Economic Sustainability of Irrigation-Dependent Ecosystem Services Under Growing Water Scarcity. Insights From the Reno River in Italy

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Abstract As water scarcity grows, irrigators are adopting modern irrigation systems to increase the proportion of water use consumed by crops and mitigate impacts on production. This increased beneficial consumption is fundamentally sourced by lower return flows-runoff and percolation-back into the environment, which are critical to sustain wetlands and other irrigation-dependent green infrastructures that supply valuable ecosystem services. We adopt an innovative multi-model ensemble of mathematical programming models to assess irrigators' responses to pecuniary compensations designed to sustain irrigation-dependent ecosystem services (Payments for Watershed Services-PWS), under multiple scenarios. The upshot is a database of forecasts that represents the range of plausible future states, which is used to assess economic performance and identify a robust adaptation strategy to growing scarcity. Our analysis of the Reno River Land Reclamation and Irrigation Board in NE Italy shows that, under most models and scenarios, the conservationist strategy has a superior economic performance than the autonomous adaptation strategy where modern irrigation systems are adopted. The conservationist strategy is also found more robust than the autonomous strategy. However, unless sensible incentives are put in place, irrigators may adopt irrigation technologies nonetheless. Under mild to moderate climate change scenarios, removing existing subsidies to modern irrigation systems is sufficient to deter their adoption. Under severe climate change scenarios, additional PWS to irrigators will be necessary to ensure the sustainability of irrigation-dependent ecosystem services.

1. Introduction

Throughout the world, the surface and subsurface return flows that leave agricultural systems following water withdrawal and application have created and sustained wetlands, forested areas and other green infrastructures that supply valuable ecosystem services (Grafton et al., 2018). Examples of irrigation-dependent ecosystem services include habitat conservation (e.g., wetlands for migrating waterfowl), climate regulation (e.g., carbon sequestration), soil retention, cultural heritage (e.g., spiritual fulfillment, intellectual development) and amenity services (such as esthetic enjoyment or recreation), among others. Since these ecosystem services are typically outside of the market, their provision by irrigated systems is not included in the valuation of agricultural production, nor is their eventual loss where the underlying green infrastructure is degraded (TEEB, 2015).

Irrigation-dependent ecosystem services are under threat by rising water scarcity and adaptive responses by farmers through modern irrigation systems, such as sprinkler or drip irrigation systems, laser leveling of fields, piped delivery systems, canal lining, and other physical rehabilitation of irrigation and delivery systems (Perry & Steduto, 2017). Modern irrigation systems are designed to increase the proportion of beneficial consumption per unit of water use. As scarcity grows, this allows irrigators to negate reductions in biophysical production from diminished water supply. This increased beneficial consumption is partially sourced by lower non-beneficial consumption (e.g., evaporation from wet fields), but the predominant source is a reduction in return flows—runoff and percolation—back into the environment, which are often reused by irrigation-dependent ecosystem services downstream. The upshot is a sustained reduction of return flows due to increased beneficial consumption, degraded green infrastructures and a deterioration of irrigation-dependent ecosystem services (Figure 1; Pérez-Blanco et al., 2020).

Exceptions will arise in cases where irrigators can participate in incentive schemes that encourage environmental performance, such as payments for ecosystem services (PES). PES are a pecuniary compensation that internalizes

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the positive externalities generated by providers (e.g., irrigators) through the protection or enhancement of green infrastructure (Asbjornsen et al., 2015). PES include both private (where individuals and/or private and non-governmental organizations are the sole buyer) and public sector payments, although there are relatively few examples of the former in the case of watershed ecosystem services, where most funding comes from supranational, national or regional governments acting on behalf of their constituency. PES have received growing attention and funding due to their perceived capacity to enhance environmental and economic performance, including through the generation of relevant co-benefits such as increased and/or more stable incomes in rural areas (Bremer et al., 2016). The literature records over 400 water-related PES (also known as Payment for Watershed Services, PWS) in more than 60 countries, rehabilitating a surface one and a half times the size of India for a total financial value of $25 billion (Bennett & Franziksa, 2016). However, "few rigorous evaluations" on the economic performance of PWS programs exist (Bhaduri et al., 2021). Lack of economic rigor is attributed to two main factors: (a) the oversimplistic representation of human agency and (b) the treatment of uncertainty (or lack thereof).

1. Most economic assessments of PWS rely on oversimplistic representations of human agency (e.g., through linear objective functions) built entirely on the basis of relationships observed in historical data (e.g., projections from baseline conditions; Bhaduri et al., 2021; Harou et al., 2009; Pérez-Blanco et al., 2021). This contravenes the Lucas Critique, after the Nobel Laureate Robert Lucas, which states that it is inadequate to predict the effects of policy shocks on human behavior entirely on the basis of relationships observed in historical data (Lucas, 1976). This is because the parameters of models elicited this way are not structural, that is, not policy-invariant, and would necessarily change whenever the policy (e.g., PWS adoption) changes. Instead, historical data should be used to reveal the micro-foundations or deep parameters (utility, preferences, resource constraints) driving agent's responses, for example, through mathematical programming models (Graveline, 2016); and use these models for prediction.

2. Available PWS economic assessments typically disregard scenario and modeling uncertainty. Scenarios in PWS studies are typically built either using simplistic point predictions or through probabilistic descriptions of plausible future states; which are then fed to a single model that is used to produce a forecast. Yet, in deeply uncertain socio-ecological systems, where researchers and stakeholders typically do not know/cannot agree on the model that relates scenarios to outputs, or the probability of these scenarios, such consolidative approach risks providing more information than what we can reasonably claim to know (Marchau et al., 2019). Disregard of scenario and modeling uncertainty becomes problematic where PWS performance is highly sensitive to future states. For example, a recent assessment of PWS in Colorado (US) found that the potential financial returns to beneficiaries was expected to be positive, but also warned that these returns would vary considerably depending on the scenario and only hold under specific model assumptions (Jones et al., 2017). Unfavorable surprises, especially those resulting in abrupt change, can lead to tipping points that significantly and irreversibly deviate expected from realized policy performance (Anderies, 2015). This is closely related to issues of permanence, that is, whether PWS will lead to sustained restoration/conservation of water-related ecosystem services, especially when future conditions (e.g., climate) may abruptly change (Rode et al., 2015).

