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Executive Summary

The EPI-WATER project consortium analysed a large number of *economic policy instruments* (EPIs) in an ex-post (Lago et al., 2012) and ex-ante assessment mode (Defrance et al., 2013; Delvaux et al., 2013; Gómez et al., 2013; Kossida et al., 2013; Skou Andersen et al., 2013; Ungvári et al., 2013). The team has developed an *assessment framework* (AF), in order to ensure that all assessments are comparable and follow the same assessment principles (Zetland et al., 2011). Based on the experiences from the assessment exercise, the AF has been revised (Zetland et al., 2012) and extended. The 30 ex-post and 5 ex-ante assessment reports summarise the analysed EPIs on more than 2,000 pages. Ancillary knowledge base is provided in the supplementary material. The large body of empirical knowledge collected by the EPI-WATER team has informed several major synthesis and advocacy reports in Europe (CEPS, 2012; EEA, 2012a; OECD, 2013) and was used as a selective knowledge base for the 2012 EU Water Policy Review (EC, 2012a). The intermediate synthesis of the ex-post assessment exercise (Lago and Möller-Gulland, 2012) revealed substandard policy design and assessment practice, non-conform with the EU principles of *smart* regulation. Only a few analysed EPIs contained a clear specification of what the instruments were meant to achieve. A comprehensive ex-ante assessment was conducted even in fewer instances and we have found no evidence of a continuing monitoring of the achievements or a periodic ex-post assessments.

The next step of the analysis focussed on an *ex-ante* assessment of prospective EPIs within specific conditions in five representative river basins. The case study addressed different water policy issues: floods and excess waters in the Tisza river basin district (RBD) sub-unit (Hungary), *droughts* and *water scarcity* in the interconnected Segura and Tagus RBDs (Spain), and in Pinios river basin (Greece); *nitrate pollution* in the Seine-Normandie RBD (France) and Odense (Denmark); and restoration of *good morphological conditions* in in the Seine- Normandie RBD (France). Each ex-ante case study analysed the environmental challenges in the respective river basin area in a holistic way and proposed a reform of the existing interplay of the policy instruments.

This report, one of a series of final project reports (Delacamara et al., 2013; Møller et al., 2013; Mysiak, 2013), summarises the key lessons learned from both ex-port and ex-ante assessment exercises and role/potential of the selected EPIs for the European water policies. The extensive body of knowledge collected is ill-suited for a synthesis in general terms. Hence, the report is organised in two parts.

PART I focusses on the principal EPIs analysed by the team, including *incentive water pricing* and *trading*, (*nitrate*) *pollution tax*, *payments for ecosystem services*, *water emission trading*, and *transient flood storage*. Each instrument is described in terms of policy challenges to which it contributes to respond, design principles and criteria, prerequisites for a successful implementation, and potential environmental and economic outcomes. The latter is based on the empirical research conducted by the EPI-WATER project consortium, complemented by a literature review and expert knowledge.

Incentive water pricing: The primary function of an incentive water pricing is to make individuals pay for the use they make of water resources, that is water consumption and emission of pollutants, or alternatively, to accept payments for actions targeted at *safeguarding* water ecosystems such as "greening" infrastructure. The use of water pricing instruments is compelled by the EU Water Framework Directive (2000/60/EC). Yet the recent EC Water Policy Review has lamented limited application of '*incentive and transparent water pricing*' (EC, 2012a, p. 10) and equalled not putting a price on scarce water resources to '*environmentally-harmful subsidy*' (ibid, p.10). Water pricing pursues multiple policy goals, seemingly at odds but reconcilable in principle: water use efficiency, that is avoiding wasteful use of water; allocation efficiency, thus maximising overall society's benefits from water uses; financial viability, meaning ability to compensate capital, skills and technology needed to ensure water services and sanitation; and social equity, standing for affordability of water as a public interest good. Flat-tariffs, taxes and fees perform better in terms of cost recovery but lack incentives for water conservation, and are unable to embrace environmental and resource costs. *Marginal water pricing* (MWP) is better suited for improving efficiency of water uses which, however, is contingent on the ability to seize the most responsive (elastic) portion of water users' demand. Short-term incentive pricing is efficient if it signals the scarcity value of water, while long-term incentive pricing is efficient if it reflects the opportunity costs associated with alternative uses of water. Multi-part tariffs that include at least one volumetric part have the potential to balance the objectives of cost recovery and water conservation: a higher fixed tariff-part favours cost recovery objective, whereas a higher variable tariff-part is more suitable as an incentive for fostering water conservation. MWP is widely used in water supply and sanitation but only in a limited way in agriculture where arguably the political will is keeping water prices low and measurement of water consumption is problematic. Flat-rate pricing schemes are typical even in countries where water abstractions for irrigation exceed 60 per cent of all withdrawn water and the ratio of irrigated over arable land is greater than 30 per cent (e.g. Greece, Malta, Spain and Italy). MWP has the greatest potential in places experiencing recurrent water shortages, where infrastructure exists for volumetric allocation, and where water volumes and profitability of use are different across the existing uses.

Trading with water entitlements is a practicable where long-term renewable water resources are unable to meet water demand, or where drought-induced water shortages are recurrent and cause substantial economic losses. Water trading is conditional on existence of well-defined, enforceable and marketable water rights, instructed flexibility of price determination between potential buyers and sellers, and information such as an adequate price-revealing mechanism. Water trading increases the economic efficiency of water allocation and is advisable, where prerequisites are met, as a part of policy portfolio aimed at regaining control over groundwater resources, and increasing resilience and adaptive capacity of water-intensive economic sectors. It is advisable to design water trading schemes as a combination of intra-basin markets – so as to improve water allocation efficiency at local levels and to enhance technical efficiency of water use - with inter-basin water market scheme enabling water transfers from relatively less water-scarce to more water-scarce areas. Assuming no transaction costs, opportunities to shift water to its most productive uses exists when the value of water is variable across water users. Where such differences exist, in principle there is an opportunity for individuals or stakeholders to engage in a bargaining process to reach mutually beneficial agreements at a price between the maximum willingness to pay (WTP) of the might-be buyers and the minimum compensation that potential sellers are willing to accept for water use rights (under temporary or permanent cession). An environmentally neutral market scheme needs to ensure that trading does not have any impact on the environment. Negative environmental effects can be avoided if potential sellers cede their water use right but the physical returns to the environment remain in the source area and only depletion (instead of water use) is transferred. How much water can be transferred between two river basins depends water losses during the transfer and other environmental constrains. Transport costs, transient losses and environmental constraints result in substantial reductions in the amounts of water that might be transferred by an efficient market and an sizeable increase of the equilibrium price. Generally, opportunities for water trading decrees with distance as transport costs increase. Experiences from countries where water trading is in place (e.g. Chile, Australia, United States, Spain) are instructive for designing effective, equitable and sustainable policy schemes.

Nitrogen Taxes: Agriculture is the largest source of reactive nitrogen pollution in surface and groundwater, causing eutrophication (and subsequently algal bloom) and harmful effects to human health, despite the Nitrate Directive (ND, 91/676/EEC) has been in place for more than twenty years. A nitrogen tax can be implemented as the *mineral fertilizer tax* (M), the *nutrient-input* tax (N) and the *nutrient-loss tax* (L). A M-tax targets nutrient inputs from artificial fertilizers only, whereas a N-tax additionally targets the nutrients in animal fodders (that in turn transform to organic fertilizers). The L-tax focuses on application residuals, and targets the *ex-post loss* from the root zone as can be established for each farm. A limited evidence shows that the use of fertilizer is relatively inelastic, but M- and N-taxes can lead to substantial reduction (up to full elimination of use) in the case of sizeable increases of the mineral fertilizer prices. The (potentially high) economic burden on farmers may be reduced through recycling of revenues (e.g. through reduction of land value taxes) so as to maintain the overall tax burden constant. Alternatively, the revenue could be used as basis for the payment for ecosystem services. Nevertheless, the additional direct costs for farmers can outweigh the monetary value assigned to environmental benefits ascertained. The L-tax scheme is often perceived as more fair in that only the losses are targeted. However, the European Court of Justice (ECJ) has declared this approach to be incompatible with the ND that specifies the allowable field application. Schemes for taxation of nitrogen could be relevant for catchments across Europe where there is excess application of fertilizers, leading to pollution of surface waters and aquifers. The complexity in curbing agricultural pollution is high due to the many uncertainties in the scientific understanding of leaching processes. The regulatory challenge is to ensure that better use is made of organic fertilizers while minimizing on use of mineral fertilizers. It is more complicated and costly to administrate organic fertilizers, and the burden is likely to fall on crop farmers rather than on livestock farmers. Comprehensive policy packages combine taxation with book-keeping of manure and fertilizers.

Payments for Ecosystem Services: Ecosystem services (ES) embody the benefits people obtain from ecosystems that are eventually translated into valuable goods. The ES have an economic value but no price. The failure to account for their true social value lead to market distortion and, ultimately, insufficient level of protection with lasting, in some cases irreversible, damage. Globally, the annual loss from land-based ecosystems alone has been estimated to 50 billion Euro (EC, 2008a). The *Millennium Ecosystem Assessment* (MEA; Millennium Ecosystem Assessment, 2005) illustrated evidence of an unprecedented speed of human-induced ecosystem changes, with '*substantial and largely irreversible loss in the diversity*'. Payments for ecosystem services (PES) allows capturing the economic value of the ecosystem services and creating an incentive for their preservation. PES are characterised as '*voluntary transaction where a well-defined environmental service* (or a land use likely to secure that service) *is being "bought" by a service buyer from a service provider*' (Wunder, 2005). *Conditionality* means that payment is contingent upon the service being continuously provided. The EPI-WATER research focused on schemes where the buyers of the environmental service are the actual beneficiaries of the environmental service ('user-financed' programs in contrast to the 'government-financed' programs).

Water emission trading (WET) is a cost-efficient way of pollution control for surface and groundwater bodies, sizeable improvement of water quality and preservation of the integrity of riverine ecosystems. Compared to traditional regulatory or price-based economic instruments, WET makes it possible to reduce the costs of achieving water quality goals by shifting pollution reduction to where the *marginal abatement costs* are lower, until these are equal for all sources. The ultimate advantage of WET is that it offers some flexibility for economic growth without compromising the environmental goals. The benefits of WET are attainable especially in larger basins, or terminal water bodies, characterised by a larger number and variety of pollution sources and different marginal abatement costs. To avoid a high concentration of pollutants in some parts of the basin, trading rules need to be carefully designed and offsets are usually permitted only between the sources at the same site or between up- and downstream sources. In practice, the text book advantages of the WET are really realised. Instead of following a *textbook* design, the existing schemes employ a combination of WET principles, making the programmes simpler and more acceptable by regulated entities and general public. Despite the growing acceptability and use of ambient air emission trading, there are only few practical experiences or tangible application plans to introduce WET in Europe. Globally, only few programmes have demonstrably led to the expected cost reduction. Under current European legislation, it is unclear whether and under which constraints the exchange of pollution permits could be granted. The Industrial Emission Directive (IED), and before the Directive concerning Integrated Pollution Prevention and Control (IPPC), require individual standards are specified for each source based on the best-available-technology. In South Sweden, the proposed CEASAR programme (Certificates for Efficient Allocation of Shares Adjusted to Retention), set to be put in practice in 2016, is targeted at point sources, and is expected to reduce their cumulative nitrogen emissions by a third (3.000 tonnes).

Transient storage of flood: Floods and wind storms are natural hazards that incur the largest economic losses in Europe, the average annual direct cost of floods have been estimated to amount to 3-4 billion Euro and annual expected damage (AED) to 5-6.4 billion Euro. Human induced climate change will in many areas across Europe increase the flood risk substantially. Progressing urbanisation and soil sealing, along with floodplain development, and wetland conversion or degradation have contributed to increased run-off and flood risk. With increasing flood damage and risk, transient storage of flood waters (TSF) on land with relatively low damage potential has gradually become a part of disaster risk reduction (DRR) strategies across Europe and beyond. The economic rationale of TSF rests on damage reduction through controlled inundation of areas with low-value land uses in order to protect areas with high-value land uses, notably urban residential or industrial developed areas. TSF is a part of natural water retention or flood attenuation measures, capable of reducing and delaying peak discharge. The economic policy instruments (EPIs) in the context of TSF may take different forms along a continuum of policy measures from a land expropriation and successive (back-)leasing through easement, up to service contracts. Compensation for land value loss and damage may take different forms. The land value loss is compensated as a portion of the market value of the property but can vary depending on the envisaged (future) land use. The damage compensation can take form of a one-off payment at the time of imposing servitude, or annual fixed rewards, or irregular damage reimbursements. The different ways of damage compensation have an implication on how the associated risk (e.g. of increased frequencies of triggering events) is shared between private and public bodies. If flood protection is recognised as a water service, as in the view of the European Commission, than the costs of TSF will have to be recovered at least to some extent either through beneficiary-pay or polluter-pay principle. Both are problematic because the flood externality created through land conversion and management upstream is both, difficult to quantify and a result (legacy) of the historical development.

PART II of this report focusses on issues relevant for choosing an EPI or building a policy portfolio; matching the chosen instrument to the existing institutional framework and the opportunity for change; and ensuring that the EPI contributes to the integration of water and other sectoral policies, or does not produce negative side-effects to these sectors. Moreover, an innovative policy mix aiming at water security is outlined. Finally, the role of environmental taxation and tradable environmental permits in Europe is discussed.

Choosing an EPI? Governance and the choice of a policy mix. A defying characteristic of an EPI is the use of private financial incentives to induce some people to change in a way which delivers a collective gain. Because the change is voluntary, an EPI exploits the differences within the water uses/users: the greater are the differences, the greater is the potential to prompt a change in a way that achieves the intended objective. For an EPI to produce expected results, it is irrelevant as to who changes, how they change, where the change occurs and when it occurs. If any of these conditions make a significant difference to the achievement of the collective good then an EPI is less likely to be appropriate. The primary design question of an EPI is hence how to induce change, or with other words, how to tailor incentives to overcome barriers in making the desired change. The simplest case is when the change could be made through a change in behaviour (e.g. turning off the tap when brushing teeth). But often changes require both an available technology and investment. There are institutional barriers as well; for example, those renting properties do not have the incentives to make permanent changes to the fabric of the property and may be inhibited by the rental agreement from doing so. There may be physical issues as well. A clear understanding of what are these barriers and what is necessary to overcome them, and consequently what is required of the EPI or other instrument is required. Making a change has two types of cost: the consequences of the change (e.g. changing cropping patterns, adopting demand management techniques, constructing local wastewater reuse); and the costs of change itself. The net consequential gains from the change have to be sufficient both for individual making the change and to the collective if the change is to occur. To provide a collective gain, the gains from the change must exceed all the costs of making the change and inducing the change. A specific problem with applying EPIs in water management is that catchment are systems and it is an innate feature of systems that making a change in one location will result in changes elsewhere in the system. Any intervention, including an EPI, must therefore be tailored so as to ensure that the performance of the system is improved. EPIs are still an innovative approach, it is in the nature of innovations that some will fail. Ideally, they should be reversible, or tried locally first.

Institutional design and sequencing. The performance of EPIs is influenced by the fit between the policy design and the institutional set-up in which they operate. Institutions are the formal rules and informal norms that define and modify the choice sets of individuals and their interactions. Most institutions are adapted to local conditions, with a purpose of balancing competing interests. Governance of common-pool resources require clear definition of boundaries of a resource system, collective action, monitoring, sanctions and conflict adjudication mechanisms (Ostrom, 1990). Management of natural resources involve legal, environmental, technological, financial and political considerations associated often with sizeable transaction costs. While some transaction costs are essential as a part of enforcement, others rather hinder institutional and economic performance. Yet, the relevance of transaction costs (especially bargaining costs required to come to an acceptable agreement with all the parties involved) has been sometimes overplayed, thus delaying the necessary water policy reform. As long as the transaction costs of the water policy reform are perceived to be larger than the opportunity costs of the status quo, the reform will not be implemented. Institutional failure may hinder the performance of otherwise theoretically sound EPIs. The analysis of the institutional setup creates the conditions to assess whether the enabling factors required are already in place or to what extent institutional change must be somehow modified to make EPIs implementable in an equitable and efficient manner. It also helps to identify whether, further to institutional change, the design of EPIs itself needs to be refined so as to enhance their efficiency and reduce their implementation costs. Both the institutional change and the potential effectiveness of EPIs are contingent on their social acceptability. *The introduction of EPIs is part of a transition in water policy*, taking. different forms: i) by improving the instrument's design, so as to reduce direct transaction costs while preserving its effectiveness; ii) by streamlining and sequencing the instrument's implementation strategy and reducing other transaction costs connected to the institutional setup; iii) by designing better strategies to avoid institutional lock-ins, while focusing on social and political acceptability; iv) by making the most out of synergies between different incentives and between EPIs and other policy instruments (e.g. information, command and control)

Integration of EPIs with policy instruments adopted in other sectors such as agriculture and energy. Decades back, the European Union's policy makers have recognised the potential granted by economic incentives and/or disincentives for driving individual and business behaviour toward achieving sustainable development objectives, including protection of healthy environment and efficient use of natural resources. Pursued through a number of statutory acts, the EPIs permeated environmental (including inland and marine water) legislation. More recently, the firm commitment to *low carbon*, *resource efficient* and *socially inclusive* growth and economy has become a cornerstone of the EU 2020 Strategy, a part of which is a budgetary-neutral shift of the tax burden away from labour and capital to consumption, property and environment. *Environmental policy integration* means placing environmental considerations at the heart of decision-making in other sectoral policies, such as energy agriculture or industry, and balancing environmental against economic and social interests. From an environmental perspective, integrating common policy aims should result in positive environmental outcomes, economic efficiency and further equity. Water policy is ultimately about making economic development and social welfare enhancement compatible with the improvement and protection of water resources. EPIs can play different functions to this end, through: i) putting in practice the polluter/user pays principle; ii) encouraging full cost recovery; iii) constituting a part of risk management; iv) triggering innovation; and v) increase cooperation across sectors. But EPIs are but one piece of the institutional change required in current water management practice. Water governance challenge consists in finding a suitable non-coercive mechanism that motivates collective action. We recommend that i) policy integration aspects are clearly expressed as a part of the EPI objective; ii) the design of the detailed mechanism is embedded in policy formulation; iii) implementation and enforcement of EPI is made by government agencies and other actors in a cross policy way; iv) implementation of the EPI properly sequenced; and v) exemptions or extensions of deadlines are allowed for sectors into which integration should take place in order to allow the sector to cope with the changes.

Pricing water security: In water scarce economy, the role of water pricing for providing water security has to be redefined. *Water security* is the result of i) reducing water demand to a level that can be sustainably covered with the existing resources, ii) promoting savings and technical efficiency so that new economic activities can take place without compromising long-term sustainability, and iii) building collective means to curb current scarcity trends, and iv) developing institutions and assets enhancing resilience in dry periods. The set of minimum changes required in water governance to reduce water scarcity and increase water security in the long term include: *Closing the river basin district*, that is accepting that any further advance in water supply cannot be met with additional water resources from the river basin. This includes that use rights are well specified and enforced, while contingent on rainfall and runoff; *Controlling scarcity trends*, which implies hindering groundwater over-drafting and allowing for the replenishment of depleted aquifers by natural or artificial means and the recovery of minimum environmental flows; *Building up collective water security*, embracing strategy to identify resources available during normal and dry periods, making allocation of water resources contingent on natural supply variability, and building flexibility in such a way that potential market and environmental damages are minimized. Security needs to be built in an anticipated and coordinated manner and the institutional framework. Scarcity comes along with increasing opportunities to price water in accordance to its increased economic value. The main opportunities to price water security in a water scarce economy include i) *willingness to pay for water security*; ii) *Managing the water portfolio*, and iii) *bridging the efficiency gap*. The efficiency gap is a clear opportunity that can be used to obtain a combination of economic and environmental benefits at the same time by producing more without further environmental degradation or to obtain the same market values with less pressures over water ecosystems. The proposed reform in water pricing is meant to make pricing an actual mechanism to match water supply and demand (contributing to the river basin closure), and assigning each water source a price depending on its role in terms of the supplied quantity and its weight for water security in the short and the longer term. The pricing of water security is introduced as an economic policy instrument (EPI) for water management through the use of a market mechanism to guarantee the existence of buffer stocks and to allow for the recovery of depleted aquifers as well as to reduce water demand. Water security as a public good that must be paid for collectively.

