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Abstract

The paper presents a rapid assessment of irrigation full cost of the Pinios Local Organization for Land Reclamation. The individual cost components (financial, environmental and resource) were estimated using the best available data and sound methodological choices. On the basis of our estimates it seems that water scarcity and its corresponding resource cost is quite important issue to be ignored. The scarcity rents falls within a range from 21% to 39% of water full cost, while the environmental cost is about 8%. The policy implications of these results are also discussed.

Keywords: WFD, Irrigation Full Cost Account, With-Without Analysis, Water Balance, Water Scarcity Rents, Pinios Local Organization for Land Reclamation.

JEL Classification: Q25, Q51,Q52

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1) Introduction

Until the advent of the Water Framework Directive (henceforth WFD), European water legislation was substantially fragmented with notable contradictions and conflicts. The radical reform of European water policy brought about by the WFD was an answer to a quickly progressive political, economic and social context. This is reflected in the rhetoric of the WFD where political actors have been substituted by stakeholders, citizens by consumers, while there is an increasing emphasis on water as an economic good which should be managed accordingly (Kaika, 2003).

The main objective of the WFD is to achieve a "good ecological status" of the European waters by 2015 (Borja and Elliott, 2007). The term European waters collectively refers to groundwater, surface waters, transitional waters, and coastal waters. The term "good ecological status" is perceived as a deviation from a reference point. The notable innovations of the new policy regime include three issues. First, water quality or "good ecological status" is not anymore determined by chemical criteria but the reference point for such comparisons is or should be biological (Bouleau, 2008; Carballo et al., 2009; Katsiapi et al., 2012). Second, water resources should be managed in such a way that consumers should bear the full cost of water uses (Unnerstall, 2007). The final innovation of the WFD refers to the crucial issue of public participation and its relevance to policy acceptability (Demetropoulou et al., 2010; Liebeskind, 2005).

The analysis of full cost recovery is generally accepted as a step towards a sustainable water management regime which may incorporate the use of economic principles such as effective pricing derived from cost recovery assessments (EUREAU, 2004). However, the full cost recovery of irrigation through effective pricing is not a desk exercise, but it has to be implemented within the local institutional, political and social constraints. Dinar and Mody (2004) argue that pricing water can be an effective tool in demand management only when appropriately implemented and regulated. In particular, the authors stress that, *inter alia*, institutional reform must ensure water pricing acceptability and transparency in water resources management.

Furthermore, it should be emphasized that the effectiveness of cost recovery pricing is enhanced if it is used in conjunction with supplementary measures (Howarth, 2009). Such measures may include water saving adjustments (devices and practices), reduction of water losses in the production supply-distribution systems and or education and public information campaigns towards prudent water use.

The objective of this paper is to provide a rapid way to assess the irrigation full cost of a local organization for land reclamation (LOLR) in central-northern Greece. The structure of the paper is as follows. The next section presents the concept of irrigation total cost. Section 3 gives a brief description of the study area, while section 4 presents the methodology and the assumptions used. The full cost estimates are summarized in section 5 and finally section 6 outlines the main conclusions of the paper.

2) Materials and Methods

2.1 On the Concept of Full Cost of Irrigation Water

According to the WFD, the main principle that European Union (EU) member-states could use, in order to achieve the stated policy objectives, is the total cost recovery of water uses. Total cost includes three components: the financial, the environmental and the resource cost.

2.1a Financial Costs

Financial costs or direct project costs according to Easter and Liu (2005) are the easiest of the three to measure, and most projects take only direct costs into account in determining cost recovery. Financial costs comprise the costs of providing and administering water services, which can be either fixed or variable. The fixed financial costs are independent of the water volume and concern the use of the fixed inputs. Specifically, they include likely rents, wages of permanent employees, fixed fees, investments and the annual expenditures (depreciation, insurance rates and interests) of the fixed assets. On the contrary, variable financial costs include all the expenses that are related to the amount of water delivered. That category includes seasonal labor costs, the value of the raw material, the variable expenditures, taxes and fees etc. Although the distinction of financial costs between fixed and variable is irrelevant in terms of cost recovery, it plays a crucial role when someone is examining water pricing methods. The financial costs do not include any accounting of the past investments in irrigation networks since they represent sunk costs (Molle and Berkoff, 2007). Ward and Michelsen (2002) explain the economic principle for ignoring sunk costs in water valuation.

2.1b Environmental costs

Environmental costs represent the damage cost that irrigation imposes on the environment and ecosystems (e.g. aquifer exhaustion by over-abstractions, increased water pollution, increased soil erosion). Baldock et al (2000) provide a comprehensive list of these impacts across Europe.

