrial, agricultural, and residential applications, and serve as real-life test cases for both individual and collective rainwater systems. In this deliverable, a cost-benefit analysis is carried out for one of the case studies (Keiberg Vossem) to assess the societal impacts of alternative rainwater management strategies.

Beyond evaluating individual cases, this deliverable aims to contribute to a broader analytical framework that can support decision-making on rainwater management strategies. We provide guidance on when collective rainwater systems may be preferable to individual solutions, by systematically comparing their hydrological performance, costs, and societal benefits. The analysis therefore not only assesses the Keiberg Vossem business park in detail but also offers insights that can inform future policy and investment decisions on decentralized water infrastructure in Flanders.

The Belgian case study directs integrated water management at three sites: two business parks and one residential area. The ambition exceeds the optimization of the regional water networks and promotes overall sustainable living and working. Rainwater collection is mandatory in Flanders. This paves the way for circular rainwater applications. The Belgian case study is built on multi-stakeholder partnerships and provides demonstration sites for industrial water use, irrigation and domestic applications.

In Figure 13, the Belgian demonstration sites are displayed. The first case study is the greyfield business park Tielt Noord, where rainwater is used for industrial activities and by farmers, supported by smart water buffering solutions. The excess rainwater is used by agriculture sector. The second case study is the newly developed residential area Agnetenpark in the city of Peer, which focuses on the use of rainwater for household purposes. The third case study is the newly developed business park Keiberg Vossem, where rainwater harvesting and use are applied for industrial use, combined with aquifer storage and recovery and blue-green infrastructure.



**Greyfield businesspark Tielt-Noord**
Rainwater use for industry and farmers
Smart water buffering

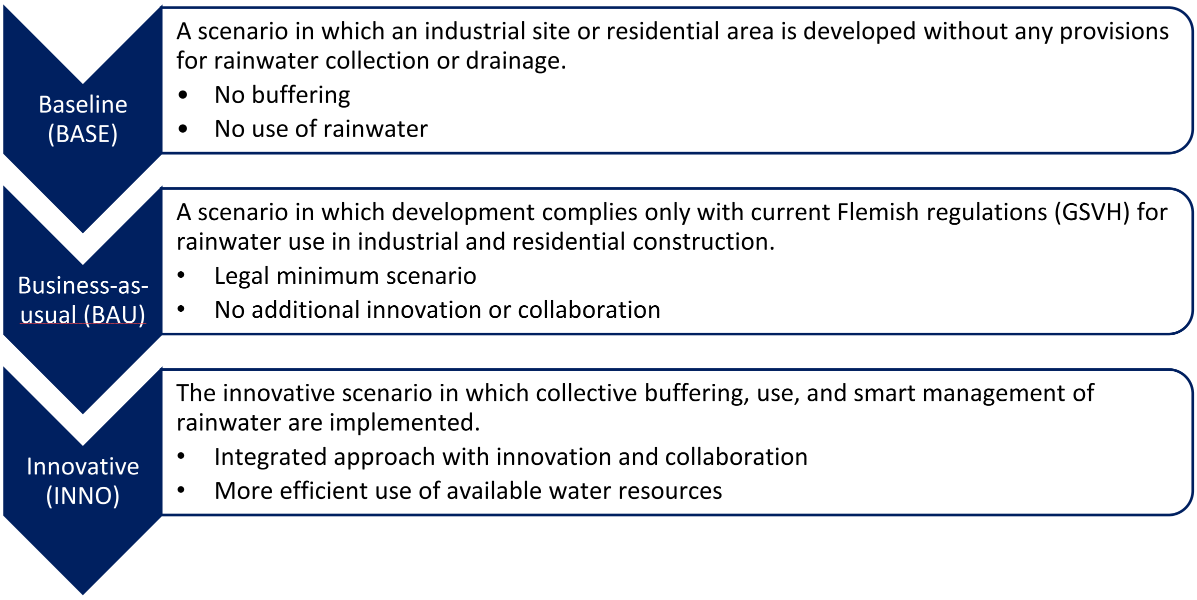
**Newly developed residential area Agnetenpark in city of Peer**
Rainwater use for households

**Newly developed business park Keiberg-Vossem**
Rainwater harvesting and use for industry
Aquifer storage and recovery / blue-green infra

*Figure 13 Overview of the Belgian demonstration sites*

### 2.6.1. Description of the model

This section describes the analytical framework used to assess the hydrological and economic performance of the different rainwater management scenarios. It first introduces the scenario definitions that form the basis for comparative analysis. Next, the water modelling component quantifies rainwater flows, storage, and use under each scenario. Finally, the cost-benefit analysis component evaluates the associated societal costs and benefits over the project lifetime. Together, these elements form an integrated



Baseline (BASE)

A scenario in which an industrial site or residential area is developed without any provisions for rainwater collection or drainage.

- No buffering
- No use of rainwater

Business-as-usual (BAU)

A scenario in which development complies only with current Flemish regulations (GSVH) for rainwater use in industrial and residential construction.

- Legal minimum scenario
- No additional innovation or collaboration

Innovative (INNO)

The innovative scenario in which collective buffering, use, and smart management of rainwater are implemented.

- Integrated approach with innovation and collaboration
- More efficient use of available water resources

*Figure 14 Scenario analysis*

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### 2.6.2. Scenario Definitions

We developed three scenarios to explore alternative approaches to rainwater management within a demonstration site. These are displayed in Figure 14. The baseline scenario represents a minimal configuration, without buffering or rainwater use measures. In contrast, the Business-as-usual scenario assumes compliance with current Flemish regulation (GSVH), but no additional innovation. Finally, the Innovative scenario introduces collective buffering, water use, and smart control of rainwater flows. These three scenarios provide a structured framework for comparing the hydrological performance, costs, and societal benefits across the three sites. This allows us to compare the net (societal) gains or losses of the innovative scenario relative to the business-as-usual scenario. In the next two sections, the water modelling and cost-benefit modelling are described in detail.

### 2.6.3. Water Modelling

Water modelling for a specific scenario and case study follows a stepwise approach. Within this section, the industrial greenfield case study is used as a reference. Three scenarios are assessed and compared: Baseline (BASE), Business as Usual (BAU), and Innovative as Built (INNO).

The majority of the water-related input data (e.g., surface runoff, local industrial water demand, etc.) is derived from datasets collected by De Watergroep for the selected case study. These datasets provide the basis for quantifying water flows and evaluating scenario-specific performance.

The overall rainwater balance varies per scenario and is directly linked to the type and extent of rainwater infrastructure implemented. In the BASE scenario, no rainwater management infrastructure is assumed, resulting in the direct runoff of all precipitation from the site into the sewer system.

The BAU scenario includes conventional rainwater management measures at the level of individual industrial plots, in accordance with the Flemish rainwater legislation (GSVH, 2023[3]). This typically involves the installation of rainwater storage tanks and infiltration facilities, at individual industrial plots. The dimensions of these systems are calculated and optimized based on local rainfall conditions and estimated water demand, using dedicated calculation software (Sirio Design[4]).

The INNO scenario closely reflects the collective as-built rainwater management concept, implemented in the RETOUCH case study. To enhance transferability and replicability, this scenario has been partially generalized, allowing the results to be applied to other sites and spatial scales (scalability being a key objective in the final phase of the project). The sizing of rainwater storage and infiltration systems in this scenario is largely based on the implemented infrastructure, recalculated in the Sirio Design software, while remaining consistent with the principles and requirements of the Flemish rainwater legislation.

The Keiberg Vossem case study is still under development; consequently, the final future water demand has not yet been fully defined. Therefore, additional estimates of rainwater consumption were applied, covering both sanitary uses and industrial process water demand. Sanitary water demand was estimated based on the expected future employee occupancy rate[5]. In addition, one significant local water consumer was explicitly included in the assessment, namely a car wash facility located within the case study area. This installation represents a relevant high-demand user and is

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[3] https://omgeving.vlaanderen.be/nl/verordeningen/de-gewestelijke-hemelwaterverordening-2023

[4] https://www.sirio.be/platform/design

[5] https://www.vlario.be/website/files/downloads/Richtlijnen-Hemelwatergebruik-Literatuurstudie-v23-06-2025.pdf

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therefore essential for realistically assessing local water demand and (collective) rainwater use potential.

For both the BAU and INNO scenarios, the overall water balance is calculated using the Sirio Design software. All required input parameters are implemented in the model, including: the number of industrial plots, the extent of sealed surfaces per plot, runoff coefficients, the volumes of rainwater storage tanks and infiltration units, local infiltration capacity, and the potential for rainwater use for both sanitary applications and industrial process water.

Based on these inputs, the software generates the key outputs required for the subsequent cost–benefit analysis. These outputs include the effective rainwater use at the plot level and for the entire industrial site, the volume of infiltrated rainwater at the plot level, and the remaining runoff from the site. The latter represents a residual water volume with potential for additional infiltration measures and/or external use, for example by local farmers for irrigation purposes.

### 2.6.4. Cost-Benefit Analysis

Cost-benefit analysis (CBA) allows us to assess infrastructure projects and policy alternatives and helps determine which option is best suited to achieving a specific goal. In this deliverable, we assess whether collective, decentralized rainwater management leads to a net societal gain compared to having no dedicated rainwater infrastructure, as well as how it compares to the use of individual rainwater tanks.

In essence, a CBA can be captured by the Net Present Value (NPV) in the following equation (Halytsia et al., 2022):

$$NPV = \sum_{t=1}^{T} \frac{B_t - C_t}{(1+r)^t} \tag{38}$$

with $t$ time; $T$ the total number of periods; $B_t$ the expected benefits at time $t$; $C_t$ the expected costs at time $t$; and $r$ the discount rate.

We implement the same discount rate recommended by the European Commission (2021) of 4% (i.e. $r$). We assess that the lifespan of the water-infrastructure projects discussed in section 2 is 40 years, so we assume that $T$ equals 40.

**Benefits calculation**

In this deliverable, several categories of benefits are considered, including avoided wastewater treatment costs, avoided drinking water production costs, and benefits related to groundwater storage. The avoided costs of water treatment or production are calculated using the following general equation:

$$AC_t = c_t \times Q_t \tag{39}$$

where:

- $Q_t$ represents the relevant water volume in year $t$ (m³), and
- $c_t$ denotes the avoided cost per cubic meter (€/m³), corresponding to either wastewater treatment costs, drinking water production costs, or groundwater abstraction/recharge values.

### 2.6.5. Data used

This section describes the data sources and assumptions used in the analysis of the Keiberg demonstration site. Section 2.6.5.1 presents the water balance data used to quantify water flows under the different scenarios. Section 2.6.5.2 details the cost data, including capital and operational expenditures, mainly based on real-life information from the implemented subcase of Keiberg (project partner De Watergroep). Section 2.6.5.3 introduces the benefit data and valuation approach, covering avoided wastewater treatment costs, avoided drinking water production costs, groundwater recharge benefits, and avoided flood damage. Together, these inputs form the basis for the subsequent water balance assessment and cost-benefit analysis.

#### *2.6.5.1. Water balance data*

The water supply and demand data are primarily based on real-life measurements and estimates from the Keiberg Vossem case study. To ensure that the results are replicable and scalable to other industrial greenfield developments, these data were partially generalized to representative reference values. On the supply side, an individual plot area of 1500 m² was assumed, with a runoff coefficient of 0.8. In addition, a collective road surface of 1 ha was included, assuming an identical runoff coefficient. These surfaces jointly determine the total rainwater supply available for storage, infiltration, and use. On the demand side, an occupancy of 15 employees per plot was assumed for estimating potential sanitary rainwater use. Sanitary water demand was calculated as 25L per employee per day, for 220 working days per year, with an additional 50% allowance to account for cleaning activities and limited plot irrigation. Furthermore, one large local industrial process water consumer was included, with an estimated annual water demand of 10000 m³/year, representing a relevant high-demand user within the business park.

Rainwater storage tanks were initially sized in accordance with the Flemish rainwater legislation and subsequently optimized using the Sirio Design software. Examples of the resulting storage volumes at both plot level and collective level are illustrated in Figure 15.

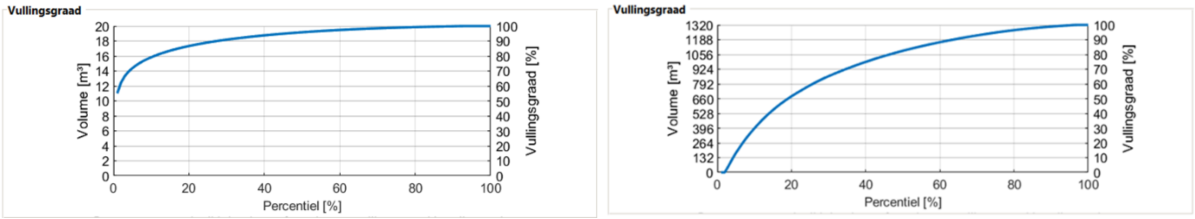

*Figure 15 Design of rainwater storage tank volume and according filling degree in percentiles of time*

For infiltration facilities, the Flemish design guideline of 330 m³ per hectare of contributing surface was applied. An infiltration capacity of 10 mm/h ($2.8 \times 10^{-6}$ m/s) was assumed, together with a safety factor of 2, to account for uncertainties in soil conditions and long-term performance.

The Sirio Design software applies a 100-year actual Belgian rainfall time series to calculate the water balance for each individual plot and infrastructure unit, as well as for the overall industrial greenfield site. These simulations were performed for both the BAU and INNO scenarios.

#### *2.6.5.2. Data on costs*

Because the subcases are already built and operated, we started from real-life and detailed reference cost data, specifically for the INNO scenarios. For subcases Keiberg and Agnetenpark, this data is provided by De Watergroep. For the subcase Tielt-Noord, the cost data is provided by Aquafin. All cost data are in real terms, with the starting year as the reference year.

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This cost data includes construction costs and operating costs. Table 8 provides detailed categories of the available cost data. The costs of the different subcases can be divided into these categories. We don't report the costs in this deliverable by category due to confidentiality.

*Table 8 Cost components*

| **CAPITAL EXPENDITURES** | |
|---|---|
| | Extraction and storage infrastructure |
| | Treatment infrastructure |
| | Distribution infrastructure |
| | Monitoring infrastructure |
| | Buffers |
| | Billing |
| | R&D costs |
| | Overhead |
| **OPERATIONAL EXPENDITURES** | |
| | Maintenance costs |
| | Operational costs |
| | Billing & administrative costs |
| | Monitoring costs |
| | Overhead |

The available cost data were primarily used as a reference for the INNO scenario, reflecting the as-built infrastructure of the case study. To enable a robust comparison with the BAU scenario, additional and more generic cost data were collected from literature, technology providers, and relevant reference projects.

Capital expenditure (**CAPEX**) data were compiled for the following components: prefabricated concrete rainwater storage tanks, excavation works, pumps, piping, water treatment units, electromechanical installations (EM), study and design costs, and overhead costs. Operational expenditure (**OPEX**) was divided into treatment operation and maintenance, electricity consumption, and technical support. The assumed water treatment configuration consists of a multi-step treatment train, including pre-filtration, activated carbon filtration, and UV disinfection. This configuration was selected as a robust and widely applied solution for both sanitary rainwater use and industrial process water applications.

At the collective infrastructure level, a total **pipe length** of 800 m with a nominal diameter of DN110 was assumed, using an indicative unit cost of 150 EUR/m. At the individual plot level, a pipe length of 40 m with DN50 specifications was considered and a cost of 80 EUR/m.

For **pumping systems**, a capacity of 5 $m^3$/h with a unit cost ranging between 3000 and 5000 EUR per pump was assumed at plot level. At the collective level, a 40 $m^3$/h pump capacity was considered, with an indicative investment cost of 25 kEUR per unit.