Environmental taxes: Although the theory underlying the environmental taxes were laid out already in the seminal work of Pigou back in 1920, it wasn't until much later that they found a practical application. The early European water effluent charges and Japan's air pollution levy proved reasonably effective in curbing emissions, which helped stimulate further reflections and proposals for making use of economic instruments around the world. Nowadays taxes addressing water pollution and use vary substantially across the EU Member States and the data collection is difficult due to highly heterogeneous and complex governance regimes. Overall, they play a small role in the environmental tax revenues dominated by transport fuel and energy taxes which in some countries such as Denmark and Ireland account for between 35 and 44 per cent. From 1990s onwards attention shifted from viewing such taxes as being useful instruments for environmental policy purposes, towards their potential under the consideration of more fundamental changes in taxation policies, whereby other taxes could be substituted with environmentally-related taxes. Today the environmental tax reform (ETR) is a part of the Europe 2020 resource efficiency initiative, aiming at among others limiting the environmental impacts of resource use while at the same time improving economic performance, hence decoupling environmental pressures from economic growth. The stated goal of the 2020 resource efficiency roadmap include 'a major shift' away from labour to environmental taxation and 'a substantial increase' of the share of public revenues from environmental taxes. Shifting of the tax burden, with more emphasis on environment related tax bases, is being considered as part of the European Semester for individual EU Member States.

Tradable environmental permits (TPs) are gaining on importance in Europe but are underdeveloped in water policy domain. Unlike price instruments, the TPs entail transactions between authorised users and enable a shift of resource to higher-value use, and/or a minimisation of compliance costs, and/or a re-distribution of scarcity rent. Property rights are social institutions that define or delimit the range of privileges granted to individuals of specific resources. The exercise of rights is qualified by rules that authorise or require specific actions to protect the resource and the interests of downstream right holders. Contrary to a layman's view, property rights do not constitute ownership of a 'resource system' but a privileged access to a resource. Water rights are hence shares of water flows and/or volumes of water that may be withdrawn for specific use. Emission rights are appropriated shares of the assimilative capacity of the environment, that is amounts of potentially harmful substances that are naturally removed or diluted without causing harm to ecosystems and humans. Development rights, a part of the bundle of land related rights, entail authorisations to create impervious surface by soil sealing and/or changing land-cover/use with substantial effects on biodiversity. TPs are enabled by unbundling alienation from the pool of other rights and impose limits on the exercise of rights through a collective-choice action. The EU environmental legislation is rather reticent about the use of TPs for managing water and other environmental resources, with few exceptions. Some EU MS moved on to exploit practically the EPs: England and Wales have pledged to reform the regime of water abstraction licences, and Sweden and Finland proposed water emission trading schemes meant to contribute to the *Baltic Sea Action Plan*. Still, the different purposes of the TPs pose diverse expectation on the legal regimes of the permits and their *security*.

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D 5.1 - Synthesis report

1 Introduction

The EPI-WATER (*Evaluating Economic Policy Instrument for Sustainable Water Management in Europe*) is a EU FP7 funded project (01.2011-12.2013) set to analyse the performance of economic policy instruments (EPIs) in Europe and beyond, and produce recommendation for a better exploitation of the potential of these instruments for achieving the EU environmental objectives (see for more detail *www.epi-water.eu*). The project consortium was composed by eleven European research institutes[1] from nine EU member States. In addition, ten academic experts from Australia, Chile, China, Israel, and United States[2] joined the team and helped to expand the evidence about the performance of some economic policy instruments not commonly applied in Europe. Throughout the project life-span, close involvement of a group of water management practitioners and policy makers[3]

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[1] Fondazione Eni Enrico Mattei, Italy; ACTeon Sarl, France; Ecologic Institute GmbH, Germany; University of Bologna, Italy; Wageningen University, The Netherlands; National Technical University of Athens, Greece; Madrid Institute for Advanced Studies in Water, Spain; University of Valencia, Spain; Middlesex University, Flood Hazard Research Centre, United Kingdom; Aarhus Univerisitet, Denmark; and Budapesti Corvinus Egyetem - Regional Centre for Energy Policy Research, Hungary.

[2] Tihomir Ancev, University of Sydney, Australia; Ariel Dinar, University of California – Riverside, United States; Guillermo Donoso Harris, Pontificia Universidad Católica de Chile, Chile; Charles Howe, University of Colorado, United States; Iddo Kan, The Hebrew University of Jerusalem, Israel; Mark Kieser, Kieser & Associates LLC, United States; Carolyn Kousky, Resources for the Future, United States; Xiaoliu Yang, Peking University, China; Andrew Yates, University of Richmond, United States; Michel Young, University of Adelaide, Australia.

[3] Bernard Barraqué, l'Institut des sciences et industries du vivant et de l'environnement, France; Viviane André, EC Directorate General Environment; Kevin Andrews, Defra - Department for Environment, Food and Rural Affairs, United Kingdom; Robert Peter, Collins, European Environmental Agency; Cristina Danes, Ministry of the Environment, Spain; Jacques Delsalle, EC Directorate General Environment; Henriette Faergemann, EC Directorate General Environment; Sarah Feuillette, Agence de l'Eau Seine Normandie, France; Eduard Interwies, Intersus Sustainability Services, Germany; Mats Ivarsson, Agency for Marine and Water Management, Sweden; Lukasz Latala, EC Directorate General Environment; Xavier Leflaive, Organisation for Economic Co-operation and Development, France; Irene Lucius, World Wide Fund for Nature, Belgium; Tania Runge, European farmers – European Agri-Cooperatives, Belgium; Cristian Rusu, Waters National Administration, Romania; Moroz Sergey, World Wide Fund for Nature, Belgium; Stefan Ulrich Speck, European Environmental Agency, Denmark.

ensured that the EPI-WATER research is conducted in a way that responds to the practical policy requirements and needs[4].

The project team first developed an assessment framework (AF), in order to ensure that all assessments are comparable and follow the same assessment principles (Zetland et al., 2011). Based on the experiences from the assessment exercise, the AF has been revised (Zetland et al., 2012) and extended. The AF makes it easier to systematically assess the effectiveness and impact of water policies. Standardisation makes it easier to compare policies, so that policymakers can sort projects from better to worse as well as understand *why* some projects or policies succeed or fail with respect to different assessment criteria. These comparisons facilitate institutional learning and adoption of best practises (Weikard and Zetland, 2013). The Framework compares the outcomes resulting from an EPI to the outcomes that would have resulted with either a "business as usual" baseline of no action or an "alternative scenario" that would have resulted from another policy intervention. The outcome-oriented criteria describe how EPIs perform. They include intended and unintended economic and environmental outcomes and the distribution of benefits and costs among the affected parties. Process criteria describe the institutional conditions (legislative, political, cultural, etc.) affecting the formation and operation of EPIs, the transaction costs from implementing and enforcing the EPI, the process of implementing the EPI, and the impact of uncertainty on the EPI (Zetland et al., 2013)

As next, the team selected representative economic policy instruments in Europe[5] (Lago et al., 2011) and beyond[6], taking into account the geographical coverage (see Annex 1), economic sectors and environmental pressures at which the instruments targeted. Each of the selected instruments was analysed separately using the AF as a guide. The ex-post assessment resulted in 30 review reports (Ancev, 2011; Branth Pedersen et al., 2011; Defrance, 2011; Dinar, 2011; Donoso Harris, 2011; Dworak, 2011; Gómez et al., 2011a, 2011b; Hernández-Sancho et al., 2011; Hernández-Sancho, F. Molinos-Senante and Sala-Garrido, 2011b, 2011a; Howe, 2011; Kan and Kislev, 2011; Kieser and McCarthy, 2011; Kossida and Tekidou, 2011; Kousky, 2011; Mattheiß, 2011; Möller-Gulland and Lago, 2011; Möller-Gulland et al., 2011; Mysiak et al., 2011a, 2011b; Rákosi et al., 2011; Sardonini et al., 2011; Schuerhoff et al., 2011; Ungvári et al., 2011; Viavattene et al., 2011; Yang, 2011; Yates, 2011; Young, 2011; Zetland and Weikard, 2011).

Intermediate synthesis of the ex-post assessment exercise (Lago and Möller-Gulland, 2012) revealed substandard policy design and assessment practice. Only a few analysed EPIs contained a clear specification of what the instruments were meant to achieve. A comprehensive ex-ante assessment was conducted even in fewer instances

[4] While we gratefully acknowledge the contributions of the international academic and policy experts, the responsibility for any error remains ours.

[5] Cyprus, Denmark, France, Germany, Hungary, Italy, the Netherlands, Spain, Switzerland, and the United Kingdom

[6] Australia, Chile, China, Israel, and the USA

and we have found no evidence of a continuing monitoring of the achievements or a periodic ex-post assessments. Some of the EPIs we analysed were reformed or replaced by other policy instruments during or shortly after we completed the assessment, without a plain explanation of the reasons leading to the policy change. Still more, the EPIs implemented after 2000 lacked a specification how they were supposed to contribute to the implementation of the EU Water Policy, embodied in the Water Framework Directive (Directive 2000/60/EC) and related water legislations. Whereas conflict resolution is a core of contemporary water management, EPIs are designed so as to enhance i) water use efficiency, that is avoiding wasteful use of water; ii) allocation efficiency, thus maximising overall society's benefits from water uses; and iii) financial viability, meaning ability to compensate capital, skills and technology needed to ensure water services; while paying due attention to social equity issues such as affordability and other distributional effects. The reviewed instruments displayed a partial contribution to one or more of the above goals.

The next step of the analysis focussed on an *ex-ante* assessment of prospective EPIs within specific conditions in five representative river basins. The inception report (Mysiak et al., 2012) of these case studies explained motivation for choosing the case study area, and the specific policy challenges experienced. The case study addressed different water policy issues: floods and excess waters in the Tisza river basin district (RBD) sub-unit (Hungary), droughts and water scarcity in the interconnected Segura and Tagus RBDs (Spain), and in Pinios river basin (Greece); nitrate pollution in the Seine-Normandie RBD (France) and Odense (Denmark); and restoration of good morphological conditions in in the Seine-Normandie RBD (France). The analysis resulted in five comprehensive reports (Defrance et al., 2013; Delvaux et al., 2013; Gómez et al., 2013; Kossida et al., 2013; Skou Andersen et al., 2013; Ungvári et al., 2013). Each ex-ante case study analysed the environmental challenges in the respective river basin area in a holistic way and proposed a reform of the existing interplay of the policy instruments.

This report, along with (Delacamara et al., 2013; Møller et al., 2013; Mysiak, 2013), makes up a set of synthesis report of the EPI-WATER project. The report is organised in the following way: The **PART I** comprises six essays, each focussing on an individual EPI including incentive water pricing and trading, (nitrate) pollution tax, payments for ecosystem services, water emission trading and transient flood storage. Each instrument is described in terms of policy challenges to which it contributes to respond, design principles and criteria, prerequisites for a successful implementation, and potential environmental and economic outcomes. The latter is based on the empirical research conducted by the EPI-WATER project consortium, complemented by a literature review. **PART II** of this report focusses on issues relevant for choosing an EPI or building a policy portfolio; matching the chosen instrument to the existing institutional framework and the opportunity for change; and ensuring that the EPI contributes to the integration of water and other sectoral policies, or does not produce negative side-effects to these sectors. Moreover, an innovative proposal for designing EPI so as to address water security is presented.

Furthermore, the role of environmental taxation and tradable environmental permits in Europe is discussed.

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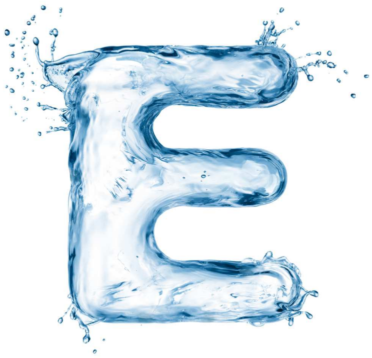
2 PART I: In-depth review of the selected policy instruments



Economic Policy Instrument (EPI) are incentives purposely designed and implemented with the objec-tive of aligning the deci-sions of individual water users with the overall ob-jectives of water policy.

The EPIs are (or should be) designed to contribute to attain predefined water policy goals.

EPI should also stand for evidence-based policy making based on well understood and analysed impacts

2.1 Incentive water pricing

Michelle Vollaro[1], Maria Molinos Senante[2], Davide Viaggi[1]

[1] University of Bologna, Italy; [2] University of Valencia, Spain

While pricing is an economic policy instrument (EPI) envisaged by the Water Framework Directive (WFD) its coverage and effectiveness is not still sufficient in Europe. The design of a pricing EPI is not an easy task since it involves key factors such as the local environmental conditions, the legislative and regulatory settings, the social and economic conditions of the targeted population as well as the infrastructural and technological endowments. The main basic condition to apply a pricing EPI is the ability to measure water availability and water use through meters. Hence, the presence/absence of cost-effective infrastructures is an essential prerequisite to implement water pricing. From a social point of view, it is crucial to take into account affordability and acceptance issues. The effectiveness of the EPI is also affected by the ability of setting appropriate water pricing rates. They should be based on marginal costs instead of average costs and environmental and scarcity costs should be also introduced in the water pricing. Regarding the outcomes of water pricing, both environmental and economic ones could vary significantly depending on how the EPI is designed and the local context in which it is applied. To guarantee the global effectiveness of the EPI, it is usually associated to other instruments, mainly regulatory ones. Nevertheless, pricing has the ability to provide reduction of water use at a higher efficiency level than regulatory instruments. Moreover, unlike subsidies, volumetric price prevent the rebound effect.

2.1.1 *Policy challenge*

Since the introduction of the Water Framework Directive (WFD, 2000/60/EC) in 2000, the conditions to date of water resources, in terms of quality and quantity, is improved across Europe, but not to the extent auspicated by the European Commission (EEA, 2012c). The distance that still persists between the objectives of the WFD and the environmental outcomes reached so far ought to be addressed by envisaging additional (complementary) measures and by removing the obstacles that impede the full implementation of the WFD (EC, 2012). To fulfil the environmental objectives of the WFD, each river water basin should undertook a program of measures (PoM) by 2009 which shall be reviewed and updated at the latest 15 years after the date of entry into force the WFD and every 6 years thereafter. Therefore, the PoM question has set the water agenda in the last 10 years and will continue to do so.

European water resources are subject to multifaceted negative pressures deriving from multiple drivers, some of which escape from the action range of the WFD. Recovering the ecological status as well as improving the use efficiency of water resources, therefore, require the reduction of pressures and, where possible, the mitigation of the drivers. Helpful support to the actions of the WFD might be provided through the adoption of Economic Policy Instruments (EPIs) that, differently from a regulatory instrument, which forces Member States (MS) to apply a specific (top-down) regulation aimed at pursuing the desired outcome, focus on the outcome – to be reached – for applying a specific (bottom-up) incentives' mechanism aimed at driving changes in behaviour of individual consumers in order to favour the desired collective action.

Pricing is the EPI envisaged by the WFD in art. 9 for inducing i) the full-cost recovery of the water services, including the environmental and resource costs, and ii) a more efficient use of the water resources, concurring to the environmental objectives, in the context of the application of the Polluter Pays (PPP) and User Pays (UPP) Principles. The diffusion of pricing mechanisms across the EU is improving, but, however, still not sufficient in terms of coverage and effectiveness, especially in agriculture and domestic sectors (EC, 2012). Substantial room for the diffusion and application of pricing mechanisms in MS is given by the art. 11 of WFD, which recognizes pricing as a "basic" measure, namely minimum requirements to be complied with. Moreover, the WFD provides River Basin Authorities (RBAs) also with the opportunity of creating ad hoc policy mixes by envisaging "supplementary" measures concurring to the environmental objectives of the directive. Therefore, several MS have chosen to apply pricing EPIs in association to other measures aiming at tackling specific environmental issues related to the qualitative and quantitative status of water resources.

Water resources in many regions of the EU are more and more exposed to the uncertainty of being insufficient with respect to the future (long-term) demand due to population and economic growth (EEA, 2012b). Abstraction rates higher than renewable capacity along with changes in rainfall patterns create conditions of frequent imbalances between demand and availability, leading to water scarcity and stress. Water scarcity, moreover, triggers a downturn in the natural capacity of water to dispose of the pollutants, reducing in turn the quality of water bodies. Increase in population density, including tourism, leakage and inefficient use of available water resources are among the principal factors inducing water scarcity. Although this issue could be addressed by envisaging measures enabling increases of water availability (new reservoirs) or reductions in water consumption (water saving technologies), like some EPIs designed under the Rural Development Plans (RDPs), the adoption of incentive pricing EPIs might work as deterrent to the inefficient use of water resources by inducing reductions in water demand. Pricing EPIs applied for inducing water demand decreases are in common use at EU level and, given their design flexibility, multiple typologies of them can be found: abstraction fees (water charges) have been implemented in Hungary (Ungvári et al., 2011), computed according to the type of water used and the use for which it is abstracted; volumetric pricing has been employed in the Netherlands (Schuerhoff et al., 2011) for the abstraction of groundwater; in the United Kingdom (Zetland and Weikard, 2011) for domestic water uses in Wales and England, and in Italy (Mysiak et al., 2011b; Sardonini et al., 2011) in some irrigation districts and for civil uses all over the Country; subsidies have been provided in Cyprus (Kossida and Tekidou, 2011) for conserving drinking water. Reduced uses of water, beyond implying water savings and diminished pressure on scarce water resources, brings also reduction in energy consumption necessary to operate the water delivery systems and sanitation.

In territories where water scarcity is or is becoming a structural (or systemic) issue, population and economic activities are also exposed to elevated risks of droughts. Southern European Countries, especially those for which critical reduction in cumulative rainfall has been forecasted, will have to engage in designing and implementing adaptation strategies able to mitigate the effects of climate change on water resources as well as all economic sectors and the environment. Lacking to adapt, or being unprepared, will leave the society exposed to the effects of high-risk low-probability events which bring about catastrophic environmental damages and economic losses. In order to protect water resources and all the related activities from droughts, specific pricing EPIs have been adopted in some countries outside Europe: in Israel (Kan and Kislev, 2011) the central water authority applied an egalitarian tariff system, while in California (Dinar, 2011) a water budget rate structure was tested for urban water uses.

Incentive pricing instruments, namely pricing mechanisms based on the marginal value of water resources[7], have in common the characteristic of providing water users with information (signals) about the internal cost of using the resource. The reaction of the users to a such pricing mechanism depends on several factors, such as, inter alia, users' elasticity of demand *in primis*, the level of the price (tariff, fee, tax…), the users' value of the resource. Hence, an incentive pricing EPI can be said to be effective if the induced reaction of the users has led to the desired outcome. Although mainly proposed to improve efficiency in water resource uses (saving, reduction of wasteful uses, improvement in allocation), pricing also pursues the objective of cost recovery, especially for compensating capital investments, and operational and management (O&M) costs. The trade-off between such multi-objective purposes is to be considered while selecting a pricing EPI. Indeed, the effectiveness of an incentive pricing EPI depends on several factors, including design and targeting, and requires reliable information. It follows that incentive pricing EPIs, whose environmental effectiveness hinges upon long-run perspectives, implemented in a context of relatively high uncertainty are more likely to fail to reach the desired outcome. This is one of the main reasons why the ability of incentive pricing EPIs to internalize also the resource and environmental costs of water uses is low (see section 2.1.5).

[7] According to Olmstead and Stavins (2009), the efficient water price is the long-run marginal cost (LRMC) of its supply. LRMC reflects the full economic cost of water supply – the cost of transmission, treatment and distribution; some portion of the capital cost of reservoirs and treatment systems, both those in existence and those future facilities necessitated by current patterns of use; and the opportunity cost in both use and non-use value of water for other potential purposes.

2.1.2 Opportunities

Improved status of water resources implies that more quantity is available for environmental, domestic and productive uses. The dynamic relations between quality and quantity of water resources is the base upon which an EPI has to be devised for concurring to the objective of improving the environmental status of water bodies. Other factors, however, are of extreme importance when an EPI is thought to be applied, such as, e.g., the ability of not debilitating economic development, the effort to avoid unfairness in the distribution of economic and financial burdens among members of the society and of the economic sectors (avoid social conflicts). This is especially true/valid for EPIs, like incentive pricing, which operate through the internalization of water uses' costs.