This paper builds on the concepts of micro-foundations (Lucas, 1976), exploratory modeling (Kwakkel & Pruyt, 2013) and multi-model ensemble (IPCC, 2014; Sapino et al., 2020) to develop an economic assessment framework for PWS that uses mathematical programming methods to elicit the deep parameters driving human behavior, while thoroughly sampling scenario and modeling uncertainty in the analysis of irrigators' responses. The ensemble includes 2 Positive Mathematical Programming (PMP) models, 2 Positive Multi-Attribute Utility Programming (PMAUP) models and 1 Linear Programming (LP) model. The assessment framework is used to simulate the economic performance (through utility) of an hypothetical conservationist strategy (i.e., no irrigation modernization and reallocation of water toward the environment) v. autonomous adaptation strategy (i.e., irrigation modernization and reallocation of water toward agriculture). By comparing the monetized foregone utility experienced by irrigators under the conservationist strategy v. the monetized foregone utility experienced by irrigators under the autonomous adaptation strategy we can obtain the minimum compensation irrigators would be willing to accept to sustain the PWS scheme. This information is subsequently compared with estimates of the economic value of ecosystem services to assess the economic performance of the proposed PWS. Repeating this process for multiple models and scenarios (climate change, irrigation modernization costs, economic values of ecosystem services) yields a database of forecasts that represents the range of plausible future states. Coupled with automated robust decision methods, this database is used to identify robust adaptation strategies that avoid
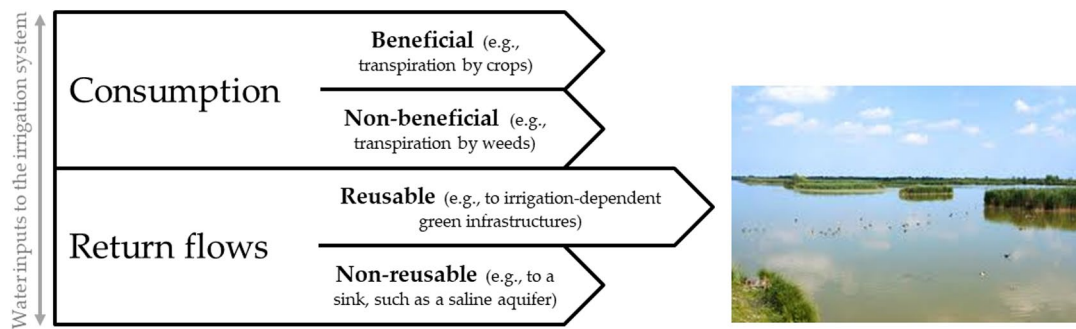
Figure 1. Water accounting balance. All water entering the irrigation system goes to either: (a) beneficial consumption, water that is purposefully converted to water vapor, primarily crop transpiration; (b) non-beneficial consumption, water that is not purposefully converted to vapor, such as through transpiration by weeds or evaporation from wet surfaces; (c) reusable return flows, water reaching a useable water body with downstream demand; and (d) non-reusable return flows, water flowing without benefit to a sink such as the sea, and therefore not useable.

unfavorable outcomes that can be identified ex-ante. Methods are flexible and replicable, and are illustrated with an application to the PWS program presently being discussed between the Regione Emilia Romagna and the Reno River Land Reclamation and Irrigation Board (Reno River LRIB, in Italian: Bonifica Renana) to conserve the ecosystem services provided by irrigation-dependent wetlands in NE Italy.

2. Case Study Background: The Reno River Land Reclamation and Irrigation Board

Since Roman times, irrigators in Northern Italy have built water retention, distribution and drainage infrastructures to expand the surface available for agricultural and urban developments. Canals, irrigation systems, detention basins, drainage systems and other water works have transformed the landscape and created a complex network of man-made gray and green infrastructures that is managed, monitored, maintained and modernized by LRIBs, a public-private partnership that brings together all the owners of land and buildings (public and private) within its area of influence.

The Reno River LRIB, located to the Southeast of the Po River Basin, manages an area of 341,953 ha across five provinces (Bologna, Firenze, Modena, Ferrara, Ravenna, Prato, and Pistoia). Of this surface, 56,067 ha are lowland alluvial plains that are drained using 1,667 km of canals, 26 detention basins (with the capacity to store over 42 million m³ of rainfall water) and 26 pumping systems. The same network of infrastructures (plus 63 additional pumping systems for irrigation) is used to support the irrigation of 18,000 ha, which are divided into 5 irrigation districts (the agents in the mathematical programming model): C001 (Dep. Bologna-Po), C002 (Po), C003 (Reno, Reno-Po, Dep. Bologna), C004 (Quaderna, Sillaro, Dep. Ozzano Emilia, Dep. Castel S. Pietro), C005 (Ghironda, Lavino, Rii Pedecollinari, Dep. Anzola Emilia, Dep. Calcara, Dep. Calderara di Reno, Dep. Padulle di Sala). To this end, the Reno River LRIB distributes on average 68 million m³/year of water for irrigation, which comes exclusively from surface water sources: 73% from the nearby Po River, via the Emiliano Romagnolo Canal (in Italian: Canale Emiliano-Romagnolo—CER); 16% from the Reno River; and the remainder from detention basins (see Figure 2; Nomisma, 2019a).

Transportation and application technical efficiencies vary across the LRIB, being estimated on average at 50% and 85%, respectively (Nomisma, 2019b). As a result, drainage and irrigation activities produce non-trivial surface return flows, which have created and sustain 160 ha of protected areas with
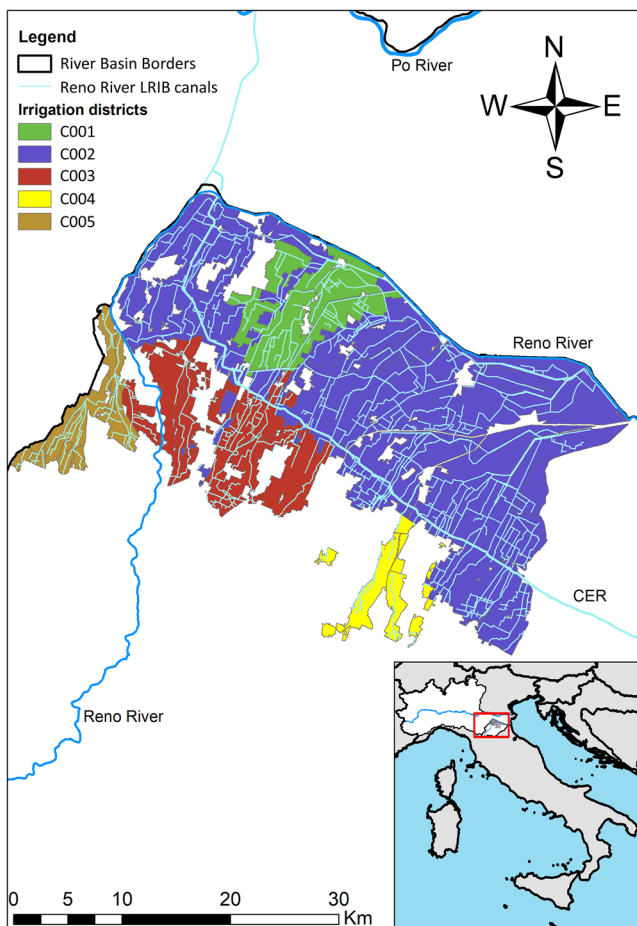
Figure 2. The Reno river LRIB.

wetlands across the LRIB. These wetlands provide valuable services beyond the conventional water supply and flood protection services typically attributed, and paid, to the Reno River LRIB, including: provisioning (e.g., food production, water storage), regulating (e.g., carbon sequestration, water purification), habitat (e.g., genetic diversity) and cultural services (e.g., esthetic, spiritual, educational and recreational; Nomisma, 2019a). Annex I in the Supporting Information S1 offers a comprehensive description of the ecosystem services provided by the Reno LRIB, which leverages on a review of the scientific literature complemented with the feedback provided by local stakeholders in a workshop held in Bologna (Italy) in April 2019.

On the other hand, in a context of diminishing water supply due to climate change, the same inefficiencies that maintain these valuable ecosystem services constrain water availability for irrigated agriculture. This has led agricultural landowners within the Reno River LRIB to call for investments toward the modernization of irrigation and drainage infrastructures, particularly the network of canals. Irrigation modernization plans have been received with caution by the government of the Emilia-Romagna Region (where most of the lowlands are located) and the Reno River LRIB itself, which are concerned of the impact this intervention would have on protected areas and their wetlands (Nomisma, 2019b). Following a series of exchanges between the regional government and the Reno River LRIB, a research project was commissioned to the authors to explore the economic feasibility and sustainability of a PWS between the regional government (buyer) and the irrigators of the Reno River LRIB (providers), whose methods, results and conclusions are reported below.

3. Methods

This paper develops an economic assessment framework for PWS that uses mathematical programming methods to elicit the deep parameters driving human behavior, while thoroughly sampling scenario and modeling uncertainty in the analysis of irrigators' responses. This mechanistic methodology is complemented with automated robust decision methods to identify a non-regret adaptation strategy.

