

Chapter 18

Environmental and Resource Costs

Assessment and the Case for Reforming the Italian System of Water Abstraction Charges



Vito Frontuto, Silvana Dalmazzone, Paolo Mancin, Alessia Giannetta, and Davide Attilio Calà

Abstract This chapter describes the design of a reform scheme for public water abstraction charges aimed at implementing the Water Framework Directive principles of (i) internalising the externalities associated with water use (or at least recover the cost of measures implemented to protect water resources); (ii) inducing an efficient allocation among competing uses; and (iii) achieving water and environmental protection without excessively hampering economic activities. We provide a simulation of the resulting water pricing systems based on data from the Piedmont Region, in north-western Italy. The reform design grounds water charges on the impacts on ecosystem services caused both by subtracting resources to freshwater ecosystems and by returning water to ecosystems, after human use, in a qualitatively degraded state. The system takes into account that the marginal damage of water uses may also depend on the quantitative and qualitative status of the concerned water body, and controls for incidence of the resulting charges.

Keywords Water pricing · Environmental and resource costs · Ecosystem services · Monetary evaluation · Affordability

V. Frontuto (✉) · S. Dalmazzone

Department of Economics and Statistics, University of Turin, Turin, Italy
e-mail: vito.frontuto@unito.it; silvana.dalmazzone@unito.it

P. Mancin · A. Giannetta · D. A. Calà

Environment Department of the Piedmont Region, Turin, Italy
e-mail: paolo.mancin@regione.piemonte.it; alessia.giannetta@regione.piemonte.it;
davide.cala@regione.piemonte.it

© Springer Nature Switzerland AG 2021

P. Turrini et al. (eds.), *Water Law, Policy and Economics in Italy*, Global Issues in Water Policy 28, https://doi.org/10.1007/978-3-030-69075-5_18

433

18.1 Introduction

The State of Water Report by the European Environment Agency (EEA 2018) shows that only 38% of the European Union (EU) surface water bodies are in good chemical status and just 40% in good ecological status – a substantial distance from the objectives of the Water Framework Directive (WFD). Compared to the 2009–2015 cycle, progress has been limited. Among a set of varied and complex underlying causes, a key role is identified in a delay by Member States to design and implement effective policy measures. While Article 9 of the WFD explicitly requires Member States to implement pricing policies that provide incentives to use water efficiently and recover costs for water services (including environmental and resource costs), virtually no Member State appears to have yet implemented comprehensive water management and pricing reforms, and in the majority of contexts there is not even sign of ongoing reform planning. Available reviews of water pricing in the EU are somewhat updated but, according to the EC (2012) *Water Blueprint*, only 49% of River Basin Management Plans (RBMPs) are preparing to change the pricing system to foster a more efficient use of water; and only 40% include measures to improve water metering – a pre-condition for any incentive-based pricing policy. This is particularly true for the most water-consumptive sectors: Rey and colleagues report that no Member State in Southern Europe has implemented an agricultural water pricing reform that integrates the principles of cost recovery, polluter-pays and affordability required by the WFD (Rey et al. 2018).

Among the very first attempts to make a step beyond this state of things, the Piedmont Region, in the North-West of Italy, introduced in July 2017 two additional ex-ante conditionalities to access funding from the EU's Common Agricultural Policy (CAP): (i) the “harmonization of the methods for quantifying irrigation water withdrawals and effective collection, communication and management of this data”, including the compulsory adoption of metering devices; and (ii) the “introduction of environmental and resource costs in the calculation of water prices”, to an extent consistent with the affordability principle (Regione Piemonte 2017). This amounts to an explicit choice by the Regional Government (consistent with the Italian regulation¹) to link access to CAP funds for agricultural firms to compliance with the principles required by the WFD – an unprecedented move towards the real implementation of the Directive.

This chapter provides a comprehensive description of the approach which has been followed to design the above-mentioned reform of public water abstraction charges. The proposed reform concerns all economic sectors that make use of public water, that is, water from out-of-network abstraction points. This includes the agricultural sector, the largest user of public water, employed mainly for irrigation; the industrial sector, which may use both network and out-of-network water supplies,

¹Decree of the Minister of the Environment no. 39 of 24 February 2015 (“*Regolamento recante i criteri per la definizione del costo ambientale e del costo della risorsa per i vari settori d’impiego dell’acqua*”).

depending on production processes and location; and the commercial sector, for which the use of out-of-network water is generally residual (e.g., fire-fighting systems, garden watering). The analysis also includes the very small share of households that hold a licence to withdraw water out of the integrated water service, because either living in remote out-of-network areas, usually served by village aqueducts holding a collective abstraction licence, or maintaining traditional rural wells in addition to the network.

The underlying principles draw directly from the WFD: water charges should (i) internalise the externalities associated with water use (or at least recover the cost of measures implemented to protect water resources); (ii) induce an efficient allocation among competing uses; (iii) achieve water and environmental protection without excessively hampering economic activities. In turn, the internalisation of externalities brings with it the principle of fairness in the contribution, that is, a system meant to proportionally spread the cost of conservation among uses according to the pressure they exert on water resources and on the environment.

Grounding water pricing on the environmental and resource costs entailed by water use requires, first, quantifying such costs in monetary terms. In the simulations presented in this chapter, we adopt a cost-based approach, consistent with the guidelines published by the Italian Ministry of the Environment,² to associate monetary values to the changes in ecosystem services impacted by water abstraction and use.

In this way, we implicitly assume that the value of avoided damages or the cost required to restore degraded ecosystems represent a proxy of the value of the services provided by those ecosystems. Monetary values of ecosystem services obtained through a cost-based approach underestimate the total economic value, since they quantify only the use value, and only that subset of it for which restoration interventions are technically feasible. This notwithstanding, we believe they convey a useful quantification of the minimum certain value for socio-economic costs associated with water use. In any case, a full recovery of environmental and resource costs would imply, in most circumstances, disproportionate costs, that is, costs to be imposed on economic activities that would be considered unsustainable and politically unfeasible (Galioto et al. 2013; Jensen et al. 2013; Klauer et al. 2016). The WFD itself admits, in the face of disproportionate costs, the possibility of derogating from the principle of full cost recovery. Moreover, from a practical point of view, cost-based valuations are also reasonably straightforward to implement in those contexts where recovery or replacement costs have actually been disbursed or at least included in RBMPs.

Among the numerous challenging questions faced by these pioneering attempts, two appear particularly crucial. The first is how to find the right balance between, on the one hand, socio-economic affordability with respect to local specificities and socio-economic features of the economic sectors involved and, on the other, proportionality between environmental costs caused by economic activities and financial

² *Ibid.*

costs imposed by water pricing. The second is the issue of whether (and how) to consider water balances, rather than simply withdrawals, as a base for water pricing. In the conclusions of this chapter we discuss both questions, with reference to the unfolding experience of its case study.

18.2 Designing Public Water Pricing

18.2.1 *The Piedmont Region Context*

The Piedmont Region, located in the northwest of Italy, is entirely part of the Po river basin district, the largest and most relevant Italian water district. The Region comprises the upstream of the Po River and other relevant water bodies (rivers, lakes and aquifers).

The most significant anthropic pressures on rivers and lakes derive from morphological alterations of the soil (63.8% of superficial water bodies are subject to significant pressures), discharges of urban wastewater (significant pressures on 36% of superficial water bodies) and release of pesticides, fertilisers and sludge due to the activities of the agricultural-livestock sector (significant pressures on 19% of superficial water bodies). For groundwater, the main factors contributing to quality deterioration are nitrates, pesticides, volatile organic compounds (VOCs) (27.5% of ground waters under significant pressures), metals such as nickel or chromium and chlorinated solvents.

The Region has historically been water-abundant, with an estimated annual average availability of 14 billion m³. Nevertheless, it has recently experienced droughts, both in winter and summer seasons, due to the combination of low precipitations and significant withdrawals from surface waters. The causes of these drought events are most likely due to climate change, but also show a clear correlation with the increasing trend of withdrawals.