There are various methods suitable for assessing environmental costs. The relevant literature on nonmarket valuation techniques is voluminous and increasingly sophisticated. These methods are usually classified as: a) cost-based valuation methods, b) revealed preference methods and c) stated preference methods (Pearce et al., 2006). Arguably, the stated preference methods represent the state of the art in the relevant literature. Recent examples of authors employing stated preference methods for assessing the environmental costs of water resources are Birol et al (2006) who used the contingent valuation method (CVM), while Martin-Ortega et al. (2011) used choice experiment (CE).

However, there are two major drawbacks of the stated preferences methods. The first one is related with the issue of aggregating the survey results across people which may lead to non-credible estimations (Pearce, 2000), whereas the second, which is particular relevant to our case study, is that those methods elicit estimates of people willing to pay for discrete classes of environmental quality (see Kataria et al (2012)). This modelling feature renders the continuous and monotonic transformation of causality extremely difficult, if not impossible.

In this paper, the environmental cost is based on the assessment of the relevant damage cost which is imposed to society by irrigation. The damage cost method simply assesses the likely costs of restoring the resource (groundwater) to baseline conditions in order to measure the benefits of reduced environmental costs. Damage costs are almost less than willingness to pay (WTP) estimates and very often it is easier to communicate them to non-economists (Dickie, 2003).

Drawing on Ando and Khanna (2004) we calculate the environmental cost of irrigation as the damages for groundwater contamination using the formula:

$$EC = V_1 * AC * \alpha \tag{1}$$

where EC is the environmental cost, V_1 is the volume of contaminated water, AC is the average cost of lost groundwater services and α is the implicit contribution of irrigated agriculture to groundwater contamination. The AC is defined as the difference between the average cost of water treatment minus the average cost of supplying drinking water. A variant of formula (1) is used for calculating damages for groundwater contamination by the state of Minnesota (Ando and Khanna, 2004).

A possible bias of the previous method, which usually leads to overestimated damages, stems from the implicit assumption that all groundwater stock is used for municipal water supply. Furthermore, environmental damages are assessed on the basis of the whole volume of contaminated water, while there might be situations under which only a fraction of the contaminated water is extracted per year. Consequently, in order to address the previous problems we may modify formula (1) as:

$$EC = V_2 * \alpha * \beta * TC \tag{2}$$

where V_2 is the volume of groundwater extracted which needs to be treated, β is the proportion of the LORL area over the sub-basin area where it belongs and TC stands for the average water treatment cost (nitrogen removal). Equation (2) encompasses two assumptions that need to be spelt out. First, the aquifer is uniformly recharged from its above area and second, the whole agricultural land of the river basin follows similar cropping pattern as the case study.

2.1c Resource costs

Resource costs represent the costs of foregone opportunities that other uses suffer due to the depletion of the resource beyond its natural rate of recharge or recovery. The term “resource costs” in the WFD jargon is equivalent to the term scarcity rents in resource economics. The standard definition of scarcity rents refers to the present value of future sacrifices associated with current use of a scarce resource (Moncur and Pollock, 1989). The estimation of the resource cost requires the estimation of the water balance in the study area. If there is a water deficit (or it is anticipated) then a resource cost exists. According to Moncur and Pollock (1988) the scarcity rent can be assessed using the following formula:

$$SR = \frac{C_2 - C_1}{e^{r(T-t)}} \tag{3}$$

where SR stands for scarcity rents, C_1 is the marginal extraction cost until some time T, C_2 is the marginal extraction cost beyond T where the public utility must use an alternative source of water (backstop technology) to cover the anticipated water deficit and r is the discount factor. If water deficit is already present then the unitary scarcity rents is given by the cost of backstop technology. Economists have long held the rule that the efficient pricing of water should include scarcity rents (Howe, 1979; Koundouri, 2004; Griffin, 2006) so that:

$$p = MC + SR \tag{4}$$

where p is the water price that achieves efficiency and MC is the marginal cost of water supply. Therefore the resource cost of irrigated agriculture is estimated as the product of the water deficit times the unit cost of the best alternative source (recycling, water treatment, etc).

3) The Study Area

The aim of this paper is to assess the full cost of irrigation water of a predominant irrigated area in Greece. A collective irrigation network in northern–central Greece, Thessaly, was chosen as case study. The irrigation network is managed by the Local Organization of the Land Reclamation (LOLR) of Pinios. The total area serviced by LOLR is 19,294.2 ha. The boundaries of study area are given in the Map 1.

INSERT MAP 1 ABOUT HERE

According to the WFD the appropriate spatial scale for designing management measures for water resources is the river basin, the so called river basin management plans (RBMP). While the river basin scale seems incontrovertible for aggregating the environmental impacts of irrigation, accounting for complex administrative and socioeconomic realities may point to a different level of analysis. In terms of this study we restricted our analysis to the local irrigation network level. The reason for such a modeling choice is that LOLR represents the lowest administrative unit for designing and applying a water policy. Put it another way, we opted for transparency and equity in the water policy design (i.e. water pricing) which typically follows a full cost assessment by sacrificing some of the accuracy of our estimates. Besides, as Keessen et al (2010) have pointed out the WFD allows notable flexibility among EU member states to follow different approaches concerning the implementation of the Directive.