3.1. Multi-Model Ensemble

While simplification through model conceptualization helps to effectively convey insights into how to better allocate resources in complex systems, it also leads to imperfections in the representation of the system and errors (Tebaldi & Knutti, 2007). Unawareness of these errors can result in misleading policy recommendations, significant deviations of expected from realized performance, and maladaptation (Hino & Hall, 2017), which can be aggravated by issues of non-convexity and irreversibility (Anderies, 2015). Despite these problems, economic performance evaluations of PES, and PWS specifically, typically rely on a single model and model setting to produce forecasts, which makes these schemes vulnerable to modeling uncertainty.

Ecological sciences have addressed modeling uncertainty through ensemble experiments that use multiple models to sample uncertainty (see e.g., Cloke et al., 2013; IPCC, 2014). Yet, ensemble experiments are under-researched in all disciplines of social sciences. In this paper, we sample uncertainty in human behavior and responses using a multi-model ensemble of mathematical programming models consisting of 2 PMP, 2 PMAUP and 1 LP models—all of which are widely used methods in the literature on economic models for agricultural water management. In these models, farmers decide on the crop portfolio so to maximize the utility provided by a set of utility-relevant variables, subject to a series of constraints (Graveline, 2016):

$$\underset{x}{\text{Max}}\; U(\pmb{x}) = U\left(z_1(\pmb{x});\;\; z_2(\pmb{x}); z_3(\pmb{x})\dots\; z_m(\pmb{x})\right) \tag{1}$$

$$s.t.:\; 0 \leq x_i \leq 1 \tag{2}$$

$$\sum_{i=1}^{n} x_i = 1 \tag{3}$$

$$\pmb{x} \in F(\pmb{x}) \tag{4}$$

$$\pmb{z}(\pmb{x}) \in R^m \tag{5}$$

where $\boldsymbol{x}$ is the crop portfolio, a vector representing the share of land allotted to each crop i; $\boldsymbol{z}(\boldsymbol{x})$ is a vector of utility-relevant attributes defined so that "more-is-better" (i.e., increasing the provision of one attribute, *caeteris paribus*, increases utility); $U(\boldsymbol{x})$ is a parameterized objective function that relates inputs (the provision of utility-relevant attributes under a given crop portfolio) to outputs (utility); and $F(\boldsymbol{x})$ is the set of constraints conforming the domain, which are common to all models in the ensemble, and whose mathematical formulation is available in Annex II in the Supporting Information S1. Of particular relevance is the water allocation constraint:

$$\sum_{c=1}^{n} \frac{w_i}{\text{eff}_i} \, x_i \le W_g \tag{6}$$

where w_i are the net water needs or evapotranspiration (i.e., excluding inefficiencies) of crop i (in m3/ha), eff_i is the irrigation efficiency (which ranges between 0 and 1), w_c/eff_c are the gross water needs or water applied to crop i (i.e., including inefficiencies), x_i is the share of land allotted to crop i and W_g is the total water allocation for the agent (m3/ha). Adopting modern irrigation systems increases eff_i and reduces w_i/eff_i.

Differences across the mathematical programming models considered in the ensemble stem from the form and calibration of the utility function (Graveline, 2016). Regarding the form, the utility functions used by mathematical programming models can be single- (the case of PMP) or multi-attribute (the case of LP and PMAUP). Single-attribute utility functions use expected profit as the sole utility-relevant attribute; while multi-attribute utility functions typically explore the relevance of expected profit, risk aversion, and management complexity aversion. A comprehensive description and mathematical formulation of the attributes explored in the ensemble (namely, expected profit, risk aversion and management complexity aversion), as well as the related data inputs, is available in Annex III in the Supporting Information S1. Utility functions can also adopt different functional forms across mathematical programming models, typically Cobb-Douglas (PMAUP), additive (LP) and quadratic (PMP).

Regarding the calibration, each mathematical programming model used in the ensemble (PMP, LP, PMAUP) has a unique calibration method, which are discussed in Annex IV in the Supporting Information S1. The calibration results for the five irrigation districts/agents in the Reno River LRIB using the five models above are presented in Annex V in the Supporting Information S1.

3.2. Exploratory Modeling and Scenarios

Exploratory modeling is a technique that uses computational experiments to study the behavior of complex systems over a set of plausible scenarios given a priori knowledge (Kwakkel & Pruyt, 2013). Exploratory modeling has been used to study structural transformations under uncertainty, and to inform the design of robust adaptation strategies (Bankes et al., 2013; Marchau et al., 2019). In this paper, exploratory modeling is used to create a set of plausible scenarios whose outcomes are subsequently tested, for each adaptation strategy (conservationist v. autonomous adaptation strategy), using the multi-model ensemble of mathematical programming models presented above. The following sets of scenarios are considered: (a) climate change, (b) irrigation modernization, and (c) environmental valuation scenarios.

Climate change scenarios. Climate change scenarios are based on the hydrologic projections for the Po River Basin in the Italian Climate Change Adaptation Plan (MITE, 2018), which are summarized in Annex VI in the Supporting Information S1. The Italian Climate Change Adaptation Plan foresees a reduction in runoff for the Po River Basin that ranges between 30% (RCP4.5) and 45% (RCP8.5) by 2,080, which will be coupled with an increase in upstream demand of up to 25% due to irrigation expansion. This will lead to increased agricultural water deficit, particularly downstream (up to 20%–40% reduction in agricultural water allocation). A total of 45 climate change scenarios were simulated using mathematical programming methods (agricultural water allocation reduction in Equation 6 from 0% to 45% at discrete intervals of 1%).

Irrigation modernization scenarios. The Reno River LRIB has designed a plan to implement canal lining and increase the average technical efficiency of transportation systems from 50% to 85%. The cost of canal lining for irrigators, based on available budget estimates, interviews with local experts (which were asked to account for overbudgeting in their responses) and subsidies (the modernization of collective irrigation infrastructures such as canals is eligible to receive direct payments from the Common Agricultural Policy (CAP) of 40%–90% of the

investment cost, on the basis of a supposedly higher environmental performance (Official Journal of the European Union, 2013)), is estimated at 90,000–120,000 EUR/km (9,000–72,000 EUR/km with subsidies); which applying the standard amortization period (50 years) and interest rate (2%; Nomisma, 2019a) yields an annuity of 2,865–3,820 EUR/km (286.5–2,291 EUR/km with subsidy). This value is then multiplied by the total length of the canals in the lowlands, divided by the total number of irrigated hectares in the lowlands, and multiplied by the number of irrigated hectares in each irrigation district to obtain the irrigation modernization cost for each economic agent/irrigation district. Irrigation modernization costs are then charged to economic agents through a flat rate (EUR/ha). Local experts advised against using a volumetric charge (i.e., EUR/m^3) because metering devices are still unavailable for most irrigators in the area. A total of 112 irrigation modernization scenarios were simulated using mathematical programming methods (irrigation modernization costs from 9,000 to 120,000 EUR/km, at intervals of 1,000 EUR/km).

Environmental valuation scenarios. The two sets of scenarios above are used to estimate the utility perceived by irrigators under the two alternative strategies considered (conservationist v. autonomous adaptation), and the minimum compensation irrigators would be willing to accept to sustain the PWS scheme. This information is subsequently compared with the economic value provided by these ecosystems, to assess the economic performance of the PWS scheme.