In the last 60 years, precipitation has not shown significant differences in mean values, while there has been an increased frequency of extreme events, both intense precipitation and drought. In 2017, total precipitations were roughly 25% less than the average cumulated value referring to the period 1971–2000, with the most critical situations in the south-eastern area of the Region. The pluviometric deficit worsened in the first months of autumn, reaching a peak of 36% in October. The pressure on water resources due to withdrawals is well-known by local authorities and stakeholders: in 2015, almost 40% of surface water bodies were subject to significant pressure. The most consistent quantitative pressure is exerted by the agricultural sector, that uses about 80% of the total volumes withdrawn. It is estimated that about 5 billion m³ of water per year are derived from surface water bodies for irrigation, most of which are used for rice fields in the north-eastern area of the Region and for the irrigation of corn in the remaining lowlands. Seasonality exacerbates the

pressure of agriculture, which peaks during summer, from April to September, when the natural water runoff of rivers and torrents is at its minimum.³

Regional authorities are in charge of issuing water abstraction rights and set the corresponding prices, with the national Government and the Po River Basin Organization playing a monitoring and coordination role among Regions within the basin. Under the current water abstraction regime, licenses are issued by the competent authorities (Provinces) for long periods, up to a maximum of 40 years. Prices are set on a per-area basis (average charge: 1.22 €/ha), or based on the average flow rate capacity of the licenced derivation (0.56 €/ls for the agriculture sector, 2.37 €/ls for households,⁴ 175.94 €/ls for industrial uses), or on the installed capacity for the hydroelectric sector (29.89–44.05 €/kW), with heavy price differences across sectors. Table 18.1 provides a complete overview of the current pricing regime.

Table 18.1 Piedmont Region, current water pricing. (Source: Regione Piemonte 2019)^a

Sector	Type of charge	UoM	Charge
Irrigation and agriculture	Charge 1	€ per l/s	0.56
	Charge 2	€ per ha	1.22
	Min.	€	31.37
Commercial	Charge	€ per l/s	11.72
	Min.	€	143.61
Household	Charge	€ per l/s	2.37
	Min.	€	59.85
Energy	> = 3.000 kW	€ per kW	44.05
	1.000 kW e <3.000 kW	€ per kW	39.86
	> = 220 kW e <1.000 kW	€ per kW	37.76
	> = 20 kW e <220 kW	€ per kW	34.61
	<20 kW	€ per kW	29.89
	Min.	€	165.49
Industrial	Charge	€ per l/s	175.94
	Min. 1	€	2357.72
	Min. 2	€	1196.81
	Min. 3	€	694.17
	Min. 4	€	

^aThe source uses a finer disaggregation of sectors. We decided to focus our attention on the most relevant ones and made some aggregations in order to replicate standard classifications and allow international comparisons. For the industrial sector, the minimum payment increases with the dimension of the licenced flow rate

³Data on quantitative and qualitative status of water bodies have been collected for 2001–2016 period by the Regional Environmental Protection Agency (ARPA) and used as the informative base for the last Water Protection Plan (Regione Piemonte 2018).

⁴As mentioned in the introduction, the almost totality of households receive water from the network, and are therefore subject to the tariff system of the integrated water service. The “Household” category appearing in this analysis refers only to the small number of families holding a public water abstraction licence.

Since their introduction in Italy in 1933, water prices have been determined mainly on the ground of the ability to pay of different sectors, without consideration of the quantities consumed – a scheme resulting in an implicit subsidisation of water-intensive sectors. Considering withdrawals and revenues from the sectors for which we have a measure of the licenced water amount, in Piedmont industrial uses account for 9% of licenced withdrawals and 55.2% of revenues; household and commercial uses are entitled to 3% of licenced water and generate 12.2% of revenues; agricultural use accounts approximately for 87% of water use and originates 32.6% of revenues. Water licences to industry, households, commerce and agriculture generate revenues for 10.7 million € annually, 18.8% of total revenues from water charges. The remaining 81.2% is generated by the hydroelectric sector, not included in the above shares because its water charges are defined in €/kW rather than in l/s, and hence the database of licences for water abstraction does not allow us to quantify withdrawals (Regione Piemonte 2019).

18.2.2 The Reform Structure

The reform presented in this chapter aims at overcoming the current system and to link charges to actual water resource use. Crucial features informing the reform design are that the system for determining charges must be: (i) standardised and replicable; (ii) strictly connected to objective criteria (official and publicly available indicators and assessments of resource state in the RBMPs); (iii) dynamically flexible to accommodate revisions and updates in subsequent planning cycles.

The first crucial requirement of the transition is moving towards a fully metered system. Water use in Piedmont is entirely metered only for households and commercial uses that are connected to the network of the integrated water service. In all other sectors, despite a mandatory requirement for large users established in 2007 (Decree of the President of the Regional Council no. 7/R of 25 June 2007), the implementation of a system for metering public water abstraction is currently at different degrees of completion depending on contexts and uses.

The reform under way calculates water charges based on the financial and environmental costs associated with the quantity of water withdrawn and the quality of the water returned. In this way, the financial and environmental costs are spread among users in proportion to the pressure they exert on water resources. In order to avoid disproportioned costs and warrant affordability, the system then sets a cap in terms of the cost recovery ratio to be obtained.

The objectives set by the WFD for Member States are in terms of a good status of water bodies to be pursued and maintained over time. One could argue, therefore, that the implementation of the WFD requires incentive-based water pricing only for water bodies that do not reach a good status: where the natural system has the capacity to assimilate pressures, one could be induced to consider externalities deriving from water use equal to zero. Such a regulation-centered approach, however, would fail to consider dynamic effects and the potential for spillovers of pressure from

water bodies with a poor status, where water charges would impose a heavier burden on economic activities, towards water bodies with higher environmental quality. A conservation-centered approach would rather suggest assigning economic value to water regardless of the state of the water body on which the pressure insists. This would avoid biases and spillovers, and aim at improving water quality where it is degraded while protecting it where it is good or higher. It is also consistent with the concept of solidarity and concurrent responsibility in resource use, which entails sharing responsibility among competing uses in reaching adequate quality status at the water body, sub-basin and basin level.

Total user costs for water resources are typically subdivided in three main components – financial, environmental and resource costs:

$$C^T = C^F + C^R + C^E \quad (18.1)$$

Financial costs (C^F) are the costs incurred for the supply and management of water uses and services (see Decree of the Minister of the Environment no. 39 of 25 February 2015). They include capital, operating and maintenance costs for water supply. For the integrated water service, financial costs are determined by the Italian Regulatory Authority for Energy, Networks and the Environment (ARERA) through a standardised Water Tariff Method (*metodo tariffario idrico* – MTI). For all other uses (irrigation, hydroelectric, out-of-network industrial), we could consider them equal to zero, since they are internalised in the users' production function.

Resource costs (C^R) are defined in the guidelines for the implementation of the WFD by the Ministry of the Environment as those generated by inefficiencies in the allocation of water resources among competing uses. They arise if the difference between the value of net benefits of current uses and the value of the best current or future available alternatives is negative (*ibid.*).

Environmental costs (C^E) are the loss in the value of water resources as a consequence of degradation in water ecosystems due to anthropic uses: “[...] the costs linked to damage that water uses may impose on the environment, ecosystems or other users, as well as costs linked to changes in water ecosystems functioning or to resource degradation due both to excessive abstractions and to lower water quality that represent a source of damage for water bodies or the welfare deriving from non-use values of the resource” (*ibid.*). Damages may weigh on two dimensions – quantitative and qualitative – and are associated to modifications with respect to the resource estimated or expected “natural values”, that is, those prevailing in the absence of human use.

The water pricing reform proposal chooses not to include in the base for calculation of user charges the cost of allocation inefficiencies. It simply quantifies the environmental cost of water use as the sum of costs linked to the quantity of resource abstracted and of the costs linked to the altered quality of the returned water due to point or non-point pressures. The estimate of the environmental cost arising from water use (C_i^E) for any of the N abstraction points is thus based on four different dimensions: (i) a physical measure of resource abstraction (Q_i^A); (ii) indicators of the qualitative and quantitative status of the source rivers and aquifers ($\alpha, \beta, \varepsilon, \gamma$);

(iii) a monetary measure of the cost of abstraction (C^A); and (iv) a monetary measure of the external costs associated to qualitative deteriorations in the returned water (C^P):

$$\sum_{i=1}^N C_i^E = (1 + \alpha_i + \beta_i + \varepsilon_i) Q_i^A C^A + \gamma C^P \quad (18.2)$$

18.2.2.1 Metering

As mentioned in Sect. 18.2.1, the incumbent system for the allocation of public water adopts different criteria depending on sectors. The quantity of water that users are allowed to withdraw is set at the stage of issuing the licence and is based on self-reported user needs validated by the regulator, and on an evaluation of the impact of new withdrawals on the water body, taking into account already issued licences.