Cost data were collected from several technology suppliers, including GEP, WAVIN, and Pipelife, and were complemented by independent construction cost information from the LIVIOS platform[6]

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[6] www.livios.be

Examples of the derived and applied cost curves (for instance for prefabricated concrete **rainwater tanks** and **excavation works**) are presented in Figure 16. These curves are based on data from reference projects, current market prices, and publicly available construction cost information (e.g. LIVIOS).

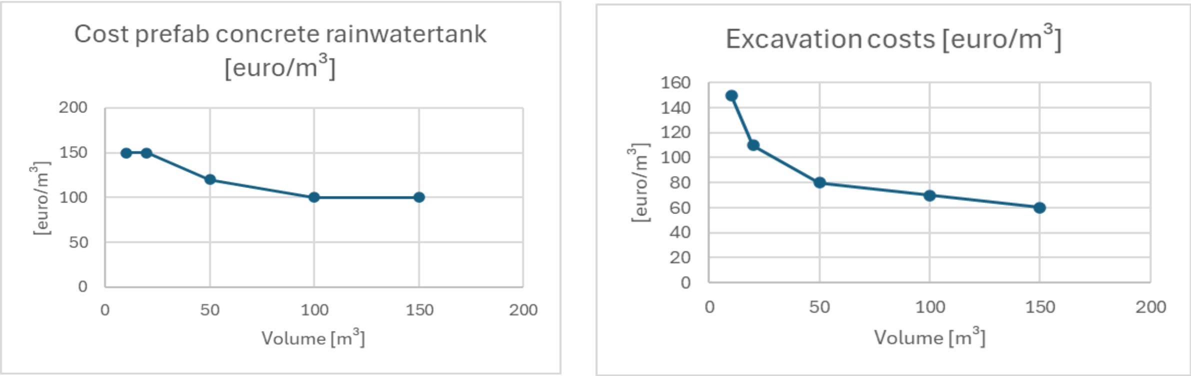

*Figure 16 Examples of derived cost curves from reference data for prefabricated concrete rainwater tanks and excavation works*

For the assumed **treatmen**t train (pre-filtration, activated carbon, and UV disinfection), a CAPEX range of 8000 to 12000 EUR per treatment unit was applied. Annual OPEX was estimated between 1000 and 1500 EUR per unit, depending on the specific operational context, such as exclusive sanitary reuse versus combined sanitary and high-volume industrial process water demand. Technology cost and performance assumptions for the treatment train were further supported by information obtained from specialized water technology providers, including Pollet Water Group (filtration, granular activated carbon and UV systems), Desotec (granular activated carbon solutions), Lenntech (filtration systems and GAC vessels), and ATB (UV disinfection systems).

In addition to the infrastructure CAPEX, supplementary costs were included for study and design, amounting to 10% of the total CAPEX, as well as an overhead cost of 10%. Electromechanical installations (EM) were accounted for with a fixed cost of 2500 EUR at plot level and 25000 EUR at collective level. Finally, a technical container to accommodate the collective treatment train was included, with an assumed investment cost of 20 kEUR.

With respect to OPEX, an electricity cost of 0.2 EUR/m³ was applied to account for pumping energy consumption, in addition to the treatment operation costs described above. Furthermore, a requirement for technical support at collective level equivalent to 0.2 FTE was assumed. No additional technical support was allocated at plot level, as only limited maintenance activities are expected for individual installations.

#### *2.6.5.3. Data on Benefits*

Table 9 gives an overview of the benefits created by rainwater harvesting in Keiberg-Vossem. The first benefit is the avoided treatment costs. When rainwater enters the wastewater treatment plants, it increases the volume that must be treated. This extra volume results in additional operational costs and may also influence the dimensioning and capacity requirements of certain treatment installations. According to the analysis of wastewater treatment costs in Flanders (Van Dijk Management, 2010), the costs attributable specifically to the additional rainwater volume represent approximately 16% of total wastewater treatment costs. The full cost-recovery tariff for wastewater treatment in Tervuren (Flanders) is €1.883 per m³ in 2025. Applying the 16% share to this tariff yields an estimated €0.30 per m³ of treated rainwater. This value is therefore used in this study as the unit cost parameter for avoided wastewater treatment when rainwater is buffered, used, or infiltrated rather than conveyed to the treatment plant.

The second benefit is the avoided costs related to drinking water production, which are derived from the VMM report (VMM, 2018). This report estimates the costs associated with water losses due to leakages. As part of this analysis, VMM provides an estimate of the production cost of drinking water. Based on the aggregated sector data, the cost of producing one m³ of drinking water is estimated at €0.41 per m³ in 2016. Indexed to 2025 using the Statbel (2025) results in a cost of €0.59 per m³, which is used in this study as the unit value for avoided drinking water production.

The third benefit is related to groundwater recharge. This generates benefits through both avoided abstraction tariffs and avoided pumping costs. The average groundwater abstraction tariff in Flanders was €0.17 per m³ in 2022 (VITO Kennispunt Water, 2025). In addition, the operational cost of pumping groundwater has been estimated at €0.23 per m³ (Bagnoli & Broekx, 2025). Taken together, these components result in a total benefit of €0.40 per m³ of water infiltrated or contributing to groundwater recharge.

*Table 9 Overview of created benefits*

| **Benefit** | **Unit cost (€/m³)** | **Source** |
|---|---:|---|
| Avoided treatment costs | 0.30 | Nature Value Explorer (VITO)<br>Van Dijk Management (2010) |
| Avoided drinking water production costs | 0.59 | VMM (2018) |
| Avoided costs regarding ground water | 0.40 | VITO (2025)<br>Bagnoli & Broekx (2025) |

# 3. Results

## 3.1. The Jucar River Basin

### 3.1.1. WEFE index

The WEFE index results reveal consistent discrepancies across climate models, SSP scenarios, and water pricing strategies, with resource access exceeding availability in all sectors. System performance progressively improves across water pricing strategies, rising from 61 in the Baseline to 63 under Dynamic Water Pricing (DWP) and 71 under Uniform Water Pricing (UWP) for the period 1979–2050 (Figure 17a).

UWP achieves the strongest gains in environmental and resource-use dimensions, with a 57% increase in the Ecosystem Index, a 21% gain in the Energy Index, and an 8% increase in the Water Index, reflecting its effectiveness in curbing water withdrawals and enhancing ecological sustainability. DWP produces more moderate improvements, with smaller increases in the Ecosystem (+20%), Energy (+7%), and Water (+1%) indices. Both strategies involve trade-offs in the food sector, reducing the Food Index by 6% (DWP) and 8% (UWP), reflecting economic efficiency losses associated with lower water-productivity crops. Overall, UWP demonstrates that simplified and transparent pricing schemes provide strong incentives for efficient resource use and ecosystem protection, but its broader economic implications, particularly for food security, must be carefully considered (Figure 17a).

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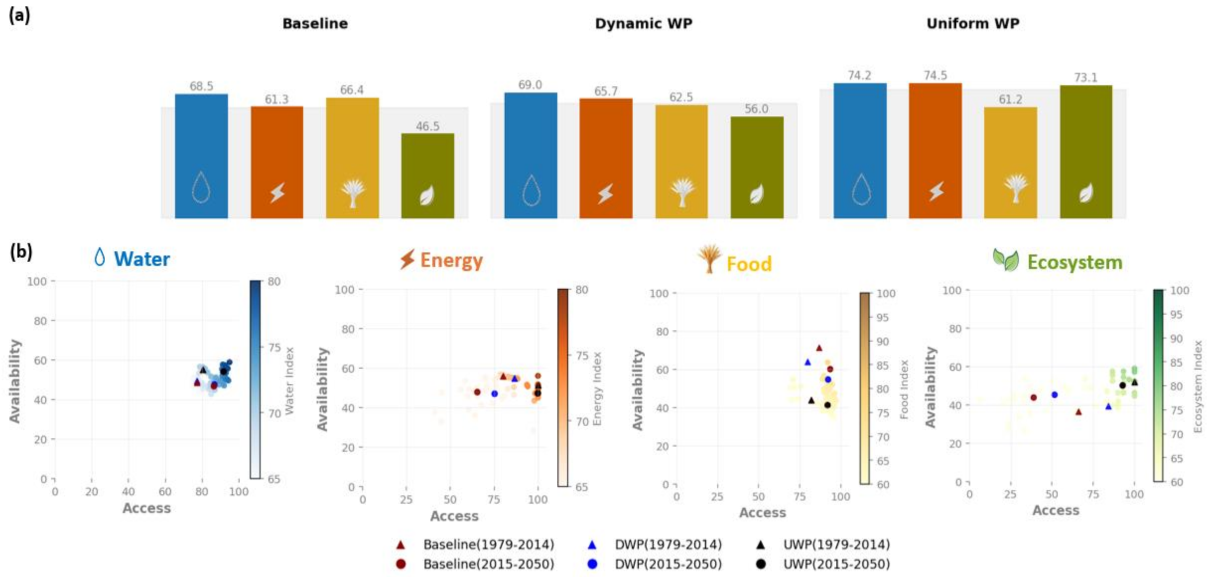


*Figure 17  WEFE Nexus Performance Index*

(a) WEFE Nexus index, sectoral performance index under different water pricing strategies for SSP3-7.0, aggregated across all climate models for 1979–2050. (b) Nexus performance index for water, energy, food, and ecosystem components across climate models, socioeconomic pathways (SSPs), and water pricing strategies in the historical (1979-2014) and projected periods (2015-2050). Each point represents the performance associated with a given climate projection, socioeconomic pathway, and pricing policy.

## Resource Availability vs. Access in the WEFE Components

Figure 17b illustrates the distribution of the performance index for water, energy, food, and ecosystems, distinguishing between access and availability across all simulations, which encompass combinations of climate models, SSPs, time periods, and water pricing strategies. The results reveal notable discrepancies both across sectors and between access and availability within each sector. For the water sector, availability is relatively constrained, ranging between 40% and 60%, reflecting limitations in the physical water supply due to hydrological variability, reservoir storage, and inflows. However, access to water for agricultural and urban demands is considerably higher, ranging from 70% to 100%, suggesting that water management measures, such as allocation priorities and infrastructure, are effective in delivering water to users even when overall availability is moderate.

In the energy sector, availability ranges from 30% to 60%, primarily driven by hydropower generation, which depends on water resources sensitive to climate variability, and by energy productivity. Access, in contrast, varies more widely, from 30% to 100%, indicating that although energy generation may be limited in some scenarios, energy delivery and consumption can be maintained or restricted depending on water pricing strategy and policy decisions. The gap between availability and access underscores that, despite constraints in physical generation capacity due to hydrological conditions, effective management and infrastructure can ensure that energy demand is partially or fully met.

For the food sector, availability ranges from 40% to 75%, reflecting constraints on production potential due to water limitations, climate variability, and changes in land use. Access to food, on the other hand, remains relatively high (70% to 100%), suggesting that while crops may face reduced water supply in some scenarios, effective irrigation management can maintain crop water delivery most of the time, reducing the duration and intensity of water stress.

Finally, for the ecosystem sector, availability ranges from 30% to 60%, reflecting limitations in environmental water flows due to climate variability and competing cross-sectoral demands. Access, measured as the percentage of habitat periods below ecological thresholds (%HPU of habitats < 0.5), exhibits extreme variability from 10% to 100%, indicating that even when some environmental water is available, ecosystems can still experience stress depending on water allocation, pricing, and policy decisions. The wide gap between availability and access highlights the sensitivity of ecosystems to human interventions and emphasizes the need for explicit environmental flow policies to balance ecological preservation with other sectoral demands.

Overall, the results indicate that access to resources does not always align with physical availability; however, effective management can help mitigate these gaps.

### Water Pricing as a Policy Lever for Sustainable Management of the WEFE Nexus

Pricing schemes influence allocation decisions, sectoral productivity, and ecosystem conservation, generating cascading effects across interconnected systems. Results indicate that dynamic water pricing produces moderate improvements across the WEFE nexus. Water access remains largely stable at around 82%, while availability increases slightly by approximately 2%, reflecting modest gains in efficiency without compromising basic needs. In the energy sector, access rises from roughly 72% to 81%, indicating better allocation for water-dependent energy activities, while availability remains around 51%, suggesting that improvements are driven by access rather than total extraction. For food production, access decreases by about 5%, and availability declines by roughly 7%, highlighting potential trade-offs where higher costs and reduced allocations could affect irrigation and crop yields. Ecosystems benefit from both improved access and availability, with increases of approximately 33% and 5%, respectively, demonstrating that dynamic pricing can support environmental flows and sustainability (Figure 18). Overall, dynamic pricing encourages more efficient water use and equitable allocation; however, careful attention is needed to protect agricultural productivity and support vulnerable users.

Uniform water pricing yields more pronounced effects across all sectors. Water access increases by about 5% to 87%, and availability rises nearly 16%, indicating that fixed pricing can strongly incentivize conservation and improved resource distribution. Energy sector access approaches near-universal levels at 100%, while availability remains around 49%, suggesting broader sectoral benefits without over-extraction. In agriculture, access improves by approximately 3%, but availability declines by around 13%, reflecting trade-offs where increased water costs may reduce irrigation potential and threaten food production if affordability mechanisms are not implemented. Ecosystems gain the most, with access nearly doubling from 51% to 94% and availability increasing by about 24%, highlighting the effectiveness of uniform pricing in securing environmental flows and ecosystem services. While uniform pricing maximizes efficiency and environmental protection, it requires careful design to strike a balance between equity and minimize negative impacts on food security.

From a policy perspective, dynamic pricing is more flexible and can balance sectoral needs while promoting efficiency, making it suitable in contexts where protecting food security and equity is a priority. Uniform pricing is effective for maximizing overall water savings and ecosystem sustainability, but it requires complementary measures, such as tiered tariffs, subsidies, or lifeline allocations, to prevent adverse impacts on vulnerable users and the agricultural sector. Adopting an integrated approach is essential, as it accounts for the interconnections among water, energy, food, and ecosystems, ensuring that pricing reforms promote efficiency, sustainability, and equity across the WEFE nexus.

| WEFE sectors | Baseline | | Dynamic WP | | Uniform WP | |
|---|---|---|---|---|---|---|
| | Availability | Access | Availability | Access | Availability | Access |
| **Water** | 47.9 | 82.1 | 49.0 | 82.4 | 55.7 | 86.6 |
| **Energy** | 50.7 | 71.9 | 50.9 | 80.6 | 49.2 | 99.8 |
| **Food** | 50.1 | 90.7 | 46.4 | 86.5 | 43.65 | 87.55 |
| **Ecosystem** | 42.0 | 51 | 44.3 | 67.7 | 51.9 | 94.3 |

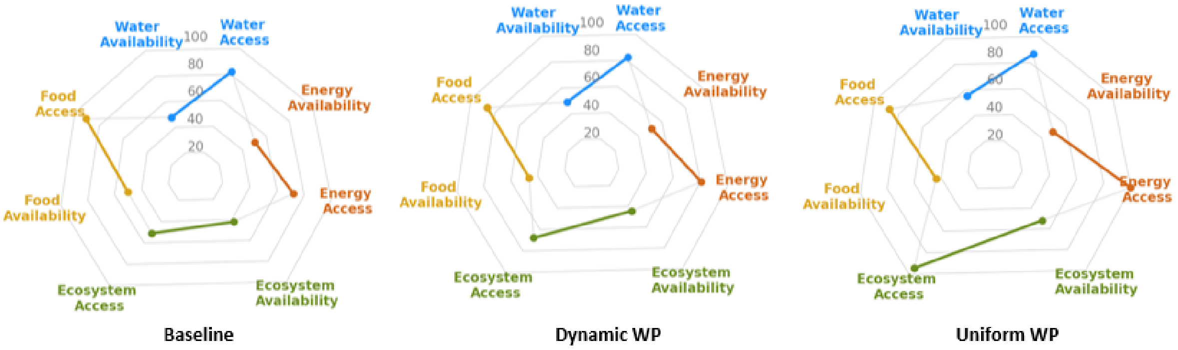


*Figure 18 Water Pricing Impact on the WEFE Nexus Performance for the SSP3-7.0*

### 3.1.2. Sectoral interactions and trade-offs

Figure 19 illustrates the influence of water pricing strategies on cross-sectoral interactions and trade-offs within the WEFE nexus for the historical and future periods under SSP scenarios, considering climate uncertainty. Both strategies reduce unsustainable water use while preserving economic benefits and improving system efficiency and water scarcity indicators. UWP substantially reduces average total water withdrawals, from 1,078 Mm³ in the Baseline to 793 Mm³ for the period 1979-2014, with reductions further amplified under future climate and socio-economic conditions, reaching 759 Mm³ under SSP5-8.5. This decrease generates energy gains and lowers system water scarcity to 0.44 for 1979-2014 and 0.56 for 2015-2050 under SSP5-8.5. However, the rigid pricing structure severely penalizes lower-productivity crops, reducing herbaceous production from 562 MT to 30 MT for 1979-2014 and potentially to 13 MT for 2015-2050 under SSP5-8.5, while moderately decreasing fruit tree and citrus production from 769 MT to 564 MT for the historical period and potentially to 499 MT under SSP5-8.5.