INSERT TABLE 1 ABOUT HERE

The major crops of the study region, in terms of land use, occupy 92.1% of the total area (Table 1), with cotton being the most important crop grown. It is clear that the area is a predominant irrigated one, since land allocated to irrigated crops amounts up to 88.5% of the total area. Irrigation requirements are mostly covered by the adjacent Pinios River (about 97%) and only a tiny proportion comes from groundwater.

4) The Methodology and Assumptions for Assessing the Full Cost of Irrigation

4.1 Calculation of financial cost

Assessing the financial cost a straightforward procedure since all the necessary data are, or should be, contained in the annual account of the LOLR's budget. In terms of this study, annual account for the year 2010 was obtained by the local authorities. The budget was checked for consistency against previous budgets, while a thorough check for accuracy was also performed, by examining whether the main expenses were included. The total figure accounted for 1,584,051.63 €. Fixed expenses represented the 63.5% while the variable expenses the 36.5% of the total financial cost.

4.2 Calculation of Environmental cost

Estimating the environmental cost is arguably the most difficult and most controversial part of the full cost assessment. As it was said before, we adopted a damage based method which requires an assessment of the likely environmental impacts of irrigation. As it is standard in the relevant literature we focused on nitrate leaching and also on drainage which transfers the nitrates to water bodies (Papaioannou et al., 2010). Other sources of water pollution were ignored, since nitrate pollution is the major driver of groundwater deterioration in the region (Ioannou et al., 2009; Stamatis et al., 2011).

The modelling framework for assessing the environmental impacts of irrigation is given in Figure 1.

INSERT FIGURE 1 ABOUT HERE

The first step was to overlay the relevant land cover data with the soil map and digital elevation map (DEM) of the area. By doing so we were able to derive the spatial distribution of crop-soil combinations for the Pinios LOLR. The next step was to feed these data to the DNDC model in order to simulate the growth of the major crops for all soil types in the region (clay, clay-loam, silty-loam). DNDC is a bio-chemistry simulation model designed for agro-ecosystem analysis[1].

DNDC has been widely used for environmental modeling to predict crop yields and fluxes of carbon and nitrate from agricultural soils (Leip et al., 2008; Neufeldt et al., 2006). The model was fed with data concerning soil characteristics, climatic variables and typical agronomic management practices. The meteorological data needed for the simulations were rainfall and temperature (min and max) corresponding to 2010. The model was calibrated for the local conditions by adjusting the main parameters concerning crop yields in accordance with the production capabilities of the study area.[2]

For irrigated crops (cotton, maize and alfalfa), the DNDC simulations produced 40 observations per soil type, each corresponding to different irrigation and fertilizer possibilities. For rain-fed crops (wheat, barley), simulations produced 30 observations per soil type corresponding to different fertilizer programs. In turn, standard meta-modelling techniques were used to estimate the functional forms of the nitrate leaching and the average drainage per crop/soil combination (Bouzaher et al., 1993).

To avoid double counting and to isolate the impact of irrigated farming on nitrate leaching, we used the “with and without” analysis (Ward and Michelsen, 2002). The analysis works as follows. First, we assess the amount of nitrate leaching of the existing cropping pattern in the region which includes both irrigated and rain-fed crops. Then we exclude the irrigated crops from our exercise and assess the nitrate leaching of the study area as if it was cultivated only with rain-fed crops (wheat and barley). The difference between these two estimates gives the implicit contribution of irrigated farming to nitrate leaching.

Table 2 gives the total DNDC estimates for the nitrate load and the percolation from agricultural land for the existing cropping pattern in the study area.

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[1] DNDC (i.e., DeNitrification-DeComposition) is a computer simulation model of carbon and nitrogen biogeochemistry in agro-ecosystems (http://www.dndc.sr.unh.edu/)

[2] These parameters concern the maximum biomass, the grain, leaf, stem and root fractions of total biomass at maturity and the maximum leaf area index.

INSERT TABLE 2 ABOUT HERE

On the basis of these estimates, the existing cropping pattern results in an average nitrate leaching of 19.52 mg/l, whereas if we assume that the whole area was cultivated with rain-fed crops, we obtain an estimate of 17.98 mg/l. Therefore the implicit contribution of irrigated farming to nitrate leaching using the "with-without" rationale can be assessed as:

$$\alpha = \frac{\left(N_{in} - N_f\right)}{N_{in}} \tag{5}$$

where α stands for the implicit contribution of irrigated farming to nitrate leaching, N_{in} is the assessed nitrate leaching of the existing cropping pattern and N_f is the estimated nitrate leaching for the hypothetical case that the entire area was cultivated with rain-fed crops. The α in our case study was found to be 7.9%.