However, these nominal quantities are very weak measures of actual water consumption and can be source of inefficiencies, in terms of allocation, incentive power and information available for policies and management. A proper incentive-based pricing scheme would require the progressive transition from the current structure to a fully metered system, with water pricing uniformly defined in €/m³.

While the adoption of meters spreads, a transitory first phase of the reform can implement water pricing on the nominal water quantities licenced to each user in each extraction point, determined on the ground of average theoretical flows, rather than on measured withdrawals. Obvious limitations of this phase include uncertainty as to the relation between licenced quantities and real water consumption (made worse by frequently oversized concessions) and a limited incentive capacity towards reducing water use and wasteful behaviour.

After a transitional period meant to allow users to adopt metering devices, water charges should start being calculated either on measured abstractions or, where meter adoption has not occurred, on maximum (rather than average) nominal flow. It has been shown in several contexts that metering alone can generate virtuous behaviour and reduce withdrawals by up to 40% (Sardonini et al. 2011). Transition from pricing on nominal flows to measurement of real abstraction is consistent with the principles of concurrent responsibility in resource use and incentive pricing.

18.2.2.2 Weighing Water Pricing by Availability and Ecological Status of Sources

Correcting the cost of abstractions as a function of the quantitative and qualitative status of the water body allows us to introduce the spatial and temporal dimensions in the water pricing system. The quantification of the physical state of water bodies is made, by expressed choice, with reference to the indices used in water planning

and environmental monitoring by regional authorities, in implementation of the WFD.

The first parameter, α , in Eq. (18.2) corrects the cost of resource abstraction on the ground of the quantitative status of the source water body. For surface water bodies, α is calculated by normalising between 0 and 1 the status of the water body measured by the Index of Alteration of the Hydrological Regime (IARI) or by the Water Exploitation Index (WEI+). WEI+ is an indicator of pressure that human activities exert on water resources calculated with reference to a specific territory (basin or sub-basin, or aggregation of basins and sub-basins). It is used to identify the areas potentially subject to water stress, with reference to the actual availability of the resource, to the observed withdrawals and to an appropriate time scale. The index, expressed as the ratio between actual withdrawals and the average natural range on the period 2000–2016, has been simulated on different scenarios of precipitation rates. In years with scarce precipitation, most cases of water stress concentrate in summer (in July, and to a lesser extent in April, May and September). In years with moderate precipitation water stress concentrates in June, July and August. In both scenarios, a strong correlation emerges between water stress and the irrigation season (Regione Piemonte 2018).

For underground water bodies the parameter α is estimated based on the quantitative status of water basins periodically reported by Regional authorities, as requested by the WFD. Monitoring over the period 2005–2017 reports, on average, a stable time path of the Region's aquifers, with a few situations of declining quantitative trends that require careful consideration.

The second parameter, β , corrects the environmental cost on the ground of the qualitative status of the water body affected by the abstraction. The determination of parameter β is based on the classification of water bodies according to the WFD: for surface water bodies it measures both the Ecological Status (ES) and the Chemical Status (CS), while for underground water bodies it measures only the CS.

The monitoring activities carried out by ARPA show that, in 2015, only 55% of Piedmont rivers had a good or higher ES, while 95% of them had a good CS. The Overall State (OS), given by the worst result between the ES and CS, was good only for about half of the rivers (134 out of 248, or 54%). With regard to lake water bodies, only 4 out of 11 (36%) exhibited a good OS, mainly due to bad ES. For groundwater, only the CS is used: a poor status was recorded in 112 points out of 268 (41.8%) for the shallow water table, while in the deep strata 32 points out of 132 (24.25%) were in poor state. If we consider the state of the superficial groundwater basin, only 2 superficial aquifers out of 17 (11.8%) had a good state.⁵

The parameters α and β are updated with the six-year classification of water bodies status as required by the WFD. They can be modified when relevant new information or new indicators become available.

⁵The aquifer model of Piedmont is able to distinguish data pertaining to superficial and deep aquifers, as it has adequately identified the groundwater divide.

The third parameter, ε , weighs on abstractions of resource designated “for human use” (deep aquifer) for purposes different from drinking use. According to the Italian law (Article 96 of Legislative Decree no. 152/2006), this triggers an additional burden on the water charge for non-priority uses of groundwater. In Piedmont, using the Regional dataset of public water licences and water charges **GE.RI.CA.** (*Gestione Riscossione Canoni*),⁶ we observe that 1871 users out of 17,885 (almost 10% of total licences) do exert withdrawals from deep aquifers for non-priority uses. The most involved sector is agriculture with 1321 licences, or 7% of the total.

18.2.2.3 The Weight of Pollution: Calibrating Water Pricing by Quality of the Returned Water

The second addend of Eq. (18.2), γC^p , captures the environmental cost linked to altered chemical-physical characteristics of the returned resource and hydro-morphological changes: for instance, the addition of solvents, nutrients, pesticides, and sediments, changes in water temperature, speed or turbulence, or any other change with respect to the qualitative features of the resource in its natural state that may cause harm to the ecosystem where water is returned.

Returning altered water does not have equal consequences in all water bodies. The parameter γ plays the role of correcting the environmental cost of returning degraded water on the basis of the quality status of the water body where the abstracted water is returned. For uses with diffuse water restitution (non-point sources), such as agriculture, the median value of the sub-basin where the abstraction takes place is used as representative value of the ecological status.

18.2.2.4 Seasonality

Since water diversion from rivers in flood season does not entail environmental costs, in contexts where the water flow regime is heavily affected by seasonality it may be appropriate to consider a variant of the above formula:

$$\sum_{i=1}^N C_i^E = \left[(1 + \alpha_i + \beta_i) F + \varepsilon_i \right] Q_i^A C^A + \gamma C^P \quad (18.3)$$

where F is a switch that water authorities may use to derogate from charging users for abstraction during water-abundant months (similar to when the regulatory obligation to safeguard the environmental flow is waived during periods of drought). In those periods, the residual price of water would include only the components

⁶**GE.RI.CA.** is the geo-localised database of licences of water abstraction in Piedmont. It is used to determine charges according to sector, type of withdrawal and any applicable reduction.

associated with the qualitative deterioration imposed on water by human use and the withdrawals for non-priority uses.

18.2.2.5 Affordability

The overall environmental cost of water use thus calculated ($\sum_{i=1}^N C_i^E$) is – together with financial and resource costs – one of the components of the total cost that the WFD requires Member States to recover through water pricing. Context-specific results should then be evaluated in light of the affordability principle. If disproportionate costs arise, the system provides for a maximum ceiling, by setting a cap in terms of the cost recovery ratio to be obtained.

18.3 Quantifying the Cost of Abstraction and Pollution: A Simulation for Piedmont

This and the following sections implement an empirical simulation of the above structural reform of water pricing using data from Piedmont. The simulation considers out-of-network industrial, commercial, household, and irrigation and agricultural uses. Energy uses are not included because water charges for hydroelectric plants are connected to the amount of generated energy rather than to the amount of water used, and there is no general conversion factor between the two, depending it on the height of the water jump between source and turbines. The Piedmont Region will address the issue of water charges for energy uses as part of the reform of hydroelectric concessions, in compliance with national Law no. 12/2019.

In Eq. 18.2, as we have seen, C^A is the monetary value of the cost associated with the abstraction of a unit of water from ecosystems. C^P measures the cost of the degradation occurring in the quality of water being used for human purposes.