The DWP follows a similar trajectory to the UWP, but with a more balanced and less pronounced reduction in water withdrawals. Its flexible structure limits negative impacts on crop production, with herbaceous crops experiencing moderate reductions, from 562 MT in the baseline to 454 MT for 1979-2014 and to 406 MT for 2015-2050 under the SSP5-8.5 scenario, while fruit tree and citrus production remain largely stable under the SSP1-2.6 and show only slight reductions under SSP3-7.0 and SSP5-8.5. This approach preserves greater resource and economic efficiency while achieving moderate environmental and energy gains, providing a more balanced trade-off across the WEFE sectors compared to the more rigid UWP.

Climate-model uncertainty generates a wide range of hydrological responses that influence the magnitude of WEFE trade-offs under both pricing strategies (Figure 20). Across the five CMIP6 models, the multi-model median supports the general performance of the two policies, while the inter-model spread (396–1238 Mm³) indicates substantial variability in annual water withdrawals and sectoral impacts.

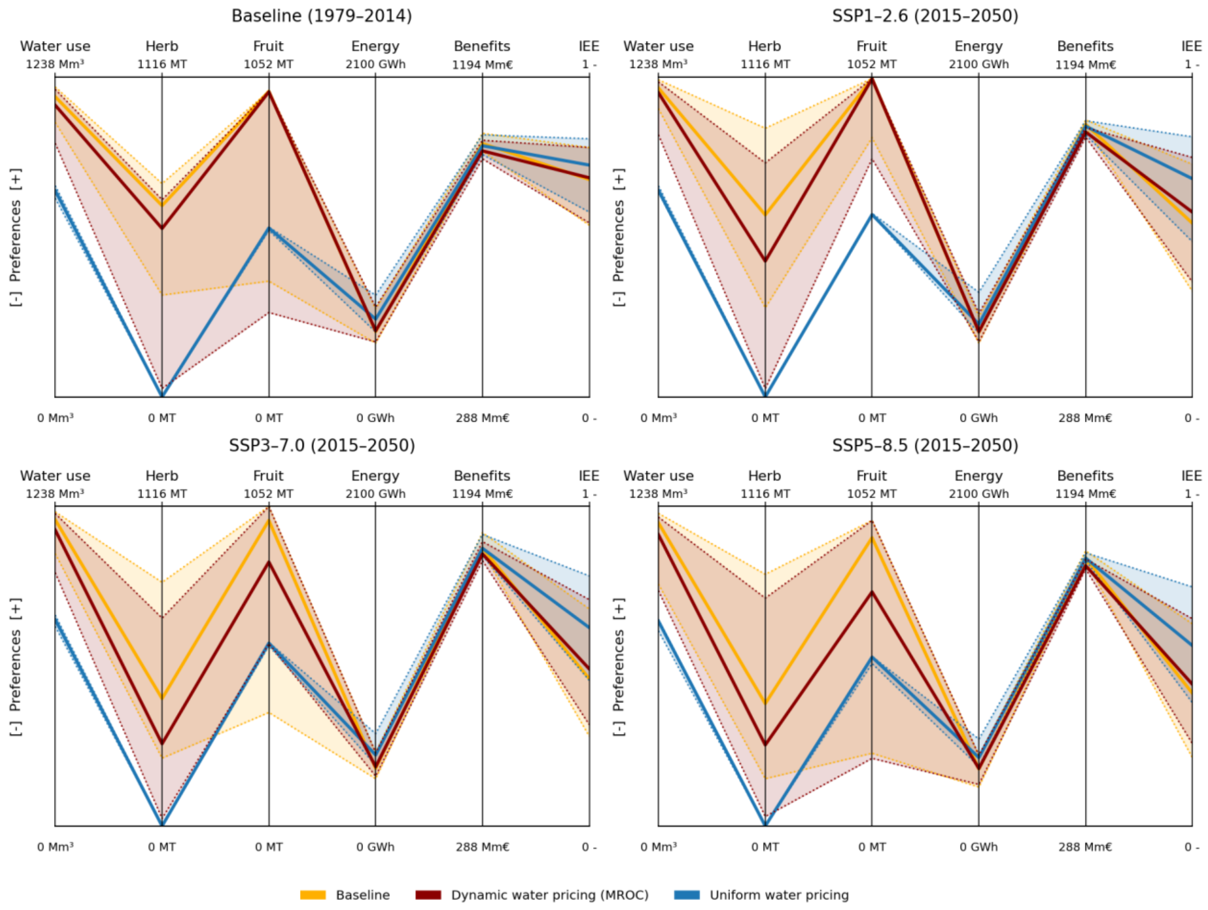


*Figure 19  Cross-sectoral trade-offs by water pricing strategies and SSP scenarios for historical and future periods, considering uncertainty across five climate models*

The performance indicators are represented on the X-axis, while the increasing preferences are on the Y-axis. The Y-axis scale is determined based on the optimization results for each performance indicator over the simulation periods and corresponds to the minimum (the bottom horizontal axis) and maximum (the top horizontal axis) values of that indicator across all scenarios. The average of the performance indicator over the simulation period is represented by a colored line for each water pricing strategy. The distribution of the performance indicator is characterized by colored areas associated with quantiles. These areas explain the response of performance indicators to changing water stress and socio-economic conditions under each policy. The area represents the range between the first quartile (25%), which corresponds to the lowest values of performance preference, and the 75th percentile.

Models projecting warmer and drier futures (IPSL-CM6A-LR and UKESM1-0-LL) produce the strongest reductions in withdrawals under UWP, with median decreases of 6% relative to the 1915–2050 baseline and a 10–90% exceedance range of 744–1229 Mm³, thereby intensifying system-level scarcity. These conditions result in more pronounced declines in herbaceous production at a 50% exceedance probability (around 162 MT under UWP and 180 MT under DWP), accompanied by moderate reductions in fruit-tree and citrus output, which become more severe under SSP5-8.5. Models with moderate climate sensitivity (GFDL-ESM4 and MRI-ESM2-0) cluster near the historical hydrological baseline, exhibiting smaller reductions in withdrawals of approximately 2% at a 50% exceedance probability and 0–1% at a 10–90% probability, with correspondingly moderate effects on agricultural production. MPI-ESM1-2-HR produces intermediate impacts relative to the other models. Despite this variability, the ranking of the two pricing policies remains stable: UWP consistently delivers the largest reductions in withdrawals and the greatest improvements in scarcity, reinforcing conservation, whereas DWP offers a more balanced cross-sectoral outcome. Even so, the inter-model percentile ranges show that the magnitude of these trade-offs depends strongly on each climate model's hydrological signal, highlighting the importance of assessing policy robustness under climate uncertainty.

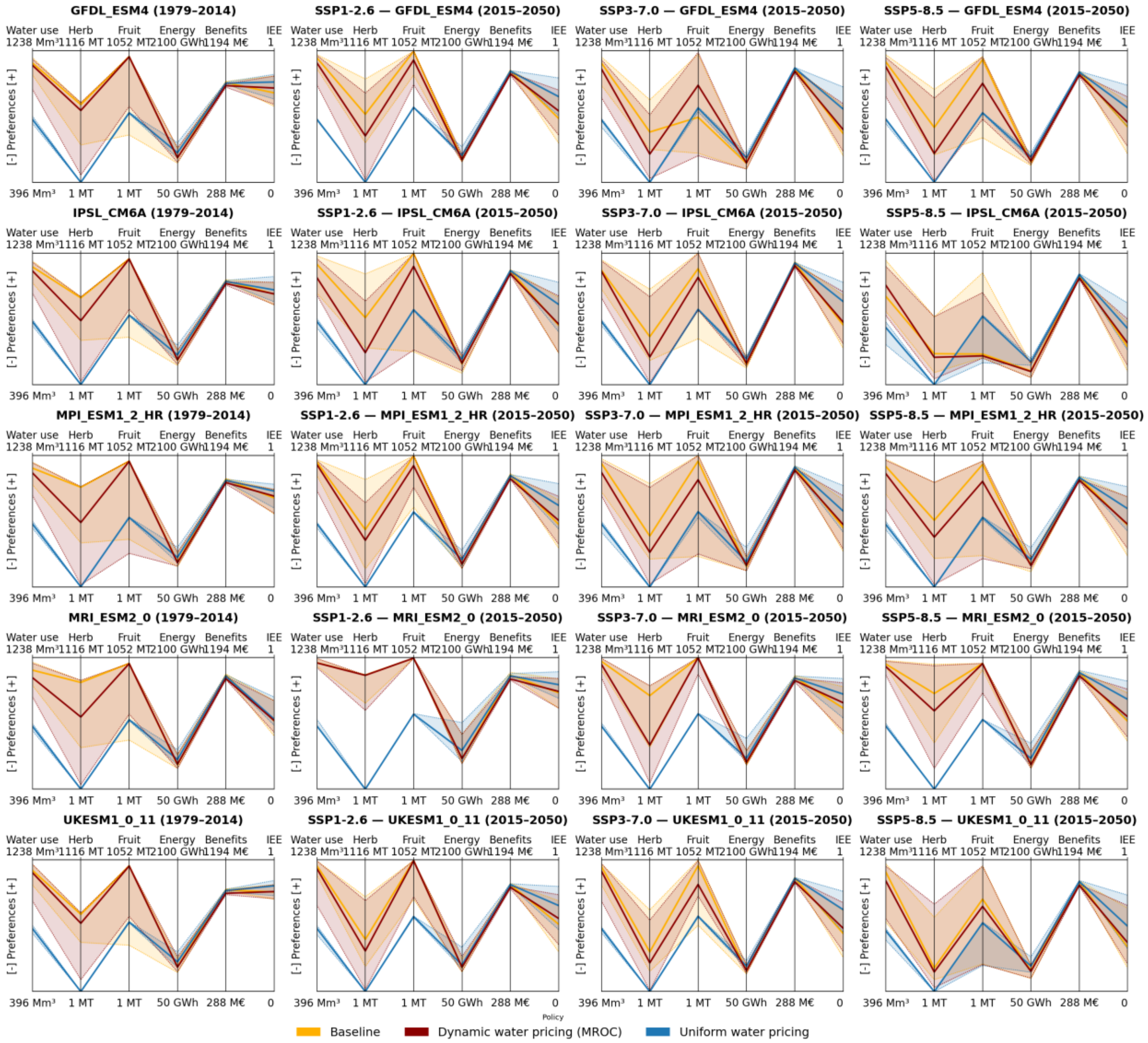
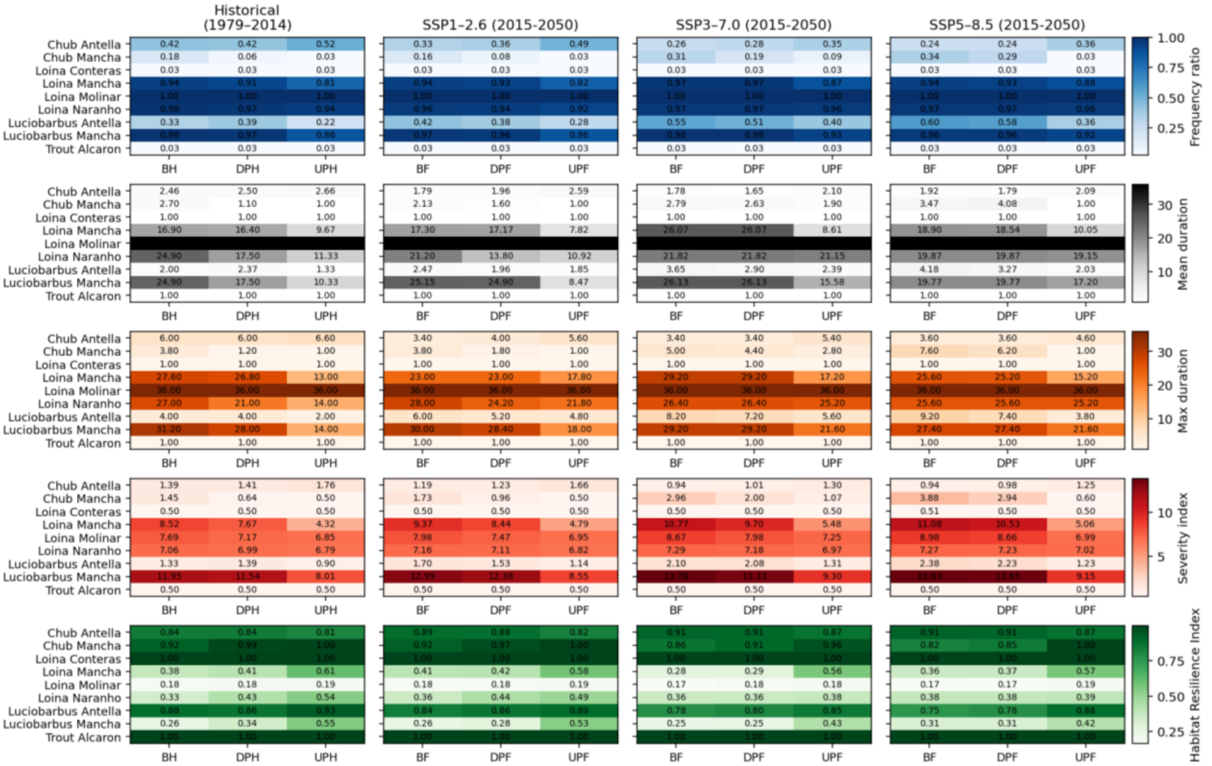


*Figure 20 Cross-sectoral trade-offs by water pricing strategies, SSP scenarios, and climate models for historical and future periods*

### 3.1.3. Water pricing as a driver of species resilience

Understanding how water pricing influences ecological resilience is crucial for assessing the broader implications of economic instruments in water management. Water pricing shapes patterns of water availability across space and time, which in turn affects the capacity of species and ecosystems to withstand periods of hydrological stress. To capture these dynamics, we assess species resilience through four complementary indicators: the frequency ratio measures how often species experience stressful hydrological conditions under different pricing scenarios. The mean duration captures the average length of these stressful episodes, while the maximum duration indicates the most prolonged event that a species must endure. Finally, the severity index determines the magnitude of the hydrological stress experienced by each species, offering a synthetic view of the cumulative stress imposed on ecological communities.