To date, there are a couple of nitrogen removal technologies available (Ahn, 2006; Birgand et al., 2007). Following Gratziou and Chrisochoidou (2011) we assume that the cost effective solution for nitrogen removal is the method which uses active sludge and denitrification (2 stages) which costs 1.3 €/m^3. According to the 2010 budget of the local public utility[3], the cost of supplying drinking water is 0.93 €/m^3. Therefore, the average cost of lost groundwater services is found to be 0.37 €/m3. Inserting these values into (1) together with the value of percolated water from Table 2, the environmental cost of irrigation in our case study is 206,725.08 €.

The urban water consumption of Larissa, the adjacent city to Pinios LOLR, is covered exclusively with groundwater. Local public utility extracts 21.3 millions m^3 from the underneath aquifer per year. Dividing the LOLR area with and the total agricultural area of the sub basin gives a value of parameter β equals 8.18%. Then, plugging these values into equation (2) the likely environmental costs of

[3] http://www.deyal.gr/oikonomika-stoixeia/proipologismos.html

irrigation for the Pinios LOLR according to the modified formula (2) are found to be 178,803.44 €.

4.3 Calculation of Resource cost

The first step for estimating the resource cost of irrigation in the study area is to construct a water budget analysis for Pinios LOLR. The fundamental principle of such an analysis is the conservation of mass (Weight, 2004). Figure 2 presents a simplified picture of the major inflows and outflows at a catchment level.

INSERT FIGURE 2 ABOUT HERE

The hydrological approach to water budget depicted in Figure 2 can be found in a number of papers such as Haie and Keller (2012), Chen and Zhao (2011) and Perry (2007). The main principle of such an approach is that total inflow equals total outflow. That is:

$$P + VA = RF + ET + \Delta S \tag{6}$$

where P is total precipitation, VA stands for the water abstracted from the river, RF is the return flows, ET is evapotranspiration and ΔS represents the change in groundwater stock. This change is given by:

$$\Delta S = DP - GA \tag{7}$$

where DP is percolation and GA is the volume of groundwater extracted.

Total precipitation can be decomposed as:

$$P = P_E + R_1 + DP_1 \tag{8}$$

where P_E is the effective precipitation, R_1 stands for the fraction of P that is returned to the river as runoff and DP_1 is the fraction of P which percolates. Effective precipitation is estimated according to the method developed by the USDA Soil Conservation Service as

$$p_e = \begin{cases} p_i \dfrac{125 - 0.2 p_i}{125}, & if \quad p_i \leq 250mm \\ 125 + 0.1 p_i & if \quad p_i \geq 250mm \end{cases} \tag{9}$$

where p_i and p_e are the monthly gross and effective precipitation respectively which were drawn from Larisa meteorological station. (Latitude: 39.38, Longitude: 22.25). Such a formula is widely used in the relevant literature (see Tsanis and Naoum (2003) and Loukas et al. (2007)). The fraction of precipitation which percolates for the local conditions falls within the range of 5% to 15% of the system's recharge (Koukidou and Panagopoulos, 2010). The midpoint of this range, 10%, it was assumed for this study.

The water diverted from the river, VA , and the water pumped from groundwater, GA, end up to the collective irrigation network:

$$VA + GA = W_A + CL \tag{10}$$

where W_A is the water available for irrigation purposes and CL represent canal losses. The quantity diverted from the river and the quantity of extracted groundwater are assessed by collecting data concerning the number of pumps which are active in the local irrigation network, the quantity of water supplied from each pump (in m^3/h), the power of each pump (in kW) and its energy consumption (in kWh). In particular, for each and every pump the time of operation was estimated on the basis of its energy consumption and its power, as:

$$operation\ \ time = \frac{energy\ \ consumption}{power} \tag{11}$$

Having estimated the time of operation and given the flow rate of each pump, the volume of water supplied from each and every pump was estimated as:

$$Water\ \ volume = operation\ \ time * flowrate \tag{12}$$

Trivial, albeit tedious, calculations of these data for all active pumps in the study area gave an estimate of 154,996,445 m^3. Surface water (Pinios river) is the main source of water supply accounting for the 97.6%, while groundwater contribution is negligible accounting only for the 2.4%.

Canal losses, CL, can be decomposed as:

$$CL = E_L + SE \tag{13}$$

where E_L are the losses through evaporation and SE stands for seepage losses which can be written as:

$$SE = R_2 + DP_2 \tag{14}$$

where R_2 represents the fraction of seepage which ends up to the river via runoff and sub-surface flow, while DP_2 is the fraction of seepage which percolates.