To quantify C^A and C^P we propose to adopt a cost-based approach, in line with what suggested by national and international guidelines – see, for example, the Common Implementation Strategy by the European Commission (EC 2003). This approach, which takes the form of methods for estimating recovery costs, replacement or more generally avoided costs, presupposes that the costs of avoiding damage or replacing degraded ecosystems represent a measure, albeit partial and hence approximated by default, of the value of the services provided by ecosystems. All cost-based methods are hence based on the supply curve for ecosystem services, so that strictly they do not measure utility. In other words, these methods do not provide information on the underlying demand curve for the relevant sets of ecosystem services, and therefore cannot provide comprehensive measures of total economic value, nor do they necessarily convey complete information about social welfare. On the other hand, they do not suffer from the hypothetical bias affecting, for example, stated preferences techniques, and are very practical, reliable and

cost-effective in contexts where restoration or replacement interventions have actually been realised or at least included in the budget of water management plans.

The value of C^A can be estimated as the unit cost of interventions required to ease the pressure on water resources induced by abstractions; we consider, in this exercise, the building cost of multi-purpose reservoirs (a cost-efficient measure). We consider 19 different projects for water storage with heterogeneous features and designs (small, large, and inter-company reservoirs) within the Po river basin. We compute the annualised value by calculating depreciation with the following assumptions:

- 2.85% remuneration of capital (the return rate of 50-year Italian Government bonds);
- different discount rates (5, 6, 7%) to test for sensitivity of results;
- cost in €/m³ of considered infrastructures calculated on the ground of the volumetric design of the single reservoir. Mean and median of these values have been calculated by excluding tails of the distribution (5°–95° and 10°–90° centile).

The resulting cost of building a water storage system is between 0.05 and 0.084 €/m³, depending on the discount rate used and on the measure of central tendency employed (Table 18.2). As said above, this estimate should be considered as only an approximation of the environmental unit cost of withdrawals and can be affected by future technological change and exogenous factors.

In order to identify the total annual cost imposed by water abstraction on ecosystems, we do not consider total annual water abstractions taking place in the Region, but only the unsustainable share of them, measured by the water deficit – the excess of water demand over sustainable water supply within each year (satisfied by reducing the stock of non-renewable groundwater in deep aquifers).

The estimate of water deficit is obtained for Piedmont by the MIKE HYDRO Basin simulation, a mathematical representation of river basins defined by including the configuration of river and reservoir systems, catchment hydrology and water user schemes (Regione Piemonte 2018). We consider two different scenarios, for years with average and scarce precipitations.⁷

Table 18.3 reports the values of C^A for different discount rates and of the total annual cost of unsustainable water abstraction for the years with average and scarce

Table 18.2 Replacement costs: simulations (€/m³)

Discount rate	5%	6%	7%
Mean (€/mc)	0.084	0.072	0.063
Median (€/mc)	0.066	0.057	0.05

⁷The outputs of MIKE HYDRO simulations are available at: www.regione.piemonte.it/ambiente/acqua/dwd/PTA/e_allegati_tecnici/II/Iih01.pdf

Table 18.3 Total cost of water abstraction (C^A) under different scenarios of water deficit

Discount rate		C^A	Water deficit (€)	
			Average precipitation year	Scarce precipitation year
5%	Mean	0.084	57,957,068	161,499,466
	Median	0.066	45,739,144	127,453,778
6%	Mean	0.07	50,039,163	139,435,937
	Median	0.06	39,490,411	110,040,461
7%	Mean	0.06	43,813,215	122,087,108
	Median	0.05	34,576,955	96,349,936

Table 18.4 Cost of water protection measures in Po RBMPs (2015–2021)

	Total cost (€)	Covered costs (€)	Cost to be recovered (€)
Integrated water service	58,565,031	58,475,031	90,000.00
Irrigation	360,824,465	353,527,165	6,754,300
All uses	16,011,403.00	398,000	15,613,403
TOTAL	435,400,899	412,002,196	22,457,703

precipitations. The resulting value of total cost is between 34.5 mln € and 161.5 mln €, depending on average annual precipitation and discount rate.

Let us now consider the cost associated with the qualitative deterioration of the water returned to ecosystems after human use. C^P can be estimated on the basis of mitigation costs specific for categories of pollutants available in the literature (for example, EC 2003). Alternatively, a convenient approximation for the externality associated with altered returned water is the cost borne by water authorities for implementing the water protection measures included in the Water Protection Plan of the Piedmont Region. Monetary measures can be updated every six years, together with the RBMPs, to take into account new adopted measures and/or variations in costs. In the simulation presented here, we adopt the latter approach. In order to implement the polluter-pays principle, we spread the cost of measures among different uses based on the volumes of withdrawn resource.

In theory, the cost of the measures implemented to protect and restore water quality converges to the monetary value of the environmental damage when the measures adopted cover all damages and are sufficient to internalise them completely. Otherwise, as in the present simulation and most real-world contexts, the obtained estimate is an approximation by default.

Table 18.4 presents the cost of measures included in the Regional Plan for Water Protection for the period 2015–2021 (Regione Piemonte 2018). These measures are designed to reduce a variety of pressures on water resources, from pesticides and nitrates abatement to watershed vegetation buffer zones, control of erosion and so on. Some of them are directed to specific uses (in particular in the fields of the integrated water service and agriculture), whereas other have general objectives and pertain to all uses.

In a cost-recovery perspective, the cost of altered returned water to be included in the calculation of water pricing is only the quota of the total cost not yet covered

by external resources (i.e., from EU or national funds) or revenues from the tariff of the integrated water service. The aggregate value of C^p turns out to be approximately 22.5 mln €. The share of it originated by irrigation is approximately 30%.

18.4 Proportionality Measures and Affordable Charges

The process of reforming water pricing cannot disregard socio-economic sustainability, through an in-depth analysis of direct and indirect distributive impacts. This requires measuring elasticity of demand and elasticity of substitution of different uses, as well as propagation of impacts within the economic system.

As mentioned in the introduction, one of the principles guiding our approach is identifying methods of analysis and policy design that are replicable without external support within local governments. With this objective in mind, we develop a simple analysis of financial sustainability and distributive impact relying on:

- (i) the Benefit-Cost Ratio (B/C) to define the proportionality of the estimated annual total cost associated with the unsustainable share of water abstraction calculated in Table 18.3; and
- (ii) the ratio between the expected variation in water charges and the average net income within each category of use in order to verify the ability of users to face any given simulated increase in water charges.

18.4.1 Assessing Proportionality

In order to verify the proportionality of the costs of the measures adopted to contain pressures on water resources, it is necessary to perform a monetary valuation of the environmental benefits resulting from their implementation.

Environmental valuation methods can be distinguished between those with a direct approach and those with an indirect approach. The indirect, or revealed preference, methods (Hedonic Prices, Travel Cost Method) are generally more suitable for assessing the direct use value, whereas techniques with a direct, or stated preference, approach (Contingent Valuation and Discrete Choice Experiments) are the only ones capable of capturing also non-use values and hence to offer a measure of total economic value.

All these methods require carrying out primary studies that are generally expensive and in certain contexts difficult to implement with a good level of reliability. For these reasons, secondary techniques, such as Benefit Transfer (BT), are increasingly employed in the economic valuation of environmental goods and ecosystem services. BT consists in transferring information available for a given context (monetary valuations from primary studies) to other contexts, after appropriate corrections to account for heterogeneity: factors such as geographical location, economic

situation (average income, employment rates) or the availability of water in a particular moment in time generally influence the valuation result.

Monetary values for the loss of ecosystem services associated with water uses have been estimated in different contexts (e.g., Bateman and Langford 1997; Brouwer 2006; Milon and Scrogin 2006; Raggi et al. 2009; de Groot et al. 2012; Ramajo-Hernández and Salazar 2012; Russi et al. 2013). The appropriate willingness to pay for individuals living in Piedmont has been identified through an extensive literature review. The study selected as the most appropriate was conducted by Raggi and colleagues on the Po and the Reno basins (Raggi et al. 2009). Their study investigates the value, as perceived by the population, of allowing water to serve ecosystem uses, specifically, avoiding withdrawals for agricultural, industry and energy uses. The estimated willingness to pay is about 38€ per family per year. Considering the number of households resident in Piedmont (2011 census) and adjusting for income differentials between the original area of study and the whole Region, the aggregate benefit (AB) can be computed as:

$$AB = \overline{WTP} \cdot WTP_0 \cdot N_{HH} \cdot \Delta_I = \text{€ } 68,414,247 \quad (18.4)$$

with \overline{WTP} the average WTP, WTP_0 the percentage of families stating zero WTP in the survey, N_{HH} the number of families living in Piedmont and Δ_I the weight to correct for income differentials.