| Frequency ratio | Historical (1979–2014) | | | SSP1–2.6 (2015-2050) | | | SSP3–7.0 (2015-2050) | | | SSP5–8.5 (2015-2050) | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | BH | DPH | UPH | BF | DPF | UPF | BF | DPF | UPF | BF | DPF | UPF |
| Chub Antella | 0.42 | 0.42 | 0.52 | 0.33 | 0.36 | 0.49 | 0.26 | 0.28 | 0.35 | 0.24 | 0.24 | 0.36 |
| Chub Mancha | 0.18 | 0.06 | 0.03 | 0.16 | 0.08 | 0.03 | 0.31 | 0.19 | 0.09 | 0.34 | 0.29 | 0.03 |
| Loina Conteras | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 |
| Loina Mancha | 0.94 | 0.91 | 0.81 | 0.94 | 0.93 | 0.82 | 0.97 | 0.97 | 0.87 | 0.94 | 0.93 | 0.88 |
| Loina Molinar | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| Loina Naranho | 0.98 | 0.97 | 0.94 | 0.96 | 0.94 | 0.92 | 0.97 | 0.97 | 0.96 | 0.97 | 0.97 | 0.96 |
| Luciobarbus Antella | 0.33 | 0.39 | 0.22 | 0.42 | 0.38 | 0.28 | 0.55 | 0.51 | 0.40 | 0.60 | 0.58 | 0.36 |
| Luciobarbus Mancha | 0.98 | 0.97 | 0.86 | 0.97 | 0.96 | 0.86 | 0.98 | 0.98 | 0.93 | 0.96 | 0.96 | 0.92 |
| Trout Alcaron | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 | 0.03 |

| Mean duration | BH | DPH | UPH | BF | DPF | UPF | BF | DPF | UPF | BF | DPF | UPF |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Chub Antella | 2.46 | 2.50 | 2.66 | 1.79 | 1.96 | 2.59 | 1.78 | 1.65 | 2.10 | 1.92 | 1.79 | 2.09 |
| Chub Mancha | 2.70 | 1.10 | 1.00 | 2.13 | 1.60 | 1.00 | 2.79 | 2.63 | 1.90 | 3.47 | 4.08 | 1.00 |
| Loina Conteras | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| Loina Mancha | 16.90 | 16.40 | 9.67 | 17.30 | 17.17 | 7.82 | 26.07 | 26.07 | 8.61 | 18.90 | 18.54 | 10.05 |
| Loina Molinar | | | | | | | | | | | | |
| Loina Naranho | 24.90 | 17.50 | 11.33 | 21.20 | 13.80 | 10.92 | 21.82 | 21.82 | 21.15 | 19.87 | 19.87 | 19.15 |
| Luciobarbus Antella | 2.00 | 2.37 | 1.33 | 2.47 | 1.96 | 1.85 | 3.65 | 2.90 | 2.39 | 4.18 | 3.27 | 2.03 |
| Luciobarbus Mancha | 24.90 | 17.50 | 10.33 | 25.15 | 24.90 | 8.47 | 26.13 | 26.13 | 15.58 | 19.77 | 19.77 | 17.20 |
| Trout Alcaron | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |

| Max duration | BH | DPH | UPH | BF | DPF | UPF | BF | DPF | UPF | BF | DPF | UPF |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Chub Antella | 6.00 | 6.00 | 6.60 | 3.40 | 4.00 | 5.60 | 3.40 | 3.40 | 5.40 | 3.60 | 3.60 | 4.60 |
| Chub Mancha | 3.80 | 1.20 | 1.00 | 3.80 | 1.80 | 1.00 | 5.00 | 4.40 | 2.80 | 7.60 | 6.20 | 1.00 |
| Loina Conteras | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| Loina Mancha | 27.60 | 26.80 | 13.00 | 23.00 | 23.00 | 17.80 | 29.20 | 29.20 | 17.20 | 25.60 | 25.20 | 15.20 |
| Loina Molinar | 36.00 | 36.00 | 36.00 | 36.00 | 36.00 | 36.00 | 36.00 | 36.00 | 36.00 | 36.00 | 36.00 | 36.00 |
| Loina Naranho | 27.00 | 21.00 | 14.00 | 28.00 | 24.20 | 21.80 | 26.40 | 26.40 | 25.20 | 25.60 | 25.60 | 25.20 |
| Luciobarbus Antella | 4.00 | 4.00 | 2.00 | 6.00 | 5.20 | 4.80 | 8.20 | 7.20 | 5.60 | 9.20 | 7.40 | 3.80 |
| Luciobarbus Mancha | 31.20 | 28.00 | 14.00 | 30.00 | 28.40 | 18.00 | 29.20 | 29.20 | 21.60 | 27.40 | 27.40 | 21.60 |
| Trout Alcaron | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |

| Severity index | BH | DPH | UPH | BF | DPF | UPF | BF | DPF | UPF | BF | DPF | UPF |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Chub Antella | 1.39 | 1.41 | 1.76 | 1.19 | 1.23 | 1.66 | 0.94 | 1.01 | 1.30 | 0.94 | 0.98 | 1.25 |
| Chub Mancha | 1.45 | 0.64 | 0.50 | 1.73 | 0.96 | 0.50 | 2.96 | 2.00 | 1.07 | 3.88 | 2.94 | 0.60 |
| Loina Conteras | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.51 | 0.50 | 0.50 |
| Loina Mancha | 8.52 | 7.67 | 4.32 | 9.37 | 8.44 | 4.79 | 10.77 | 9.70 | 5.48 | 11.08 | 10.53 | 5.06 |
| Loina Molinar | 7.69 | 7.17 | 6.85 | 7.98 | 7.47 | 6.95 | 8.67 | 7.98 | 7.25 | 8.98 | 8.66 | 6.99 |
| Loina Naranho | 7.06 | 6.99 | 6.79 | 7.16 | 7.11 | 6.82 | 7.29 | 7.18 | 6.97 | 7.27 | 7.23 | 7.02 |
| Luciobarbus Antella | 1.33 | 1.39 | 0.90 | 1.70 | 1.53 | 1.14 | 2.10 | 2.08 | 1.31 | 2.38 | 2.23 | 1.23 |
| Luciobarbus Mancha | 11.95 | 11.54 | 8.01 | 12.99 | 12.38 | 8.55 | 13.78 | 13.33 | 9.30 | 13.83 | 13.65 | 9.15 |
| Trout Alcaron | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 | 0.50 |

| Habitat Resilience Index | BH | DPH | UPH | BF | DPF | UPF | BF | DPF | UPF | BF | DPF | UPF |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Chub Antella | 0.84 | 0.84 | 0.81 | 0.89 | 0.88 | 0.82 | 0.91 | 0.91 | 0.87 | 0.91 | 0.91 | 0.87 |
| Chub Mancha | 0.92 | 0.99 | 1.00 | 0.92 | 0.97 | 1.00 | 0.86 | 0.91 | 0.96 | 0.82 | 0.85 | 1.00 |
| Loina Conteras | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| Loina Mancha | 0.38 | 0.41 | 0.61 | 0.41 | 0.42 | 0.58 | 0.28 | 0.29 | 0.56 | 0.36 | 0.37 | 0.57 |
| Loina Molinar | 0.18 | 0.18 | 0.19 | 0.18 | 0.18 | 0.19 | 0.17 | 0.18 | 0.18 | 0.17 | 0.17 | 0.19 |
| Loina Naranho | 0.33 | 0.43 | 0.54 | 0.36 | 0.44 | 0.49 | 0.36 | 0.36 | 0.38 | 0.38 | 0.38 | 0.39 |
| Luciobarbus Antella | 0.88 | 0.86 | 0.93 | 0.84 | 0.86 | 0.89 | 0.78 | 0.80 | 0.85 | 0.75 | 0.78 | 0.88 |
| Luciobarbus Mancha | 0.26 | 0.34 | 0.55 | 0.26 | 0.28 | 0.53 | 0.25 | 0.25 | 0.43 | 0.31 | 0.31 | 0.42 |
| Trout Alcaron | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |

*Figure 21  Water pricing impacts on aquatic species and habitats' resilience under SSP scenarios*

BH: Baseline for historical period; BF: Baseline for future period; DPH: Dynamic water pricing for historical period; DPF: Dynamic water pricing for future period; UPH: Uniform water pricing for historical period; and UPF: Uniform water pricing for future period.

Results indicate that both water pricing strategies reduce the frequency ratio, the mean, and the maximum duration of habitat below the ecological threshold, as well as the severity index, thereby strengthening the resilience of most species in the river basin. The most resilient species are Chub, Loina contreras, Luciobarbus antella, and Trout, consistently maintaining a high percentage of Habitats Provisioning Units (HPU) close to 1.00 across historical and future scenarios, with marginal gains under both pricing strategies. Sensitive species, such as Loina mancha, Loina molinar, and Loina naranjo, exhibit more pronounced improvements in %HPU, with reduced severity and increased resilience under both Dynamic and Uniform Water Pricing, reflecting their high sensitivity to water allocation management (Figure 21). These findings demonstrate that water pricing not only alleviates the impacts of water scarcity but also strengthens ecosystem stability, providing targeted benefits for vulnerable species while sustaining the persistence of naturally resilient populations.

### 3.1.4. Key insights and policy recommendations

Sustainable water management is critical for ensuring resilience across the WEFE nexus and for achieving the Sustainable Development Goals (SDGs). Addressing water scarcity, ecosystem degradation, and competing sectoral demands requires integrated strategies that balance efficiency, equity, and environmental protection. Economic instruments, such as water pricing, represent a key tool for guiding allocation, incentivizing sustainable practices, and strengthening system resilience under current and future socio-economic and climatic conditions.

Water pricing strategies have demonstrated the potential to improve system performance and resource efficiency. UWP delivers the strongest environmental gains, significantly improving ecosystem, energy, and water indicators, while markedly reducing unsustainable water withdrawals.

This reduction mitigates water scarcity and enhances ecosystem stability, providing clear benefits for sensitive species. However, the rigidity of UWP imposes substantial trade-offs in agricultural production, particularly for low-productivity herbaceous crops and, to a lesser extent, perennial crops, indicating potential risks to food security. DWP, by contrast, achieves more moderate environmental improvements while maintaining higher agricultural outputs, providing a more balanced distribution of benefits across sectors and supporting adaptive responses under changing conditions.

These results underscore the importance of designing water pricing schemes that are transparent, flexible, and sensitive to sectoral trade-offs. Policies integrating economic incentives with environmental safeguards and productivity considerations can optimize resource use, enhance biodiversity, strengthen ecosystem services, and support sustainable water management.

## 3.2. The Upper Main River Basin

### 3.2.1. Water Quantity

Water quantity represents a central dimension of the WEFE Nexus, as it directly governs the allocation of water between agricultural production, environmental flow requirements, and competing societal demands. In the UMRB, climate-induced changes in precipitation patterns, evapotranspiration rates, and groundwater could alter the magnitude and seasonal distribution of available water resources.

Figure 22 illustrates the long-term average water balance of the Upper Main River Basin under the baseline climate scenario (RCP26). Precipitation constitutes the primary input, with a large share partitioned into plant and canopy evapotranspiration and soil evapotranspiration. A substantial portion of incoming precipitation contributes to groundwater natural recharge, which in turn sustains baseflow and Environmental Flow Requirements. Green water availability plays a key role in meeting the minimum cereal water requirement, highlighting the strong dependence of cereal production on soil moisture rather than on extracted blue water. Blue water withdrawals originating mainly from groundwater extraction are relatively limited and are mainly directed toward public and private supply systems that serve household and industrial demands. Overall, the baseline scenario illustrates a relatively balanced system in which ecological flow requirements and agricultural needs are largely met without excessive pressure on extractive water sources.

Figure 23 represents the basin’s water balance under a more extreme climate scenario (RCP85). In this scenario, total precipitation and the evapotranspiration components are reduced. Green water availability remains significant but is less sufficient to meet the minimum cereal water requirement, leading to a larger gap and the appearance of remaining green water constraints. Blue water extractions (surface and groundwater extraction) for cereals increase, generating competition with other uses.

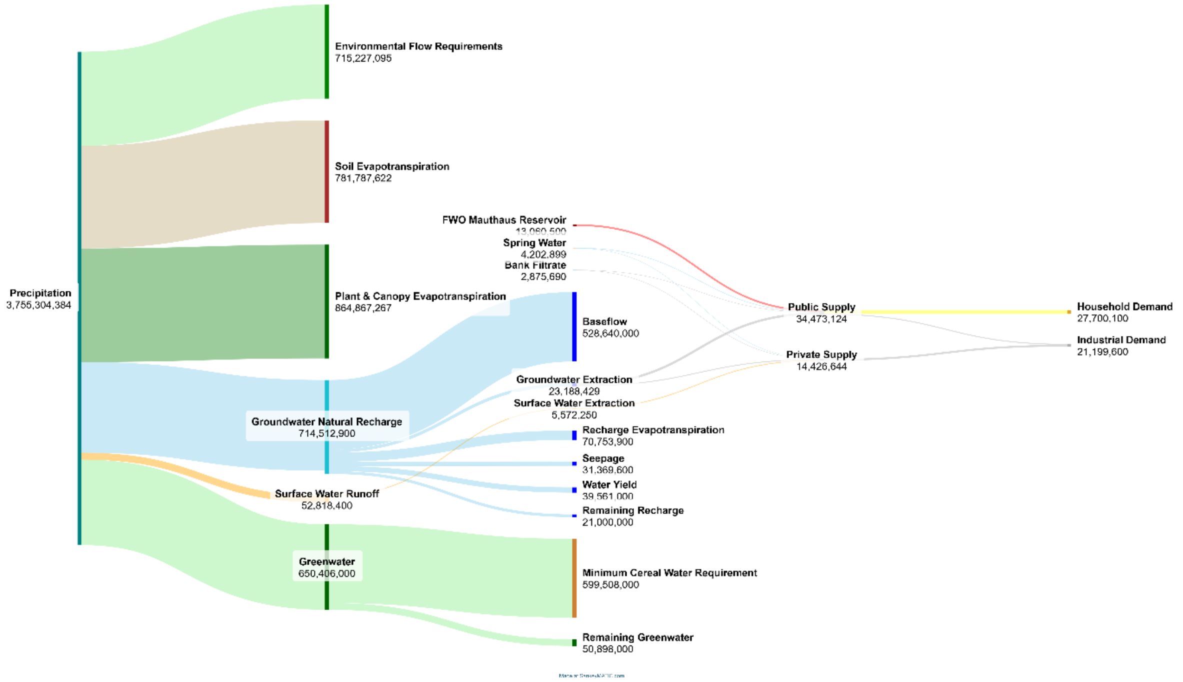


*Figure 22 Sankey Diagram with the inflow-outflow of the hydrological system in the baseline case of RCP26 Low MCWR, with water volume in m³. Source: Twohig (2023).*

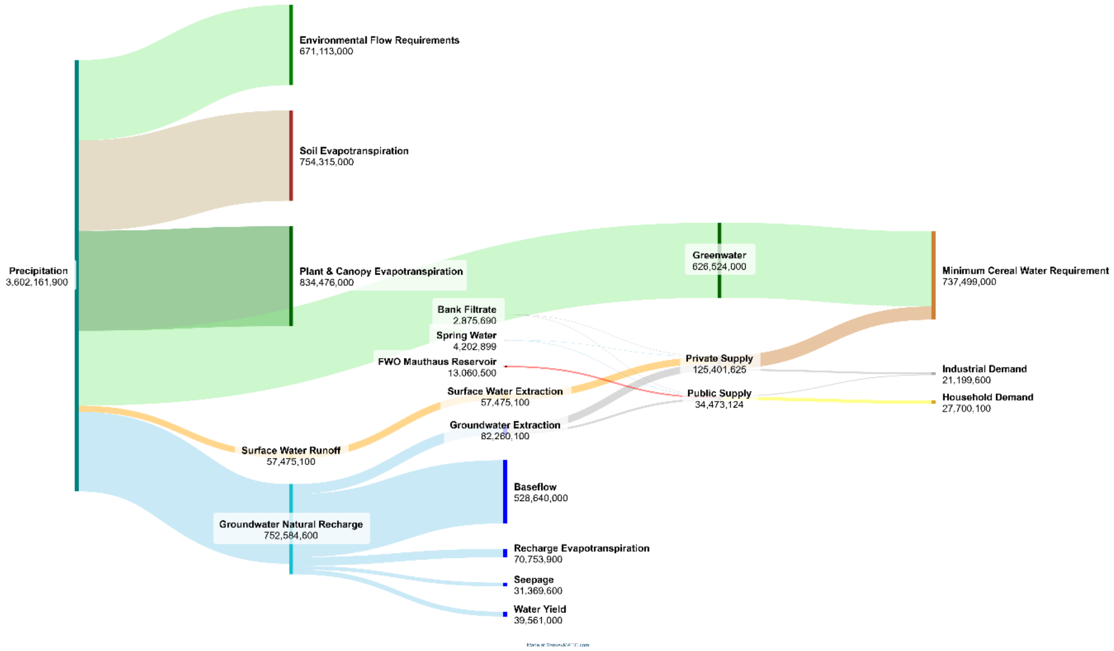


*Figure 23 Sankey Diagram with the inflow-outflow of the hydrological system in the extreme case of RCP85 High MCWR, with water volume in m³. Source: Twohig (2023).*

Figure 24 presents simulated agricultural water demand from 2000 to 2050 under three Minimum Cereal Water Requirement (MCWR) scenarios: low, medium, and high. Each panel displays annual demand trajectories alongside a smoothed trend line, illustrating both interannual variability and longer-term patterns. Under the **low MCWR scenario**, agricultural demand remains minimal throughout the period, fluctuating at very low levels with occasional short-lived peaks, indicating limited irrigation needs when crop water requirements are modest or largely met by green water. The **medium MCWR scenario** exhibits substantially higher demand levels, with pronounced variability and a modest upward trend toward mid-century, reflecting increasing pressure on water resources as crop water requirements rise. In the **high MCWR scenario**, agricultural demand is consistently elevated, exceeding 100 million m³ in most years and showing frequent peaks and sharper oscillations, indicative of heightened sensitivity to climatic variability and more frequent reliance on supplemental water inputs.