The primal function of an incentive pricing EPI is to make individuals paying for the use they make of water resources, mainly water consumption and discharge of pollutants, or to receive payments for enabling actions devoted to safeguard water ecosystems, such as "greening" infrastructures and water saving technologies. The application of an incentive pricing EPI for specific objectives can be twofold: direct, when the triggered behavioural changes have an immediate effect on water resources (e.g. water saving); indirect, but less incentivizing, when it creates the necessary conditions for the implementation of other measures (e.g. earmarked revenue collection for financing water safeguard infrastructures). According to the policy challenges to be tackled and to the site-specific objectives to pursue, different opportunities, in terms of environmental, economic and societal benefits, for the application of specific incentive pricing EPIs can emerge. On one hand, flat-tariffs, taxes and fees are pricing EPIs typically applied for inducing a rationalization in the use of both pollutants and water resources, but mainly for guaranteeing the financial sustainability of water-related services (cost recovery of O&M and rarely capital). The application of these types of pricing EPIs is usually complementary (subsequent) to the implementation of regulatory instruments aimed at setting general water resources quality-quantity standards. On the other hand, the implementation of incentive pricing for tackling water scarcity has a direct effect on reducing water demand as long as the price signals provided are able to induce efficient uses of water resources. Given the different ability to pay for water resources among users and sectors, opportunities from adopting incentive pricing are also configurable in benefits accruing from allocations of water resources toward more profitable uses, the reduction of leakages, especially through metering, and the secondary effect of incentives provision for the adoption of water saving technologies in all economic sectors.

Pricing EPIs associated to the registration of abstraction entitlements (to set a cap on water use) have been chosen in Hungary (Ungvári et al., 2011) to, mainly, recover costs and reducing water use; in the Netherlands (Schuerhoff et al., 2011) a volumetric "green" tax has been set for groundwater in order to induce a reduction in over-abstractions; installation of meters for signalling household water consumption has been applied in Wales and England (Zetland and Weikard, 2011) and volumetric pricing introduced for inducing families to use water more wisely. Volumetric pricing through metering has been applied also in Emilia Romagna (Italy) (Sardonini et al., 2011) for reducing unfairness in cost recovery among irrigators and non-irrigators (irrigators pay for the actual water they use) and for increasing social benefits through improvements in irrigation water use and allocation.

Inducing a desirable behavioural change that favours less harmful environmental impacts (or improvements in environmental status) is possible also through EPIs providing subsidies. Subsidy schemes guiding individual behaviours towards desirable collective actions have been applied in Cyprus (Kossida and Tekidou, 2011) providing incentives for diversifying water sources for uses different from drinking. The benefits accruing to society for safeguarding (saving) drinking water have been redistributed to the population for inducing the use of alternative water sources, such as ground and recycled water, for non-drinking purposes and to reduce the need for costly desalinated water, especially in drought periods.

2.1.3 Design

Incentive pricing can be defined as an EPI when it is able to trigger a change in individual behaviour through the provision of specific incentives. Although this is a sufficient condition, the necessary requirement for incentive pricing (and other tools) to be defined as an EPI is to induce a desired collective action. Given the heterogeneity of individuals using water resources, the design of incentive pricing needs to be framed such as to maximize participation (as in the case of subsidies) or to minimize targeting errors (e.g., inclusion vs. exclusion, definition of tariff systems), given the objectives.

Setting a price on a good essentially means to inform consumers, or users, about the scarcity of such a good and, therefore, to induce an indirect selection of consumers who are able and willing to afford the consumption of that good or a reduction in the aggregate demand. The opposite mechanism holds for subsidies. However, the intended modification in behaviour is highly influenced by both the pricing structure, the level ("rightness") of the price and the degree of response of each single consumer to the signal that the price provides for. The elasticity of demand (the per cent variation in consumption induced by a unit per cent variation in price or, in other words, the responsiveness of users to price variation) is the main factor determining the desired modification in consumption. For water resources, the elasticity of demand is very low, given that water is an essential good for life and has no substitutes (at least for a standardized level of quality), especially for domestic uses. However, when water is used beyond the essential needs and is provided at low costs, the demand in those points is more elastic and so incentive and also other pricing instruments, such as flat-tariffs, taxes and charges, are able to induce a balanced reduction in consumption if calibrated to catch/operate rightly on the more responsive portion of consumers. Indeed, for such reason and in order to improve the targeting, pricing EPIs, both incentive and others, are usually designed to reflect the policy objective and the ability to pay of users as well as to guarantee the polluter/user pays principles. As regards the objectives, the design of a pricing EPI could involve also crucial factors, such as the environmental conditions of the areas in which the EPI will be implemented, the legislative and regulatory settings, the social and economic conditions of the targeted population as well as the infrastructural and technological endowments. Another important aspect to consider in the design stage is the cost-efficiency of the measure, especially in the implementation stage, which is mostly related to targeting, to internal managerial (administrative) complexity and eventual implementation and transaction costs (e.g., installation of meters and volume reporting) as well as the time dimension of the objectives (short vs. long term).

Different examples of design exist throughout the EU Countries, and, according to the policy goals, the pricing EPIs proposed, which not necessarily embed the incentive-provider feature , are based on specific factors. In Hungary (Ungvári et al., 2011) the water resource fee, applied to reduce over-abstraction, was based on a complex set of modifying multipliers that takes into account specific uses and the type of available water resources. The groundwater tax in the Netherlands (Schuerhoff et al., 2011) was charged to affect "resource consuming behaviours" of the large set of water extractors, excluding farmers totally and industries partially, and was computed according to the volumes extracted, that are self-reported by the users. A volumetric (incentivizing) approach was also applied in Wales and England (Zetland and Weikard, 2011), where the introduction of meters for linking consumption to costs deters inefficient use of water and improves water allocation towards more profitable uses. Water metering also provides precise information about leakage, inducing therefore reduction of water losses and related costs, including energy costs. Indeed, metering in households provides information about leakage in urban areas because water companies or water utilities quantify both the volume of water treated in drinking water treatment plants and the volume of water consumed in households. Therefore, the difference between them is the water loss in the networks. At the same time it must be noted that, given investment and transaction costs, a limited adoption rate of metering will be optimal where water scarcity is not severe (Zetland and Weikard, 2011). In Italy (Sardonini et al., 2011), a three-parts tariff, including a volumetric component, replaced an area-tailored (per ha) tariff for allowing a more equitable distribution of recovery costs among irrigators (users) and non-irrigators as well as a generalized reduction in water use. For the residential water services and sanitation, Italy applied a water tariff system designed to efficiently manage all the services related to urban water uses, from the distribution to the sanitation. A fix tariff component for each priced service was associated to an increasing (volumetric) block part for covering the costs related to variable consumption. In particular, in Emilia Romagna Region (Mysiak et al., 2011b) the domestic tariff also takes into account the affordability of users on the basis of households' size and income, such to set a cross-subsidization from wealthier to poorer families (social tariff).

Incentive pricing EPIs providing subsidies aim at inducing actions that would have not been operated otherwise because of the high costs to sustain, either by private or society, for purposes that are considered as not profitable. The level of the subsidy (remuneration) is extremely important in the design stages of the EPI, especially for targeting, for it defines a trigger level upon which private people or companies elaborate the decision to engage or not in the policy. This is the case of Cyprus (Kossida and Tekidou, 2011) in which subsidies have been provided for inducing water users to choose among the options of building boreholes or installing water recycling systems in order to reduce the demand of costly desalinated or drinking freshwater. The EPI was designed such as to provide a fix subsidy for each option and a cap for groundwater extraction.

Outside the EU, two experiences in designing pricing EPIs are worth to report. After a national reform aimed at improving efficiency of water management at all levels, the central water authority of Israel (Kan and Kislev, 2011) applied a tariff system to all water uses, based on cost-recovery objectives at national and municipal levels, implemented through an egalitarian principle for setting the tariffs at municipal level (identical municipal end-users tariffs). In California, USA (Dinar, 2011), instead, in order to reduce the demand of residential water uses in drought prone areas and to guarantee at the same time adequate cost-recovery, efficient allocation and sufficient/fair provision of water to all households, the water authorities applied an (experimental) pricing EPI designed as a water budget rate structure (WBRS), which is based upon a marginal cost pricing. Under the WRBS, the fix costs of service are politically distributed in a certain percentage of the total rate (irrespective of water use by the customer) and the remaining percentage as variable, computed on the amount of water used. In this way, the fixed cost part is kept at a both a reasonable level for the customers and the water utility, while the variable costs consist in several increasing tiers (between 4 and 6), depending on the water utility. The first and second tiers are set such as to represent reasonable use of water and they refer, respectively, to indoor water use and to outdoor water use.

2.1.4 Prerequisites

In terms of institutions and governance systems, the potential application and use of incentive pricing EPIs for conservation purposes is now rather considered and accepted, respectively. Exceptions concern cases in which water is managed by institutions that do not contemplate water use regulation among their statutory objectives (e.g. irrigation boards may have to recover costs, but not necessarily to regulate water quantity).

The main basic condition for the application of a pricing EPI is the ability to measure water availability, e.g. at district level, and use, e.g. at final user level preferably through meters, at low costs such to have, at least, a reliable informative (water accounting) framework useful for the development and the implementation of the instrument. This may depend on presence/absence of infrastructures or infrastructure type and relative cost-benefit considerations. For example, the lack of suitable distribution infrastructures may depend on high costs due to geographic/physical conditions (e.g. disperse settlements or undefined water abstraction points) or path-dependency, as the development of new infrastructures may take decades.

Social/governance prerequisites may be related on two sides. On the one hand, prerequisite may relate to affordability or acceptance under fairness considerations, which may be addressed by fine-tuning the design of the instrument (e.g. block tariff). On the other hand, in order to assure the process of the EPI application is rightly closed (completed), a prerequisite is the ability to monitor and enforce both measurement of the water used, especially at the end-user level, and the relative payment.

Criteria matching needs and prerequisites for the application of the instrument can be identified on a local scale (i.e. within the most detailed administrative units available from EU or national statistics). The experience shows that the large share of domestic/industry uses nowadays employ volumetric pricing with the aim of rationalizing water uses and allowing the application of the polluter/user pays principle, while this is rarely used in the agricultural sector for which, historically, the political will has been to keep water prices low and problems with measurement are higher. Among agricultural uses, however, several areas are already using this system, such as the irrigation district Tarabina (Sardonini et al., 2011); the most suitable for developing volumetric pricing in the future are those agricultural areas that: a) already have infrastructures allowing volumetric pricing; b) have high concentration of high water use and high value added crops; c) quantities used and profitability of use are very heterogeneous across uses, which makes flat rate payments less acceptable.

A key aspect that affects the outcome of the instrument is the ability to set the (theoretically) appropriate level of price. Several issues may contribute to failures in this respect, as: a) prices are often set based on costs of previous periods, so anyway different by those of the year in which they affect uses; b) accounting costs are used, in which there is often a component of arbitrariness in costs allocation (e.g. depreciation, fixed personnel costs) and cross subsidies may intentionally or not intentionally remain; c) average cost, instead of marginal cost, is used as a proxy for pricing when differentiation of users is difficult to account for or not known; d) environmental costs remain difficult to estimate; e) scarcity rent is often not accounted for.

A second range of aspects concern the reactivity of users to changes in prices. This may be limited by several issues, including slow technical change, lack of perception of changes, transition and transaction costs in adapting procedures or in elaborating new information coming from the price, risk aversion etc.

An issue may arise when interpreting the results of pricing EPIs: for example lack of effectiveness may be due to the fact that, at the set price, the price does not justify reaction; in this case it must be clarified whether the aim is to reach the efficient water abstraction level or reduce water abstraction whatever the cost.

2.1.5 Outcomes

Considering that pricing EPIs may affect multiple dimensions of water management at the same time, the environmental and economic outcomes of their application could vary widely, depending especially on the way they are designed and on the local context in which they are applied. In assessing the outcomes, particular attention is to be given to potential trade-offs that could emerge between the different dimensions considered, such as reduction in water uses versus degree of cost recovery or fairness in costs distribution versus efficient allocation or income transfers (cross-subsidization) versus water conservation. Further, in order to guarantee a global effectiveness of pricing EPIs, they are usually associated to other instruments, mainly regulatory (prescriptive and command-and-control), providing for a suitable framework able to prime (activate) and control their incentive potentials as well as to corroborate them to work for reaching the desired outcomes. Nevertheless, a clear-cut distinction between effectiveness of regulatory and incentivizing pricing policies for water conservation, in terms of predictability, cannot emerge unless statistical analysis are available (Olmstead and Stavins, 2009). In this regard, Howe (1997) stated that water conservation (in the short-run) takes place only under moral suasion or direct regulation, while Olmstead and Stavins (2009) assert that using prices to manage water demand is more cost-effective than implementing non-price conservation programs. For these reasons, clear and intelligible cause-effect relationships between pricing EPIs and specific outcomes are difficult to established and, moreover, hard to generalize.

From a theoretical perspective, the effectiveness of pricing EPIs in accomplishing water saving objectives, as already said, is fundamentally related to the ability of the pricing instrument to catch the most responsive (elastic) portion of water users' demand (Dinar and Subramanian, 1998; Huffaker et al., 1998; Olmstead, Hanemann and Stavins, 2007). However, many other factors influence the effectiveness of such instruments. One of this is time: short-term incentive pricing EPIs are efficient if designed to properly signal the scarcity value of water resources, while long-term incentive pricing EPIs are efficient if opportunity costs of water resources is well known, otherwise the anticipated individual reactions won't trigger and the desired collective actions won't be realized. The efficiency of long-term pricing EPIs are also affected by the rebound effect for which the relationship between elasticity in water use and technology adoption could imply different and unknown water saving outcomes (Huffaker et al., 1998). In turn, as for non-pricing instruments, another limiting factor through which assessing the outcome of an EPI is information.

Environmental and economic outcomes depend upon the type of pricing EPIs applied. Unique and undifferentiated pricing (flat-tariff, taxes, fees, charges) proved to work better for revenue collection and cost recovery, but not to be efficient in terms of total (capital and environmental) cost recovery and water saving because of the lack of incentives to reduce consumption. Marginal value pricing and multi-part tariff, including at least one volumetric part, instead, including increasing and decreasing block tariff, have the potential to accomplish the objectives of cost recovery and water saving, although multi-part tariffs are subject to trade-offs: higher fixed tariff-part is more suitable for financial purposes (cost recovery and cross-subsidization), while higher variable tariff-part is more suitable for environmental and economic purposes (provides incentives for reducing use and fostering water conservation) (Dinar and Subramanian, 1998). Such situation is actually prevailing in the EU agricultural sector where incentive pricing is rarely used and cost recovery is not fully accomplished (ARCADIS, 2012). Indeed, according to the most recent reports on water pricing in the EU agriculture, in countries where the ratio of water abstraction for irrigation purposes on total abstraction is higher than 60% and the ratio of irrigated over arable land is greater than 30%, such as in Greece, Malta, Spain and Northern Italy, mainly flat-rate pricing schemes are applied and the level of the tariffs is not sufficient to achieve full cost recovery (capital, environmental and resource costs are not accounted for in the tariff rates). However, the trend is growing for shifting to incentive pricing (mainly marginal pricing – volumetric) in countries, such as Cyprus, Portugal and Southern (including islands) Italy, where water stress is becoming a major issue, but also in countries, such as Spain and Malta, where investments in water delivery infrastructures has been realized for rationalizing the allocation (through pressure-pipe systems) and use (through meters) of water resources (ARCADIS, 2012; INEA, 2011). Variability of pricing EPIs among users and sectors, beyond regulating use's conflicts and fostering efficient allocation towards more valuable uses of water resources, also pursue social objectives, especially income distribution from wealthy to poor families or industry to other sectors. Nevertheless, clear-cut effects of pricing EPIs on equity and social justice issues still remain unclear.

According to the cost recovery principle, when an individual or group imposes an extra costs on the system, this individual or group should be accountable for the extra cost. This situation often occurs in water-scarce regions with large seasonal water demand. In this context, water prices should reflect the opportunity cost of the resources used to deliver water. From a theoretical point of view, it has been reported that peak load pricing (PLP) strategy is a good approach to deal with the objectives of cost recovery and demand control in areas affected by high seasonal water demand. PLP involves applying differential tariffs that reflect the different costs associated with providing services during peak and off-peak periods. The great advantage of this strategy is that it encourages conservation when water is at its most valuable (CSIRO, 2001). Under a PLP strategy, rates are reduced or normal during off-peak water demand periods, and increased in peak months when demand is higher, and water availability can periodically lower. Despite the usefulness of the PLP, currently, most water rates are static, i.e. are temporally invariant. Some exceptions can be found in Hampshire, parts of West Sussex and Medway in United Kingdom where a seasonal tariff was implemented. The experience of Hungary (Ungvári et al., 2011) does not convey net information about the effects of the Water

Resource Fee (WRF). Its application, in association to abstraction licenses, coincided with the period of transition to a market-based economy which induced increases in energy and water provisions and a contingent general economic downturn. Industries, however, faced the increased tariffs by improving technology and saving on water consumption. The fishery sector seemingly responded to increases in WRF by diminishing the amount of water used during the period 2000-2005. However, such period has brought business downturn for the fishery sector as well, so the effect of the WRF on water use reduction cannot be singled out. A positive outcome, instead, can be recorded for the general national budget, which largely benefited from the revenue collected through the WRF, especially from bulk users like drinking water districts and industries.

The groundwater volumetric tax (GWT) implemented in the Netherlands (Schuerhoff et al., 2011) recorded different environmental outcomes. Industry and drinking water sectors reacted to the GWT by reducing extraction of water from water-tables, but substituted it with surface water. As a result, tap water tariff increased for domestic users, but they covered only the 10% of the cost of the GWT. Most of the GWT burden rested on the industrial sector. Beneficiary of such GWT has been the agricultural sector which was largely exempted. As a consequence, the ground water extraction for irrigation purposes increased.

The environmental outcomes of the introduction of water meters for domestic water use in the UK (Zetland and Weikard, 2011) are still unknown. Families who opted for the meter installation were probably the ones that used less than the average water, while those who did not opted have not changed their habits as well. However, while metering produced a cost reduction for those who installed meters, more financial burden has been transferred to non-metered families. Reduced consumption has been recorded for areas in which metering was mandatory. Issues of social justice remain unresolved because the introduction of meters and related volumetric pricing reduced the cross-subsidization mechanism embedded in the previous rateable value (RV) system.

In Emilia-Romagna Region (RER), in Italy (Mysiak et al., 2011b), the volumetric pricing EPIs applied for managing the integrated water and sanitation system (WSS) produced an average decrease in leakage as well as in abstraction and consumption of water, except for tourist places along the Adriatic sea coast. All over Italy, between 2000 and 2009, a reduction of 11% of water consumption has been recorded in front of increasing tariffs. However, no information is available as regards the effects of such reduction in water use on the environment. In terms of efficiency improvement and cost recovery, the information available does not allow to establish the role of the pricing EPIs on such dimensions. In the period considered, no significant modernization and extension of the water system has been recorded.

Still in RER (Sardonini et al., 2011), the application of a volumetric tariff system in an irrigation district produced a significant level of water saving of about 50% in six years and a fairer distribution of costs between irrigators and non-irrigator with respect to the previous flat-rate tariff. However, no information is available about strategic behaviour of farmers in terms of irrigation technology adoption and changes cropping patterns. Decrease in water use has been mainly recorded for farms providing for aquatic areas for hunting purposes (called chiari), which, however, represented ecological areas for several species of birds. Reduction in water use implied also a significant reduction in energy costs, given that water in such irrigation district is totally delivered through pressure pipe systems.

The subsidy program for incentivizing use of water sources different from drinking water in Cyprus (Kossida and Tekidou, 2011) is not possible to evaluate for lack of information and, especially, because such program has been implemented through a wide policy mix, including metering, pricing, leakage reduction and awareness raising campaigns. The installation of subsidized boreholes in households has not been monitored adequately and likely the cap imposed on extraction has been frequently breached.

The egalitarian tariff system applied at municipal in Israel (Kan and Kislev, 2011) along with the reform of the water management at national level implied reduction in water use and leakage. The increase of tariff levels at municipal level induced a reduction of freshwater use of about 18% during the period 2004-2010. Freshwater for irrigation purposes also decreased by 32% in the same period in front of an increase of recycled water of 22%. Overall, there has been a significant increase in desalinated water with consequent increase in energy consumption.

The water budget rate structure (WBRS) applied in California (Dinar, 2011), in the USA, to tackle the effects of droughts events on water availability, proved to be successful in several dimensions. Reduction in: outdoor irrigation and related run-off; indoor uses; low-quality water import from Colorado river and relative energy for treatment. The fixed cost recovery has been stable over drought periods and cost distribution has been considered fair. Generalized social acceptance of the pricing EPIs has been recorded.