As all estimates of environmental benefits, this value is affected by the general limitations of environmental monetary valuation techniques, and particularly of those based on the elicitation of individual's willingness to pay – incomplete information by the survey respondents, hypothetical bias, protest bias, scope effects (Hanley and Czajkowski 2019).

Table 18.5 presents the results of the B/C analysis: benefits are the outcome of the BT method described above, whereas costs of abstraction are the aggregate value from Table 18.3. The results show that the precipitation scenario turns out to be a crucial assumption. In the case of a year with moderate precipitation, the ratio B/C is always greater than one, implying that the benefits overcome the costs. In years with scarce precipitation, the ratio is always below 1, independently of the discount rate used – implying (given our BT estimate of benefits based on individual willingness to pay) disproportionate costs.

Table 18.5 Proportionality measure: benefits over costs

			B/C	
Discount rate		C ^A	Average year	Moderate year
5%	Mean	0.084	1.18	0.42
	Median	0.066	1.50	0.54
6%	Mean	0.07	1.37	0.49
	Median	0.06	1.73	0.62
7%	Mean	0.06	1.56	0.56
	Median	0.05	1.98	0.71

18.4.2 Assessing Affordability

The affordability of different levels of water charges is assessed by computing the expected change in average income that would occur as a result of the calculated cost-recovery water charges.

For agricultural users we employ the EU-RICA dataset (INEA 2014),⁸ which reports average value added, net income and costs of production per hectare for different crops in the Region. We assume a standard agent with a withdrawal rate of 1 L/s per hectare for the irrigation season (one semester per year). The calculated incidence ranges between 22% and 36% of net income (Table 18.6): abstraction charges recovering the whole environmental costs of water use, although approximated by default as done on this exercise, would not be affordable for agricultural firms.

The analysis of incidence for the industrial and commercial sectors faces heavy informational requirements: it would require data on average incomes for different sub-sectors with very different water intensity; in addition, the water licence database does not allow us to match the licenced water amount with the sub-sector of the corresponding company. We therefore use the agricultural sector, which would face both the heavier load and the most significant revision in water charges, as a benchmark for the overall affordability analysis.

An in-depth literature review on the affordability of water pricing (among others, Galioto et al. 2013, which referred to the context of the Po river basin) returns as affordable levels of incidence for productive and commercial sectors values ranging between 0.5% and 3% of net income. Table 18.7 reports the corresponding levels of annual water charge per L/s. We selected as affordable the incidence level of 1.5% of net income. In the agricultural sector this would correspond to an annual water charge equal to 54.86€ L/s (0.014€/m³ on average).

This rescaled value of water charges, although very far from the level ensuring a full cost recovery (estimated in 788.80 € L/s for farmers and 1577 € L/s for the other sectors) is 100 times higher than the current charge paid by the agricultural sector.

Table 18.6 Incidence analysis (agricultural sector)

Discount rate		C ^A	Full cost recovery (€/L/s)	Incidence on average net income of farmers (€)
5%	Mean	0.083	1322.17	36%
	Median	0.066	1043.44	29%
6%	Mean	0.072	1141.54	31%
	Median	0.057	900.89	25%
7%	Mean	0.063	999.51	27%
	Median	0.050	788.80	22%

⁸Data are available at: www.regione.piemonte.it/agri/area_statistica/agridata/dwd/webpie12.pdf

Table 18.7 Simulated annual water charges per L/s versus incidence on net income (agricultural sector)

Incidence level	0.5%	0.75%	1.00%	1.25%	1.50%	1.75%	2.00%	2.50%	3.00%
Mean	€ 18.29	€ 27.43	€ 36.57	€ 45.72	€ 54.86	€ 64.01	€ 73.15	€ 91.44	€ 109.72
Median	€ 10.69	€ 16.03	€ 21.37	€ 26.71	€ 32.06	€ 37.40	€ 42.74	€ 53.43	€ 64.11

Table 18.8 Piedmont region, revised water pricing

Sector	Type of charge	UoM	Current charge	Revised charge			
				C ^A	C ^P	Total	Minimum
Irrigation and agriculture	Charge 1	€ per L/s	0.56	54.86	1.42	56.42	100
	Charge 2	€ per ha	1.22	54.86	1.42	56.42	
Commercial	Charge	€ per L/s	11.72	110	0.89	110.89	
Household	Charge	€ per L/s	2.37	110	0.89	110.89	
Industrial	Charge	€ per ls	175.94	110	0.89	110.89	

For the other sectors, assuming a new charge of 109.72€ (obtained using the same water price, for the case of a licence of 1 L/s per firm, but for the entire year), the charge increases around 10 times for commercial activities and 55 times for the domestic sector. Only in the case of the industrial sector the new charge would be lower than the current one.

The final rescaled water charges are shown, by sector, in Table 18.8. The total revised charge is the sum of C^A , as simulated in Table 18.7, and C^P , that is the cost of water protection measures (Table 18.4) spread among uses according to their water use.

Finally, minimum payments are standardised for each use at 100 €/year. This value has been identified by the Regional authorities as the annual administrative cost for the management of each licence.

18.4.3 Revenues Simulation Under the Water Charges Reform

Using the database containing the complete population of the water abstraction licences and water charges of the Piedmont Region (GE.RI.CA.), we calculate the revenues under the current scheme of water charges and the reformed one. This is done here as a static exercise, without considering the elasticity of water demand. In reality, the very objective of reforming water charges is inducing a water-saving change in behaviour; the change in the burden that water charges impose on economic activities is therefore expected to increase in reality less than calculated in this section. A simulation of distributive impacts taking into account also water consumption and crop portfolio adjustments is done by Sapino and colleagues (Sapino et al. 2020).

Using the average flow rate of current licences as a proxy of 2017 water abstraction levels and under the assumption of inelastic demand, annual revenues would increase, in aggregate, from 57 mln € to 82 mln € (Table 18.9), potentially generating a substantial fiscal space to finance new measures for the protection and restoration of water resources.

Agriculture, responsible for the largest use of water, is the sector most affected by the charge increase. Annual revenues collected from irrigation licences would increase from 3.5 mln € to 26 mln €. On average, agricultural companies would face an increase in annual water charges of +17.12% with respect to 2017. The strongest increase affects consortia or farmers associations holding licenses for large quantities of water then shared among associates. The burden increases very slowly, due to the system of minimum payments, for farmers up to the 70th percentile in terms of water consumption (Fig. 18.1).

The burden of out-of-network water charges, in terms of overall collected revenues, remains almost unchanged with respect to 2017 for the other sectors considered here. For the industrial sector we measure an average increase of 0.43%, 0.67%

Table 18.9 Current and simulated revenues

Sector	Revenues 2017 (€)	Simulated revenues (€)	Average difference (€)	Average variation (%)
Irrigation and agriculture	3,512,602	25,973,720	6395.45	17.12
Commercial	282,301	318,646	19.68	0.13
Household	1,030,374	2,970,673	1170.97	0.67
Energy	46,266,146	46,266,146	0	0
Industrial	5,944,309	6,367,245	390.31	0.43
Total	57,035,732	81,896,430	1522.87	13.68