The literature on ecosystem services does not prescribe a single technique to measure their economic value, and several methods can be used to this end (TEEB, 2015). With sufficient time and resources, original environmental valuation studies (such as contingent valuation or contingent ranking) are typically preferred (Arrow et al., 1993). However, original environmental valuation studies demand large research teams, and necessitate careful study design and data analysis before methods and results can be validated. Alternatively, benefit transfer methods can be used to approximate the economic value of ecosystem services through estimates obtained by other studies performed elsewhere. The benefit transfer approach, which is adopted here, transfers an "estimate from another study/studies to a different context", usually by multiplying the mean economic value for a person/family of the ecosystem service(s) *X* in location A by the population/number of families in location B, so to obtain the value of the ecosystem service(s) in B (Rosenberger & Loomis, 2003). Use of benefit transfer has the additional advantage of generating multiple plausible environmental valuation estimates for ecosystem services (one per study in the sample), instead of one point prediction as original valuation studies would do. This can be used to create multiple environmental valuation scenarios that more thoroughly sample scenario uncertainty.

Annex VII in the Supporting Information S1 presents the outcome of a literature survey from the Environmental Values Reference Inventory (www.evri.ca) on the economic benefits of the ecosystem services generated by (irrigated) agriculture. The relevant studies for our research were screened in three stages: (a) a review of the gray and academic literature concerned with the measurement of the total economic value of the ecosystem services provided by water-dependent ecosystems was performed, which led to 323 studies; (b) of this list, those studies that focused on at least 4 of the ecosystem services of relevance for the Reno River LRIB (see Annex I in the Supporting Information S1) were selected, which led to 47 studies; (c) the list was further reduced to account only for the most recent studies (last 15 years, 2007–2021), studies estimating annuity values (instead of lump sum values, to avoid discount rate uncertainties) and studies providing pecuniary values (i.e., qualitative studies were excluded), which led to 9 studies. Estimates in the original studies are reported in Annex VII in the Supporting Information S1 in current year's values in foreign currency, either per person, family or unit of surface, and were converted to 2,020 values using exchange rate and GDP deflator data from World Bank (2020). This resulted in an annuity value of ecosystem services in the Reno River LRIB that ranges between 57 and 372.4 EUR/ha/year, with a median of 126 EUR/ha/year.

3.3. Managing Uncertainty Through Robustness

Arguably, model selection techniques could be used to choose among candidates the model that performs better, for example, through minimization of calibration errors (see Annex V in the Supporting Information S1), instead of relying on an ensemble. Nonetheless, assessing model performance is controversial and goes beyond a straightforward comparison of calibration errors. Notably, models in our ensemble are designed as a substitute for direct experimentation, which means that we cannot evaluate the predictive performance of the models within the ensemble, a critical step in model selection (Konishi & Kitagawa, 2008). It may occur that a model with a relatively low calibration error performs poorly against non-observed data as compared to alternatives (poor predictive

performance; Pindyck, 2015). Moreover, calibration errors are not directly comparable among different models, since modeling errors are independent (Cloke et al., 2013). Alternatively, multi-model ensemble modeling can be used to generate a probability distribution function that combines all models to generate a point prediction that avoids model selection bias. Yet, this is challenging due to the subjectivity involved in defining prior assumptions about the distribution and the accuracy and weight attributable to each model (Tebaldi & Knutti, 2007). Besides, a populated ensemble including several models is necessary to infer an accurate probability distribution function, and this requires a large amount of resources (computational, personnel, etc.) that may not be available. A similar argument could be made for consolidative v. multi-scenario analysis, since scenarios are typically the result of model predictions (e.g., climate models, environmental valuation models).

Therefore, rather than selecting those models/scenarios that better predict or using a weighting approach, which may artificially reduce uncertainty (Hino & Hall, 2017), this work considers multiple scenarios/models and an un-weighted approach. The result is a database that offers information on uncertainty regarding model design through the ensemble spread, as well as on scenario uncertainty through exploratory modeling. It has been argued that when "probabilistic information is not considered, each potential vulnerability is equally important on the overall robustness, which can also be interpreted as an implicitly equal weighting" (Taner et al., 2019). Yet, as noted above, in our case we cannot claim that each scenario/model has an equal weight, because these weights are essentially unknown. In this context, robustness is advised in decision making, so to minimize potential regret.

A robust decision can be informed through heuristic (i.e., inductive reasoning, building on the expertise of decision makers) and/or mechanistic methods. Since the design of the PWS in the Reno River LRIB is still in an exploratory stage, and a formal and structured discussion of the results that allows to articulate heuristic-based robust decision methods is ahead in time, this research adopts two widely used mechanistic robust decision algorithms to inform PWS performance: Minimization of maximum loss (Minimax) and Minimization of Maximum regret (MinMax regret; Aissi et al., 2009), two conservative decision making approaches that choose the strategy that minimizes the potential loss (Minmax) and regret (Minmax regret) under the models and scenarios considered.

4. Results

4.1. Simulation Results

The methods proposed above are used to assess the performance of the conservationist strategy (i.e., no irrigation modernization and reallocation of water toward the environment) v. autonomous adaptation strategy (i.e., irrigation modernization and reallocation of water toward agriculture) under multiple scenarios and models. The upshot is a database of simulations informing on the expected irrigators' choices and related economic performance of the two strategies (including profit, employment, Gross Value Added and, most notably, utility). Figure 3 informs on the crop portfolio choices of irrigators under alternative climate change scenarios/reductions in water allocations, for the conservationist strategy (Figure 3a) and the autonomous adaptation strategy (Figure 3b), in each model of the ensemble. Note that since irrigation modernization costs/PWS are charged/paid through a flat rate on a per hectare basis, they do not alter the relative position in terms of utility return among alternative crops, and do not affect crop portfolio responses.