Altogether, though outcomes from pricing EPIs application are not suited for being generalized, they provide insightful information that may help in analysing suitable preconditions and in designing objectives-tuned mechanisms given the specificities of local conditions.

2.1.6 Comparison

From a theoretical perceptive, as compared to regulatory instruments, pricing has the ability to provide reduction of use at a higher efficiency level, notably allocating the reduction of use to the users with lower marginal costs of adaptation. To some extent it also solves some asymmetric information issues related to the differentiation of profits/utility across different water uses and users. It also allows higher flexibility of adaptation to changing economic conditions affecting the opportunity cost of reducing water use.

In comparison with subsidies (either volumetric or investment-related), volumetric price prevents the rebound effect. At the same time, if using marginal prices, it over-allocates costs to users. Anyway, the distributions of private gains comparing pricing and subsidies tend to be the opposite, which may imply several economic and distributional considerations, as well as making them differently suitable with respect to sectors undergoing different sectorial policies.

Comparisons with water markets and insurance instruments are more difficult to establish, as they address, strictly speaking, issues different from water saving or conservation. For such objectives, water markets may be effective when water rights can be withdrawn or bought for environmental purposes. This implies a different distribution pattern of costs and benefits with respect to pricing (users are gaining rather than paying), but evidence about relative effectiveness is in most cases not available (Johansson et al., 2002).

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2.2 Water trading

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Water trading essentially involves the voluntary exchange of rights or entitlements to use water (Hanak et al., 2011; Rey et al., 2011; Young, 2010). To achieve the desired status of water bodies, quantitative constraints on abstraction must be set and converted into property rights over the use of water (Howe, 2000). There is therefore an overall cap on rights to use water. Water users can then trade these rights within the limits defined by the water authority. Water trading, though, is different to water transfers despite implying the diversion of water.

2.2.1 Policy challenge

Water trading can be used to help address water scarcity and droughts. It is especially relevant at a local level, when transport costs are low and all use rights are defined over the same water source (Rey et al., 2011). Allocating water to its more valuable uses is a means to increase economic values without harming the environment (i.e. producing more with the same), and also represents an opportunity to reduce water abstractions (i.e. producing more with less) (Garrido and Calatrava, 2009), provided that other complementary instruments are put in place.

Experience with water markets shows their significant role in finding mutually beneficial agreements between buyers and sellers, thus increasing the production of goods and services and making water trades a convenient instrument to promote different economic activities (Young, 2010). These development objectives were actually the main driver in the original adoption of current water trading schemes and concerns on their environmental outcomes are still an emerging issue (Howe, 2000).

The implementation of water trading schemes is also advisable, under certain conditions, as part of a policy mix aiming at regaining control over groundwater resources and harnessing the potential of water resources to provide higher levels of resilience and adaptive capacity for economic development.

Voluntary trading can play a critical role in stabilizing the economy and in providing an effective drought management alternative, provided all stakeholders are involved and provisions are made to compensate for third party effects (as in the Tagus river basin district, Central Spain).

2.2.2 Opportunities

Besides the evident interest of maybe buyers and sellers in a water market, the real opportunities of water trading lie on its potential to take advantage of the existing incentives to trade water in order to obtain benefits for the purposes of water policy.

In other words, success in water trading might not be judged by the number or the volume of transactions or against paid prices, but according to their contribution to curb scarcity, reduce drought exposure or increase resilience and/or adaptability. As below, designing and implementing a water trading scheme consists in transforming a good business opportunity, for those directly involved in the bargain, into a good water governance opportunity for anyone.

These opportunities of water trading to improve water governance can be summarised in the following intermediate targets to which water trading is expected to make a significant contribution to:

- Attain a Pareto improvement in the current water allocation among market activities in order to enhance the potential of the economy to increase the provision of goods and services within the limits of available water resources (such as in water markets in Chile, Australia and Colorado, USA).
- Improve adaptability by making water allocation to alternative uses contingent to available resources anytime in order to reduce welfare losses and provide a better response to droughts (as in the Tagus RBD).
- Reallocate risks so that the vulnerability of water uses exposed to scarcity and droughts is diminished.
- Create opportunities for water saving and conservation, thereby providing an alternative to traditional supply-side approaches to water management, deterring, for instance, costly investment in water infrastructures.
- Indirectly create incentives for research, technological development and innovation in water technologies and processes.
- Create a framework in which water users can make decisions based on local conditions and ad-hoc needs.
- Create a framework in which water users can independently adapt their practices to emerging issues (without relying on government action, but subject to its command).
- Show water users the opportunity costs (i.e. those associated to foregone alternative choices) of some of their decisions on water use.

Box 1: Water Trading in EPI Water: A multi-level water use right trading scheme

Water use right trading has been proposed in order to improve resilience in the economy and harness the potential of water for economic development. Specifically, we suggested a combination of intra-basin markets to improve water allocation efficiency at local levels and to enhance water usage technical efficiency, combined with an inter-basin water market scheme to transfer water from relatively less scarce to more scarce areas. While environmental concerns are more likely to be minor at local scales, wider concerns apply for inter-basin trades.

The above-mentioned opportunities are higher in the EU where potential for water trading is largely unexploited. However, although water trading generally increases the efficiency of water allocation, this admits countless nuances. Previous experience in non-EU countries represents a real learning experience that may help improve design and implementation to make water trading a more effective, equal and sustainable instrument of water policy as well as to avoid failures and reduce the risks involved. The implementability of water markets is conditional on the existence of marketable water rights, freedom to agree on prices, and information such as an adequate price-revealing mechanism (as it can be seen in case studies from the Murray-Darling Basin in Australia, Chile, and the Northern Colorado Water Conservancy District in the US, or even in the very incipient trades in the Tagus River Basin, in Spain). The lack of these structural requirements prevents the formation of water markets (as in China).

Assessing opportunities to transfer water is still an open research issue. Assuming zero transaction costs, opportunities to reallocate water to its most productive uses would exist when the value of water is variable across water users. Should these differences exist there would then be an opportunity for individuals or stakeholders to engage in a bargaining process to reach mutually beneficial agreements at a price that must be set in between the maximum willingness to pay (WTP) of the might-be buyers and the minimum compensation that potential sellers are willing to accept for water use rights (under temporary or permanent cession). In studies about the opportunities for water trading this is overall the approach followed as a first step to identify the maximum potential for water markets (Rey et al., 2011).

However, these basic calculations do not answer the basic question: how much water can be negotiated and transferred between two river basins taking into account all the possible costs (transport and interim losses from infiltration and evaporation) and taking into account all the environmental constraints. Basic studies do not usually take into account basic operational costs. Even ignoring other transaction costs this is already an important outcome showing that opportunities for water trading decay with distance as transport costs increase.

Besides that, from a social perspective the key question to be solved is if what is in the best interest of contracting parties is also in the best interest of all the potential people affected by the spatial reallocation of water. Experience with water trading shows that finding the answer to this question is very challenging as it requires due knowledge of all the likely effects and their welfare outcomes over the so called third parties. Reviewing the existing experiences with water trading would serve only to ratify that these effects have not been always taken into account (Hanak et al., 2011; Young, 2010). Rather, they are often overlooked (along with the environmental impacts of water use right trades).

It is also well known that although all water rights are legally defined, norms do not have a full picture of all the possible contingencies and all the ways decisions over water use and diversion might affect others' opportunities not to mention all the environmental services potentially delivered by the water ecosystems along the entire river basin (Howe, 2000).

None of these problems (be it transport cost or potential third-party effects) are important when water is traded on a local basis among users of the same kind, as it might happen when all farmers within the same irrigation district negotiate the individual rights received so that the more productive ones can get a higher share of water instead of the quota allocated by the water authority (Albiac et al., 2006). This guarantees a better allocation of the overall water available without transactions being conditioned by significant transport costs or any relevant third-party effects.

Negative environmental effects can be avoided if potential sellers of water cede all their water use right but all the physical returns to the environment (whatever its form) remain in the area and only depletion (instead of water use) can be transferred. The reduction in water use that would need to be taken to allow one unit of depletion to be transferred would depend on the technical efficiency with which water is used in the ceding basin and, in economic terms, this will result in an increase in the cost of water effectively transferred. The effect of taking this kind of caveats over water costs will be higher the lower the technical efficiency (Rey et al., 2011). Noteworthy, third-party effects over other activities dependent on former water users, such as local agro-industry, cannot be so avoided.

An environmentally neutral water market needs to anticipate these effects so that water trading does not have any impact on the environment. All these elements (transport costs, transient losses and environmental constraints) result in substantial reductions in the amounts of water that might be transferred by an efficient market and an important increase in the (still hypothetical) equilibrium price.

Box 2: Evidence of informal trading in the Segura river basin district

Experience shows that although water trading (rather than water use right trading) is not expressly permitted, farmers are willing to engage spontaneously in such a kind of bargain (Estevan and Lacalle, 2007; WWF, 2006). Evidence does exist that informal water markets may be trading substantial amounts of water in the Segura every year (Hernández-Mora and De Stefano, 2013). Since these transactions are uncontrolled, illegal trading might be putting in the market water resources in excess of allowed quotas and this might be one of the emerging factors driving overexploitation of groundwater.

If this is the case, water trading might be based on the capacity of some farmers to obtain additional (ground) water in excess of what they are permitted to use, instead of their willingness to use smaller amounts of water than those specified in their water entitlements.

Informal water transactions in the SRB may be trading significant amounts of water at the highest prices in Spain: evidence collected by Hernández-Mora and De Stefano (*ibid*.) shows that water prices during drought events hit 0.70 €/m^3 in the agricultural district of Campo de Cartagena (SRB).

This analytical approach to the opportunities of water trading consists in identifying the maximum amount of water that can be traded in a scenario where only the financial interest of the trading parties is considered (see Figure 1, below). Following a similar line of argument one may also build an extremely conservative scenario to consider the maximum water that might be traded in the presence of any possible provision to avoid detrimental environmental impacts or third-party effects (see Box 3).

Box 3: Assessing the Potential for Interbasin Water Reallocation between the Tagus and the Segura River Basins

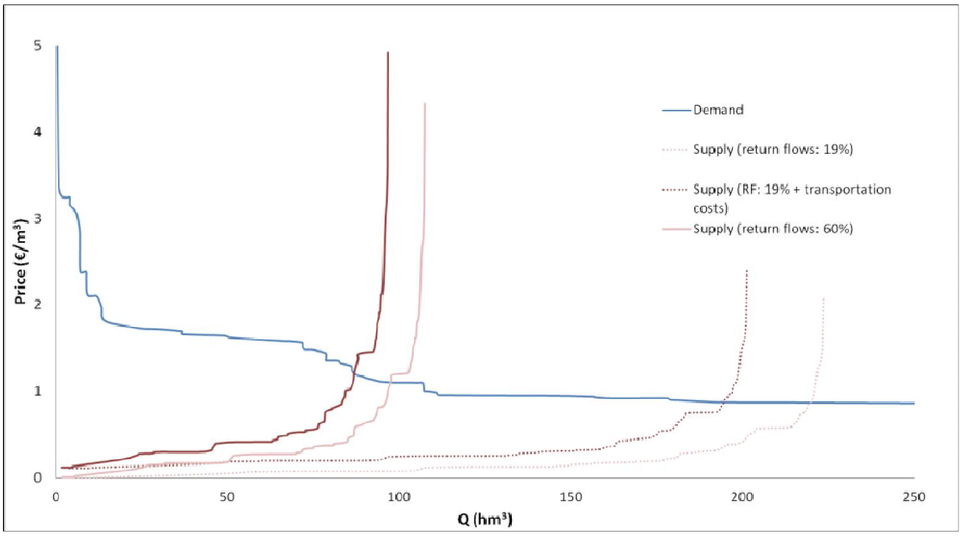

Figure 1: Focus on the environment: water trading in environmentally neutral markets

Source: (Gómez et al., 2013)

When transport costs and losses are factored in and only depletion instead of use is considered, the amount of water that might be efficiently transferred is only 85 hm^3 instead of the 240 hm^3, assuming no transaction costs and no environmental impact. In the absence of public intervention the market-clearing price may even be higher than the local financial cost of desalinated water.

This analysis has important consequences on the possibility of opening the option to trade with water as an incentive to enhance water efficiency. In fact the water saved by installing more efficient devices (such as modern irrigation systems) might reduce water use but will not result in reduced consumption, which at best will remain constant. If the number of tradable water rights issued after a user (e.g. a farmer) proves to have upgraded the technology she uses is equal to the (actual or presumed) reduction in water consumption, this might be an effective way to put more water into the market. Yet, if the criterion to issue tradable water rights is the change in use then efficiency will never be a means to put more water into trade. This is but one example of how the criteria of expanding water trade as much as possible might be in contradiction with the criterion of guaranteeing that any water transaction should have at least a neutral effect over the water environment.

2.2.3 Design

Water trading may adopt different forms (Hanak et al., 2011; Rey et al., 2011; Young, 2010):

- *Spot water markets*, both informal and formal (i.e. under legal arrangements), are common to transfer surface or groundwater resources for short-term trades in the context of a single basin. Spot, as opposed to long-term exchanges, stands for transactions in which water delivery is immediate or is meant to occur in the very near future.
- *Water banks* are central institutions acting as a clearinghouse mechanism for users willing to purchase or sell water. A clearinghouse is an organization that collects and gives out information on supply and demand of water rights. Water is then sold at a price with a mark-up (i.e. an amount of money added onto the price) to cover the operating costs of the bank, which are often borne by the buyer.
- *Bulletin Boards* are a type of water bank in which the price is not set by a central institution but rather the result of buyers and sellers posting bids and requests for water use rights at a central bulletin board (i.e. irrigation district authority) or through electronic platforms.
- *Auctions* are used to allocate rights between two or more users who compete for the same use right. Whereas in spot markets buyers and sellers occasionally interact, auctions allow as many trades as possible at a common price. In double-auction markets, buyers and sellers submit sealed bids for specific amounts of water right. In all-in-auctions, bids are ordered during the auction session so that bidders see when their offer is accepted and have the opportunity to enter more bids.
- *Derivative markets* are those based on long-term agreements (i.e. water is not to be delivered neither now nor in the near future). In the so-called option markets, one type of derivative markets, buyer and seller agree on the quantity of water and the date of delivery and both must comply. Under the so-called forward contracts, the buyer may decide to forego the purchase before the expiration date; hence a deposit is paid as compensation to the seller.
- *Environmental leasing and purchase programs* are usually meant to increase in-stream flows for environmental purposes. They include water trusts, governmental leasing and purchase of use rights, and buyback programs.

Designing and implementing a water-trading scheme require performing a number of assessments and activities. In terms of design, the following key steps are:

- On the basis of hydrological balances, defining and quantifying a quantity of water (allowing for variance) that can be obtained from surface and/or groundwater, by time and place.
- Excluding environmental flows (e-flows) that are necessary to uphold or attain the good ecological status of water bodies, according to the Water Framework Directive; that is to say, the quantity of water that nature needs for the good

ecological status to be achieved and the provision of ecosystem services to be maintained.

– Defining water entitlements and rights. This includes how they relate to the physical resource and how to ensure a sustainable yield (temporally and spatially) that can be subject to trade.
– Setting up an institutional arrangement (i.e. legal reform, specific bodies within water authorities, official registry, arbitration procedures, etc.) to manage the legal entitlements.
– Setting up an effective monitoring system, including metering and other devices to measure individual water use;
– Ensuring the enforcement of water use rights over all water sources;
– Setting up appropriate safeguard mechanisms (i.e. legal provisions, assessment procedures, etc.) to (i) guarantee the environmental outcomes, (ii) protect third-party potentially affected interests, (iii) regulate the possibility to carry over water between years, and (iv) prevent hoarding of rights and speculation practices.

The structure and features of water rights affect the manner in which markets perform. Systems that limit marketable volumes to consumed water curb externalities and environmental threats (USA, Spain). Systems that allow the transfer of nominal entitlements without considering effective and consumptive use face problems of over-allocation and, most importantly, externalities (something evident in Australia and Chile). In addition, Chile faced problems of water monopolization in non-consumptive use rights. If streamlining and concentrating the approval process for transfers in a single body with adequate management powers (as in Colorado, US), transaction costs may be reduced. On the other hand, limiting transactions to agents already holding water rights (the so-called market incumbents) and to uses ranking higher than the seller limits the performance of markets (for instance in Spain). The structure and performance of markets need to be assessed and regulated as a process (Australia) based on experience and trial and error (USA approach).

To be effective, water trading requires making water use more flexible through allowing buying and selling to be an option instead of the strict use of water rights in the amounts, places and particular uses for which they are issued by the water authority. The definition of tradable water rights is a major change of the current institutions in place (as in Maziotis et al. (2013)) where, contingent to the availability of water at each moment in time, individual users are granted with usufructuary rights that, unless an intricate authorization process is followed, cannot be used for another purpose or in another site than that authorized by the water authority.

Regarding water scarcity and the other objectives of water policy, the main questions around the effectiveness of water trading have to do with guaranteeing that trading water (use rights) may be environmentally neutral. In other words, what it is in the interest of specific individuals or parties agreeing on a transaction over water should not harm the socially agreed interest of preserving water sources.

A particular threat that would need to be avoided in order for water trading to gain social and political acceptance is the perception that instead of reducing water scarcity, trading might open the door for current scarcity trends in a particular place to expand to the rest of the territory making water scarcer elsewhere.

Evidence shows that trading schemes may have increased pressures over water resources (by putting into use water that might not have been used in the absence of markets). This has been the case of the Murray-Darling basin in Australia and Chile, where available resources are claimed to be over-allocated (although there is no empirical evidence on this for Chile, where this statement would accept a number of non-minor nuances). On the other side, physical interactions between water bodies along a river basin and externalities that may arise still make it difficult to find a set of property rights that can be efficiently traded. For instance, in Chile increased activity in consumptive water use markets has boosted conflicts with downstream users due the effects of water use rights over return flows.

2.2.4 *Prerequisite*

Water trading can only work if:

1. Opportunities do exist to reallocate water and to improve welfare at the same time. This is the case when:
   - There is a high differential among marginal returns from water among uses and places (i.e. profits obtained from water use), and when infrastructures can transfer water at a competitive cost;
   - Water use efficiency and the contribution of water to social welfare can be substantially improved;
2. Property rights, in particular water use rights that become tradable, are properly defined and enforced. A critical issue in the implementation of markets is a clear but nonetheless full definition of water rights or entitlements and their associated risks. The structure and features of water entitlements affect the way in which those markets perform. Systems that limit marketable volumes to consumed water (effective use) limit externalities and environmental threats. Systems that allow the transfer of nominal entitlements without considering effective use face problems of over-allocation and, most importantly, externalities. In addition, evidence shows that trading schemes may have increased pressures on water resources (putting water into certain harmful uses that might have not been used otherwise).
3. Environmental externalities and third-party effects are considered. It is also important to account for the interactions between surface and groundwater resources (no specific provisions can be found in many of the assessed systems). Main concerns, though, remain on third-party effects (for instance, linked to the definition of rights on water return flows) and other environmental externalities.

   Local water trading, e.g. among the members of a given irrigation district, does not raise important environmental concerns as far as all the parties directly involved in the agreement comply with the overall amount of water entitled to the group (such as in a cap and trade scheme).

This might not be the case when imperfectly controlled groundwater sources are involved. In this case, some farmers might be able to sell additional amounts of water without reducing their use accordingly. If that happens the demand of water for trading is not covered with the resources already available but rather through increasing short-term supply at the expense of higher water scarcity, and lower resilience to droughts, in the future.