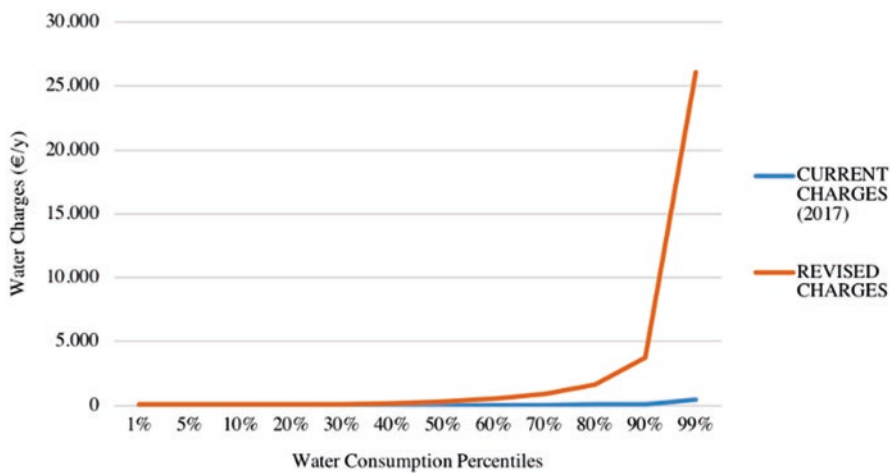


Fig. 18.1 Annual water charges by consumption percentiles

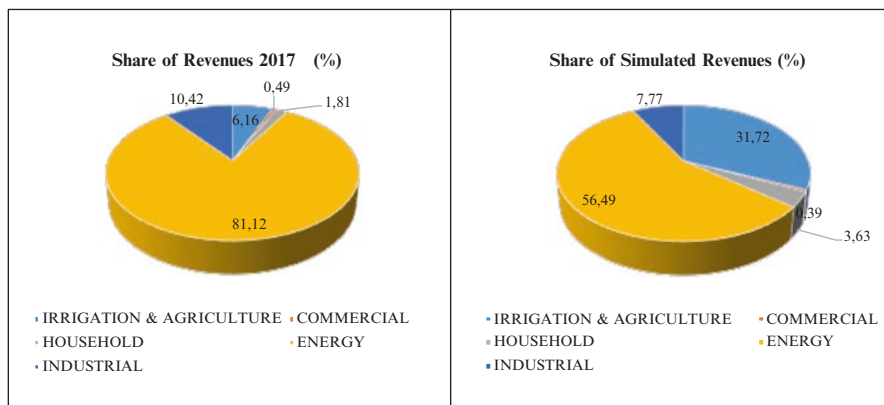


Fig. 18.2 Current and simulated revenues

for households, and 0.13% for commercial uses. As mentioned in Sect. 18.3, the simulated reform is not implemented on the energy sector, the revenues from which in Table 18.9 and Fig. 18.2 are kept constant to 2017.

As discussed in previous sections, the reform proposal described in this chapter is inspired by the objective of linking water charges to water use in order to provide an incentive to virtuous and sustainable behaviour. The shares of revenues from water pricing resulting from the revised system would indeed reduce the distance between withdrawals and contribution characterising the status quo. In 2017, irrigation and agriculture accounted for 6% of total revenues from water charges; under the reform scenario, with the same level of use they would account for almost 32% of revenues. If water charges for energy uses remained unchanged, the relative weight of the energy sector on total revenues would be reduced from 81% to 56.5%; it should be recalled however that also a reform of hydroelectric concessions is under way, in compliance with Law no. 12/2019. All other sectors, that would be affected negligibly in terms of payments, would observe a reduction in the share of total revenues they generate (Fig. 18.2).

18.5 A Spatial Analysis of Pressures and Revenues

We developed a comprehensive geo-localised dataset of water abstractions, water quality and revenues from water charges in order to conduct a spatial analysis of the status quo, in terms of both withdrawals and revenues, and to compare it with the reform scenario. Specifically, we analyse the spatial correlation between pressures and revenues to assess the consistency of the reform design with the user/polluter-pays principle. The spatial analysis also provides some insight into the appropriate scale for water management policies. We show that the introduction of the proposed scheme for determining water charges would substantially improve compliance

with the WFD, particularly with respect to the recovery of environmental costs connected with water resource use.

Results from the spatial analysis are then validated by a statistical analysis that demonstrates a correlation between nominal withdrawals and overall payments for water higher under the revised scheme than with the current scheme of water abstraction charges.

Figure 18.3 portrays the spatial distribution of public water licenses (abstraction points). We notice that the greatest concentration of withdrawals is in the plains in the southwestern part of the Region, where we find the urban area of Turin and agricultural land devoted in particular to the intensive cultivation of corn. In the northeast of the Region, where we find a dominance of rice crops, we observe a lower concentration of abstraction points. Nevertheless, if we move from the number of points to the intensity of use (the flow rate per point, Fig. 18.4a), the picture changes substantially.

We employ an interpolation technique, the Inverse Distance Weighting (IDW) method, to obtain a snapshot of the spatial distribution of water withdrawals at the

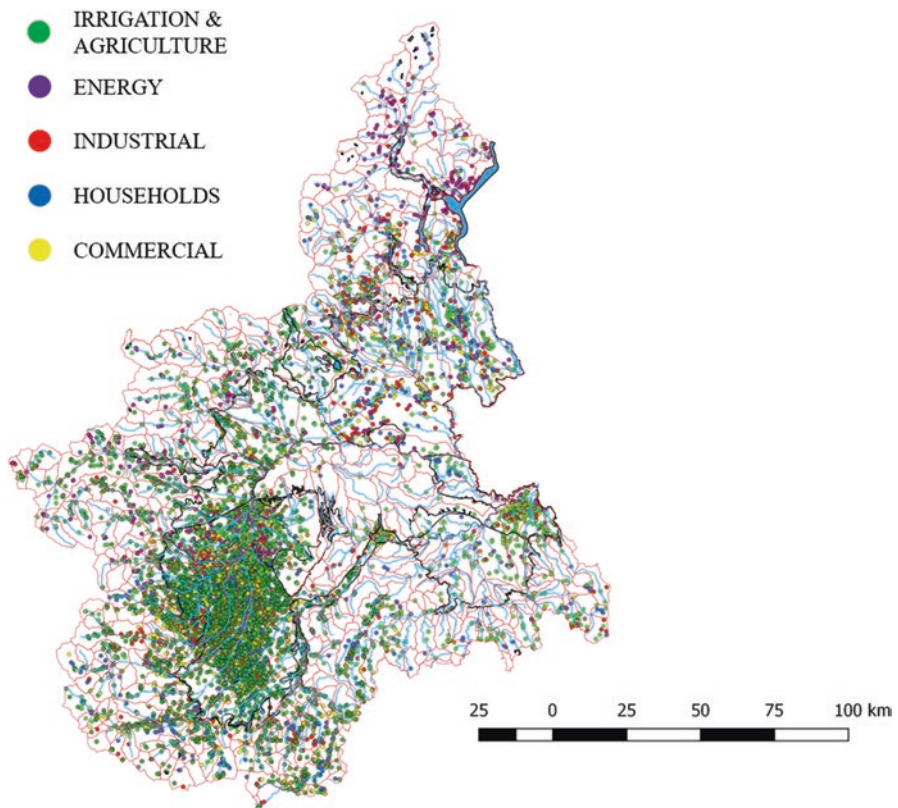
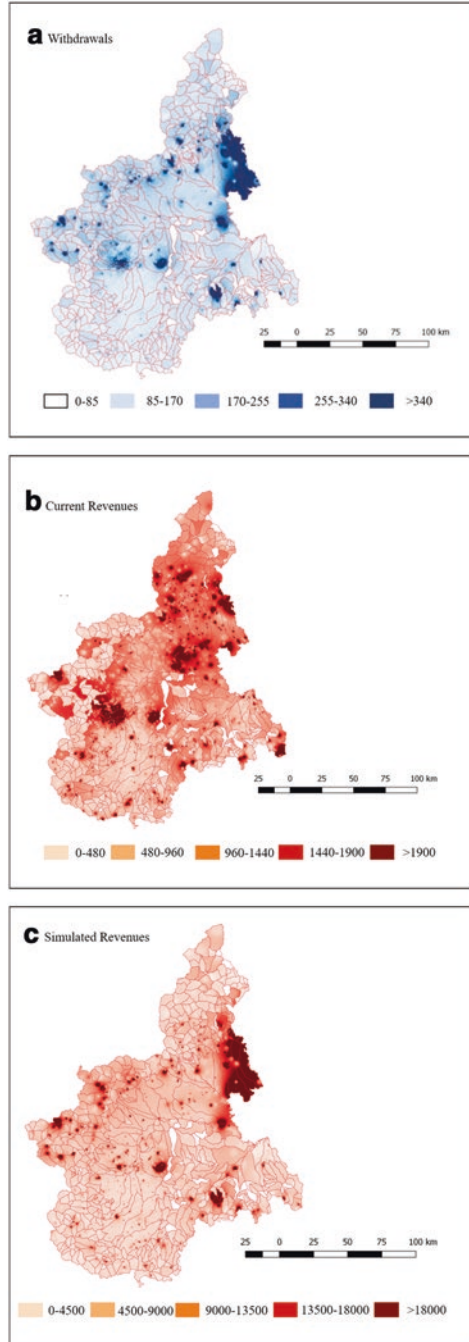


Fig. 18.3 Spatial distribution of abstraction points

Fig. 18.4 (a) Spatial distribution of withdrawals, L/s; (b) current revenues, €/year; (c) simulated revenues, €/year



Regional scale. Immediately evident is the very large use of water resources in the northeast, near the border with Lombardy, in correspondence with the rice production area cultivated with the traditional seasonal flooding technique.