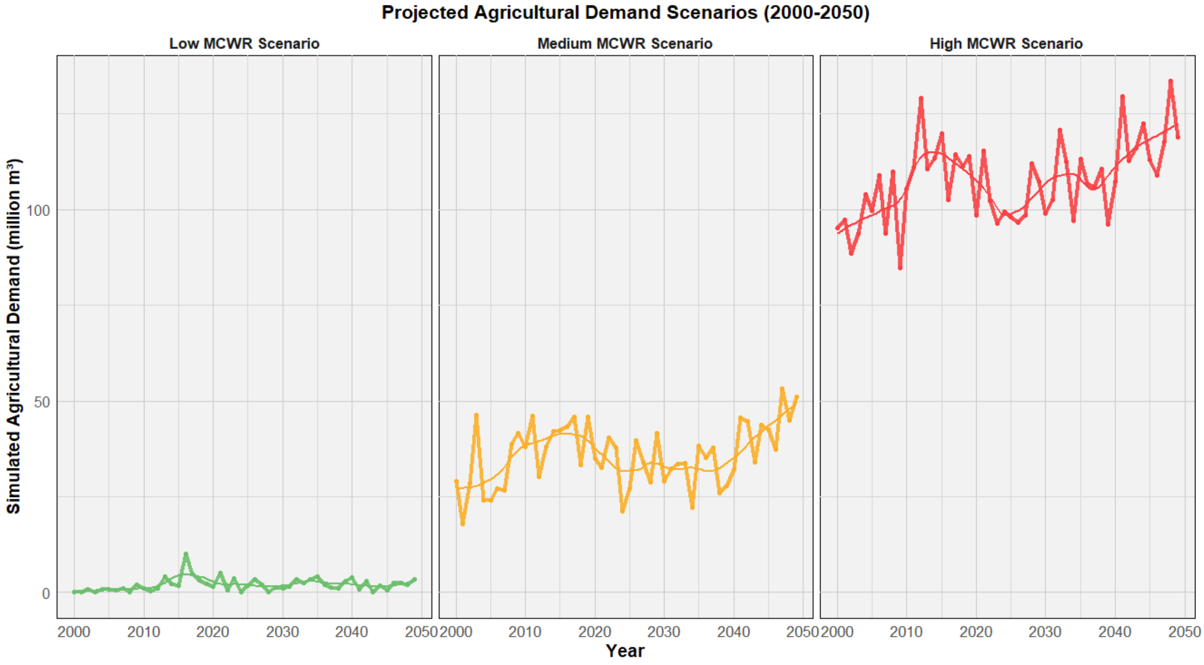


*Figure 24 Projected Agricultural Water Demand Scenarios. Source: Twohig (2023).*

### 3.2.2. Water quality

In intensively cultivated regions such as the UMRB, nutrient runoff and leaching pose significant risks to both surface and groundwater resources. These pressures are further exacerbated by climate change, which can intensify nutrient mobilization through altered hydrological pathways and higher irrigation demand. The water quality results presented in this section explore how different fertilizer-use trajectories and agricultural water requirements affect nitrate concentrations over time, and how declining water quality feeds back into agricultural productivity. By capturing these interactions, the analysis underscores the importance of integrated nutrient and water management strategies to safeguard ecosystem health and maintain sustainable food production.

Figure 25 illustrates projected water quality dynamics, measured as nitrate concentration (mg/L), under different combinations of climate conditions, minimum cereal water requirements (MCWR), and fertilizer-use pathways from 2000 to 2050. Results are shown for two fertilizer scenarios: decreasing fertilizer use and stable fertilizer use, with each panel displaying low, medium, and high MCWR trajectories. Across both panels, nitrate concentrations initially rise in the early 2000s before diverging over time depending on the imposed scenarios. Under decreasing fertilizer use, nitrate levels generally stabilize or decline, particularly for the low and medium MCWR cases, while the high MCWR scenario maintains higher concentrations and shows a slight upward tendency toward mid-century. In contrast, the stable fertilizer-use scenario exhibits systematically higher nitrate concentrations across all MCWR levels, with a pronounced upward trend over time, especially under high and medium MCWR conditions.

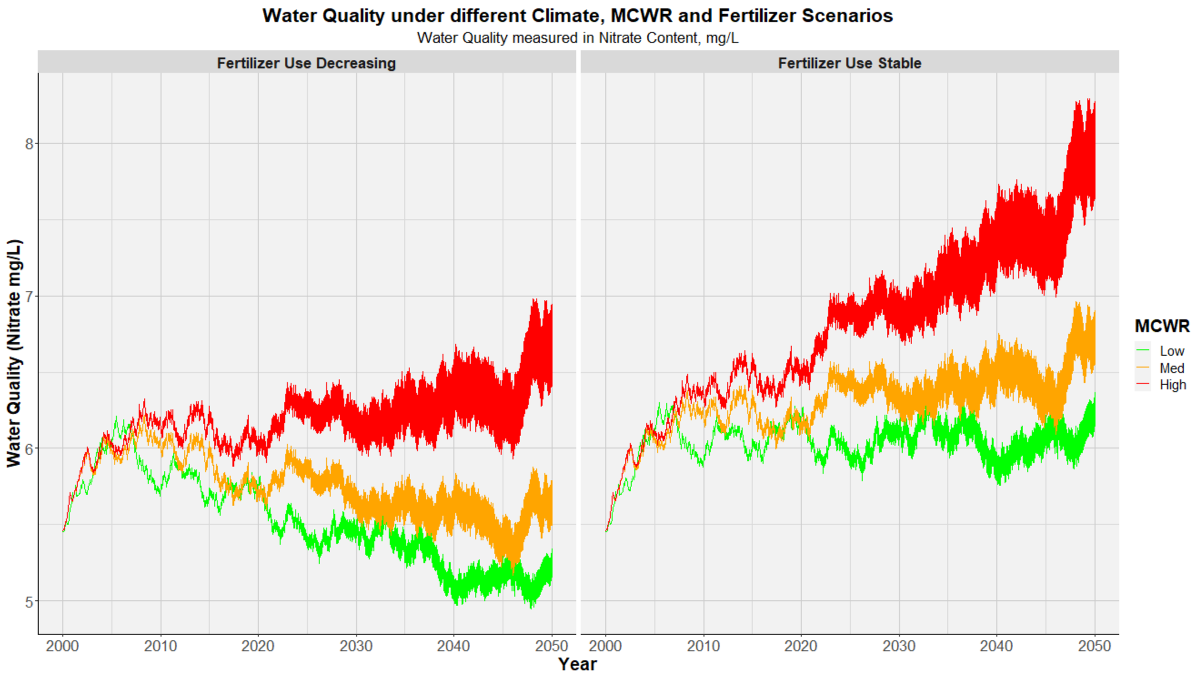


*Figure 25 Projected water quality under different MCWR & Fertilizer scenarios, 2000 – 2050. Source: Twohig (2023).*

### 3.2.3. Key insights and policy recommendations

The results presented in this study highlight the increasing pressure on water resources in the Upper Main River Basin arising from the combined effects of climate change, agricultural intensification, and water quality degradation. The integrated analysis of water quantity, water quality, and economic scarcity reveals that future challenges cannot be addressed through sectoral policies alone. Instead, the interactions identified within the WEFE Nexus framework point to a need for coordinated and adaptive governance mechanisms that simultaneously account for hydrological constraints, agricultural productivity, ecosystem protection, and economic efficiency.

### 3.2.4. Water pricing initial results

Economic instruments provide a critical mechanism for translating physical water scarcity into decision-relevant signals for resource users and policymakers. Shadow pricing of water captures the marginal economic value of an additional unit of water, reflecting scarcity conditions, competition among sectors, and constraints on availability. The results in this section assess the evolution of water shadow prices under alternative crop water demand scenarios.

Figure 26 shows the evolution of the average shadow price of water from 2000 to 2050 under different Minimum Cereal Water Requirement (MCWR) scenarios. Three trajectories are presented, corresponding to low, medium, and high MCWR levels. Across all scenarios, the shadow price of water increases steadily over time, indicating a growing marginal value of water as availability becomes more constrained and demand intensifies. Under the low MCWR scenario, water prices rise gradually, remaining consistently lower than in the other cases. The medium MCWR scenario exhibits moderately higher shadow prices and slightly greater interannual variability. The high MCWR scenario displays the steepest increase and the highest values throughout the period, with prices surpassing €0.30 per cubic meter toward the end of the projection horizon.

Figure 27 illustrates the projected effect of poor water quality on cereal production under different climate scenarios from 2000 to 2050. Four trajectories are shown, comparing production outcomes under RCP2.6 and RCP8.5, both with and without accounting for water quality index (WQI). Across all scenarios, cereal production exhibits a gradual downward trend over time, reflecting increasing climatic and environmental pressures. Under RCP2.6, production levels remain comparatively higher, with the inclusion of water quality index leading to a modest but persistent reduction relative to the baseline case. In contrast, the RCP8.5 scenario shows substantially lower production levels, with pronounced interannual variability and a steeper decline over the projection period. Accounting for poor water quality further exacerbates these reductions, resulting in significantly lower cereal output by mid-century.

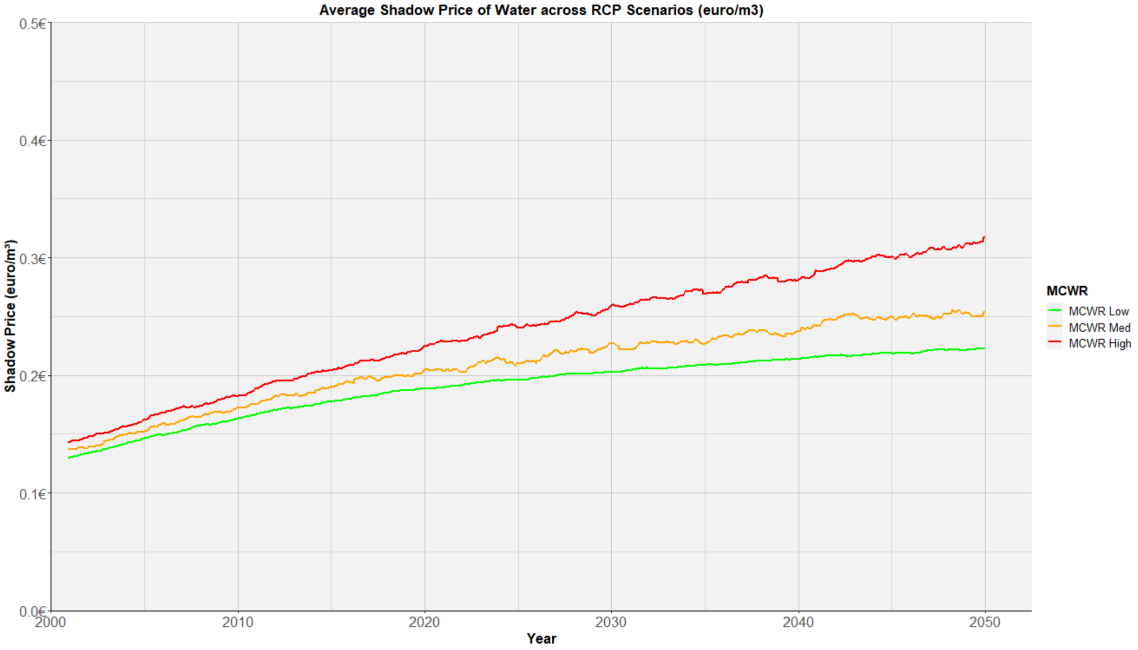


*Figure 26 Projected shadow price of water under different MCWR & Fertilizer scenarios, 2000 – 2050. Source: Twohig (2023).*

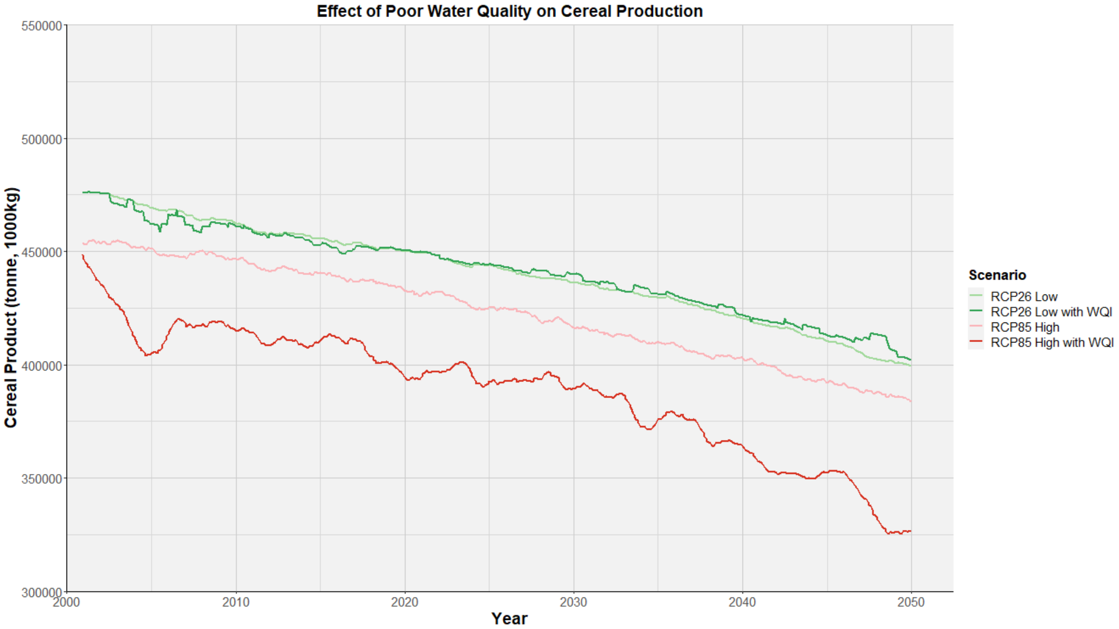


*Figure 27 Projected cereal production with and without accounting for the impact of water quality, 2000 – 2050. Source: Twohig (2023).*

## 3.3. Malta

### 3.3.1. Water, energy, and economic results

In this simulation, a time period of 50 years is considered (T = 50). The development of the three state variables for energy, water, and economic capital is presented in Figure 28 below.