Irrigation modernization under the autonomous adaptation strategy increases the proportion of beneficial consumption per unit of water allocated for irrigation, which increases efficiency (eff_i) and allows irrigators to reduce the gross water needs of each crop i, or w_i/eff_i (see Equation 6), and thus negate/reduce the impacts of climate change and water scarcity on yields. As a result, under the autonomous adaptation strategy, the crop portfolio remains largely stable (although marginal changes are observed) under all scenarios and models, until water allocation is reduced by >30%. When water allocation is reduced by 30% or more, those irrigated crops with a lower utility return (mostly corn) are partially replaced by rainfed cereals (mostly wheat or barley, depending on the model). On the other hand, under the conservationist strategy where no irrigation modernization plan is implemented, the reduction of water allocation constrains water availability and leads to a substitution of relatively low return irrigated crops (corn, sugar beet and pasture) by rainfed wheat or barley (depending on the model) from the onset (>0% water allocation reduction). The surface of high return irrigated crops (vegetables, fruit
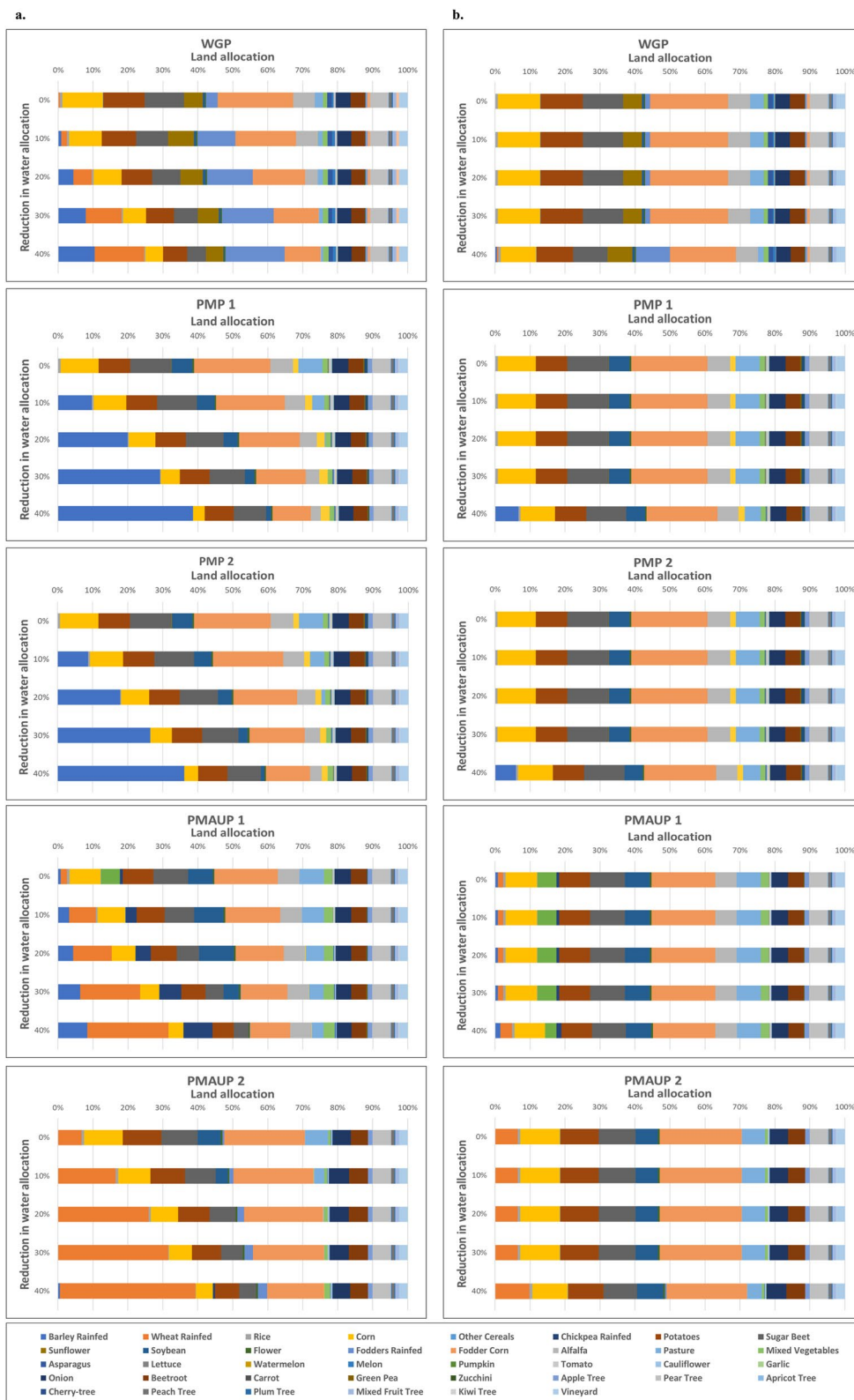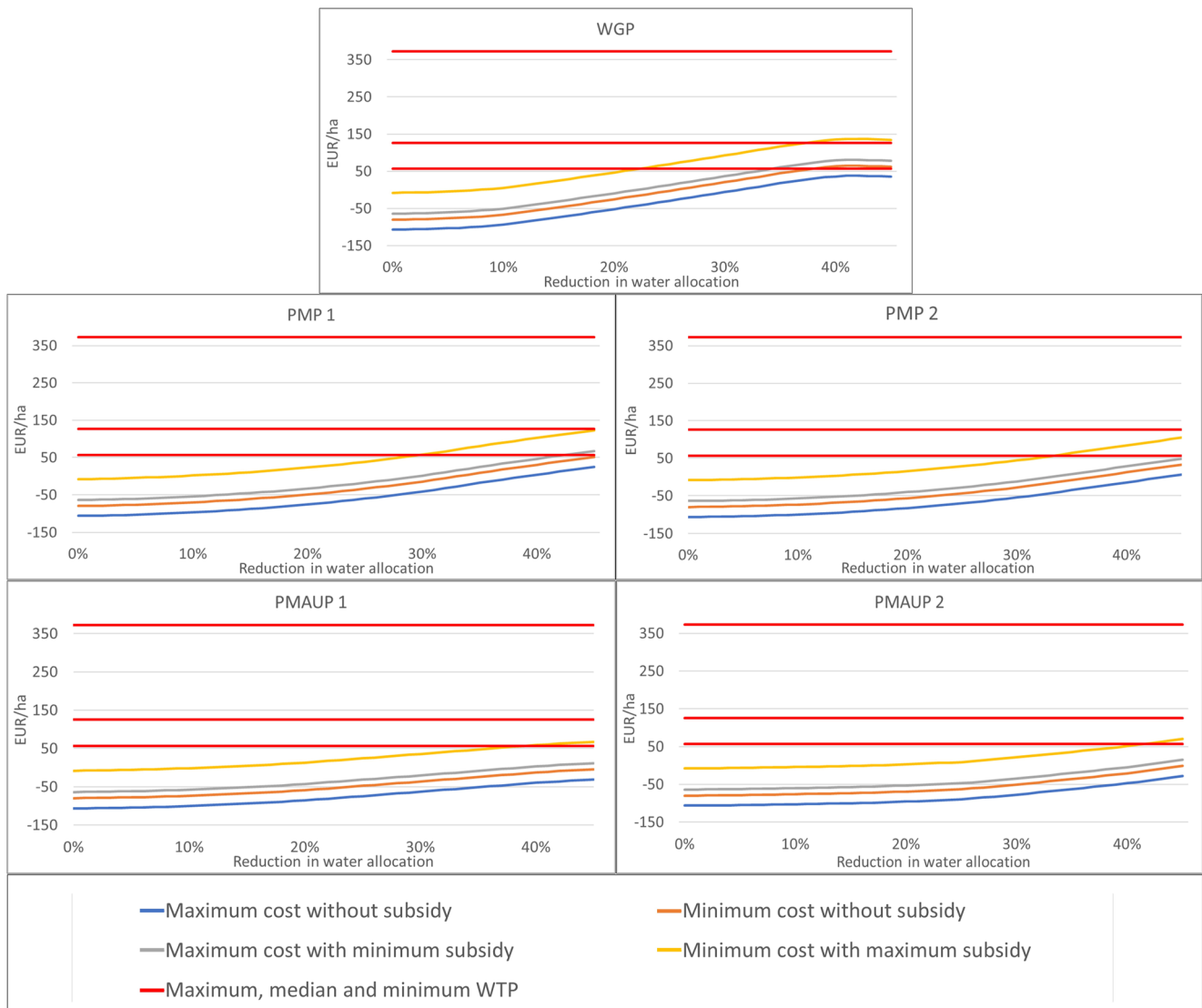
Figure 3. Crop portfolio choices: conservationist strategy (Figure 3a) v. autonomous adaptation strategy (Figure 3b).

Figure 4. Willingness to accept v. Willingness to pay for PWS in the Reno LRIB for (a) each model of the ensemble and (b) all models of the ensemble.

trees) remains constant in the initial simulation steps (<20% water allocation reduction); and is progressively substituted by less water intensive irrigated and/or rainfed (wheat or barley) crops with a lower return when water allocation is reduced by >20%.