Figure 18.4b presents the spatial distribution of revenues with the current system of water charges (excluding energy), highlighting a concentration of large contributors in the industrial districts of the metropolitan area of Turin in the centre of the Region (automotive), of Biella in the northeast (chemical and textile) and of the Crescentino area in the east (mainly chemical). Figure 18.4c (simulated revenues with the proposed water pricing system), compared with Fig. 18.4a (water withdrawals), highlight the move towards a closer compliance with Article 9 of the WFD.

This insight is statistically confirmed by a correlation analysis between water withdrawals and the current and reformed revenues (Table 18.10). The Pearson index shows that there is a strong positive correlation (0.944) between withdrawals and the revised water charge.

An alternative way to measure the degree of compliance of the water pricing system with the user/polluter-pays principle is by constructing Lorenz curves relative to average values (at the water basin level) of nominal withdrawals, and of current and revised revenues (Fig. 18.5). A perfect alignment of the revenue curve with withdrawals (in black) would represent a perfect correspondence between the pressure exerted by users on water resources and their payments for water abstraction.

The distribution of the proposed revised charge (in green) moves closer than that of current charges (in red) to the distribution of abstractions. A perfect alignment is

Table 18.10 Correlations measures and user/polluter-pays principle

	Current charge	Revised charge
Pearson index	0.284	0.944
Confidence interval (95%)	[0.270; 0.298]	[0.942; 0.945]
<i>p</i> -value	0,000	0,000

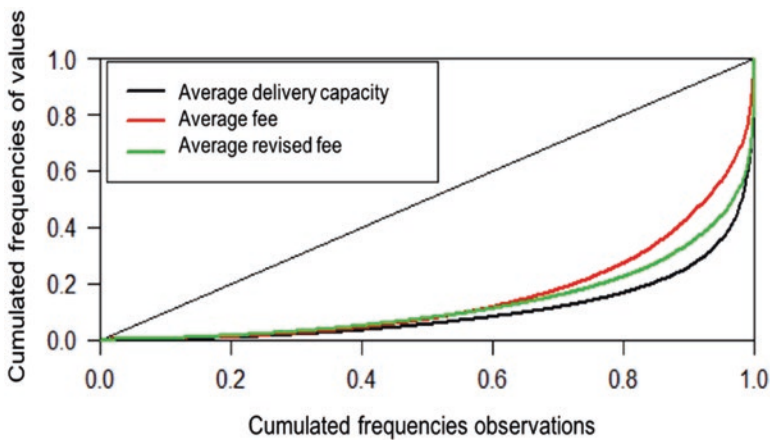


Fig. 18.5 Lorenz curves

prevented by the presence of minimum payments and the weights introduced in Eq. (18.2) to take into account the qualitative status of water bodies.

18.6 Conclusions

This chapter describes the pioneering attempt by the water authorities of the Piedmont Region, in the northwest of Italy, to envision a system of public water abstraction charges based on the assessment of the environmental costs entailed by water use. The reform design endeavours to face and explore the practical difficulties and constraints that arise when implementing Article 9 of the WFD in the real world, and to work as a testing ground for more general reforms of the Italian system of water pricing.

The fundamental point in the proposed system is to make the burden of water pricing proportional, for users, to the impact their use of water imposes on the environment. Such impact arises from the environmental cost both of subtracting resources to freshwater ecosystems, where they provide a variety of supporting, regulatory, and recreational services; and of returning water to ecosystems, after human use, in a state qualitatively degraded with respect to the original one, harming aquatic life and environmental quality. In addition, the reform design takes into account that the marginal damage caused by abstracting water and by returning it polluted also depends, in turn, on the quantitative and qualitative status of the concerned water body.

Precondition for implementing any incentive-based pricing reform is the diffusion of flowmeters to measure the actual abstraction by each licence-holder: efficiency of water use is attainable to the extent the pricing method affects the demand for water.

Water abstraction controls, technically feasible and in place in several parts of the world, still have little diffusion in Italy. In Piedmont, some progress is under way after they have been made compulsory by a Regional regulation issued in 2017 as ex-ante conditionalities to access funding from the EU's CAP. There is probably scope for EU and national level policy guidelines in this domain.

Reforms of abstraction charges such as the one presented here, however, do not necessarily need to wait for a complete adoption of metering devices before being introduced. A transitory phase with water charges calculated on nominal withdrawals rather than on the actual quantities withdrawn would not be able to elicit the full potential incentive to use water efficiently, but would still represent a progress towards the internalisation of the externalities of water use. In addition, it could itself serve the purpose of fostering the diffusion of meters – for example, by using the maximum rather than the average flowrate of licences as the base for calculating water charges for users who have not installed a metering system, or by allowing charge reductions as incentive for the transmission of water abstraction measurement data in real time.

Moving from the theoretical determination of an efficient water charging scheme to its practical implementation requires quantifying the value of the environmental damage associated with water use, and we have shown with this work that, despite data limitations that forced several approximations, it is doable. More sophisticated options for monetary valuation are available, and can be explored in further research and case studies. Here we have chosen, at several junctions, the option that would lead to approximate environmental costs by default. This notwithstanding, the estimate of the costs imposed on the environment by the current patterns of water use points to monetary values that would be considered, if we decided to recover them through water pricing, a disproportionate share of net incomes, particularly for agriculture. The same result was found by Galioto and colleagues for the case of the Emilia-Romagna Region (Galioto et al. 2013).

On the ground of the affordability principle, the final step of the designed water pricing system provides for a rescaling of the environmental-cost-recovering water charges compatible with the maximum incidence a community deems appropriate. In our simulation, we set a cap for the incidence of water charges at 1.5% of net annual incomes. It is very unlikely that this would represent an unaffordable burden for companies using public water as an input in their production process, in any sector. Yet, it represents a move towards recognising water as an essential collective asset of high economic value, which requires to be protected from depletion and degradation, and whose use in economic activities should respect the same logic of efficiency and minimisation of wastage that apply to the use of energy or any other productive input.

The significant rescaling made necessary by the affordability analysis conserves nonetheless the proportionality of users' contribution to the level of damage caused by their own extractions and by the qualitative state of the water as they return it to the environment. Where the state of the returned water is affected by non-point forms of pollution, as for example nitrates and pesticides, and hence environmental monitoring authorities cannot ascertain a direct connection with specific sources, the median value for the ecological status of the sub-basin where the abstraction takes place is used to determine water charges for users in that area. This introduces a principle of joint responsibility among users of a common resource.

The objective of revising water pricing is not raising revenues, but introducing a well-defined and specific incentive to conserve and protect a precious resource. For it to be effective, a crucial point is the elasticity of demand for water. A correct estimate of how the demand for water reacts to price increases is necessary to better investigate the distributive impacts, but even more to understand how large the margin for improving water efficiency is. If water requirements and polluting discharge were substantially a rigid constraint for the economic activities involved, then the proposed system would merely imply a redistribution of resources from users/polluters to the larger society. This would still represent a move towards the implementation of Article 9 of the WFD and would help generate fiscal space for environmental protection and restoration measures, but would have limited incentive potential for changing water use behaviour.