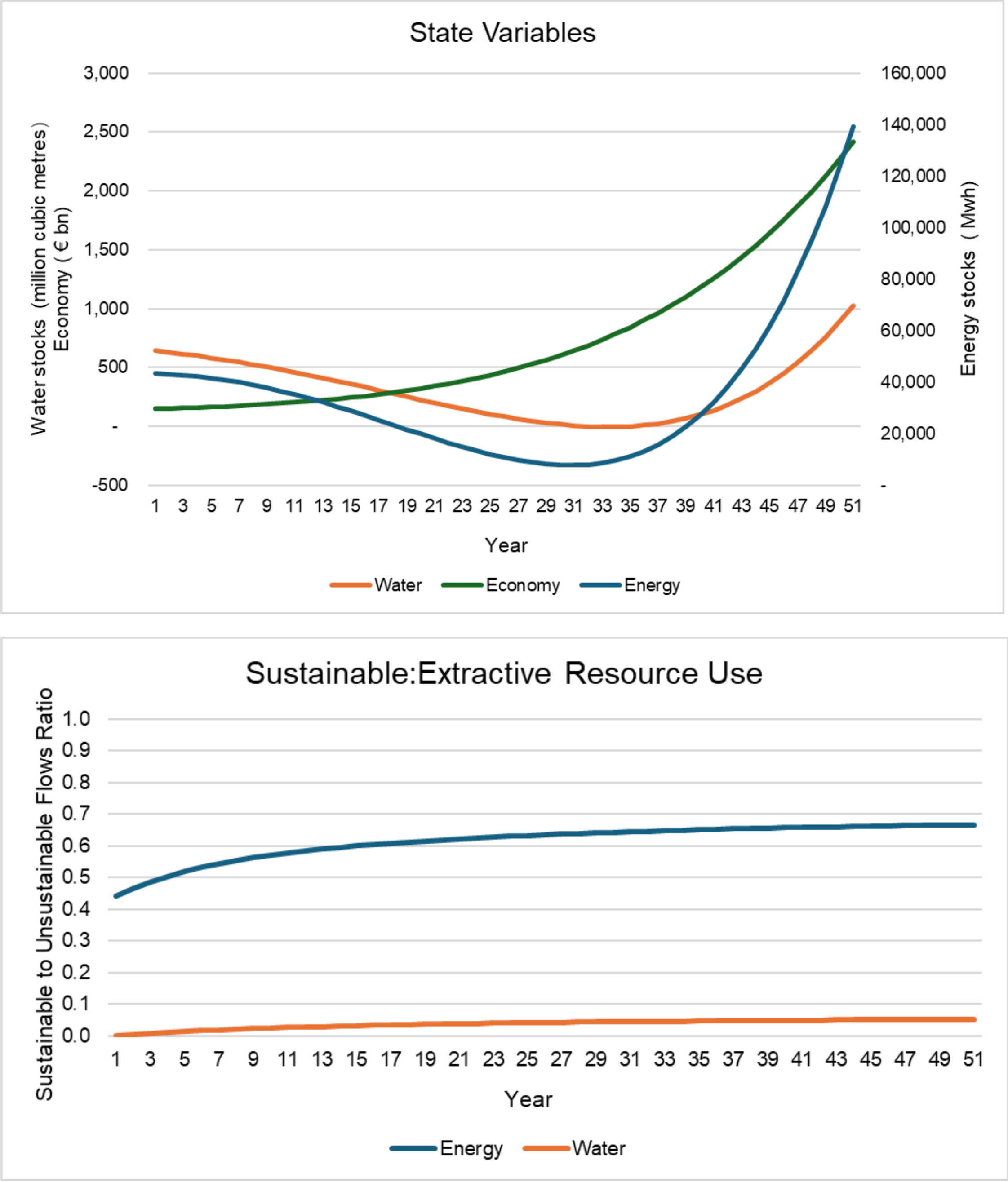


*Figure 28 Calibrated Results for the first scenarios.*

The results indicate a future scenario for Malta where economic capital and output continue to grow by 5.71% and 5.41% annually. This is a validation of the use of the production function approach utilised but can still require further elaboration possibly through the introduction of a population variable which would convert all values into per capita terms, and act as a driver for both economic output as well as demand for resources (which at the present stage is treated as an exogenous variable). It is thus the case that an implicit assumption in these results is the continued growth in population as observed in the past ten years.

Despite these positive economic outcomes, and to an extent also because of them, the stock of energy and water resources is being depleted up to the 35th year of the simulation. In the case of water, it reaches zero value. The average annual growth rate of energy and water stock is 2.36% and 0.94%, respectively. The later recovery would take place as the economy grows sufficiently to organically generate sufficient resources for the investment in the stock of these resources, and as it strives to meet the constraints for sustainable resource use in the last decade of the simulation. Despite this, the scenario is clearly unsustainable and calls for decisive action to increase investment immediately to ensure a sustainable path over the coming years, especially the more immediate ones.

Sustainability can be restored by, among others, increasing the share of output invested annually in energy and water resources. A scenario here is considered where such investment is increased by 15% in the case of energy and 30% in the case of water. The results are presented in Figure 29.

These indicate a much lower rate of depletion in the stocks of resources in earlier years and a better rate of recovery in later years. While economic capital and output do not suffer because of this as they still grow by 5.71% and 5.41% respectively, it can be expected that the level of consumption would be affected.

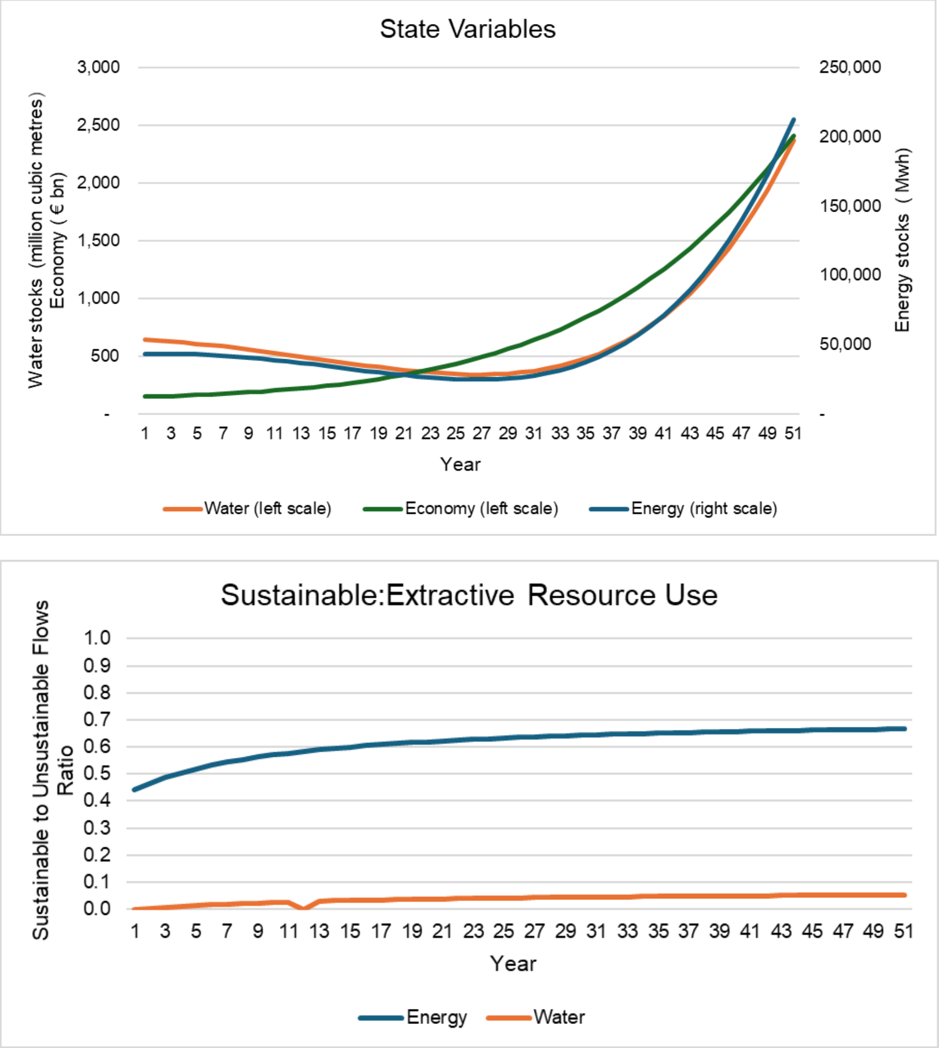


*Figure 29 Calibrated Results for the second scenario*

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### 3.3.2. Next steps for the modelling framework

This report details the present results of conceptual research undertaken to explore the relationships between economic variables, energy and water resources from the perspective of dynamic sustainability. It details the development of the conceptual model in algebraic form, showing the dynamic relations and constraints. It presents an application to the Maltese economy using calibrated values.

These results show that economic growth can coexist with rapid early depletion of water and energy stocks. Therefore, without increased investment, Malta risks unsustainable trajectories particularly in water within 3 decades. This implies that moderate increases in reinvestment are needed to improve sustainability while maintaining growth.

The intention is to continue enhancing the model, which currently focuses only on energy and water. To fully represent the WEFE Nexus, future development will integrate local food production by including agriculture and ecosystem components. There is also room to gather additional relevant data, refine model coefficients and expand the model’s relationships to achieve more detailed and accurate results.

## 3.4. EEIO Model of the Slovak Economy

### 3.4.1. Economic and Water-Use Profile of Slovakia (Baseline)

The baseline scenario represents the Slovak economy in 2021, capturing sectoral outputs, value added, employment, and water consumption as per main sectoral aggregates, reflecting both production and supply-chain processes.

In the structure of the Slovak economy, services and manufacturing sectors are dominant in terms of economic output and employment, while agriculture and the food industry play a relatively minor role economically but exert significant pressure on water resources (Table 10, Figure 30). Agriculture generates only 5.3 billion USD in output (around 2% of the total) but consumes 123 million m³ of water, representing almost 29% of total water use. Similarly, the food industry, with an output of just $5.1 billion, uses 69 million m³ of water. These figures highlight a structural imbalance, where sectors with low economic contributions (particularly agriculture) are disproportionately water-intensive, underscoring the importance of targeted policies to improve water efficiency and manage demand in the agricultural and food production sectors.

*Table 10 Baseline – Slovak economy in 2021*

| | **Total economy** | **Agriculture** | **Food industry** | **Manufacturing** | **Services** |
|---|---|---|---|---|---|
| Output (mil USD) | 270 007 | 5 339 | 5 108 | 112 462 | 147 098 |
| Value added (mil USD) | 105 156 | 2 059 | 1 228 | 25 712 | 76 157 |
| Employment (th ppl) | 2 710 | 110 | 35 | 581 | 1 985 |
| Water consumption (mil m³) | 429 | 122 | 68 | 152 | 87 |

Funded by the European Union. Views and opinions expressed are however those of the author(s) only and do not necessarily reflect those of the European Union or REA. Neither the European Union nor the granting authority can be held responsible for them.

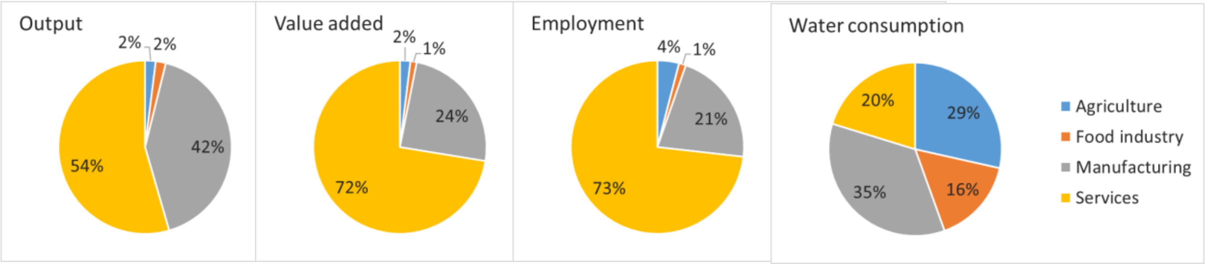

*Figure 30 Baseline – Slovak economy in 2021*

The water footprint analysis reveals distinct sectoral contributions to the production of different categories of goods and services (Figure 31). Agri-food products account for the largest share of water use, at 179 million m³, and this category of goods is almost entirely associated with water used in agriculture (109 million m³) and the food industry (66 million m³). In contrast, industrial goods require 154 million m³ of water, with about 88% of the total (136 million m³) originating from the manufacturing sector, while agriculture and services contribute only marginally. Services exhibit a total water footprint of 96 million m³, most of which stems from water use within the service sector itself (76%) and a smaller share stems from the manufacturing sector (14%), reflecting operational and infrastructure-related requirements rather than extensive supply-chain inputs. The findings, therefore, reflect the distinct ways in which food, industrial goods, and service provision create pressure on water consumption in agriculture, manufacturing, and the service sector. Accordingly, policies aimed at improving water sustainability should be informed by the distinct pathways through which final demand generates water consumption pressures across sectors.

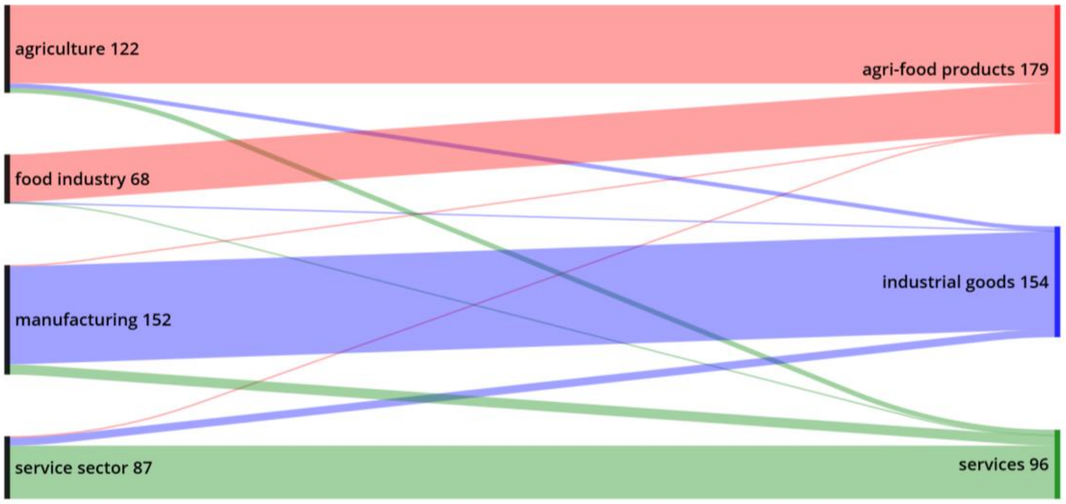

*Figure 31 Water footprint of goods produced in the Slovak economy in 2021*

### 3.4.2. Changes in Food Demand Composition (Scenario 1)

Under Scenario 1, which assumes a dietary shift toward more plant-based foods (final demand for cereals, fruits, vegetables +20%, meat -5%), total water consumption in the total economy increases by 0.99% compared to the baseline. The most pronounced effect is observed in agriculture, where water use increases by 4.19%, reflecting the additional water requirements associated with the expansion of crop production. By contrast, the food industry experiences a 1.37% decrease in water consumption due to reduced meat processing. This indicates that although plant-based foods create pressure on water resources in primary production, they are less water-intensive than animal-based products in the processing stage. Changes in manufacturing and services are negligible (0.02% and

0.04%, respectively), confirming that the water impacts of this dietary shift are concentrated in agriculture and, to a lesser extent, food processing. Overall, while lower demand for meat reduces water use in processing, the increase in crop production leads to a net rise in total water consumption, as many plant-based foods, particularly fruits and vegetables, remain water-intensive in the primary production stage.