Figure 4 assesses the economic performance of the PWS scheme by comparing the willingness to accept (WTA) to the willingness to pay (WTP) for an hypothetical PWS in the Reno LRIB, under alternative combinations of scenarios and models. The WTP is obtained from a literature review using benefit transfer methods (see Annex VII in the Supporting Information S1). The WTA is obtained in two stages using mathematical programming methods. First, we use the utility function calibrated in Equations 1–6 to calculate the monetized foregone utility (through the compensating variation, CV) experienced by irrigators in every possible scenario g under each strategy s, as follows:

$$\text{CV}_{g,s} = \frac{e\left(U_{g=0,s}, W_g\right)}{\text{Surface}} \tag{7}$$

Where e is an expenditure function representing the minimum amount of money agents would need to attain the utility level they experience in the baseline scenario $g = 0$ ($U_{g=0}$), where there is neither climate change nor irrigation modernization, starting from an alternative scenario g.
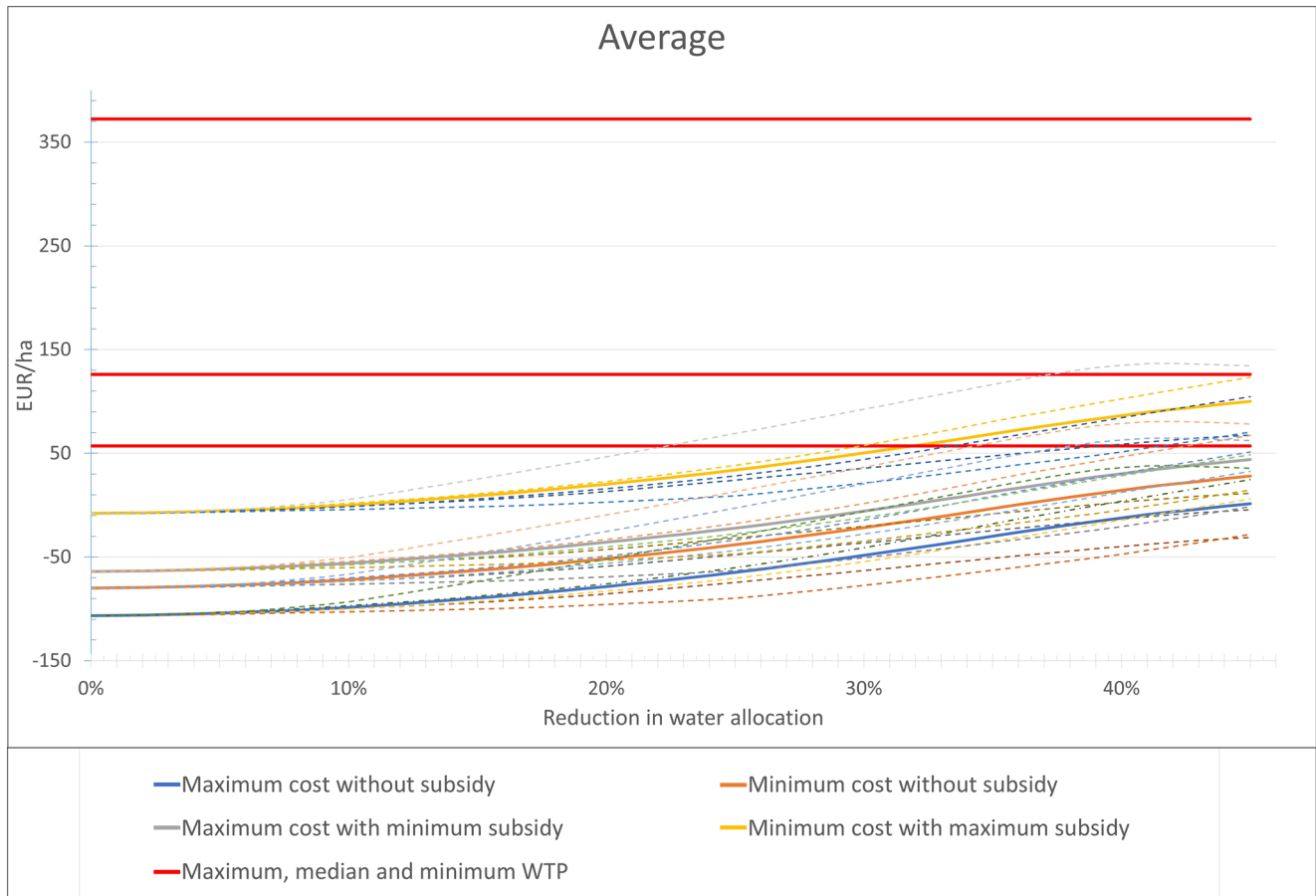
Figure 4. (Continued)

Second, by comparing the CV of the autonomous adaptation strategy ($s = A$) v. conservationist strategy ($s = C$), the WTA in every possible scenario g is obtained as follows:

$$WTA_g = CV_{g,C} - CV_{g,A} \tag{8}$$

A positive WTA denotes a preference for the autonomous adaptation strategy; while a negative WTA indicates that irrigators would experience a higher utility loss from the adoption of irrigation modernization v. the conservationist strategy, and therefore the latter would be preferred even in the absence of pecuniary compensations to irrigators through PWS.

When reductions in water allocation are null or low (<9%), the WTA is negative for all models and scenarios considered. At this stage the costs of irrigation modernization exceed the benefits from additional water availability, which is mostly used to irrigate crops with a low utility return. This results in a lower utility and higher CV under the autonomous adaptation strategy relative to the conservationist strategy, and a negative WTA. As climate change strengthens the water allocation constraint and threatens the irrigation of crops with a higher return, utility (CV) under the autonomous adaptation strategy increases (decreases) relative to utility (CV) under the conservationist strategy, and the WTA for the PWS scheme increases. Beyond a water allocation reduction of 9%, the WTA starts exceeding a value of 0 EUR/ha for some models and scenarios—meaning that irrigators will demand a compensation for not adopting irrigation modernization and conserving the Reno LRIB green infrastructures and ecosystem services instead. Beyond a water allocation reduction of 22%, the WTA starts exceeding the lower threshold of the WTP for some models and scenarios, indicating that irrigators would prefer the autonomous adaptation to the conservationist strategy even in presence of a pecuniary compensation (minimum WTP) through PWS. Eventually, the median WTP is also surpassed, although this only happens for severe climate change
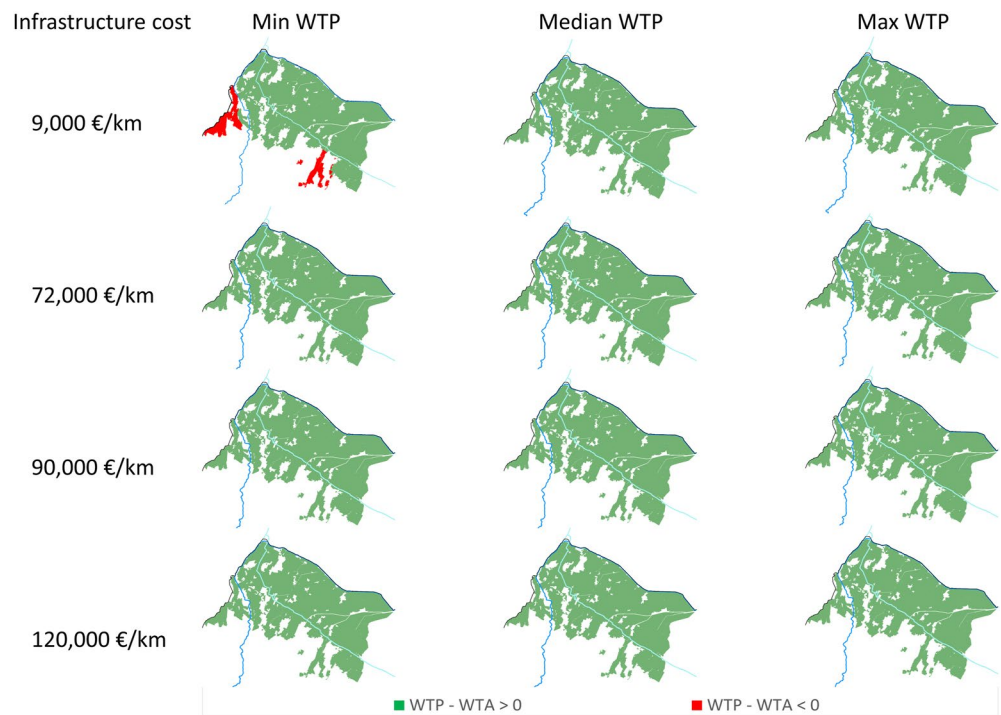
Figure 5. Economic performance (measured as WTP minus WTA, in EUR/ha) of PWS and spatial asymmetries. (a) 25% of water allocation reduction; (b) 40% of water allocation reduction.

scenarios (water allocation reduction >38%) and only in a few models and irrigation modernization scenarios. No combination of models and scenarios leads to a WTA that exceeds the maximum WTP.