From field measurements, canal losses have been estimated to account for the 40% of water diverted into the irrigation network (Makropoulos and Mimikou, 2012). Evaporation represents 1% of these losses. The fraction of percolation was estimated using the aggregated infiltration rate for the area, which is 14.96% (Makropoulos and Mimikou, 2012).

Return flows, RF , equals the runoff from agricultural land plus $R_1 + R_2$. Equally, total deep percolation, DP, equals the deep percolation from agricultural land plus $DP_1 + DP_2$.

The evapotranspiration parameter in equation (6) was estimated as the aggregate daily crop evapotranspiration, ET_c , over all crops for the reference year:

$$ET_c = K_c ET_0 \tag{15}$$

where ET_c is the crop evapotranspiration (mm d^{-1}), K_c stands for the crop coefficient (dimensionless) and ET_0 is the reference crop evapotranspiration (mm d^{-1}).

The reference evapotranspiration was estimated using the FAO-56 Penman-Monteith method (Allen et al., 1988). Typical values for the crop coefficient K_c and the duration of the crop growth stages were taken by Papazafiriou (1999). The sowing day was determined according to the local agricultural practices. The aggregate ET_c estimates for the study area are given in Table 3.

INSERT TABLE 3 ABOUT HERE

In turn, Table 4 presents the major components of the local water budget which indicates a magnitude of water deficit of 5,821,030.08 m^3.

INSERT TABLE 4 ABOUT HERE

In turn, the resource cost of water uses was estimated by multiplying the water deficit with the unit cost of the best alternative source of irrigation water. Arguably, water treatment is the best available alternative source, given that desalination is out of question and water transfer is controversial and not politically accepted (Close, 1998; Margaris et al., 2006). Drawing on the “i-adapt” project, the (site–specific) unit cost of water treatment varies from 0.083 €/m^3 to 0.195 €/m^3 according to the available technical solutions and the likely irrigation needs (Makropoulos and Mimikou, 2012). Table 5 present the likely resource costs on the basis of the assumptions used.

INSERT TABLE 5 ABOUT HERE

On the basis of previous assumptions, Table 6 gives the estimates of the resource cost per different technical methods of overcoming the water deficit.

5) Results

By aggregating the financial, the environmental and the resource cost we can assess the likely total cost of irrigated agriculture in the study area. However, given that there are two scenarios for assessing the environmental cost and two scenarios for assessing the resource cost, the full cost assessment has to be calculated for the whole four scenarios. The latter is not particular useful, so we summarize our findings for the two polar cases, as the best and the worst scenarios.

The best scenario refers to the least cost (cost-effective) combination of the available methods for environmental and resource cost assessment. In particular, environmental cost is estimated using formula (2) and the resource cost involves water treatment without the construction of reservoir and restricted irrigation. The full cost estimate under the best scenario is 2,246,000.57 €. By contrast, the worst scenario corresponds to the most expensive combination of these methods, namely formula (1) for the environmental cost and the construction of reservoir and unrestricted irrigation for the resource costs, which results in an estimate of 2,925,877.57 €. It is noteworthy that the worst scenario is 30.3 % more expensive than the best scenario. The next figure displays the cost compositions of these polar cases, the best and worst scenarios.

INSERT FIGURE 4 ABOUT HERE

There are some interesting observations regarding Figure 4. First, the composition of the full cost of irrigation displays considerable variation across the scenarios examined. Not only is the absolute figure of the full cost conditional to the specific measures taken to improve water quality and reduce water scarcity but its composition is also affected. Second, water scarcity is clearly more important than the cost of contaminated groundwater. Third, the cost component with smaller variation is the environmental cost which accounts about 8%.

6) Conclusions

The paper presents a rapid assessment of irrigation full cost of Pinios LOLR. The individual cost components (financial, environmental and resource) were estimated using the best available data and sound methodological choices. On the basis of our analysis emerges that water scarcity and its corresponding resource cost is quite important issue that has to be addressed by local authorities. The scarcity rent falls into the range from 21.5% to 38.8% while the environmental cost is about 8% of the total irrigation cost. The typical farmers' adjustments to cope with water scarcity involve primarily deficit irrigations and private wells. Given that an increasing number of private wells are illegal, it seems that an unknown but arguably excessive pressure on groundwater is currently exercised. Such a situation represents a clear irrational obstacle towards effective management of water resources.

Finally, if we accept that Pinios LOLR has a balanced account, in the sense that the prevailing land-based pricing system is capable of recovering its financial cost, the existing pricing system achieves a cost recovery ratio of 70.5% under the best scenario, and only 54.1 % under the worst scenario. Presumably, such figures should trigger a policy respond towards a radical reform of the water pricing policy in order to restore efficient water uses in the region. Typical pricing reforms may include volumetric charges, on monitored or estimated water uses, and pumping taxes for the case of private wells. Nevertheless, the efficacy of such a pricing reform depends on the appropriate water supply management regime.