*Table 11 Scenario 1 – changes in food demand composition*

| % change rel. to baseline | **Total economy** | **Agriculture** | **Food industry** | **Manu-facturing** | **Services** |
|---|---|---|---|---|---|
| Output | 0.08% | 3.41% | -0.51% | 0.01% | 0.03% |
| Value added | 0.11% | 4.60% | -0.43% | 0.01% | 0.02% |
| Employment | 0.15% | 3.50% | -0.79% | 0.01% | 0.03% |
| Water consumption | 0.99% | 4.19% | -1.37% | 0.02% | 0.04% |

### 3.4.3. Improved Water Efficiency in Agriculture (Scenario 2)

Scenario 2 reveals a 5% improvement in the water efficiency of crop production (i.e., 5% less water required per unit of output) produces exclusively environmental effects, with no changes in output, value added, or employment (Table 12). This outcome confirms that efficiency improvements do not alter economic activity in the short term. As a result of more water-efficient crop production, agricultural water use decreases by 3.90%, resulting in a 1.11% reduction in total water consumption at the economy-wide level. Other sectors, including the food industry, manufacturing, and services, remain unaffected because the efficiency gain applies solely to crop production processes. The results indicate that technological and management improvements in crop production can meaningfully reduce water use without compromising economic performance.

*Table 12 Scenario 2 – improved water efficiency in agriculture*

| % change rel. to baseline | **Total economy** | **Agriculture** | **Food industry** | **Manu-facturing** | **Services** |
|---|---|---|---|---|---|
| Output | 0.00% | 0.00% | 0.00% | 0.00% | 0.00% |
| Value added | 0.00% | 0.00% | 0.00% | 0.00% | 0.00% |
| Employment | 0.00% | 0.00% | 0.00% | 0.00% | 0.00% |
| Water consumption | -1.11% | -3.90% | 0.00% | 0.00% | 0.00% |

### 3.4.4. Water-Use Restriction in Agriculture (Scenario 3)

Scenario 3 simulates the effects of a 5% reduction in water availability for the agricultural sector, representing a situation where water acts as a limiting factor for economic activity (Table 13 Scenario 3 – water-use restriction in agricultureTable 13). In the results table, water consumption for the aggregate "Agriculture" decreases by 4.96% rather than 5% because this aggregate also includes forestry and fishing, where no restriction was applied. As water is considered a factor determining production, restricted water availability for agriculture reduces its output and propagates both upstream and downstream through the economy, affecting sectors linked by supply chains. As a result, agriculture experiences a decline in economic output of 4.43%, and the total economic output of all sectors decreases by 0.39%. Corresponding reductions are observed in value added and employment. The food industry is moderately affected due to a downstream effect: reduced agricultural production limits the availability of raw inputs for processing, while the economic impacts on manufacturing and services are smaller but measurable. Total water use in the economy falls by 1.71%, reflecting not only lower water use in agriculture but also the interlinked effects that lead to reductions in water consumption across the food industry, manufacturing, and services. Overall, this scenario highlights how limitations in agricultural water availability can have ripple effects across multiple sectors, with the largest impacts, unsurprisingly, concentrated in the agricultural sector itself.

*Table 13 Scenario 3 – water-use restriction in agriculture*

| % change rel. to baseline | **Total economy** | **Agriculture** | **Food industry** | **Manu-facturing** | **Services** |
|---|---|---|---|---|---|
| Output | -0.39% | -4.43% | -0.78% | -0.14% | -0.43% |
| Value added | -0.45% | -5.50% | -1.48% | -0.19% | -0.38% |
| Employment | -0.57% | -5.83% | -0.91% | -0.12% | -0.40% |
| Water consumption | -1.71% | -4.96% | -0.28% | -0.34% | -0.62% |

### 3.4.5. Economy-Wide Water Constraint (Scenario 4)

Under Scenario 4, total water availability in the economy is reduced by 5%, representing an economy-wide water rationing scenario. Competition for water among sectors is a general challenge, and because sectors differ in their water intensity, reductions in output are not uniform: high-water-intensity sectors shrink more significantly, while low-water-intensity sectors shrink less. In other words, the reduction in output is, in line with economic principles, distributed proportionally across sectors according to water intensity. This scenario illustrates how the economic structure may shift toward less water-intensive activities, highlighting potential trade-offs between water conservation and overall economic performance.

The impact of the restriction is highly uneven across sectors. The food industry experiences a dramatic decline in water use (-30.24%), reflecting its strong reliance on water-intensive operations and highlighting the vulnerability of processing activities to water scarcity. In contrast, agriculture sees only a minor reduction in water consumption (-0.08%), as its water use is largely constrained by efficiency and pre-existing allocations. The substantial reduction required in the food industry signals potential risks for food supply chains and economic stability if water restrictions are applied uniformly across the economy without accompanying efficiency improvements.

*Table 14 Scenario 4 – economy-wide water constraint*

| % change rel. to baseline | **Total economy** | **Agriculture** | **Food industry** | **Manu-facturing** | **Services** |
|---|---|---|---|---|---|
| Output | -0.02% | -0.07% | -0.62% | 0.00% | 0.00% |
| Value added | -0.02% | -0.09% | -1.28% | 0.00% | 0.00% |
| Employment | -0.04% | -0.09% | -1.03% | 0.00% | -0.03% |
| Water consumption | -5.00% | -0.08% | -30.24% | -0.04% | -0.65% |

### 3.4.6. Key insights and policy recommendations

The results highlight several important insights regarding water use and economic activity. Moving toward plant-based diets (Scenario 1) increases overall water use (if not compensated by a substantial decrease in demand for meat), indicating that, while plant-based foods are often considered healthier, they can remain highly water-intensive. The macroeconomic effects are minimal, but agriculture and food processing experience notable shifts, underscoring sectoral vulnerabilities. Improvements in water efficiency (Scenario 2) reduce water use in agriculture and total water consumption in the economy, without affecting output, value added, or employment. This demonstrates that technological interventions can deliver significant environmental benefits without economic trade-offs. Restricting water availability in agriculture (Scenario 3) also reduces total water consumption, but the effects are uneven: agriculture bears the largest reductions, while the other sectors experience secondary effects due to interlinkages in input and output flows. Overall economic impacts are small but concentrated, revealing the vulnerability of supply chains under water stress. Economy-wide water rationing (Scenario 4) illustrates that not all sectors are affected equally. Water-intensive sectors, particularly the food industry, face disproportionate constraints, while other sectors remain relatively insulated. This highlights the importance of considering sectoral water intensity when designing policies for national water limits.

Policy recommendations emerging from these scenarios include:

- Promote water efficiency in agriculture through improved irrigation, technology, and management practices as a cost-effective way to reduce water stress without compromising economic performance.
- Pair dietary shifts with efficiency measures, since increasing demand for plant-based foods may still raise agricultural water consumption.
- Develop sector-sensitive water allocation strategies to prioritize essential production and avoid severe disruptions, particularly in food processing, under resource constraints.
- Integrate water footprint metrics into food and agricultural policy, ensuring sustainability goals reflect both consumption patterns and production efficiency.
- Support innovation in water-efficient food processing technologies to mitigate risks under water scarcity and safeguard supply chains.

Taken together, these measures suggest a coordinated approach that combines technological improvements, informed dietary guidance, and strategic water allocation to manage water demand while maintaining economic resilience.

## 3.5. The HHNK case study

In this section, we highlight three key sets of results from the HHNK survey and choice experiment that will be showcased in an online interactive dashboard. First, we present descriptive statistics on farmers' experiences with drought, their current adaptation measures, and their attitudes towards cooperation. Second, we show how often farmers choose policy alternatives that involve investment in water storage, rather than the status quo, and how this behavior varies across farmer groups defined by survey responses. Third, we report willingness-to-pay (WTP) estimates for the main policy attributes, such as support for own storage, cooperation arrangements, and abstraction bans. Together, these three components provide a concise but comprehensive picture of how farmers in the HHNK area perceive drought risks and respond to different drought-management and water-storage options.

### 3.5.1. Descriptive insights from the survey

A first set of outputs consists of pie charts (and related summary tables) that describe key survey outcomes, see Figure 32. These plots show, for example, how many farmers report having experienced (serious) hindrance from drought in recent years, which drought measures they already apply on their farm, how much storage capacity they currently have available (if any), and how they perceive future drought risks. Other descriptive figures summarize attitudes towards cooperation with HHNK and with neighboring farmers, perceived responsibilities for drought management, and views on the feasibility of investing in additional storage.

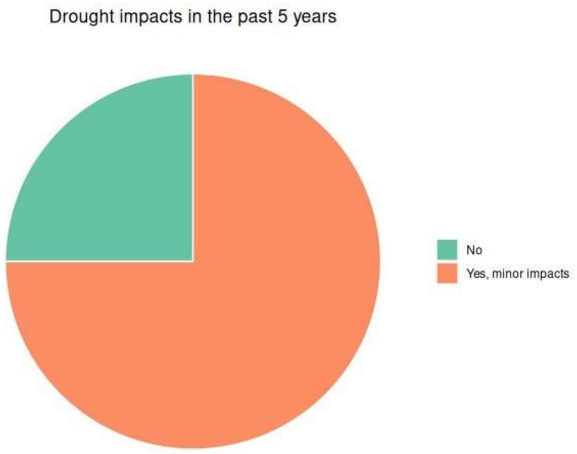

*Figure 32 Illustrative example of descriptive survey insights*

These descriptive results provide a factual baseline for the case study: they document how farmers currently perceive and experience drought, which adaptation measures are already in place, and how open they are to different forms of cooperation and support. They also help to interpret the choice experiment results by showing how representative the sample is and where there may be important heterogeneity in exposure, constraints, and attitudes.

### 3.5.2. Choice patterns and links with survey outcomes

A second set of dashboard outputs focuses on the choices made in the choice experiment. Bar charts display, for each respondent and for selected subgroups, the share of choice tasks in which one of the alternatives that involve investment in water storage (A or B) is chosen instead of the status quo option. An example is provided in Figure 33. This provides a simple yet informative indicator of how willing respondents are, on average and by subgroup, to move away from the current situation towards some form of investment in water storage and associated drought-management arrangements.

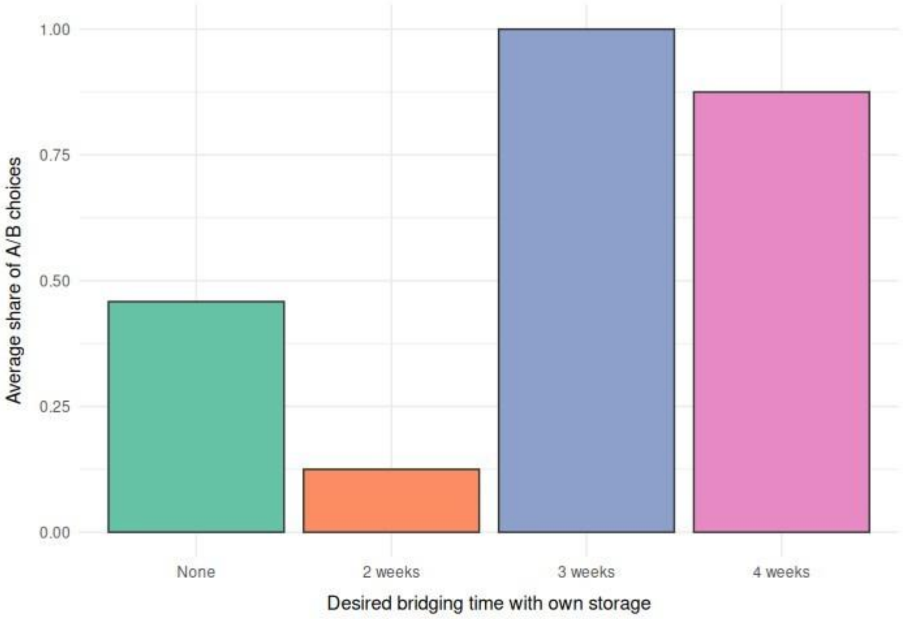

*Figure 33 Illustrative example of linking survey outcomes to choice patterns*

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These bar charts can be conditioned on survey variables to explore how choice behaviour varies with drought experience, perceived hindrance, existing storage, farm type or attitudes towards cooperation. For example, the dashboard can contrast the share of A/B choices between farmers who report serious drought impacts and those who do not, or between farmers who already have storage and those who do not. Such comparisons help to visualise how underlying experiences and constraints shape the propensity to invest in adaptation and to accept associated restrictions or cooperative arrangements.

By combining descriptive survey outcomes with these choice-based indicators, the dashboard gives a first, largely model-free, picture of how different farmer groups respond to the proposed policy and investment options.

### 3.5.3. Willingness-to-pay estimates and implied trade-offs

The third key output of the dashboard consists of willingness-to-pay (WTP) estimates derived from the random utility model. For the full sample, and where relevant for subsamples or alternative model specifications, the dashboard reports WTP for changes in the main policy attributes: extending bridging capacity with own storage, moving from individual to cooperative investment modes (with or without HHNK), and changing the number of weeks per year with an abstraction ban. These WTP measures translate the estimated marginal utilities into monetary terms and indicate how attractive different components of a policy package are to farmers.

The WTP results highlight several general patterns. Farmers, on average, value support for bridging or expanding their own storage, indicating that measures which reduce the financial burden and perceived risk of investing in storage are attractive. Cooperation attributes tend to be viewed positively when they facilitate coordination and information exchange, although some reluctance emerges towards more complex or binding collective schemes, reflecting concerns about autonomy and perhaps transaction costs. Abstraction bans are acceptable to many respondents only when combined with adequate bridging support (for example through access to stored water), consistent with the idea that strict demand-management instruments need to be embedded in a broader adaptation package rather than applied in isolation.

From a WEFE perspective, these patterns point to policy packages that can simultaneously limit peak abstractions and protect water levels (Water), support stable production and farm income (Food), manage additional energy use for storage and pumping in a cost-conscious way (Energy), and reduce stress on aquatic and peatland ecosystems during dry periods (Ecosystems). At the same time, the results underscore that high cost burdens or inflexible abstraction rules may reduce participation and undermine both economic and environmental objectives, particularly for more vulnerable or capital-constrained farms.

For each main attribute change (for example extending bridging capacity from one to three weeks, switching from no cooperation to cooperation with HHNK or with other farmers, or reducing the length of an abstraction ban), we compute the implied WTP in euro per hectare per year from the random utility model. In the dashboard, these WTP values are plotted in a radar chart with one axis per attribute and separate lines for the full sample and for selected subgroups (such as farmers with and without serious drought hindrance, or farmers with and without existing storage). An example is provided in Figure 34.

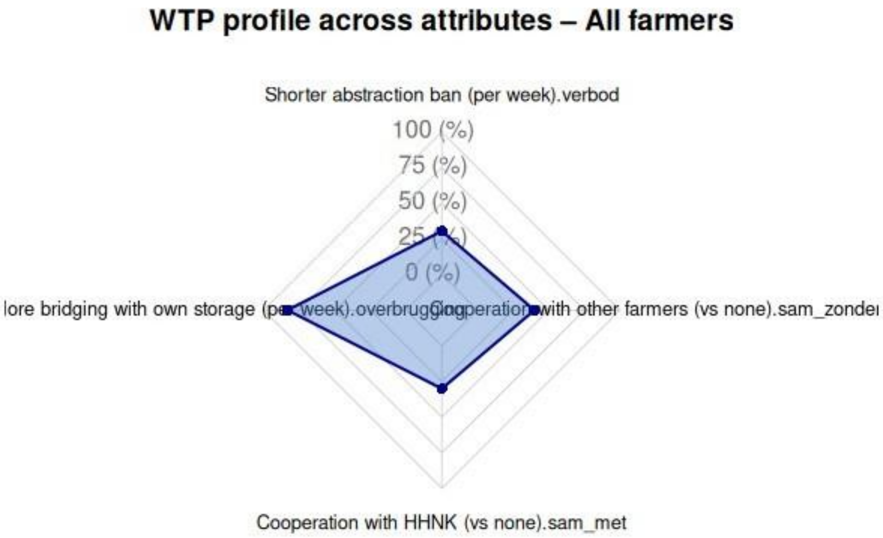

*Figure 34 Illustrative example of the WTP spider chart*

This visualisation highlights which attributes are most valued overall and how the pattern of preferences differs across farmer groups, without requiring any additional information on costs. It also makes clear where confidence intervals overlap with zero, signalling attributes for which WTP is estimated imprecisely or is not statistically different from zero. In this way, the WTP profile complements the descriptive figures and choice-pattern charts by providing a concise, model-based summary of how farmers prioritise different elements of drought-management and water-storage policy packages.