Importantly, in the absence of irrigation modernization subsidies the WTA only becomes positive beyond a water allocation reduction of 26%; and exceeds the minimum WTP beyond a water allocation reduction of 35%. Neither the median nor the maximum WTP are surpassed by the WTA in those scenarios without irrigation modernization subsidies.

Figure 5 compares the WTP to the WTA for each agent/irrigation district under selected scenarios. This information is relevant to identify and redress potential asymmetries in the implementation of PWS (e.g., through direct payments to those who experience losses and/or water reallocations among farmers). The WTA is calculated here as a simple average of all the models in the ensemble—sometimes referred to as best estimate (IPCC, 2014).

Those irrigation districts with a larger share of high return crops (mostly vegetables) are more likely to experience losses from the adoption of PWS, especially under severe climate change scenarios. Under a water allocation reduction of 40%, the irrigation district C004 is better off adopting the autonomous adaptation strategy, provided infrastructures are subsidized and the minimum or median WTP is paid. The irrigation district C005 is better off adopting the autonomous adaptation strategy even if no subsidy toward the adoption of modern irrigation infrastructures is paid.

4.2. Robustness

Robustness is assessed using two mechanistic robust decision algorithms: Minimax and Minmax regret.

The performance indicator used in the case of the Minimax algorithm is the gain/loss experienced in each scenario as compared to the baseline scenario g = 0 (no climate change, no irrigation modernization). Under the autonomous adaptation strategy, the Minimax performance indicator equates $-CV_{g,A}$; while under the conservationist strategy, the Minimax performance indicator equates $WTP - CV_{g,C}$. The Minimax performance indicator is obtained for every model, scenario and strategy considered. Subsequently, for each of the strategies, the
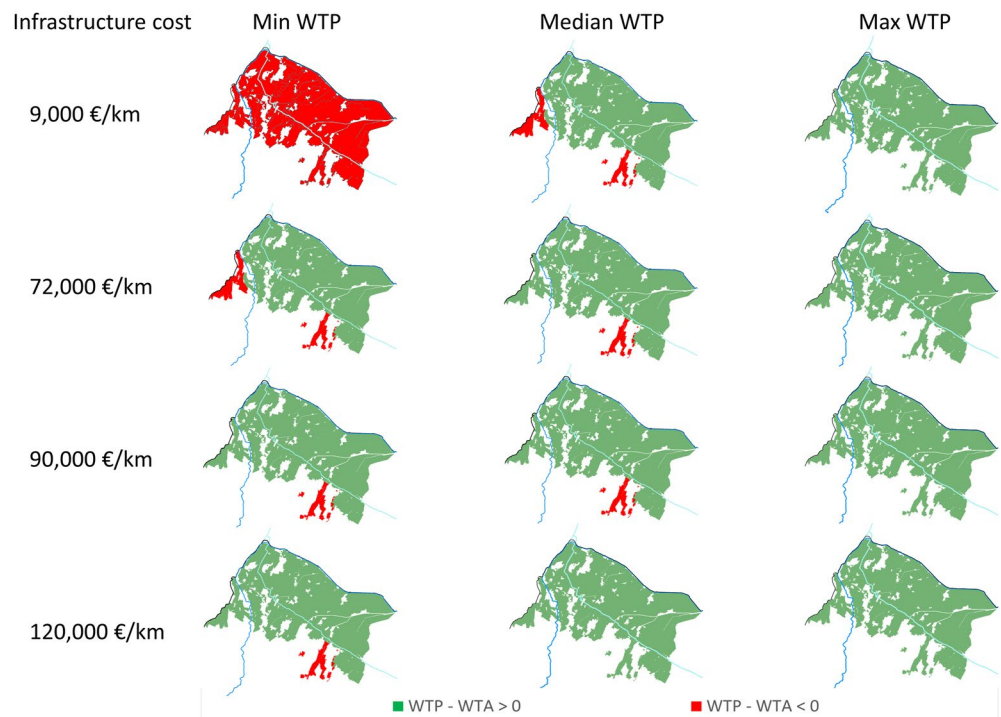
Figure 5. (Continued)

maximum loss is obtained across all models and scenarios. The strategy that minimizes the maximum potential loss (Minmax) is found to be more robust; which in our case is the conservationist strategy.

The performance indicator used in the case of the Minmax regret is obtained in two steps. First, the best performance indicator for each scenario among the Minmax performance indicators (see above) is obtained. Then, a regret indicator is obtained by subtracting actual gain/loss (again obtained as $-CV_{g,A}$ for the autonomous adaptation strategy, and as $WTP-CV_{g,C}$ for the conservationist strategy) from the best performance indicator under a given scenario. The strategy that minimizes regret is then chosen; which is found to be, again, the conservationist strategy.

4.3. Discussion

Farmers are in charge of managing the land and are given the responsibility to protect it. An important part of this stewardship role involves the conservation of natural resources, such as water, and the environmental assets and ecosystem services that depend on them. Historically, landscape stewardship has included environmentally friendly adaptive strategies such as no-till, planting cover crops, collecting water runoff to reduce nutrient load to water bodies, integrating crop and pasture rotations, and others. However, without adequate rules and incentives, adaptation strategies may as well be unsustainable (e.g., water theft, aquifer overdraft).

We show that as climate and water resource allocations change, farmers may decide to deploy modern irrigation systems that increase agricultural water consumption to mitigate/negate production losses, while reducing water availability for other uses—including valuable ecosystems and their services. This autonomous adaptation strategy is being encouraged by ill-designed incentive schemes—most notably the subsidization of modern irrigation systems (Perry & Steduto, 2017). In our study in the Reno LRIB we find that, under mild to moderate climate change scenarios (water allocation reduction <26%), removing infrastructure subsidies is sufficient to prevent maladaptation through modern irrigation technologies that deplete environmental water allocations. This critical result aligns well with research at the global (Pérez-Blanco et al., 2020) and private irrigator level (Adamson & Loch, 2021), which has shown that when subsidies are removed, expected water savings are often insufficient on their own to motivate private irrigation modernization investments.

This is likely to change under moderate to severe climate change scenarios, when water scarcity will affect increasingly valuable crops and new schemes of incentives such as PWS may be necessary to prevent autonomous (mal)adaptation. In our study in the Reno LRIB we find that, if no infrastructure subsidies are applied, PWS would have a cost (measured through the WTA) that is below the minimum WTP for irrigation-dependent ecosystem services until a water allocation reduction of >35%; and below the median and maximum WTP for all scenarios considered. In the presence of subsidies, the cost of PWS can exceed the minimum WTP earlier (water allocation reduction >22%), while the median WTP can be also surpassed, albeit only at severe climate change scenarios (water allocation reduction >38%). This suggests a satisfactory economic performance of PWS for most scenarios and models considered. Applying automatic robust decision-making methods, the PWS is found to be a more robust strategy than the adoption of modern irrigation systems.