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Table 1: Major crops per soil type for the study area**)**

Crop	Soil type*	Area (ha)	Proportion of crop per soil type	Proportion of major crops over total land
Alfalfa	C	459.08	32.3%	
	CL	625.01	44.0%	
	SiL	336.29	23.7%	
Total		**1420.38**		8.0%
Winter wheat	C	430.33	29.6%	
	CL	640.97	44.1%	
	SiL	380.70	26.2%	
Total		**1,452.00**		8.2%
Corn	C	107.49	31.2%	
	CL	148.25	43.1%	
	SiL	88.49	25.7%	
Total		**344.24**		1.9%
Cotton	C	4,906.73	33.7%	
	CL	6,059.01	41.6%	
	SiL	3,593.21	24.7%	
Total		**14,558.94**		81.9%
Total land of major crops		**17,775.56**		92.13%
Total area		**19,294.20**		

* C:Clay, CL:Clay-Loam, SiL: Silty Loam

Source: Authors calculations on LORL data,

Table 2: Estimated drainage and nitrate load for the study area

	Cultivated area (ha)	NO_3 (kg/ha)	NO_3 (kg)	Average percolation (m^3/ha)	Total percolation (m^3)
Wheat (Clay)	430.33	3.28	1411.48	340	146,312.20
Wheat (Clay-Loam)	640.97	6.41	4108.61	350	224,339.50
Wheat (Silty-Loam)	380.70	10.79	4107.79	400	152,280.00
Total wheat	1,452.00		9627.88		522,9321.70
Maize (Clay)	107.49	14.68	1,577.95	384	41,276.16
Maize (Clay-Loam)	148.25	15.20	2,253.21	434	64,340.50
Maize (Silty-Loam)	88.49	21.89	1,937.05	529	46,811.21
Total maize	344.24		5,768.40		152,427.87
Cotton (Clay)	4,906.73	7.37	36,162.60	357	1,751,702.61
Cotton (Clay-Loam)	6,059.01	2.28	13,814.54	397	2,405,426.97
Cotton (Silty-Loam)	3,593.21	13.49	48,472.40	478	1,717,554.38
Total cotton	14,558.94		98,449.55		5,874,683.96
Alfalfa (Clay)	459.08	12.77	5,862.45	331	151,955.48
Alfalfa (Clay-Loam)	625.01	13.92	8,700.14	342	213,753.42
Alfalfa (Silty-Loam)	336.29	28.84	9,698.60	470	158,0546.30
Total alfalfa	1,420.38		24,261.19		523,765.20
Total	17,775.56		138,063.82		7,073,808.73

Source: Pinios LOLR and DNDC estimates

Table 3: ET_c estimation for the Pinios LOLR

Crop	Evapotraspiration (m^3/ha)	Land area	ET
alfalfa	6370	1420.38	9047820.6
cotton	5983	344.24	87110991
maize	6210	17775.56	2137730.4
wheat	2830	1420.38	4109160
TOTAL		17775.56	102,405,702.00

Table 4: The Water budget for the Pinios LOLR in Hm^3

Inflows		Outflows	
Effective precipitation	79.446	Evapotraspiration	103.103
Water diverted from the River	90.742	Return flow	58.927
Groundwater extracted	2.255	Percolation	16.311
Total	172.444	Total	178.265
Deficit		-5.821	

Table 5 Irrigation Resource cost in € per different technical options for overcoming water deficit

Irrigation Option / Technical Option	Restricted Irrigation		Unrestricted Irrigation	
	Unit cost (€/m^3)	Total (€)	Unit cost (€/m3)	Total (€)
with reservoir construction	0.141	820,765.24	0.195	1,135,100.86
without reservoir construction	0.083	483,145.50	0.089	518,071.68