4. Transaction costs are internalized. Transaction costs should be minimized but not neglected, since they play no minor roles in some occasions.
5. Gaining consensus and making water trade politically acceptable requires transparent and effective encompassing measures to safeguard the environmental objectives of water policy. Otherwise the interest on water trading will not be able to overcome the risks associated to being an instrument to put more water into use, instead of reducing scarcity, and to extend scarcity throughout space and from scarce to less water scarce river basins. Worsening overexploitation and scarcity trends is likely if water use rights do not match available water resources. This implies additional pre-conditions such as:
   - Removal of excess (nominal) property rights when rights are allocated in excess of available water resources in order to avoid social disputes.
   - Convincing and tested evidence on the dynamics of the water resource to show awareness and capacities to avoid undesirable effects over environmental flows, groundwater or other water related ecosystems (social perception on the potential effects of water trading over the environment depends on people's trust on the institutional capacities to detect and control potential environmental threats).
   - Trust in the ability to monitor and enforce water property rights. Risks are larger when monitoring and enforcement is poor and non-controlled or illegal rights are put into the market.
6. Additional safeguards are required to guarantee that the water trading scheme in place has properly addressed the following well known threats of trading with water:
   - Leading to speculation with water rights when they are accumulated and not used. This can be addressed via charging permit fees for unused water, limiting applications for water use rights to the original needs and through water rights subject to forfeiture if not used.
   - Reinforcing social disparities and reducing spatial cohesion, as water is re-allocated to more valuable uses. This can be addressed through the proper integration in water planning decisions and specific assessments of major water diversions.
7. Water trading can only be part of the solution if the deficits that are covered in the importing basin are compatible with the closure of the exporting one; that is

to say, if the capacity of the ceding basin to yield the surpluses that can be transferred at any time are compatible with the maintenance of environmental objectives.

8. Finding water-trading alternatives that fulfil all these conditions might be challenging but the real issue is that without transparent information showing that this is not happening, the social acceptance of water trading will remain a difficult, or even impossible, task.

Box 4: Controlling outlawed overdrafts: a critical requirement for water trading

An important question around the pervasive evidence of water trading in the Segura RBD (Hernández-Mora and De Stefano, 2013) is not whether they are means of local users to avoid transaction costs imposed by prevailing regulations that prevent water from finding its more valuable use in the economy (which might be a legitimate function of markets), but a means to encourage outlawed water abstractions (which is a way to go deeper into the current unsustainable trends of water withdrawal).

2.2.5 Outcomes

Experience with water trading shows how important water markets have been in helping find mutually beneficial agreements between buyers and sellers, thus increasing the production of goods and services and making water trading a convenient instrument to promote agriculture, manufacturing, hydropower, and other economic activities. These development objectives were the main driver in the original adoption of current water trading schemes and concerns on their environmental outcomes (although not completely absent in origin) are still incipient in most cases.

Evidence also shows that trading schemes may have increased pressures over water resources (by putting into use water that might not have been used in the absence of markets). It is usual that in surface and groundwater systems where water entitlements and allocations are not tradable, a significant share of the entitlements issued are not be used. Reasons for non-use in the case of groundwater include holding resources as a reserve to face drought events. This has been the case of the Murray Darling basin in Australia (Crase, 2012), where the emergence of a complex and profitable water market has resulted in over-allocation that threatened the fulfilment of environmental goals. In order to solve this problem a series of measures have been implemented, including command-and-control policies (first via a decision to secure 500 GL of water for the environment under a Living Murray Initiative and second by the transfer of basin wide water planning responsibilities to an independent Murray Darling Basin Authority) and financial instruments (the commitment of A$3.1 billion for the purchase of water entitlements from irrigators and the commitment of A$5.8 billion for investment in so-called water saving projects).

However, these policies may not be enough to face the challenge of an increasing demand. In Chile, for instance, the whole river flows have been fully allocated since at least three decades ago, which has led to the deterioration of aquatic ecosystems in semiarid and arid regions of the country (with significant problems in the region of Copiapó, in northern Chile). This can be said to be gradually changing with a series of reforms implemented since then (such as forfeiture for non-use), but positive environmental outcomes are still to be proved, at best.

On the other side, physical interactions between water bodies along a river basin (including in-stream uses and the connection between surface and groundwater, the definition of property rights that can be efficiently traded in a market is still a challenge) and externalities (and third-party effects) that may arise, still make it difficult to find a set of property rights that can be efficiently traded in existing water markets. For instance, in Chile increased consumptive water use market activity has generated further conflicts with downstream users due to the effects of water entitlements over return flows. Almost all consumptive water use right holders generate significant return flows (leakage and seepage water) that are used by downstream customary right holders, but it is not known how these customary use rights are dependent on return flows (Ward and Pulido-Velazquez, 2008).

Voluntary trading can thus play a critical role in stabilizing the economy and in providing an effective drought management alternative, provided all stakeholders are involved and provisions are made to compensate for third-party effects. This was evidenced in the Tagus River Basin in Spain, where voluntary agreements allowed the optimization of existing water resources without building additional infrastructures, engaging in massive groundwater abstractions or significant political costs.

Water trading is supposed to be a means to increase the overall allocation of water amongst places and economic activities. Provided transaction costs are not unreasonable (both the Chilean and Australian markets have a similar system of pro-rata share of water stocks, intended to reduce transaction costs and to eliminate opposition to transfers), the participation condition is more likely fulfilled when there are important differences in the marginal value of water among potential buyers and sellers and mutually beneficial agreements are feasible (so that the participation condition is met). In all case studies on water markets under EPI-Water, one may have expected major differences in water prices across uses and that these differences might persist beyond what can be explained by asymmetries in conveyance costs and water quality, suggesting that water markets may have not developed fully enough to optimize efficiency gains.

Nevertheless, in many water right trading schemes, incentive compatibility is not guaranteed. Representative examples show that the option to trade water may put into use a substantial amount of resources that in the absence of trading opportunities would have remained in the environment. In this case, water markets can paradoxically contribute to increased water scarcity and to spread water scarcity along the territory. This is already shown in the water transfers in the Middle Tagus in Spain (and it is even more evident in the Henares irrigation district, possibly the first formal experience in Spain), but it has also been proven, at a much larger scale, in the Murray Darling basin in Australia. This is the well-known rebound effect. As noted by Jevons (1865) some 150 years ago, increasing efficiency of resource use may even increase the pressure on a resource.

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2.3 Nitrate tax

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2.3.1 Policy challenge and opportunities

The EPI-WATER research on water quality (Skou Andersen et al., 2013) explored implications of introducing a nitrogen tax on mineral fertilizers in a specific catchment (Odense River Basin ORB, Denmark). The analysis demonstrated discernible environmental benefits, but only when considering the full cycle of nitrogen reductions.

The implications of the nitrogen tax would be a substantial price increase for mineral fertilizers, leading to a reduction in their use (up to full elimination). As organic fertilizers can be used to substitute for mineral fertilizers, crop yields would decline relatively less however. Organic fertilizers are a waste product of animal husbandry and while their availability would not be affected by a nitrogen tax, their trade value would increase. With a nitrogen tax applying to both types of fertilizers some of that value could be returned across the board with full revenue-recycling, e.g. allowing for a reduction of land value taxes.

Nevertheless there would be additional direct costs for farmers and these costs could outweigh the monetary value assigned to environmental benefits in the analysis. A proportionality test might be requested according to WFD art. 4. For instance, in one ORB scenario of 'high' ecological water quality the costs would be disproportionately high, whereas for the ORB scenario of 'good' ecological water quality there is a reasonable balance between costs and (environmental) benefits.

Schemes for taxation of nitrogen could be relevant for catchments across Europe where there is excess application of fertilizers, leading to pollution of surface waters and aquifers (map cf. European Commission, 2013). Nitrate concentrations measured in surface waters echo agricultural practices. Hydrological modelling can be used to estimate concentrations in root zone (see MITERRA results in figure 2), while figuring out the actual nitrate pulse to groundwater aquifers and surface waters requires more comprehensive calibration of models with monitoring.

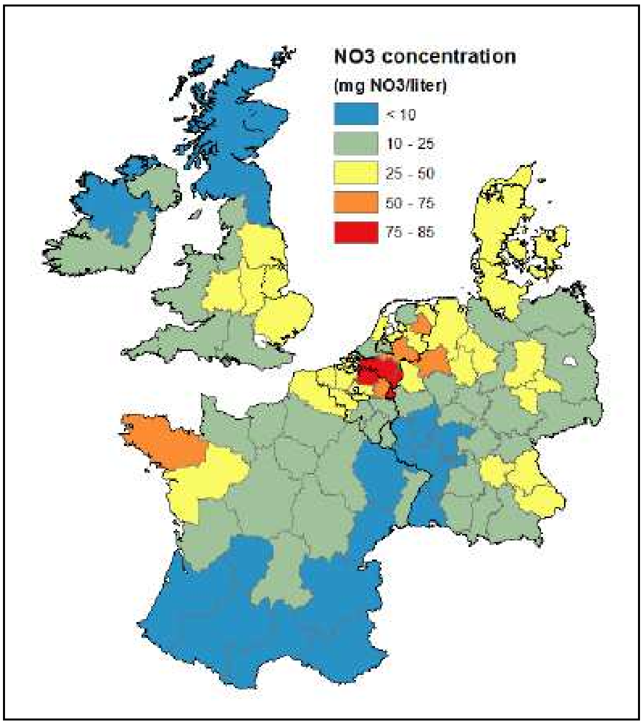


Figure 2. Mean nitrate concentration in root zone at NUTS2 level 2008 (MITERRA model in van Grinsven et. al., 2012).

In this paper we address a possible up-scaling of the nitrogen tax. In doing so it becomes relevant to consider environmental and economic implications but even more so to reflect on the policy framework. A nitrogen tax can be designed in various ways, targeting mineral fertilizers as well as organic fertilizers.

2.3.2 Design

The discussion on design of nutrients taxation has taken place mainly among specialists in tax administrations and can be found summarized in several government reports. The complexity in curbing agricultural pollution is high due to the many uncertainties in the scientific understanding of leaching processes. There is a spill-over from this complexity to design considerations for nutrients taxation. Comprehensive policy packages combine taxation with book-keeping of manure and fertilizers. Analyses further address the recycling of revenues, which can be used to mitigate the distributional implications within the sector as well as to maintain the overall tax burden constant satisfying the principle of revenue neutrality.

For analytical purposes it is useful to focus on the EPI only and to separate the revenue recycling issue from the principal choice of tax base. This leaves us with only three principal variations of a nutrients taxation scheme (cf. NITROTAX project; see van Zeijts, 1999): the mineral fertilizer tax, the nutrient-input tax and the nutrient-loss tax. The three tax schemes can be understood as interrelated extensions.

A Mineral fertilizer tax (M) targets nutrient inputs from artificial fertilizers only, whereas a Nutrient input tax (N) additionally targets the nutrients in animal fodders (that in turn transform to organic fertilizers). The nutrient Loss tax (L) focuses on application residuals, and targets the *ex-post loss* from the root zone as can be established for each farm.

Mineral fertilizer taxes have been discussed for a long time and were in place in Finland, Austria and Sweden for up to two decades before they joined the EU in 1995. Rougoor et al (2000) report that fertilizer use was relatively inelastic (-0,1 to -0,5) in response to these taxes, but nevertheless estimated the presence of significant impacts, in particular in Austria with a tax rate at 70% of the fertilizer price. Prevailing fertilizer prices were relatively low at this time contributing to widespread over-fertilization.

A nutrient input taxation scheme has been introduced in Denmark for phosphorus. The gradual saturation of agricultural soils with phosphorus, and the potential for leaching, explains the tax base chosen. Traded animal fodders are subject to the tax rate of €0.5 per kg of P. It is expected to provide a stimulus for improving fodders and adding of enzymes that allow for increased natural uptake. A 20 per cent P-reduction could be observed within 3 years from the start in 2005 (Vinther and Poulsen, 2008).

A leaching tax was in operation in the Netherlands from 1998-2005. To calculate the farm-specific losses a comprehensive mineral accounting scheme (MINAS) was introduced. Farmers were obliged to book-keeping and were taxed accordingly. Tax rates were increased in steps from low initial levels and in the final years amounted to €5/kgN and €20/kgP, which is 5-10 times the market price for mineral nitrogen fertilizer for example. Still, only surplus losses of nitrogen and phosphorus were addressed, with tax-exempted allowance thresholds of 40 kgN/ha and 10 kgP/ha. The European Court of Justice (EJC) in its decision on the Dutch implementation of the Nitrate Directive assessed the compatibility of this taxation scheme with the Nitrates Directive and raised a question mark over leaching taxation due to the inherent uncertainties and the discretion to book-keeping.

The OECD/EEA database on economic policy instruments reports no current application of nitrate taxes in EU Member States, with duties on ammonia nitrogen in Czech Republic and Bulgaria as possible exceptions. Still, Croatia is reported to tax mineral fertilizer nitrogen at a rate of 16 eurocents/kgN.

2.3.3 Prerequisites

Farm surpluses of nutrients can be calculated by subtracting outputs contained in farm products from inputs in fertilizers and fodders. Figure 3 provides a plot of field application rates and surpluses, considering different farm types including ecological farming. Some farms have low losses close to 50 kgN/ha while others are recorded for losses above 250 kgN/ha. While surpluses do increase with application rates, there are considerable variations among farm types. Intensive livestock farms often have challenges in disposing of manure. Ecological farms may have high losses too. Crop growers prefer to use mineral fertilizers, which is easier to administrate.

In Denmark losses have been reduced over time due to improvements in fertilizer management practices, in particular the reduction of mineral fertilizer use (with 50%) and substitution with use of organic fertilizers. Still, rates of fertilizer application are too high to meet WFD-requirements. In the European Union mineral fertilizer use has been reduced on average, but is increasing in new Member States.

The regulatory challenge is to ensure that better use is made of organic fertilizers while minimizing on use of mineral fertilizers. It is more complicated and costly to administrate organic fertilizers, and the burden is likely to fall on crop farmers rather than on livestock farmers.

Revenue recycling of fertilizer taxation proceeds per unit of farmland is a method whereby compensation can be granted to land owners, although this approach could be questioned from the point of view of the polluter-pays principle. Alternatively the proceeds could be used as basis for the payment for ecosystem services (PES), see case study for Seine-Normandie, France (DeFrance et. al., 2013).

A tax scheme where only nitrogen surpluses is subject to taxation is often perceived as more fair in that only the losses are targeted. Still, ECJ in the MINAS case declared this approach to be incompatible with the Nitrate Directive that specifies the allowable field application. Stressing the principle that pollution should be rectified at the source (prevention principle cf. TFEU) ECJ disapproved of MINAS (ECJ, 2002). On this background Andersen et. al. 2013 consider a tax scheme where only mineral fertilizer is taxed.

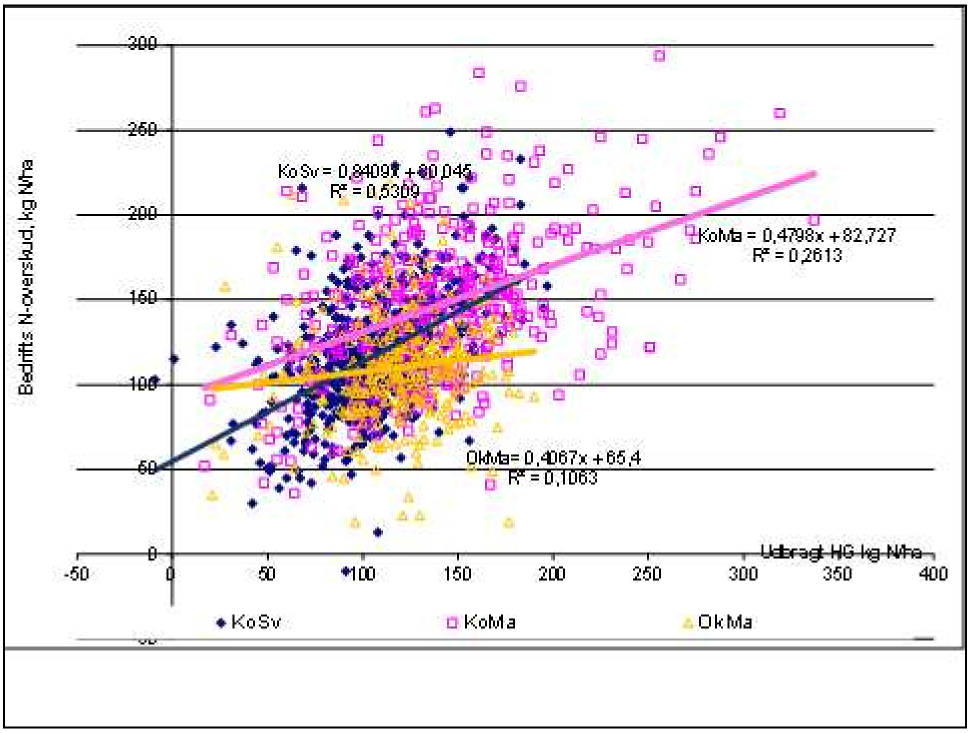


Figure 3: N-surplus in relation to field application of fertilizer nitrogen (kgN/ha) for livestock pig (KoSv), livestock cattle (KoMa) and ecological farms (OkMa). Source: Sillebak et al, 2003.

2.3.4 *Modelling framework: impact pathway approach*

Implications of a mineral fertilizer tax are challenging to predict, because so many variables are at play. In the EPI-WATER ex-ante study (Skou Andersen et al., 2013), the modelling approach relied on a catchment-specific agricultural model to capture farmers' responses to the price signal from the tax, from which the results were used to feed a biophysical modelling chain established in accordance with the principles of the impact pathway approach.

The results indicate that the most significant benefits relate to improved protection of drinking water and to reductions in ammonia evaporation. Both are considered to have potential health effects for humans. Additional benefits could be estimated for reduced GHG emissions. Finally there are benefits from clean surface waters, although these in comparison are relatively minor.

Table 1 provides an illustrative assessment of the external costs of nitrogen for six European catchments for which data is available based on EXIOPOL and EUROHARP projects. In the absence of catchment-specific agricultural models it is not possible to simulate a range of mineral fertilizer tax rates. However, a mineral fertilizer tax that is sufficiently high to be prohibitive to the use of mineral fertilizers (as in the T1 scenario of WP4) could be predicted to result in benefits comparable to the range of avoided costs related to mineral fertilizer nitrogen as indicated in Table 1.

External costs related to drinking water are illustrative and are based on the exposure-response functions derived from available epidemiological studies in EXIOPOL (Andersen et al. 2011) (According to IARC, the International Agency for Research on Cancer, nitrogen must be regarded to have possible cancer effects). For ammonia the external costs are based on the MS-specific unit costs resulting from a modelling study for the European Commission under the CAFÉ (Clean Air for Europe) cost-benefit assessment (AEA, 2005). The external costs for GHG refer to N2O and are based on a carbon price of €20 per ton CO2-equivalent.

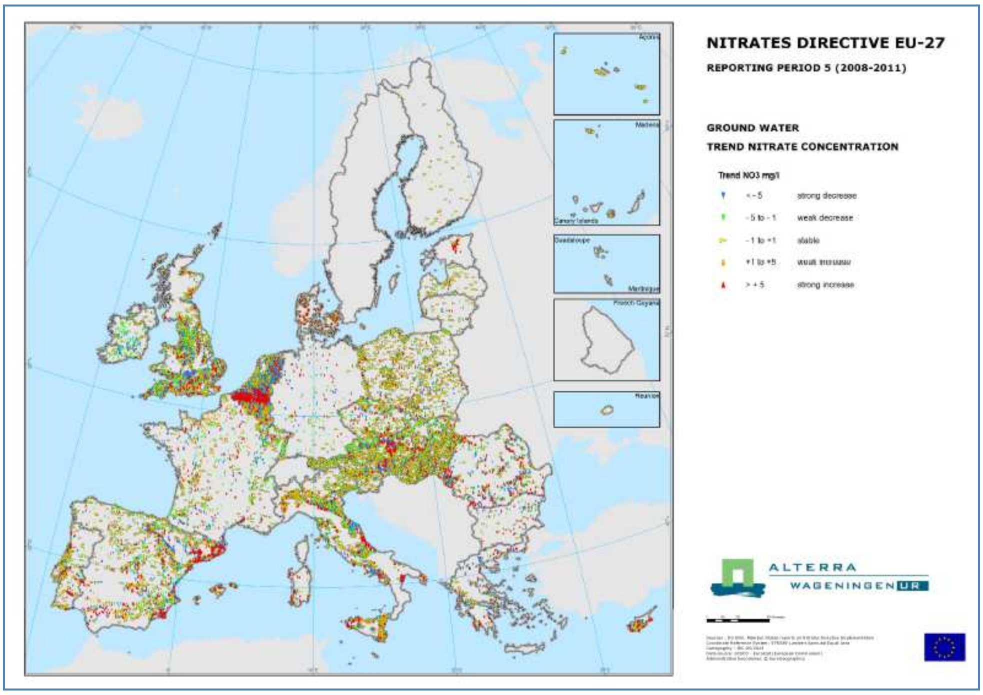
The nitrogen pulse to groundwater aquifers is weaker than to surface waters, which explains why the origin of potable water supply influences externality estimates for the various catchments. Where groundwater is the dominant source for water supply, externalities for ammonia dwarf those for drinking water. Surface waters on the other hand are immediate recipients for leaching. The Czech catchment appears to be a hot-spot, as leaching affects the Zelivka water reservoir, a main water supply for Prague.