Our findings have relevant implications for water policy design in the EU and in other areas where modern irrigation systems are being adopted, often with the support of public subsidization programmes (Pérez-Blanco et al., 2020). In the EU, the Common Agricultural Policy (CAP) subsidizes up to 90% of the investment cost of modern irrigation systems, on the basis of a supposedly higher environmental performance of these technologies (Official Journal of the European Union, 2013). According to CAP reasoning, if an irrigation system *X* with 50% technical efficiency is substituted with an irrigation system Y with 75% technical efficiency, the original water needs can be satisfied with a fraction (50/75) of the original water applied (e.g., 100 v. 66.67 units of water), and therefore 33.33 units of water will be saved. This confuses water applied with water consumed and assumes that economic agents will behave the same way (planting exactly the same crop portfolio) after the modernization, as before—two widespread but incorrect assumptions among policymakers. In reality, unless water use is curtailed following the adoption of modern irrigation systems, we should expect the farmer to increase consumption, reduce return flows, and limit water availability for third party uses, including green infrastructures, so to increase farm income (Grafton et al., 2018; Pérez-Blanco et al., 2020; Perry & Steduto, 2017). Therefore, subsidies to modern irrigation technologies are not only ineffective to save water—they can also exacerbate water scarcity by increasing the consumed fraction of water applied.

In light of the overwhelming available scientific evidence showing that modern irrigation systems increase consumption and aggravate scarcity, why do policymakers continue to subsidize them to save water? First, despite the growing consensus among scientists that modern irrigation systems increase water consumption (Grafton et al., 2018), there is a widespread belief among non-experts that modern irrigation systems will save water. Once a belief has been established, individuals are more likely to accept (or even build) arguments that conform to that belief (Nickerson, 1998; Shermer, 2011), even when more recent information discredits it (Johnson & Seifert, 1994). This makes very challenging to debias and debunk the belief that modern irrigation systems will save water (Lewandowsky et al., 2012), particularly among policymakers that are not familiar with the behavioral drivers explaining farmer's responses to modern irrigation systems. Second, those who benefit from modern irrigation systems (e.g., adopting farmers, equipment suppliers) exert political pressure and other lobbying efforts to obtain public subsidies that develop new resources and increase farm income—often at a marginal cost that exceeds marginal value. This is visible in our study in the Reno River LRIB, where in the absence of public subsidies, the autonomous adaptation strategy is preferred to the conservationist strategy only under severe climate change scenarios (water allocation reduction of >35%); while with subsidies, modern irrigation technologies are preferred to PWS following a water allocation reduction of >22% (moderate climate change).

Since modern irrigation systems worsen rather than alleviate water scarcity, it is necessary that policymakers abandon the preconceived idea that these technologies will almost always save water and start adopting new frameworks that contribute to align individual farmer choices with collectively agreed policy goals, such as alleviating water scarcity while mitigating and potentially reverting income losses under climate change (i.e., sustainable growth). A prerequisite to achieve this goal is to conduct debiasing and debunking exercises among policymakers to put to rest the belief that modern irrigation systems will almost always save water (examples of debunking and debiasing exercises are available in Cook et al., 2018; Lewandowsky et al., 2012; Linden et al., 2017). Additionally, achieving sustainable growth under increasing water scarcity necessitates sensible water reallocations that conform to basic economic principles, including:

- The theory of economic policy, which argues that in order to meet a number of goals, an equal number of instruments are necessary (Tinbergen, 1952). Thus, if the objective is to save water (objective 1) while

enhancing/protecting farm income (objective 2), two instruments will be necessary (e.g., decoupled subsidies to farmers to enhance/protect income and quotas to save water)

- The Assignment Principle, which argues that each instrument should be assigned to the objective to which it is best suited, and that this instrument should not be used to pursue another objective (Mundell, 1962). The Assignement Principle complements Tinbergen's (1952) work and can be interpreted as a warning against water panaceas or "win-win" solutions, where a single instrument is adopted to pursue two (often conflicting) objectives

- A framework for the effective design of interventions, where the objectives and the instruments set by policymakers do not directly affect behavioral responses (in our case, the decision of whether to adopt or not modern irrigation systems; Ciriacy-Wantrup & Bishop, 1975). For example, instead of subsidizing modern irrigation systems (which directly affects behavioral responses by promoting the adoption of modern irrigation technologies by farmers), policymakers should set the objectives to be met (e.g., ecological flows) and the instruments to achieve them (e.g., quotas, pricing), and let farmers respond to these new conditions through changes in inputs and technology (e.g., reduced water use, modern irrigation technologies)

These basic economic principles suggest that the failure of modern irrigation systems to save water is the consequence of flawed policy design. Policymakers promoting subsidies to modern irrigation systems "talk" about saving water but "dream" about increasing production (Connell, 2007), and thus violate the Tinbergen Principle (two objectives, one instrument). Moreover, scientific evidence shows that there are "much more cost-effective" alternatives to save water than modern irrigation systems (Qureshi et al., 2011), such as quotas or pricing, meaning modern irrigation systems also violate the Assignement Principle. Finally, coupled subsidies such as subsidies to modern irrigation systems directly affect the operational decisions by farmers, instead of setting objectives and instruments farmers have to comply/deal with. These basic economic principles further underpin the findings obtained using our quantitative framework, namely, that the conservationist strategy (sometimes complemented with PWS) has a superior economic performance than the (subsidized) autonomous adaptation strategy where modern irrigation systems are adopted under most models and scenarios, and is also more robust.

Does all the above mean that modern irrigation systems are always ineffective toward saving water? No. Modern irrigation systems can yield savings while protecting and/or enhancing agricultural income if they are complemented with effective water saving policies (such as quotas or pricing) that strengthen water allocations to farmers, so to ensure that any additional agricultural consumption following the adoption of modern irrigation systems is equal or lower than the foregone non-beneficial consumption and non-beneficial return flows (see Figure 1). Under conventional return flow regimes where return flows are beneficial, the water savings and/or additional farm income that can be achieved this way are rather marginal, and typically do not justify investments in modern irrigation systems (Adamson & Loch, 2021). This is not the case under (infrequent) escape flow regimes where return flows are non-beneficial and can be appropriated by farmers at no economic cost (i.e., higher farm income without reducing water availability to third party users; Huffaker, 2008).

5. Conclusions

This paper develops a multi-model and multi-scenario method to assess irrigators' responses to, and the economic performance of, pecuniary compensations designed to sustain irrigation-dependent ecosystem services through PWS. We find that, under most models and scenarios, the conservationist strategy (sometimes complemented with PWS) has a superior economic performance than the autonomous adaptation strategy where modern irrigation systems are adopted. The conservationist strategy is also found more robust than the autonomous strategy.

We envision several ways in which our model and research could be improved. First, the ensemble of mathematical programming models used in this paper could be expanded by including other models available in the literature, so to more thoroughly sample uncertainty. Second, a sensitivity analysis is also warranted to further sample uncertainty. This could be done exploring additional attributes in the multi-attribute models, PMAUP 1 and PMAUP 2; or considering alternative crops and adaptation strategies in the portfolio, for example, allowing for continuous yield functions and deficit irrigation instead of fully irrigated v. rainfed crops (Graveline & Mérel, 2014). Third, our ensemble focuses on the microeconomic aspect of a human system, which is one of the two components of complex human-water systems. Future research should study the interconnection of our human system multi-model ensemble with existing multi-model ensembles that represent the water system (see

e.g., Cloke et al., 2013). This would make possible to model the two-way feedbacks between human and water systems and their consequences (e.g., how the value of ecosystem services increases/decreases during periods of scarcity/abundance; Sivapalan et al., 2014), and sample the cascading uncertainties across them. Fourth, the representation of the human system could be also enhanced by modeling the interconnection between the micro level explored in this paper and the macro level, for example, through price feedbacks, which can be modeled endogenously in macroeconomic models (Parrado et al., 2020). Finally, efforts to produce more comprehensive and robust models should be complemented with debiasing and debunking exercises—a prerequisite to overcome locked-in policy failures that subsidize ineffective water saving policies (Cook et al., 2018; Lewandowsky et al., 2012).

Data Availability Statement

The data used for the calibration of the ensemble of mathematical programming models and for the benefit transfer exercise is available free of charge at the online supplementary material and in an online repository at the following link: https://doi.org/10.5281/zenodo.5578968.

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