Map 1: The Study Area of the Pinios LOLR

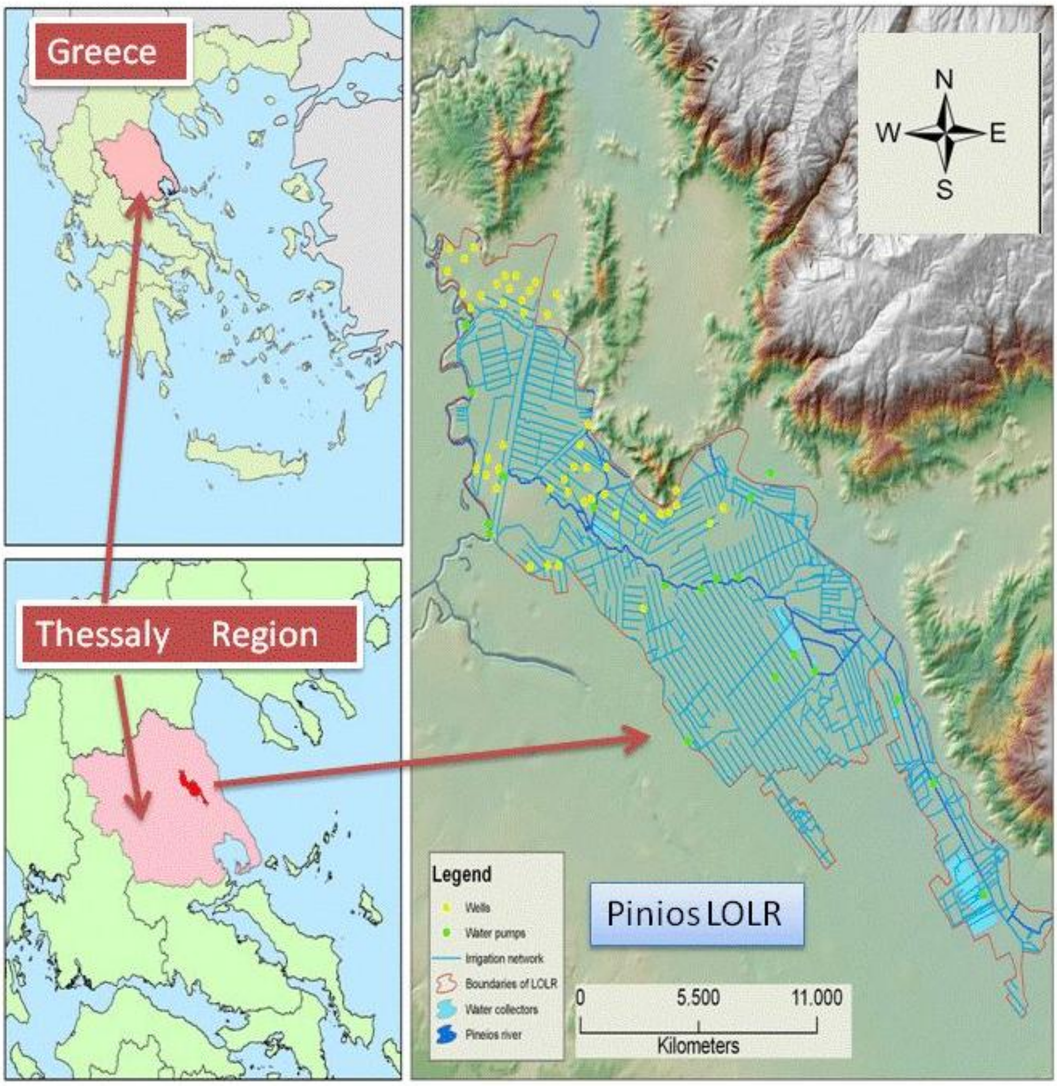


Figure 1: Modeling Framework for assessing the environmental impacts of irrigation