External costs appear in most cases to exceed marginal abatement costs (MAC) with water tap nitrogen filters. To identify the optimal level of nitrogen pollution it would be necessary to consider also N-related MAC for ammonia and GHG.

Catchment (population)	CZ: Zelivka (900,000)	DK: Odense (60,000)	IT: Enza (293,000)	LU: Attert (12,800)	NO: Vansjö (60,000)	UK: Ouse, York (200,000)
Impact externality	*€ per kgN /year*	*€ per kgN /year*	*€ per kgN /year*	*€ per kgN /year*	*€ per kgN /year*	*€ per kgN /year*
Ammonia	0.30	0.12	0.17	0.38	0.09	0.26
Drinking water	28.48	0.08	3.05	0.11	(0.94)	0.11
Eutrophication	-	0.02	-	-	-	0.01
GHG	0.09	0.09	0.09	0.09	0.09	0.09
Sum	28.87	0.31	3.31	0.58	(1.13)	0.47
MAC control	5.93	0.45	1.10	0.16	0.97	0.17
Benefit of EPI (T1)	*€ per hectar /year*	*€ per hectar /year*	*€ per hectar /year*	*€ per hectar /year*	*€ per hectar /year*	*€ per hectar /year*
Ammonia	15	9	24	39	11	30
Drinking water	1424	6	451	12	(114)	13
Eutrophication	-	2	-	-	-	1
GHG	5	7	14	10	11	11
Sum	1443	25	489	60	(136)	55
MAC control	296	35	163	16	117	20

Table 1 External cost of mineral fertilizer-N for six European catchments. Marginal abatement costs (MAC) relate to potable water.

The figures presented in Table 1 must be regarded as illustrative in that they result from an attempt to apply the impact pathway approach (conventionally applied for air pollution) to water management but could be further improved and consolidated. It should also be noted that the figures are presented as averages for each catchments, whereas in reality further spatial disaggregation could support differentiation of estimates according to the specific influence on drinking water supply.



Map: Trends in groundwater nitrate concentrations between the reporting periods 2004-2007 and 2008-2011 (European Commission, 2013).

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2.4 Payments for ecosystem services

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ACTeon, France

In 2005, experts contributing to the Millennium Ecosystem Assessment revealed to decision makers and citizens that ecosystem have been changed more rapidly and extensively for the past 50 years than in any comparable period in the human history. An approach not only considering one environmental issue but more comprehensively ecosystems and their direct and indirect contribution to human well-being (ecosystem services - ES) is becoming a guiding idea for most of environmental policies. Indeed, they underlined that the degradation of ecosystems and associated ecosystem services had reached an alarming rate: approximately 60% of the ecosystem services examined by the MEA (2005) were being degraded or used unsustainably in some cases leading to abrupt and irreversible changes associated to nonlinear changes in ecosystems. From the economic perspective, payments for environmental services schemes (PES) can be instrumental to this approach and have experienced a growing interest in the last decade from policy makers. Indeed the TEEB (2011) promotes that economic assessment of ecosystem services can be a tool to guide biodiversity management by revealing pro-biodiversity investments as a logical choice. They also consider PES as one of the mechanism allowing to capture the value of ES and capable of creating the right incentive. PES schemes have also been promoted in the EU Biodiversity Strategy to 2020 as one of the tool required to its implementation. In addition, the potential of PES schemes is highlighted in the Roadmap for a Resource Efficient Europe. However, such instruments are not well developed in Europe to date (Bennett et al., 2013), while the limited number of implementation seems to be promising. The EPI-Water project explored a few existing examples and conducted an ex-ante assessment in France to better understand its potential within a European context.

2.4.1 Policy challenge

One of the key challenges is that of diffuse pollution across Europe and especially the pollution by Nitrates. Indeed, diffuse sources of pollution are considered to pose most obstacles in achieving good status in EU waters (EU, 2013) with the Nitrates Directive objectives identified as one of the key measures to achieve in the recent Blueprint to Safeguard Europe's Water Resources.

Agriculture is the predominant source of nitrogen discharged into the environment in Europe. As shown in Figure 4, this pressure concerns mainly the western and central Europe. Regarding the numbers of cattle, pigs and sheep, the pressure from agriculture has decreased in the period 2008-2011 compared to 2004-2007 (EU, 2013).

This reduction of pressure resulted in an improvement of the fresh surface water quality regarding the nitrate pollution. However, the situation for groundwater only shows a slight improvement due to varying time lag between changes and results.

Over the 2008-2011 period, 14,4% of the 33,493 groundwater monitoring stations reported nitrates concentration exceeding the threshold of 50 mg/l, and 5,9% were between 40 and 50 mg/l.

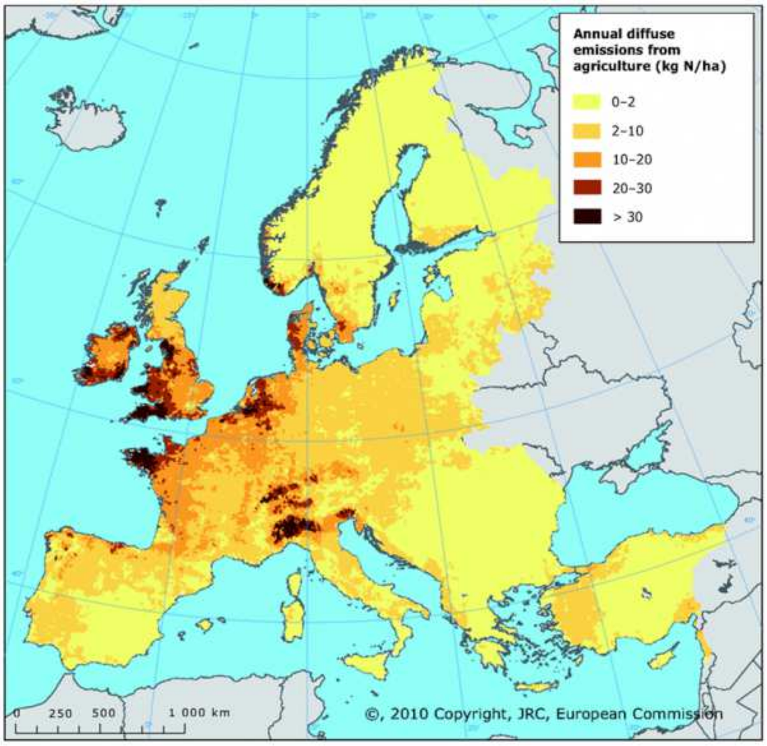

Figure 4: Annual diffuse agricultural emissions of nitrogen to freshwater in 2009 (kg N per ha of total land area) Source: Bouraoui et al (2009)

While most of Member State predict a further reduction in nitrate concentration both in surface and Groundwater, prediction are subject to uncertainty (EU, 2013). Accounting for these, the case of France could be seen as representative of European worst cases (France was subject to 2 of the ten infringement cases opened by the European commission in 2013 in relation to insufficient implementation of the Nitrate Directive). For instance, on the Seine-Normandy river basin district, nitrates concentration in groundwater has increased since the 1950s and the PIREN-Seine measured an increase of 0.64 mg/l in average between 1970 and 2000 (Ledoux et al., 2007).

Nitrate pollution remains thus a major issue in the basin leading to high treatment costs and the closure of a few water abstraction sites. However it is difficult to estimate the current cost of nitrate treatment in Europe. Still, the case of La Bassée-Voulzie shows that, nitrates treatment costs might be multiplied by 10 in 50 years (Defrance et al., 2013).

PES schemes can be one additional tool to address this issue. In addition, PES schemes are, as local EPI, even more appropriate in a context where the main concerns are the remaining hotspots of pollution in Europe, where specific land use or geological conditions make the currently implemented programme of actions against nitrate pollution inefficient (EU, 2013).

2.4.2 Opportunities

There are three main sources of opportunities for PES to develop in Europe. The first is linked to the approach followed by key water policy instruments. The promotion of the ecosystem service approach at EU level most evidently in the Water Framework Directive (2000/60/EC) or the Flood Directive (2007/60/EC) opens the way to increase the relevance of PES schemes as complementary responses to increasingly demanding regulation for economic agents.

In addition, the nature of PES as offering win-win situations around ecosystem services can be complementary to existing regulation if the policy mix is well defined. Given its emphasis on mutually beneficial agreements, PES is particularly suited to address pressures that are not necessarily covered by regulation (although PES can support or ease the enforcement of regulation), aiming at providing additionality to any other existing policy instrument. Regulation defines certain rights (i.e. polluter-pays principle, land use restrictions, etc.). However, economic agents do retain certain rights with regards to their practices which need to be recognised and valued if additional limitation to their rights is to be envisaged.

Finally, there is also a third, more opportunistic possibility offered by the current reforms in the CAP and the financial crisis. A similar role is currently (but only partially) played by agri-environmental measures (AEM), while the two instruments are different in terms of flexibility (type of ecosystem services considered, amount of the payment, duration). The superposition of PES in areas benefiting from the AEM could be seen as redundant although having a purpose with different geographical and temporal scope. The probable scrap of the AEM in the foreseeable future offers a series of opportunities to explore for the future development of PES schemes. Moreover, the current context of financial crisis is pushing for deeper involvement in from private partners as beneficiaries and contributors, which would partially alleviate public budget obligations.

2.4.3 Design

Defining Payment for Environmental Services (PES) schemes is a challenging exercise. Muradian et al. (2010) explain that PES schemes are designed to be incentives and aim at changing individual (or collective) behaviour that otherwise would lead to excessive deterioration of ecosystems and natural resources. PES schemes are "a transfer of resource between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources" (Muradian et al., 2010).

A narrower approach is the handy and concise definition of Wunder (2005) who defines PES through five criteria as "(i) voluntary transaction where (ii) a well-defined environmental service (or a land use likely to secure that service) (iii) is being "bought" by a service buyer (iv) from a service provider". The fifth criterion defining a PES scheme refers to conditionality, where the service provider secures service provision, this PES scheme needs to be truly contingent upon the service being continuously provided.

While the literature considers a diversity of PES schemes, the research under the EPI WATER project focused on PES schemes where the buyers of the environmental service are the actual beneficiaries of the environmental service ("user-financed" programs in contrast to the "government-financed" programs). In addition, two characteristics of PES schemes should be considered: i) level of the payment for PES schemes is defined considering both the costs borne by the producers to offer the environmental service (ES) and the benefits of the buyer; ii) it is a flexible instrument that can be adapted to local conditions and stakeholders through a negotiation process.

In a PES scheme, the level of the payment is defined considering both the producer of ES losses and the benefits of the buyer(s). This eventually leads to higher payments than just compensation of additional costs or the loss of income associated to practice change as it is the case for agri-environmental measures (AEM). Figure 5 illustrates the example of a buyer paying farmers to change their practices so to reach a certain environmental objective.

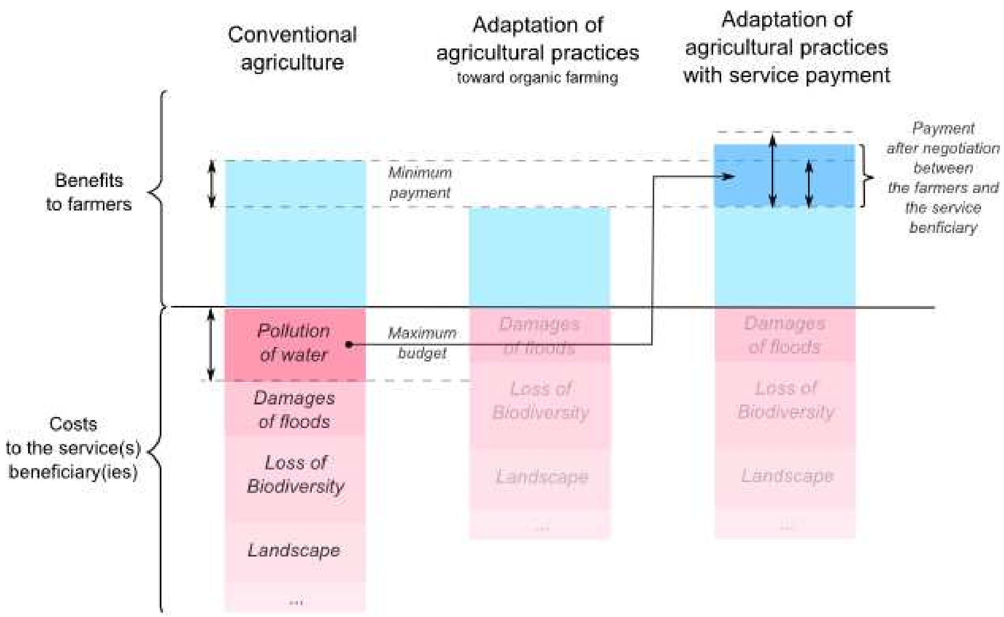

Figure 5: The logic of a payment for one environmental service. Source: Defrance et al., (2013), adapted from Engel et al., (2008).

PES schemes can be adapted to local conditions and stakeholders through a negotiation process. The type of environmental services considered, the duration of the payment, the nature of the payment, as well as the monitoring system can be discussed by the buyers and the providers. Flexibility contributes to the diversity of PES schemes around the world and to the difficulties in proposing one clear and common definition for this economic instrument.

2.4.4 *Prerequisites*

PES schemes are viable if few conditions are met. First of all, the environmental issue must be **clearly identified** in terms of ecosystem services (one main ES and eventually secondary ES). This condition is comprehensively described in the literature (UNECE, 2007; FAO, 2007). Once this is done, the **relevant area to be characterized can be delineated**. Indeed, Fisher *et al.* (2009) classify each ecosystem service depending on its relationships between service production and where the benefits are realised. In the case of groundwater quality, the service could be either directional (benefits can be found in specific location due to the flow direction) or omni-directional (benefits can be found around the provision location without directional bias) depending on the hydro-geological conditions.

Both the service and the area must be **well known** from a scientific point of view. Indeed PES schemes are more likely to succeed when there is a good understanding of the situation and/or good informational system at local scale (in the case of water quality issues, information on groundwater, its functioning, estimation of time lag between a change of land use and a change in pollutant concentration, etc.). On the one hand, **land use characteristics and opportunities** must be well understood and defined. What are the current practices, objectives and constraints of producers? What are the costs and environmental outcomes related to current and future land management practices? This can help understanding the **supply function for ecosystem services**. On the other hand, stakeholders must have (or be able to produce) a good understanding of **the ecosystem service and its functioning**. At the end, it has to be possible to apply the Driving Forces-Pressures-State-Impacts-Response framework (DPSIR). Modelling (hydrological at least) often contributes to this understanding as it is the case for Evian (Defrance, 2011) and for the Bassée-Voulzie (Defrance *et al.*, 2013). This can help understanding the **demand function for ecosystem services.**

In addition, information should be **produced and shared by an intermediary** to be seen as neutral and acceptable. However, both creating, and sharing information are resource consuming. Such costs, along the subsequent costs of negotiating and enforcing agreements can be described through the transaction costs analysis. **While transaction costs are identified as essential in the literature referring to the development of EPIs** (McCann et al., 2005; Krutilla and Krause, 2010)**, they are very rarely quantified and analysed for the design or implementation of PES schemes**. The ex-post assessment of the Evian case (Defrance, 2011) estimated that transaction costs were of the same order of magnitude as the amount of the payment. Actually, they are probably higher as the cost of understanding of the service functioning (hydrological modelling) was not factored in. The size of the transaction cost were not an obstacle to the scheme given the benefits associated to it by the buyer (a multinational company), however it may be so under less favourable conditions.

As regards stakeholders' characteristics, three pre-conditions should be highlighted. At first, the main service buyer must be **financially impacted** by the preservation or degradation of the ecosystem service (facing losses) and she should be a "primary buyer" (private organisation who benefit directly from improved ES provision). In the second place, the beneficiary should have a few basic characteristics: **dynamism, local legitimacy,** being appreciated by the community and willingness to involve other stakeholders and share information with them so to take the project into its final stages. Finally, one must be able to identify and mobilize **a local "champion" known and recognised** by the providers to make PES work, as this will help building trust and thus reducing transaction costs.

Last but not least, the **institutional set up** must be clear and **adequate guidelines** should be accessible to public bodies both at national and local level on what is feasible in terms of involvement in such schemes (i.e. contracting, intermediation, act as buyers or sellers, etc.). While local PES schemes focusing on groundwater quality emerged in Germany without this condition (thanks to learning by doing and the dissemination of such lesson learned), the development of PES schemes is severely constrained in Europe due to institutional uncertainty in this matter. Europe seems always to be ranked last in worldwide reviews of PES schemes or incentive and market-based mechanisms being used to protect the natural infrastructure of watersheds (Bennett *et al.*, 2013). **A clear and transparent definition and legal framework remain to be defined** at EU level (considering the diversity of situation in Europe) to promote the implementation of PES scheme as water-related EPI[8].

2.4.5 Outcomes

PES schemes can deliver both in theory and in practice as estimated by the WP4 ex-ante assessment (Defrance et al., 2013) focusing on a rural area with agriculture as the dominant land use (Bassée-Voulzie hydrographic Unit) and demonstrated by the WP3 ex-post assessment in Evian, France (Defrance, 2011) and New-York, US (Kousky, 2011). The case of Evian is quite similar to the one of Vittel (France) as both are related to Natural Mineral Water management. The Evian Company initiated in the late eighties a promising multisectorial water protection policy tackling wastewater collection and treatment and agriculture among others. It aims at protecting the Evian Natural Mineral Water (NMW) by promoting a sustainable development of its catchment area. An association gathering the main stakeholders was thus created to smoothen the process of negotiation and helped reducing transaction costs. While environmental outcomes are difficult to assess with certainty due to the hydrogeological context, the overall assessment of the EPI focusing on agriculture is largely positive. The protection of the New York City's drinking water coming from the Catskill-Delaware watersheds system is also well known. It has become a reference because the cost of the prevention policy was significantly lower than the cost of building a filtration plant.

In these case studies, the following impacts have been underlined:

[8] http://ec.europa.eu/environment/water/water-framework/objectives/pdf/Work%20Programme%202013-2015.pdf

1. PES schemes can increase the provision of environmental service and potential co-benefits (complementary ES produced). For instance, changing agricultural practices to maintain and restore water quality could lead to an increase of biodiversity (Case of Evian (WP3 CS19), (Defrance, 2011)). In addition, cooperative agreement between farmers and water utilities has also been assessed in Dorset (UK) within the EPI-Water project (WP3 CS3) (Viavattene *et al.*, 2011). However, measuring and monitoring environmental outcomes is always a challenge;

2. These EPI can also increase the revenue of land owners securing or increasing the production of environmental services, thus leading to win-win situations. However acceptability can be discussed when PES scheme focus on a reduction of pressure instead of the production of "new" services (Polluter-Pays Principle);

3. Finally, implementation of PES schemes is associated to the creation and sharing of information. Both sharing information and structuring the transaction contribute to reinforce the political voice and legitimacy of stakeholders involved.

Similar outcomes can also been found in the now famous French case of Vittel, owned by Nestlé (Perrot-Maitre, 2006; Depres, 2008). Compared to AEM experience, the participation of private organisations can increase performance.

However, extrapolating or up scaling these results is a difficult task for three main reasons. First of all, most PES schemes tend to be local economic policy instruments, embedded in their context. Moreover, the focus in the EPI WATER project was on "user-financed" programs related to water quality which may strengthen the local dimension of such schemes.

Moreover, agricultural characteristics vary greatly between and within EU Member States (van Grisven, 2012): land use, agricultural practices, distribution of crops and even the psychology of farmers on which depend the link between the incentive and the outcomes are site specific. General guidelines can and have been produced on PES but the acid test generally lies with a very local negotiation process.

Finally, institutional conditions can vary significantly among Member States. For instance, the development of PES schemes in France is held back by the fact that diffuse pollution affects primarily local towns which do not have access to the financial and human resources required for this kind of process, while they would be allowed to contribute (Barraqué *et al.*, 2005; Defrance, 2011)). Conversely, the legal institutional framework allows national financial support to farmers to only cover compensation of additional costs or a loss of income that are not covered by the first pillar of the CAP. The payment for environmental services can thus only be done through agri-environmental measures-like mechanisms. The water agencies or similar basin-level institutions could be relevant intermediaries in the design and implementation of PES schemes, in particular for collecting and redistributing financial resources. However, their budget has been assimilated to the budget of the French Ministry of Environment at the dawn of the 21th century and thus fall into the second category of government financed type.