The system design aims at being simple enough to facilitate implementation and transparency, but also adaptive to specificities of local contexts. The calculated licence charge can be modulated through reductions for particular categories of activities, such as, for example, agricultural activities classified as High Nature Value Farming or with organic certifications, industrial activities with environmental certifications (e.g., EMAS), activities of particular socio-economic significance (e.g., activities in mountain environment or other marginal areas, or contribution to management of flood events).

Last, but not least, are the questions of consumptive versus non-consumptive uses and of the relationship between irrigation and groundwater recharge. In Piedmont the most employed irrigation techniques are irrigation by flow (66.62%) and submersion (29.89%) (INEA 2011). Moreover, 96.5% of the Regional irrigation is considered to be of low efficiency. There appears to be potential for increases in water efficiency through a reform of water pricing. However, the transition from a low-efficiency to high-efficiency irrigation systems optimizes the use of water by plants and leads to water saving, but in doing so it decreases the quantity of water that percolates into the aquifers and contributes to their recharge. This, in turn, in some contexts, may impact negatively on feasible groundwater abstraction rates and reduce the dilution of groundwater NO₃ concentrations. The ideal system of water pricing, towards which we should be working in the future, ought to be formulated in terms of water balances, rather than withdrawals – although maintaining a focus on the qualitative features of the returned water.

References

- Bateman, I. J., & Langford, I. H. (1997). Non-users willingness to pay for a national park: An application and critique of the contingent valuation method. *Regional Studies*, 31(6), 571–582.
- Brouwer, R. (2006). Valuing water quality changes in the Netherlands using stated preference techniques. In D. Pearce (Ed.), *Environmental valuation in developed countries (Case Studies)* (pp. 132–147). Cheltenham: Edward Elgar.
- de Groot, R., et al. (2012). Global estimates of the value of ecosystems and their services in monetary units. *Ecosystem Services*, 1(1), 50–61.
- EC. (2003). *Common implementation strategy for the water framework directive (2000/60/EC)*. Brussels: European Commission.
- EC. (2012). *EU blueprint to safeguard Europe's water resources* (Communication No. COM (2012) 673 Final). Brussels: European Commission.
- EEA. (2018). *European waters assessment of status and pressures* (EEA Report No 7/2018). Luxembourg: European Environment Agency.
- Galioto, F., et al. (2013). An assessment of disproportionate costs in WFD: The experience of Emilia-Romagna. *Water*, 5(4), 1967–1995.
- Hanley, N., & Czajkowski, M. (2019). The role of stated preference valuation methods in understanding choices and informing policy. *Review of Environmental Economics and Policy*, 13, 248–266.
- INEA. (2011). *Rapporto sullo stato dell'irrigazione in Piemonte*. Programma interregionale Monitoraggio dei sistemi irrigui delle regioni centro settentrionali. www.crea.gov.it/
- INEA. (2014). *RICA database*. www.rica.inea.it/public/it/index.php. Accessed 20 Sept 2018.

- ISTAT. (2019). *Reddito netto familiare medio e mediano per fonte principale di reddito e anno*. www4.istat.it/it/piemonte/dati?qt=gettable&dataset=DCCV_REDNETFAMFONTERED&dim=5,0,9,0,5,10,24,6,5,3,182,12,13,0&lang=2&tr=0&te=0. Accessed 17 July 2019.
- Jensen, C. L., et al. (2013). A practical CBA-based screening procedure for identification of river basins where the costs of fulfilling the WFD requirements may be disproportionate – Applied to the case of Denmark. *Journal of Environmental Economics and Policy*, 2(2), 164–200.
- Klauer, B., Sigel, K., & Schiller, J. (2016). Disproportionate costs in the EU Water Framework Directive – How to justify less stringent environmental objectives. *Environmental Science & Policy*, 59, 10–17.
- Milon, J. W., & Scrogin, D. (2006). Latent preferences and valuation of wetland ecosystem restoration. *Ecological Economics*, 56(2), 162–175.
- Raggi, M., et al. (2009). *AQUAMONEY – Development and testing of practical guidelines for the assessment of environmental and resource costs and benefits in the WFD – Pilot case study PO Basin*. Available: www.ivm.vu.nl/en/Images/D42_Case_study_report_Po_Italy_tcm234-188875.pdf
- Ramajo-Hernández, J., & Salazar, S. (2012). Estimating the non-market benefits of water quality improvement for a case study in Spain: A contingent valuation approach. *Environmental Science & Policy*, 22, 47–59.
- Regione Piemonte. (2017). Condizionalità ex ante riferita al settore delle risorse idriche prevista dall'Accordo di Partenariato ai fini dell'accesso ai fondi europei relativi al Programma di Sviluppo Rurale 2014–2020. Implementation of DGR n. 43–4410, 19 dicembre 2016.
- Regione Piemonte. (2018). Piano di Tutela delle Acque (PTA) – Revisione 2038. Implementatio of DGR n.28–7253, 20 luglio 2018.
- Regione Piemonte. (2019). *GE.RI.CA – Gestione Riscossione Canoni*. www.regione.piemonte.it/web/sites/default/files/media/documenti/2019-02/canoni_2019.pdf. Accessed 15 July 2019.
- Rey, D., et al. (2018). Role of economic instruments in water allocation reform: Lessons from Europe. *International Journal of Water Resources Development*, 0, 1–34.
- Russi, D., et al. (2013). *The economics of ecosystems and biodiversity for water and wetlands*. London/Brussels/Gland: IEEP/Ramsar Secretariat.
- Sapino, F., et al. (2020). A microeconomic multi-model ensemble experiment to assess socio-economic effects of agricultural water pricing reform in the Piedmont Region, Italy. *Journal of Environmental Management*, 267, 110645.
- Sardonini, L., Viaggi, D., & Raggi M. (2011). EPI-WATER WP3 (Ex-Post Case Studies), Deliverable 3.1. (Review reports). Water tariffs in agriculture – Emilia Romagna case study. Project Report. http://www.feem-project.net/epiwater/docs/d32-d6-1/CS7_Emiliana%20Romagna.pdf

Vito Frontuto holds a Master Degree in International and Diplomatic Sciences and a Ph.D. in Economics from the University of Turin. He is Adjunct Professor in Environmental Valuation Methods and Research Fellow of the Department of Economics and Statistics of the University of Turin. He is involved in many research projects on environmental resource management, water management, renewable energy and fiscal reforms of national and international interest. He is author and/or co-author of more than ten publications in international journals. He is member of the Scientific Committee of the Turin School of Local Regulation and of the Fondazione per l'Ambiente T. Fenoglio.

Silvana Dalmazzone obtained her Ph.D. from the University of York and is Associate Professor of Economic Policy at the University of Turin, where she teaches Environmental and Natural Resource Economics. She served as policy consultant for UNEP and other national and international environmental agencies, as Vice-President of the Italian Association of Environmental and Resource Economists, as board member for national and international conferences and scientific coordinator or member of national and international research projects. Her main publications are

in the fields of economic-ecological modelling, economics of biodiversity loss, decentralisation in environmental governance, environmental accounting, resilience, valuation of ecosystem services, and green economy reforms in developing countries.

Paolo Mancin is Head of Unit at the Water Protection Department of the Piedmont Region, where his main tasks are the implementation of Directive 2000/60/EC, technical legislation for water uses and related fees and the coordination of working groups for the Regional law on hydro-powerconcessions. He is a designated member in Regional, national and international Committees. He took part in various European projects and worked, as a Reviewing Expert, at Basin Authorities in Sweden (Kalmar) and Spain (Barcelona). He published technical reports on environmental impacts on water, river restoration, and participatory processes for water management.

Alessia Giannetta holds a degree in Biological Sciences from the University of Turin. She is a functionary at the Environment Department of the Piedmont Region, where she deals with issues related to water protection. She collaborates with the river basin Authority on the Po River Basin Management Plan, in application of the Water Framework Directive; she also takes part in the elaboration and implementation of the Regional Water Protection Plan required by national legislation. She is involved in the implementation of European projects related to the governance and requalification of river ecosystems.

Davide Attilio Calà holds a degree in Statistics from the University of Turin. He is a functionary at the Environment Department of the Piedmont Region, in charge of planning and managing the quantification and collection of fees for public waters use and mineral waters abstraction. He is involved in policy design for water management and the recovery of environmental costs.