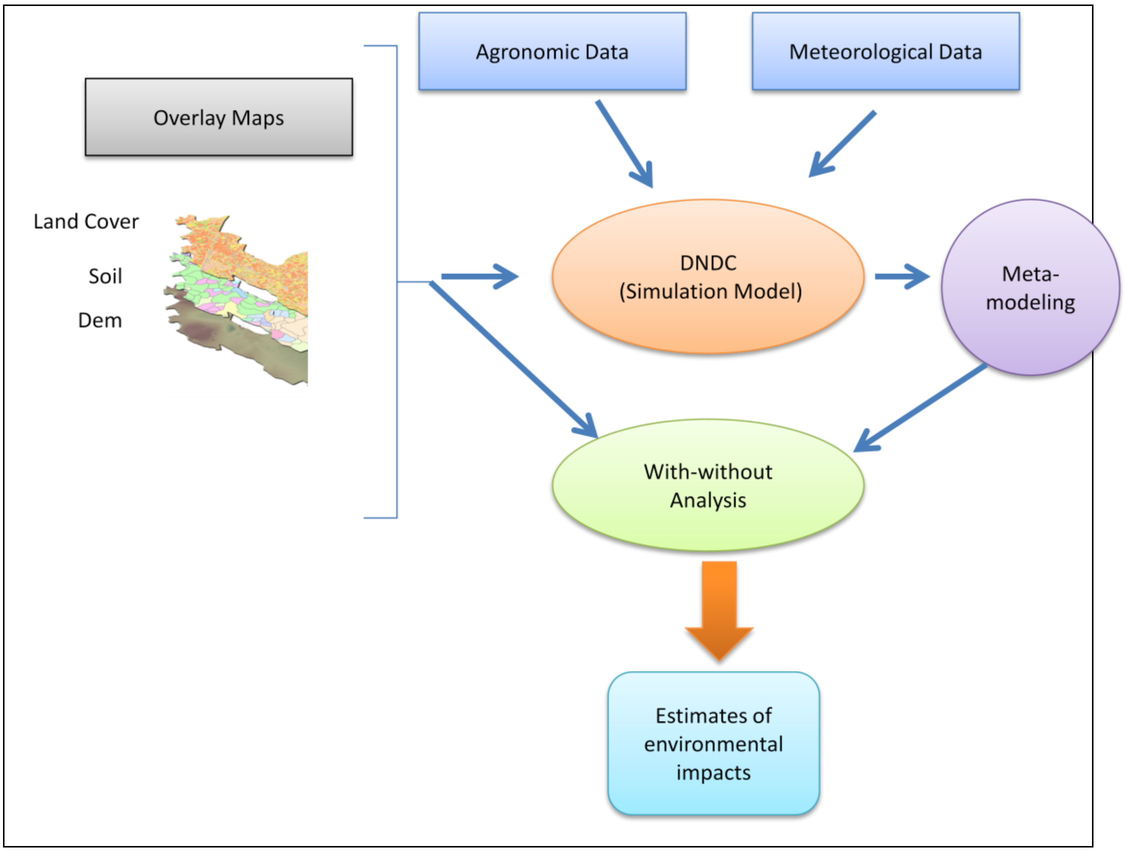
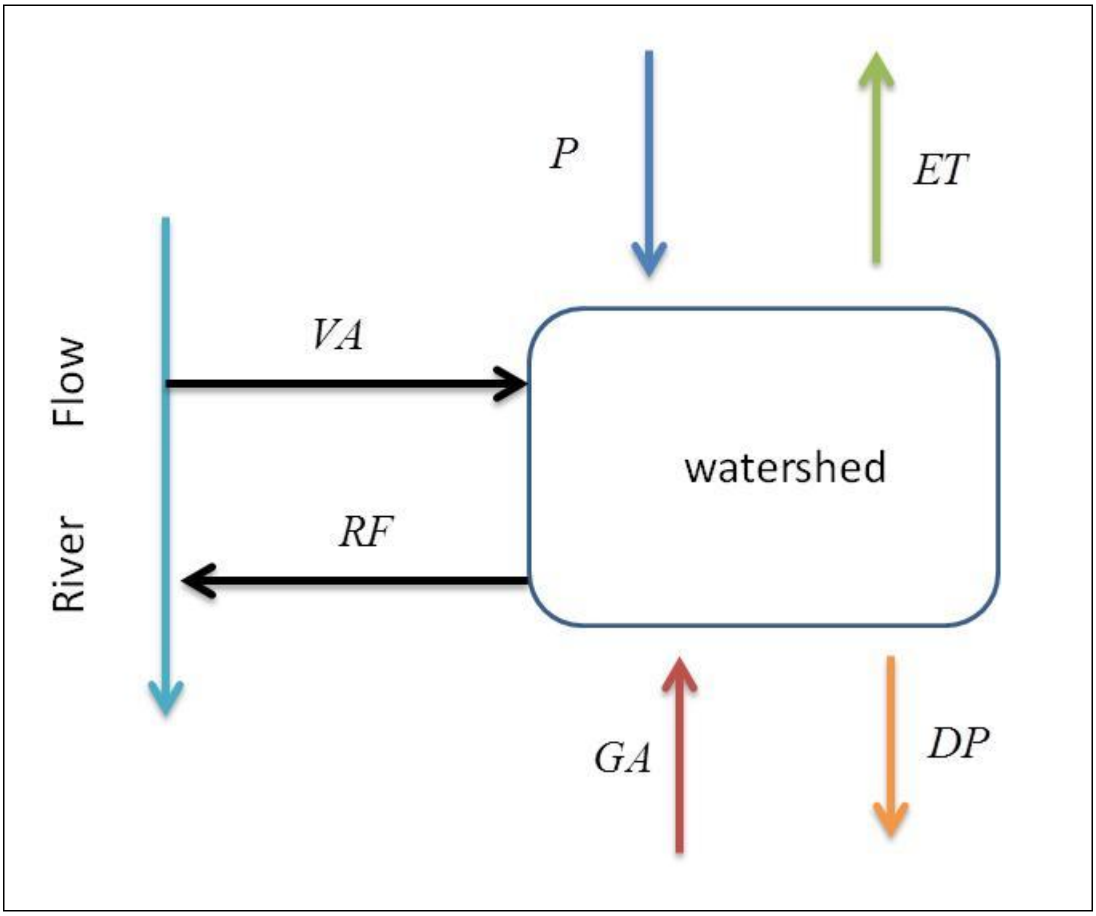


Figure 2: The Water Budget Rationale



Source: Haie and Keller (2012)

Figure 4: The composition of Irrigation full cost for the Pinios LOLR for the worst and best scenario