2.4.6 Comparison

Water quality was traditionally tackled through remediation measures such as water treatment. The case of New-York city illustrates the possible switch between the old system based on remediation and a new approach trying to combine both remediation and preventive measures. While New York City gets its drinking water from three watersheds that are grouped into two systems, the Cronton system and the Catskill-Delaware, they decided in the 1990s to filter the Cronton system and to develop a voluntary agreement with farmers on the Catskill-Delaware system, to address a decline in water quality for New York City (Kousky, 2011).

PES schemes, as well as other EPI, should not be considered as a solution in themselves. Actually, this type of EPI ought to be seen as an additional tool to protect or restore the quality of water and more generally the preservation of environmental services. In the Evian case study, the voluntary agreement is considered as a tool for the preservation of the Natural Mineral Water complementary to (i) the natural geological protection, (ii) the legal protection (the "Declaration of Public Interest") that is mostly conceived to maintain the integrity of the impermeable cover of the aquifer and (iii) the technical protection (design and protection of the spring catchwork such as using stainless steel pipes, high quality grouting, but also protection at the surface: close buildings above the spring, alarms).

The *Bassée-Voulzie* case study allows the assessment of the coupling of the nitrate tax with the PES scheme, considering EPIs not only as alternatives but as complementary following the recommendation of Bourgeois (2012). The assessment concluded that the combination would enable lowering the level of the tax and the PES for the same environmental outcome compared to the effects of each EPI considered separately. The combination thus helps reducing the negative economic impacts of the tax and the deadweight effects of the PES. In addition, such a policy mix contributes to increase acceptability creating a system in which specific issues of acceptability for each EPI symmetrically cancel out one another: a tax on nitrogen fertilizers will faced an outcry from farmers and fertiliser industries, while it fits to the Polluter-Pays Principle (PPP); on the other hand, a PES scheme can be designed to be acceptable for farmers, while it is contradictory to the PPP.

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2.5 Water emission trading

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Water emission trading (WET) is widespread in Anglo-Saxon countries, while gaining some interest in Europe, driven by the accomplishments in the air pollution control. Similarly to air quality emission trading, notably the EU Emission Trading System (ETS) for greenhouse gases, WET makes it possible to reduce the costs of compliance with ambient water quality standards and regulations. Trading with emission permits shifts pollution reduction to where the marginal abatement costs are lower, until these are equal for all sources. Contrary to carbon dioxide, however, water pollution (such as nitrate or phosphorus) is not uniformly mixed and hence the harm caused is location-specific. This poses substantial restrictions on trading with water emission permits, and the market is often thin. The experience shows that WET schemes, where implemented, become more effective and accepted over time. In Europe, WET may become a practicable instrument for nutrient pollution control involving non-point sources, as other upfront pollution control measures at the point sources have already been exploited to large extent. Our analysis draws upon the research conducted in the EPI-WATER project and literature review.

2.5.1 *Policy challenge*

At latest in 2012 it became evident that the ambitious goals of the Water Framework Directive (2000/60/EC) would not be met by 2015 for about a half of water bodies in Europe (EEA, 2012). Water pollution causes immense yet not fully understood social losses. For comparison, the costs of not complying with the *environmental acquis* has been estimated to 5-20 billion Euro/year (COWI, 2011).

Agriculture is the largest source of reactive nitrogen pollution in surface and groundwater (~60 per cent), causing eutrophication (and subsequently algal bloom) and harmful effects to human health, despite the Nitrate Directive (ND, 91/676/EEC) has been in place for more than twenty years. Recent review (EC, 2013) of the progress achieved since the introduction of the ND testified modest improvements. On average, 20.3 per cent of EU-27 groundwater and 4.6 percent of surface monitoring stations show concentration of nitrate above or slightly below 50 mg nitrate per litre, which is a value of concern according to the ND. Moreover, some 80 per cent of European freshwater bodies exceed the threshold of 1.5 mg of nitrogen per litre deemed detrimental to biodiversity (Sutton and Bleeker, 2013). Across the EU, the use of nitrogen fertilisers declined as a result of EU Common Agricultural Policy reform in 1992, introducing mandatory set-aside, and the secondary biological wastewater treatment, mandated by Urban Wastewater Treatment Directive (UWWD, 91/271/EEC). Annual average application of nitrogen fertilisers amount to 85-90 kg per ha (Stoumann Jensen et al., 2011), while the highest rate permitted by ND is 170 kg per ha. Almost a half of nitrogen input in agriculture stems from mineral fertilisers (EEA 2012). The social costs of excess nitrogen in European water bodies alone have been estimated to 5–24 Euro per kg of reactive nitrogen (Brink et al., 2011).

2.5.2 Opportunities

Water emission trading (WET) is a cost-efficient way of pollution control for surface and groundwater bodies, sizeable improvement of water quality and preservation of the integrity of riverine ecosystems. Compared to traditional regulatory or price-based economic instruments, WET makes it possible to reduce the costs of achieving water quality goals by shifting pollution reduction to where the marginal abatement costs are lower, until these are equal for all sources. As a result, the overall compliance costs are lower and the abatement pace more rapid. In addition, WET offers relative certainty with respect to the achievement of the qualitative targets. Moreover, further ancillary environmental benefits are obtained if the nonpoint sources from agriculture are included in the trading, ether as regulated or non-regulated entities, and a source of inexpensive pollution reduction. The ultimate advantage of WET is that it offers some flexibility for economic growth without compromising the environmental goals (Moore, 2012).

These benefits are attainable especially in larger basins, or terminal water bodies, characterised by a larger number and variety of pollution sources and different marginal abatement costs. To avoid the high concentration of pollutants in some parts of the basin, trading rules need to be carefully designed and offsets are usually permitted only between the sources at the same site or between up- and downstream sources. This and other design conditions reduce somehow the efficiency of the trading programmes.

In theory, the cost efficient abatement of the pollution is independent of the initial allocation of the permits (as explained further down). Because the regulator does not need to know the marginal abatement costs of the regulated sources, the transaction costs are lower, especially if the regulator facilitates the trades. Indirectly, WET creates incentives for research and development in water technologies and processes (dynamic efficiency).

In practice, the text book advantages of the WET are hardly realised. Instead of following a textbook design, the existing schemes employ a combination of WET principles, making the programmes simpler and more acceptable by regulated entities and general public. Despite the growing acceptability and use of ambient air emission trading, there are only few practical experiences or tangible application plans to introduce WET in Europe (see also section 5). Globally, only few programmes have demonstrably led to the expected cost reduction (Shortle, 2013).

One reason for this is that the European environmental legislation leaves little scope of water emission trading. The WET programmes in Europe would operate in the context of the Water Framework Directive (WFD), the Directive on Industrial Emissions (IED, 010/75/EU; which will in January 2014 repeal the Directive concerning Integrated Pollution Prevention and Control, 2008/1/EC IPPC, and other directives), the Nitrate Directive (ND, 91/676/EEC), the Urban Waste Water Directive (UWWD, 91/271/EEC), and indirectly the EU Common Agricultural Policy (CAP). Especially the IPPC and IED pose substantial challenges to WET as they prescribe instalment-specific pollution target and use of best available technology (BAT).

2.5.3 Design

Water emission trading consists of exchanging pollution permits (allowances or credits, see below for the difference) among the pollution sources (e.g. industrial, sewage treatment plants, agriculture holding). Each source can comply with the mandatory requirements either by reducing own emissions, up to or beyond the given limit, or by purchasing additional permits from other sources with lower marginal abatement costs. The water emission trading leads to a cost efficient solution, the total costs of pollution abatement is lower than they would be otherwise in the case of conventional form of regulation. The WET schemes differ with respect to:

What and who is regulated: Most frequent water pollutants are fertilisers (nitrate and phosphorus) and trading also exists for salinity, temperature and organic matter (measured as biochemical oxygen demand BOD). Persistent bio-accumulative toxics (e.g. mercury) are presently not traded (Willamette Partnership, 2012). Some schemes enable cross-pollutant (e.g. phosphorus-nitrogen) trading. Trading exists between point sources (PS), between point and non-point sources (NPS), and to lesser extent between non-point sources. Most frequently the PS are regulated whereas the NPS are not. In such a case the NPS, characterised usually by low marginal abatement costs, generate emission reductions which are used to offset emissions of PS.

What is traded; credits for emission avoided or allowances for emission generated: In a baseline-and-credit scheme, which is in principle an extension of traditional regulatory approach (Ellerman, 2003), each pollution source is assigned specific emissions limits to be met. The sources may reduce own emissions beyond this limit (and hence over-comply with the mandatory limits) and sell the credits to other sources which face higher marginal abatement costs of meeting its own emissions limits. The certified credits are exchanged between sources that over-comply and sources that under-comply with the regulatory limits.

In a cap-and-trade scheme, the pollution control authority determine an absolute cap (maximum allowable emissions) and allocate pollution allowances among the different sources so that the limit is not exceeded. The allowances can be allocated for free (grandfathering), based on the historical rates of emissions; or auctioned. Other allocation schemes are possible but rare. Initial allocation of allowances may constitute state aid and hence need to be communicated to the European Commission (Philippe, 2012). The economic efficiency of WET, in theory, is achieved independently of how the permits are allocated (REF). In practice there is evidence (REF) that the efficiency is compromised by the nature of uniformly mixed water pollutants and the design criteria preventing hot-spots pollution.

Type of market structure: Trading can take different forms. In bilateral trades, characterised for high transaction costs, each transaction is negotiated between seller and buyer individually. In the case of sole-source offset there is no trading in the narrow sense, the individual sources may relax the limits in some places while tightening it in other places. A clearinghouse is a single intermediary between sellers and buyers. It buys the pollution offsets and sells them to the potential buyer. Exchange markets are public fora with transparent bidding and price building. The trades can be facilitated by third parties (e.g. brokers, credit banks), which is sometimes seen as an additional market structure.

2.5.4 Prerequisites

Water emission trading requires effective, enforceable and verifiable regulatory targets and an enabling institutional framework. The geographic scope of the water emission scheme has to be determined, the pollutants specified, and sources to-be-regulated decided. Economic efficiency of WET exploits the variable marginal pollution control costs across the sources. Hence sufficiently large differences must exist in the target area. A reliable inventory of pollution sources and understanding of the propagation of the pollutants are critically important for this end. The regulatory target is expressed in terms of allowable emission of substances whose accumulation or concentration in water bodies may lead to pollution. The target can be defined in absolute or relative (source specific) terms. In some cases analysed in the EPI-WATER project (e.g. Kieser and McCarthy, 2013), the trading scheme was introduced before the regulation was endorsed and the regulatory intention was sufficient to drive the engagement in the programme. Voluntary WET programmes also exist (see section 5) but suffer from negative selection and reduce substantially the cost efficiency of the programme (Schneider and Wagner, 2003).

Based on the pre-determined regulatory target, the polluters receive permits to release pollution causing substances. In theory, the permit constitutes a transferable right but many programmes stop short of declaring the permit a secure right. Often, public perception of a right is not reconcilable with environmental bad. The concept of WET is sometimes misinterpreted as 'paying polluter' principle. Instead a value neutral term offset or certificate is used to gain public support. Cap-and-trade systems have the advantage that the permits can be allocated without the regulatory authority knowing the sources' marginal costs of abatement. For equity reasons, sometimes not all permits are allocated at the onset of the programme in order to facilitate entry of new entities in the market. For similar reasons the permits holders may be required to place a proportion of their permits for sale. The baseline-and-credit schemes specify the permits taking into account the source-specific performance and historical emissions.

Some flexibility in environmental regulations and legislations is needed to introduce a WET programme. Under current European legislation, it is unclear whether and under which constraints the exchange of pollution permits could be granted. The Industrial Emission Directive (IED), and before the Directive concerning Integrated

Pollution Prevention and Control (IPPC), require individual standards are specified for each source based on the best-available-technology. It is generally believed that although some space for WET exists, a greater deployment necessitates a revision of European legislation.

The regulatory framework includes effective trading rules and trades. The rules include the temporal validity of the permits (which may be issued in perpetuity or specified trading period), and restrictions that prevents market's distortion. For example, trading rules may be designed to prevent the concentration of high number of permits in the hands of few sources, that else would exercise market power and obstruct entrance of new entities. The programs may also include a mechanism regulating the price of the permits, through caps or other safety mechanism (charges or taxes on the pollution exceeding the limit). A cap on permit prices reduces the uncertainty concerning the cost of compliance but reduces the efficiency of the scheme.

The trading rates address the uncertainty with respect to actual reduction of emissions which is important especially for the PS-NPS trading. Furthermore, it accounts for effects of nutrient transport, equivalency between multiple pollutants and in some cases buyer risks. Selman et al. (2009) distinguishes between:

Delivery ratio is a parameter that accounts for losses or attenuation during the transport in the basin. The closer are the under-and over-complying sources, the more equivalent are the efforts to reduce or avoid the emission, and the delivery ratio is closer to 1:1. With other words, delivery ratio is a spatial discount factor that takes into account natural pollution sinks, that is the processes through the polluting substance is absorbed or otherwise transformed. Uncertainty ratio expresses the certitude or our confidence with the estimation of the efficiency of NPS reduction measures, which may depend on weather or other environmental factors. Hence, uncertainty ratio expresses the equality between the pollution reduction at the sources. Equivalency ratio is used in cross-pollutants trades and expresses the degree to which the different polluting substances contribute to the observed environmental problem/harm. Retirement ratio is used when a proportion to credits are retired in order to ensure net quality benefits. The retired credits are withdrawn and cannot be used for a future offset. Insurance ratio is a proportion of the credits used as an insurance pool for the case the offsetting source would not delivered the expected reduction. The PS when buying credits from non-regulated sources, notably NPS, retain liability and may face sanctions in case the NPS do not deliver the expected reduction. Hence the credits assembled though the use of insurance ratio are used as compensation and the pool is managed by are managed by the regulator.

The existing programmes differs in the use of the above ratios in determining trading rates. An inappropriate trading rates can lead to higher concentrations of pollutants (hot spots) in some parts of water bodies, and hence to deterioration of the water quality, or to market barriers preventing a better deployment of the WET programmes in practice. Fisher-Vanden and Olmstead (2013) provide an overview of the trading rates in the active programmes in the US. They found a variation between 1.1 and 4:1 (7.9:1 in one case related to phosphorus).

2.5.5 Outcomes

Pollutant discharge into water bodies has been traditionally controlled in Europe and many other OECD countries by ambient water quality or technology standards. WET schemes have been implemented, to various degree of success, in Australia, Canada, New Zealand, United States, and explored in Finland, Sweden and some other EU Member States (Shortle, 2012). In the 1990s a precursor of WET for nitrate pollution has been put in practice in the Netherlands but was later converted into a tax (Wright and Mallia 2008) and finally phased out after negative ruling of the European Court of Justice.

In the EPI-WATER project we have analysed WET for nitrate and phosphorus pollution control in Kieser and McCarthy (2011), nitrate pollution in Yates (2011) and salinity offsets in Ancev (2011). In addition, the reverse-auction-driven land-use change programme analysed in Ungvári et al. (2013), although different in scope, is applicable for soil erosion control and hence nutrient pollution and sedimentation. In addition, Mysiak et al. (2011) have analysed the tradable green energy certificates and their impact of overexploitation of water resources. Alternative EPIs for pollution control, taxes and charges, analysed in the EPI-WATER project are briefly discussed in the section 6.

Kieser and McCarthy (2011) analysed a baseline-and-credit trading in the Greater Miami River Watershed (GMRW) of Ohio, arguably one of the most successful WET in US. The N-NPS exchange was implemented while awaiting completion of the regulatory limits which did not happen until after the ex-post review study was completed. The PS, only seven out of 300, enrolled in the programme to secure themselves a favourable exchange rates once the new regulation will have become mandatory. Each credit, worth of a pound (ca. 0,454 kg) of total nitrogen (TN) or total phosphorus (TP), was secured by implementing agricultural practices reducing pollution load. To facilitate the trading, the regulatory authority managed a clearinghouse operated on the bases of reverse auctions. Trades were permitted only between upstream NPS and downstream PS, disadvantaging the PS located higher upstream (meaning that the pool of bidding NPS was smaller for them). Although the transaction costs were deemed low and not preventing the trade, some 100 meetings were organised with and between regulatory and regulated entities and public, necessary to build trust and confidence for the scheme. The scoping analysis revealed high potential for trading and very favourable benefit-costs ratio, even without accounting for all ancillary benefits. After ten rounds of trading, discharge of estimated 339 tonnes of TN and 130 tonnes was avoided.

Yates (2011) analysed the nitrogen WET based on cap-and-trade PS-PS scheme in Neuse river basin, North Caroline. The regulatory limits in the form of total maxim daily loads (TMDL) were enacted but NP sources have been given the opportunity to form an association and ensure that their collective limits are met, while opening the initially allocated permits for trading within the association. Although the nitrogen discharge has declined dramatically, only few trades were registered. The reason for this fact may lie in considering the trading as a sort of protection against non-compliance. The PS apparently opted for reduction of own emissions, in light of the expected increases in pollution load. The Neuse scheme enabled sole-source, bilateral and clearinghouse trading. The price of the permit was capped by imposing fines on pollution exceeding the given limits. The whole basin area is considered a single trading zone, whereas the author suggest that establishing several zones and permitting the exchange only within each zone would reduce the risk of high concentration of pollution at least in principle. Yates (2011) characterises the scheme as failing to produce cost-saving compliance. From the background of the earlier discussion (section 3) this may be but an initial stage of the market development.

Ancev (2011) has analysed three salinity-offset scheme in Australia, differing in the context in which they have been applied but leading to similarly encouraging results. The first scheme, applied in the southern Murray-Darling Basin, Coleambally Irrigation Area (CIA), was pursued through the net recharge offsetting policy and the irrigation cooperative statutes. The policy enabled NPS-NPS scheme operating on the basis of baseline-and-credit. The second scheme introduced in the Central West of the New South Wales through environmental protection licence. In the reference area there is an operative PS-PS trading scheme in place, set to reduce the salt load from point sources during the low river flow. However, because the Ulan mine could not participate in the above scheme, it opted for a sole-source offset scheme based on PS-NPS exchange. The low-salt water produced though mining is disposed on 250 pasture land owned by the mine. To offset the salt load, the mine implemented land-use changes and management practices on the other agricultural land it owns. The third scheme, based on NPS-NPS in the Southern Australia, is hypothetical one, exploring the benefits of WET to foster new irrigation development in high salinity impact areas, provided the additional salt load was compensated. The environmental benefits estimation of the analysed schemes was obstructed by drought and subsequent reduction of irrigation volumes. Nonetheless, the economic efficiency of the analysed schemes has been confirmed.

Several reviews of existing or planed WET programme exists in US and globally (Selman et al., 2009; Stanton et al., 2010), the most recent one (Willamette Partnership, 2012) has found 24 active PS-NPS programmes (as in 2011) and the trade volume in the US between 2000 and 2008 amounted to more than USD 52 million. Most trades however were concentrated in few programmes.

From among the programmes not analysed in the EPI-WATER project, it is worth to mention the Dutch manure quota scheme. In the late 1980s and throughout 1990s, a tradable manure quota scheme was in operation in the Netherlands. The scheme introduced farm-level manure production rights from all animal sources of up to 125 kg of phosphate per hectare of land (Wossink, 2003). The rights were grandfathered based on actual manure production. Initially, new right acquisition was only possible with farm-land expansion. Since 1994 the rights became tradable, with restriction across categories of manure from different animal species. The transfer of rights was permitted from surplus to deficit zones. In 1998 the programme was modified by the introduction of the Minerals Accounting System (MINAS), a tax on surplus nutrients (Shortle, 2012). The MINAS was found not in line with the provision of the Nitrate Directive by the European Court of Justice in 20033.