All results presented in the dashboard are preliminary, as data collection is still ongoing and model specifications may be refined. They should therefore be interpreted as indicative patterns that will be updated and tested for robustness once the full dataset is available.

### 3.5.4. Key insights and implications

The HHNK case study provides an initial, farmer-centred view on drought adaptation and on-farm water storage in a low-lying, supply-constrained region. Using a stated-preference random utility framework combined with a rich survey, we document both how farmers currently experience and perceive drought risks, and how they trade off costs, restrictions and cooperation arrangements in hypothetical policy packages.

Three broad insights emerge from the preliminary results. First, many farmers have already experienced drought-related impacts but have so far made only incremental adjustments and remain hesitant to undertake larger, lumpy investments in storage, reflecting financial, behavioural and coordination barriers. Second, there is clear willingness to pay for measures that reduce the cost and risk of investing in on-farm storage. Third, cooperation and collective solutions have potential but must be designed in ways that keep transaction costs manageable, preserve sufficient autonomy and recognise heterogeneity across farms.

The results underline that poorly designed cost-sharing or restrictive measures risk low participation and limited impact. As data collection and model refinement continue, the interactive dashboard will be updated and can serve as a practical tool for HHNK and RETOUCH partners to explore alternative policy designs, communicate trade-offs to stakeholders and link farmer preferences to more detailed governance or hydrological assessments.

## 3.6. Belgian Case Study

This section presents the results for the Keiberg case study and is structured in two parts. Section 5.1 reports the water balance results, providing insight into how the different scenarios affect water flows, abstraction, and use within the system. Section 5.2 presents the results of the cost-benefit analysis, in which the BASE, BAU, and INNO scenarios are compared from an economic perspective. Together, these results allow for an integrated assessment of both the hydrological and economic impacts of the proposed rainwater harvesting and management solutions, while acknowledging the uncertainties and limitations inherent in monetising all potential benefits.

### 3.6.1. Water balance results

Based on the water balance methodology (Section 2.6.3) and the input data described in Section 2.6.5.12, water balance results were calculated for the three scenarios applied to the selected industrial greenfield reference case study of Keiberg Vossem. The BASE scenario was calculated directly from connected surface areas and runoff coefficients, while both the BAU and INNO scenarios were modelled using the Sirio Design software.

The main aggregated water balance results are presented in Figure 35 and form the basis for the subsequent cost-benefit analysis. As expected, the BASE scenario results in a substantial loss of rainwater from the site, which must be conveyed and treated externally. In contrast, both the BAU and INNO scenarios enable on-site industrial rainwater use. However, in the BAU scenario, only 12% of the available rainwater is effectively used, primarily due to the limited buffering capacity at individual plot level. In the INNO scenario, collective infrastructure significantly increases rainwater use, reaching 46% of the available rainwater, which corresponds to the total estimated demand for rainwater consumption.

The remaining excess rainwater can be managed through on-site infiltration, either at plot level or via collective facilities, using conventional infiltration systems or more advanced treatment and injection solutions, as implemented in the real Keiberg Vossem case. Alternatively, surplus rainwater can be stored in a collective buffer and supplied to local farmers for irrigation purposes. Both infiltration and irrigation use options contribute to reduced groundwater abstraction, thereby generating benefits within the Water–Energy–Food–Ecosystem (WEFE) nexus. These benefits are quantified and discussed in the following sections.

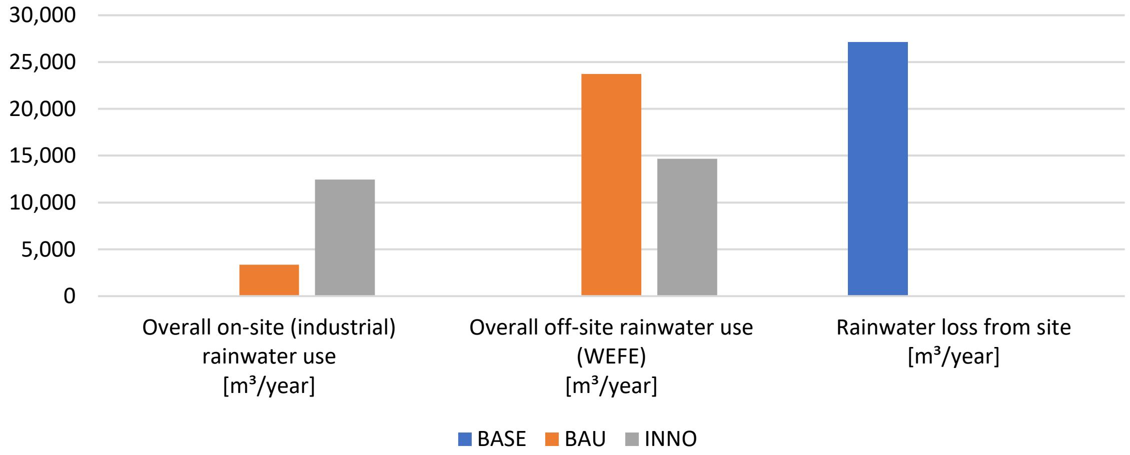

*Figure 35 Rainwater use in three scenarios*

### 3.6.2. Cost-benefit analysis

Table 15 presents the results of the cost-benefit analysis by comparing the BASE, BAU, and INNO scenarios using different reference scenarios. The reference scenario is indicated in the rows, while the columns show the cost differences of the alternative scenarios relative to that reference. For each reference scenario, the value of the reference itself is zero. Positive values indicate higher costs compared to the reference scenario, while negative values indicate cost savings. The row *"Total comparison with reference scenario"* aggregates the individual cost components (groundwater extraction and tariff costs, drinking water production costs, flooding costs, CAPEX, and OPEX) into an overall cost difference for each scenario.

When comparing scenarios using BASE as the reference, the scenarios with rainwater harvesting (BAU and INNO) show that costs are higher than benefits. This result needs to be interpreted with caution: several broader environmental, social, and long-term system benefits (e.g., such as ecosystem services, increased water security under extreme climate conditions, and wider resilience effects) are not explicitly included in the calculations. As a result, the quantified benefits presented here likely represent a conservative estimate of the true value of the benefits.

A more policy-relevant comparison is therefore the comparison between INNO and BAU. When BAU is used as the reference scenario, the INNO scenario results in a net cost saving of 242k euro. This improvement is mainly driven by lower capital and operational expenditures, as well as substantial reductions in drinking water production costs. Although groundwater extraction and tariff costs increase, these additional costs are outweighed by the overall savings.

*Table 15 Results of CBA for three different scenarios with a time horizon of 40 years and real discount rate of 4%*

| Reference | Effect | BASE | BAU | INNO |
|---|---|---|---|---|
| BASE | Groundwater extraction & tarrif costs | 0 | -189.311 | -116.976 |
| BASE | Drinking water prodution costs | 0 | -39.707 | -146.552 |
| BASE | Wastewater treatment costs | 0 | -161.661 | -161.644 |
| BASE | CAPEX | 0 | 873.924 | 731.280 |
| BASE | OPEX | 0 | 407.078 | 341.406 |
| BASE | Total comparison with reference scenario (BASE) | 0 | 890.322 | 647.515 |
| BAU | Groundwater extraction & tarrif costs | 189.311 | 0 | 72.335 |
| BAU | Drinking water prodution costs | 39.707 | 0 | -106.845 |
| BAU | Wastewater treatment costs | 161.661 | 0 | 18 |
| BAU | CAPEX | -873.924 | 0 | -142.644 |
| BAU | OPEX | -407.078 | 0 | -65.672 |
| BAU | Total comparison with reference scenario (BAU) | -890.322 | 0 | -242.808 |
| INNO | Groundwater extraction & tarrif costs | 116.976 | -72.335 | 0 |
| INNO | Drinking water prodution costs | 146.552 | 106.845 | 0 |
| INNO | Wastewater treatment costs | 161.644 | -18 | 0 |
| INNO | CAPEX | -731.280 | 142.644 | 0 |
| INNO | OPEX | -341.406 | 65.672 | 0 |
| INNO | Total comparison with reference scenario (INNO - Generic) | -647.515 | 242.808 | 0 |

Overall, while the comparison between INNO and BASE is difficult to interpret due to structural differences between the scenarios and the incomplete valuation of benefits, the comparison with BAU provides a clear and robust conclusion. The results indicate that the INNO scenario performs better than the BAU scenario from an economic perspective, even under conservative assumptions. This suggests that innovative rainwater management solutions offer a promising and cost-effective alternative to business-as-usual approaches in the context of water-stressed regions such as Flanders.

### 3.6.3. Key insights and policy recommendations

The analysis confirms that collective rainwater management solutions can outperform business-as-usual approaches (i.e., the legal minimum scenario) from a societal perspective, even when assessed under conservative benefit assumptions. While not all benefits can be fully monetised, the comparison between the INNO and BAU scenarios shows that innovative, collective systems can deliver net economic gains. These findings underline the importance of moving beyond minimum regulatory compliance and adopting integrated, forward-looking water management strategies. From these results, we suggest the following recommendations.

- **Promote collective rainwater systems where joint solutions create added value**

Policymakers should actively encourage collective rainwater harvesting, buffering, and use systems in locations where scale effects are present, such as business parks, industrial zones, and dense residential developments. Comparing individual and collective decentralised systems is essential to identify cases where joint investments deliver greater societal value and improved economic performance.

- **Use BAU, not BASE, as the benchmark for decision-making**

When evaluating rainwater management investments, policy assessments should compare innovative solutions against realistic business-as-usual scenarios rather than minimal baseline configurations. This ensures that decisions reflect actual regulatory requirements and future system pressures, leading to more meaningful economic comparisons.

- **Provide guidance on choosing between individual and collective systems**

Public authorities should develop clear guidelines to support developers and municipalities in deciding when collective rainwater systems are preferable to individual solutions. These guidelines should consider site characteristics, water demand profiles, flood risks, and opportunities for shared infrastructure.

- **Align financial incentives with societal benefits**

To accelerate uptake, subsidies, tariffs, and regulatory instruments should better reflect the societal benefits of collective rainwater management, including avoided flood damage and reduced pressure on drinking water systems. Targeted incentives can help overcome higher upfront investment costs.

- **Assess the scalability and transferability of collective rainwater solutions**

Future research should explicitly assess how the performance of collective rainwater management systems scales across different spatial contexts, levels of urban density, and demand profiles. Understanding whether societal benefits persist, increase, or diminish at larger scales is essential for strategic planning and regional roll-out. The assessment of scalability and transferability will therefore be addressed in the next phase of the project, building on the analytical framework developed in this study.

- **Analyse the distribution of costs and benefits across actors**

While the present analysis adopts a societal perspective, future research should examine how costs and benefits are distributed among different actors, such as developers, residents, businesses, water utilities, municipalities, and farmers. Identifying potential mismatches between who bears the investment costs and who captures the benefits is essential for designing fair, feasible, and effective governance and financing arrangements. A detailed actor-based distributional analysis will therefore be included in the next phase of the project to support the development of appropriate cost-sharing mechanisms and incentive structures.

- **Integrate WEFE considerations in rainwater management strategies**

Rainwater that is harvested but not used by local actors should be explicitly considered within a broader WEFE perspective. Surplus rainwater that is kept local can contribute to groundwater recharge, supporting ecosystems and mitigating drought impacts, or be made available for agricultural use. This reduces pressure on conventional water sources. Policy frameworks should recognise these cross-sectoral benefits, ensuring that rainwater management contributes not only to urban water objectives but also to ecosystem health and food production.

# 4. Conclusions

The six European case studies presented in this document illustrate the diverse challenges and opportunities for sustainable water management within the WEFE nexus framework. By applying a consistent nexus-oriented approach across heterogeneous hydro-climatic, economic, institutional, and governance contexts, these studies demonstrate how quantitative modelling can inform the design and evaluation of economic instruments to achieve efficient and equitable water allocation. Each case study employed tailored quantitative or qualitative methods to capture the interdependencies among water, energy, food production, and ecosystems.

Hydroeconomic modelling was employed in the Jucar River Basin (Spain) to develop and evaluate different water pricing strategies, systematically compared against a baseline scenario reflecting current allocation rules and tariff structures. A combination of hydrologic and system dynamics modelling was developed for the Upper Main River Basin (Germany) to capture complex feedback loops and dynamic interactions across the WEFE nexus. This integrated framework simulated the interplay between water quantity, quality, sectoral demands (particularly agriculture), and socio-economic responses, while incorporating economic scarcity signals through shadow pricing to reflect the marginal value of water under hydrological constraints and competing uses.

In the Malta case study, an island system highly dependent on energy-intensive non-conventional water sources such as desalination, a dynamic sustainability model was developed to explore the interrelationships between economic variables, energy consumption, and water resources from a long-term sustainability perspective. The model was formulated in algebraic terms to represent dynamic relations, feedback loops, and resource constraints, and then calibrated and applied to the Maltese economy. The results revealed that continued economic growth can coexist with rapid depletion of water and energy stocks in the early phases, leading to potentially unsustainable trajectories. More scenarios are being analyzed for the island. An environmentally extended input-output modelling framework was applied to the Slovak case study to trace resource flows and virtual water embedded across sectors and the economy. This approach captured intersectoral linkages and indirect effects, enabling the evaluation of policy scenarios including shifts toward plant-based diets, improvements in water efficiency, restrictions on agricultural water availability, and economy-wide rationing.

In Flanders (Belgium) a cost-benefit analysis was applied to evaluate the societal impacts of innovative decentralized rainwater harvesting and management solutions within the WEFE nexus framework. This approach quantified both monetary and non-monetary costs and benefits across hydrological, economic, environmental, and resilience dimensions, supporting the Flemish government's Blue Deal initiatives to enhance water security through circular practices such as rainwater buffering, reuse, and infiltration. Finally, in the HHNK case study (Netherlands) a choice experiment based on farmer surveys was implemented to elicit stakeholder preferences and incorporate social, behavioral, and governance dimensions into WEFE nexus policy evaluation. Choice data revealed patterns in preferences, showing how frequently farmers opted for policy scenarios featuring supported water storage infrastructure over maintaining current practices, with variations across subgroups defined by risk perceptions, farm types, and cooperation willingness.

Together, these complementary methods generated comprehensive evidence on the effectiveness of context-specific economic instruments. The results consistently show that nexus-oriented policies can reduce resource conflicts, enhance ecosystem protection, and improve overall system resilience.

The interactive web-based dashboard developed within the project further amplifies the impact of these findings by providing transparent, user-friendly access to model outputs, scenario comparisons, and policy implications across all case studies. This tool supports stakeholder engagement and informed decision-making at local, regional, and European levels.

In conclusion, this work advances European water policy by delivering transferable insights and practical tools for sustainable nexus management. As water scarcity and competing demands intensify, the rigorous application of integrated quantitative approaches—grounded in real-world European contexts—will remain essential for designing adaptive, evidence-based strategies that secure the long-term viability of water resources and the interconnected WEFE systems they underpin.

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