In South Sweden, the proposed CEASAR programme (*Certificates for Efficient Allocation of Shares Adjusted to Retention*), set to be put in practice in 2016, contributes to the fulfilment of the objectives set in the Baltic Sea Action Plan (BSAP) and Marine Strategy Framework Directive. It is targeted at point sources – wastewater treatment plans – in southern Sweden, expected to reduce their cumulative nitrogen emissions by a third (3.000 tonnes). The programme is a baseline-and-credit scheme, which allocates to each source treatment quotas rewarding them for their past performance in terms of nitrogen removal. The quota determines the level of pollution abatement; sources with higher treatment ratio will be allocated smaller treatment quota. The sources will have to deliver an equivalent number of certificates (nominal value 1 kg of nitrogen load avoided) at the end of each year. They may decide whether to generate certificates through own nitrogen load abatement, or to fulfil the obligation through purchase of additional certificates. The certificate serves as a certified pollution reduction, and not as right to pollute (SEPA, 2013). Back in 2009, SEPA proposed a more articulated WET programme, consisting of three interconnected trading mechanisms. First, a cap is imposed that curbs the pollution load consistently with the BSAP. The cap is translated into permits allocated among the regulated sources. A source emitting more pollutants than initially allocated is penalised by a fee. The revenues collected though these fees are used to finance offset intervention on a market with compensatory measures, a clearinghouse like maker structure operated by SEPA. The permits can be freely traded on a secondary market (SEPA, 2009).

A study funded by Finish Ministry of Environment (Green Stream Network, 2008) has explored a Baltic Sea wide nutrient trading programme with the similar scope as SEPA (2013). The study found a large trading potential across the Baltic Sea's riparian countries whose anthropogenic load of nitrogen equivalent amounts to 1.1 million tonnes. The proposed programme foresees four implementation phases, starting with voluntary abatement of nutrient discharge and only wastewater treatment plant participating. Successively and building upon the previous phases the programme is extended to other progressively regulated PS and NPS. The baseline-and-credit like programme is intended to allow trade across pollutants with a rate 1 kg of nitrogen equivalent to 0.14 kg of phosphorus.

The analysis of the above WET programmes, complemented by the fast growing literature knowledge concerning active or proposed programmes, confirms the potential WET possess to reduce the compliance costs with the rigorous European legislation concerning water quality. WET programmes may become a part of the policy toolbox, not a universal solution, in places in which the prerequisites (see subsection 2.5.4) are met. The implementation of the instrument requires substantial time and hence it cannot be expected that the WET programmes may accelerate the achievement of the WFD objectives by 2015. There is a legal uncertainty concerning the applicability of WET in the context of the current legislation (WFD and IED). The European Commission may help to discern these doubts by taking a clear position and giving advice to Member States on how to harness the potential for the sake of better environmental performance of water quality regulation.

2.5.6 Comparison

Nitrate and phosphorus pollution are addressed most frequently by price-based instruments such as taxes and pollution charges (OECD, 2012). The application of nitrate taxes has been explored in the EPI-WATER in Skou Andersen et al. (2013) and Defrance et al. (2013), and is analysed in depth in Skou Andersen et al. (2013b). Emission charges were analysed in (Möller-Gulland et al., 2011) (Hernández-Sancho et al., 2011) and (Rákosi et al., 2011). In theory, the advantage of the WET programmes over the traditional economic instruments is a higher control over the environmental outcomes and reduction of costs of compliance. In practice, the existing WET programmes are very heterogeneous (Willamette Partnership, 2012) and comparison is difficult. Sound and enforceable regulation, as explained earlier, is a key prerequisite of the WET programmes. In many respects, WET programmes especially baseline-and-credit schemes are closer to traditional regulatory approach, while allowing some flexibility in compliance, than to environmental taxes. The sole-source mechanism is a case in point. Taxes or charges can be used to cap the price of the permits and reduce the compliance cost uncertainty, particularly in case of non-linear environmental damages and uncertainty about the abatement costs (Johnstone, 2003). In principle similar permit price regulation is possible by using the reserve permits. In addition, taxes and charges generate revenue and limit the wealth transfer when the allocated permits are grandfathered.

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2.6 Transient flood storage

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With increasing flood damage and risk, transient storage of flood waters (TSF) on land with relatively low damage potential has gradually become a part of disaster risk reduction (DRR) strategies across Europe and beyond. The economic rationale of TSF rests on damage reduction through controlled inundation of areas with low-value land uses in order to protect areas with high-value land uses, notably urban residential or industrial developed areas. TSF is a part of natural water retention or flood attenuation[9] measures, capable of reducing and delaying peak discharge. The economic policy instruments (EPIs) in the context of TSF focus on an adequate and cost-effective compensation (flowage easement), and may include incentives for provision of additional socially beneficial services. If flood protection is recognised as a water service, as in the view of the European Commission, than the costs of TSF will have to be recovered at least to some extent either through beneficiary-pay or polluter-pay principle. Both are problematic because the flood externality created through land conversion and management upstream is both, difficult to quantify and a result (legacy) of the historical development. This synthesis, first, reviews the challenges generated by floods, providing estimates of flood related costs across the continent. Next, the opportunity offered by flood water storage schemes is reviewed, emphasizing the cost saving potential compared to other arrangements, especially dikes. Then policy instruments that can be deployed to ensure efficient operation of flood water storage options is discussed, with special attention to different

[9] There are multiple terms used to some extent interchangeably when referring to the transient storage of flood water. Stella Consulting (2012) refers to *natural water retention measures* (NWRM) as '*measures that aim to safeguard natural storage capacities by restoring or enhancing natural features and characteristics of wetlands, rivers and floodplains, and by increasing soil and landscape water retention and groundwater recharge*'. JBA Consulting (2005) uses the term *natural flood attenuation* encompassing the overall impact of the '*floodplain to change the shape of the hydrograph (reduce flood peak and increase flood duration) during out-of bank events due to a combination of storage and resistance to flow*'. Dawson et al. (2011) use the term *runoff reduction and storage*, whereas Wilkinson et al. (2010) refer to '*runoff attenuation features*'. For the purpose of this article we use *transient storage of flood* (TSF) water intentionally with reference to the areas used for other purposes, notably agricultural purposes, isolated for irregular flooding. Morris et al. (2005) refer to w*ashland* as '*an area of the floodplain that is allowed to flood or is deliberately flooded by a river or stream for flood management purposes, with potential to form a wetland habitat*'. UN (2002) uses the term floodway as a '*portion of the flood- prone area that is required to pass the design flood event without a significant rise in water levels compared to undeveloped conditions*'. Hawley et al. (2012) use the term (flood) *detention basins* (Hawley, 2012).

instrument designs. Much detail will be devoted to a range of questions that are important from the perspective of successfully using flood storage, such as preconditions of use, transactions costs, or uncertainties. Under extrapolation it is showed that in a good number of European countries flood risk is evident, TSF has an important role, and short summaries of actual TSF related EPI use are presented.

2.6.1 Policy challenge

Floods and wind storms are natural hazards that incur the largest economic losses in Europe[10]. During the recent decades, the average annual *direct* cost of floods have been estimated to amount to 3-4 billion Euro (Barredo, 2009; EEA, 2011) and *annual expected damage* (AED) to 5-6.4 billion Euro. These estimates are at best the lower bounds of actual social costs of floods as they rely on incomplete records and neglect indirect and intangible losses (EEA et al., 2013). Since its establishment in 2002 and until September 2012, the European Union's Solidarity Fund has been mobilised for 29 flood events and the provided aid exceeded 1.34 billion Euro (42 per cent of all disbursements).

Many of Europe's large cities and conurbations are located close to major rivers in the middle or lower reaches of river basins (EEA, 2012b). As a rough estimate, in absence of a faithful account of current flood protection measures, around 20 per cent of cities are classified as vulnerable to fluvial floods[11]. Several review and large scale assessments, notably IPCC SREX report (IPCC, 2012), have found that the increasing economic losses from weather- and climate-related disasters are to a large extent due to population growth and wealth accumulation. In other words, the observed increase of losses is caused by more people living where they may be adversely affected by disasters. Still, human induced climate change will in many areas across Europe increase the flood risk substantially. The existing studies project an increase of annual flood-related economic losses in Europe by the end of the century by 7.7-15 billion Euro (Ciscar et al., 2011; Feyen et al., 2012) for the IPCC-SRES scenarios A2 and B2, and by 50 billion for the scenario A1B (Rojas et al., 2013).

Progressing urbanisation and soil sealing, along with floodplain development, and wetland conversion or degradation have contributed to increased run-off and flood risk. Since the mid-1950s the total surface area of cities in the EU has increased by 78 per cent, whereas the population has grown by only 33 per cent (EEA, 2006). Over

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[10] Europe is used hereafter in reference to 39 member and cooperating countries of the European Environmental Agency (EEA). The EEA member states are the (28) EU Member States, four countries of the European Free Trade Association (EFTA: Iceland, Liechtenstein and Norway and Switzerland), and Turkey. The EEA cooperating counties include Albania, Bosnia and Herzegovina, the former Yugoslav Republic of Macedonia, Montenegro, Serbia as well as Kosovo under the UN Security Council Resolution 1244/99.

[11] The assessment is based on percentage of city's area, whereas city is defined by urban morphological zones within the core city, that would be flooded if the water levels rise by 1 m.

the period 1990-2006 the settlement areas increased by 15.000 sq.km (9 per cent) (EC, 2012). The urban population is expected to grow by 2020 to 80 per cent (EEA, 2010a).

The demand for greater protection of highly vulnerable areas occurs at the same time as the civil and common law doctrines of flood liability tending to converge gradually to *no adverse impact*[12] principle and governments are increasingly held liable for flood damages including the "residual" flood risks from dikes, levees, storm water systems and other flood reduction structures (Kusler, 2011).

2.6.2 Opportunities

The TSF offers an opportunity for cost-effective reduction of flood risk, not as a stand-alone instrument but as a part of a policy bundle. The appeal of flood storage rests on reducing potential flood damage in highly vulnerable environments such as residential and industrially developed areas, through controlled flooding of areas with a lower damage potential (i.e. low-value land uses, notably agriculture). Compared to other water retention measures, the TSF is also an attractive part of the flood risk reduction strategy because it does not require permanent change of the land use. Instead, it serves as a buffer capacity: land which is managed in a certain productive way is '*called to service*' only when this is useful.

To give an example of the opportunities of TSF, a 2006 cost-benefit analysis of flood defence strategies on the river Tisza in Hungary identified that during the next 100 years the present value of the total costs (investments, defence operations and damage) of building a system of 6 large flood storage reservoirs was about 60 per cent below the total costs of a benchmark, where only the currently existing flood protection dikes are maintained. Raising the height of dikes by 1 meter all along the river would be even cheaper (80 per cent cost saving compared to the benchmark) but this strategy would require enormous initial investment costs, placing a high burden on state finances. A slightly less expensive option is creating a system of 11 reservoirs, with both the present value of all future costs, and the initial investments costs lower than that of just raising the dikes.

2.6.3 Design

The EPIs may take different forms along a continuum of policy measures from a land expropriation and successive (back-)leasing through *easement*, up to service *contracts*. Land expropriation entails an *adequate* compensation. This would normally be equivalent to full land market price but in justifiable cases it may be lower. Their nature of *being voluntary* is what distinguishes EPIs from regulatory instruments such as expropriation. As a special case, the land is leased back to the previous owner to pursue activities compatible with the land destination for TSF. Leasing contract will set the rules of an eventual compensation for damage occurred from occasional

[12] *No Adverse Impact* floodplain management is an approach which ensures that the action of one property owner does not adversely impact the properties and rights of other property owners (Kusler and Thomas, 2007)

floods. *Lease-back* in this sense is a special form of a contract, as described below, in which the role of public and private bodies is exchanged.

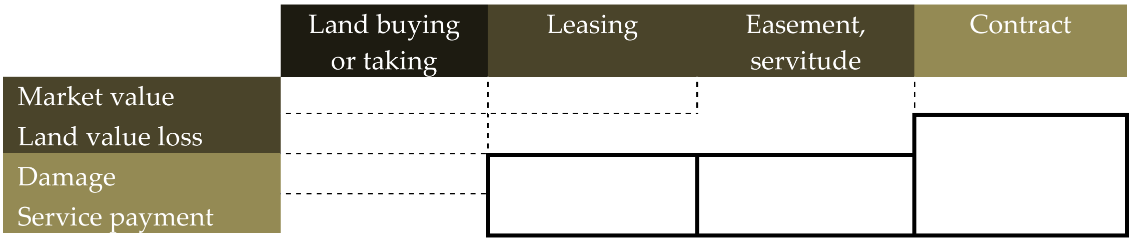


Figure 6: Forms of transactions underlying the TSF on privately owned land. Only a contract is an EPI in a narrow sense since voluntary. Land taking for overriding public interest is an EPI only if the land owner are offered the option to lease-back the land. Easement is a restriction of land tenure rights which is less intrusive than land expropriation. Contract is a negotiated agreement. The transactions and the compensation highlighted in dark allow little or no space for a EPI. The highlighted area shows the cases in which either land transaction or compensation scheme generate some space for an economic incentive.

(Flowage) *easement* is a restriction of the land tenure rights. In the case of TSF, the easement imposes an obligation to accept occasional flooding of the land, in exchange of a fee or compensation. The compensation reflects the loss of land value and the expected flood damage. A service contract is a public-private or private-private negotiated agreement. The economic incentive assumes some form of a(n ecosystem) service payment. River restoration and water retention on agricultural land are among the measures supported by the Rural Development Programmes (RDP), under the second pillar of the Common Agricultural Policy. The payments are predetermined or negotiated by the specific programme, but can also be determined via auctions.

Compensation for land value loss and damage may take different forms. The land value loss is compensated as a portion of the market value of the property but can vary depending on the envisaged (future) land use. The damage compensation can take form of a one-off payment at the time of imposing servitude, or annual fixed rewards, or irregular damage reimbursements. The different ways of damage compensation have an implication on how the associated risk (e.g. of increased frequencies of triggering events) is shared between private and public bodies.

Costs of litigation in case of land taking and imposed servitude can results in substantial delay in the development process. To avoid such situations the EPIs can provide a more satisfactory outcome.

2.6.4 *Prerequisites*

A first step of designing a flood storage area is to find appropriate sites for implementing flood storage. Defining suitable sites requires a thorough investigation of the site and significant engineering considerations, however key factors can be considered at an early stage (Ackers & Bartlett, 2009). An adequate storage volume is a first condition. Storage volume will depend on the landscape profile of the floodplain. A narrow part of a valley or a large flat area will facilitate the storage of larger volume at lower cost (embankment) size. The soil and geological conditions are also essential as an impermeable and suitable foundation is essential for the flood storage structure. In most cases a local source of materials will be sought to build the embankment. Slope stability, seepage through and under the embankments and settlement are key elements in the design depending on the ground conditions (EA, 2009).

Capturing and releasing the flood is dependent on the shape of peak flow time of the considered river and its tributaries. Indeed the reader needs to keep in mind that the main aim of the flood storage is to limit the maximum flow of the river by attenuating the flood peak. Attenuating the flood peak changes the response time of the catchment and therefore may increase the flow downstream at the confluence of sub-catchments. Three general strategies may be considered in implementing flood storage (Ackers & Bartlett, 2009). The first is to capture the flood volume close to the protected urban area. The advantage is that one flood peak can be captured, the disadvantage is that a large volume of storage is required that might need to be released quickly in order to capture successive peak flows. A second strategy is to capture the water on a sub catchment before the confluence of two main rivers. The third strategy consists of distributing the total flood storage volume across a number of smaller on-site storages closer to the source of the tributaries.

In all cases a good understanding of the catchment and associated river flow is required as well as the different hydro-meteorological conditions for an integrated management of the flood storages. One of the pre-requisites is also to have a good telemetry and a good modelling of the hydrological system.

On the site certain conditions may simply prohibit the storage of water. The presence of settlements, existence of infrastructure such as gas and oil pipelines, cultural and heritage sites, environmental sites have to be considered. Previous land use and the potential risk of contamination of the water also need to be considered. Pollution of the river and underlying ground water bodies as well as the formation of anaerobic conditions should also be carefully considered. Other impacts outside the flood storage area may also occur due to for instance the existence of drainage network or irrigation.

If flooding an area is possible, a key criterion is the land use value put at risk. The yearly potential flood damages for crop losses range in UK from 60-225 €/ha (grass), 380-780€/ha (cereals), 600-1500 €/ha (arable land) (Penning-Rowsell et al., 2013). Similar range of values was observed in France (100€-3000€; Cepri, 2008). Targeting grassland rather than arable land is recommended. Yet the reader must be aware that if the revenues from the flooded area constitute a large part of a farm’s economic activities, overall damages may exceed the simple land use value.

High farmland prices and price fluctuation may be a barrier to TSF. Over the past two decades, the agricultural land purchase prices increased in most EU countries, in some cases substantially so (250 per cent in Ireland; Swinnen et al., 2008). The reason for this is that the subsidy granted from the Common Agricultural Policy is capitalised in the farmland prices, and even more so in the rental prices (ibid). CAP subsidy however is not the only and perhaps not the largest driver of land prices. (EEA, 2010b) attested different price drivers for agricultural and urban land, and substantiated the influences of land taxes (purchase, sales and ownership) on the price. In the context of high farmland prices, farmers may be less willing to commit their land to TSF, or if they do, only if the compensation takes into account land value losses associated with the easement. The regulation of the land trades and rentals has a distinct influence on land prices and transaction.

One consideration, however, is that as the price of agricultural land rises, the value protected through TSF in urban areas is also likely to rise therefore the relative advantage of flooding farmland continues.

Land is not a homogeneous economic good. Besides environmental (e.g. soil and climate) and geographic (e.g. proximity to urban centres) criteria, the cultural attitude towards land possession and management often play an important role.

In addition, land committed to TSF may get progressively degraded by the water borne pollutants and invasive species and contaminated sediments transported by the floodwaters.

EPIs could be appropriate solutions in those intermediate situations where the feasibility of land use change highly depends on the local conditions and the initiative of the local stakeholders. In case of low probability inundations (low expected frequency) the cumulated damage is not necessarily large enough to allow the conversion of the instrument into an annual payment that triggers adaptation to change the recent land use practices[13], because the high annual payment would result in a higher expenditure for the public than the replaced cash flow from compensation of damages. On the other side of the spectrum the high probability of inundations make the most conventional agricultural activities (intensive crop production) unsustainable. In these cases there is no other way than the purchase of the right of land use in the form of an easement or the purchase of the land itself. Between these two extreme situations are the cases where EPIs may have a role.

If more than one area satisfies the requirements of the flood protection scheme, then EPIs are the convenient instrument to support the process of reaching an agreement with land owners. If only one specific area is suitable for the flood water storage scheme, then the acceptance of an easement contract possibly with extra incentives for the provision of ecosystem services would be the role of the EPIs.

There is another opportunity for EPIs in the outlined instrument space. This is the development of the lease contracts that follow the public purchase of the lands in question. This could be another appropriate field for EPIs to arrange the delivery of a package of ecosystem services in a cost efficient way for the public.

[13] This, nevertheless, depends on the relative disadvantage of the adaptation option. If that is not large enough, the trigger could still be strong.

1. *Transaction costs*. In cases where there are multiple potential sites for flood water storage, a range of factors can be considered before selecting the location. From the perspective of transaction costs a number of smaller, fragmented reservoirs may be less attractive than one or two large storage sites. A small number of large land owners will make it easier to reach a brokered, negotiated agreement (also, lower transaction costs). A large number of small land owners may challenge resources and lengthen the negotiation process. In this case expropriation could be considered as an alternative to signing up for the agreement[14], especially if a necessary bargaining incentive is needed where complete participation of all the landowners is required to complete the storage area.

In case of competing locations for reservoirs, site selection may be optimised through auctions in which farmers make bids for the level of compensation payment, but transaction costs in this case may steeply rise, it should be considered if the advantage of reaching more optimal location of the sites is worthwhile.

2. *Need to understand the conditions in which individual farmers are operating and particularly the constraints*. When designing the EPI, policy makers need to consider that some of the land is rented to farmers under long-term contracts, and thereby, long term obligations on the part of the land owner. In case of land rental, flood related crop damage generally registers with the farmer (and not the land owner), therefore the EPI should put more emphasis on the farmers' position. Likewise, incentives to land use change may be less effective in the short run in case the majority of the land within the reservoirs is rented. Similarly, in the Hungarian case study it became clear that there are some vertically integrated farms with animal husbandry outside the flood water storage reservoir relying on feedstock from within the reservoir. These enterprises suffer much higher flood related damages than what is apparent at first sight, due to the vertical integration of their activities they face difficulties of replacing their own produce (lack of liquid crop markets, transportation costs) to shift to alternative crops with a lower damage profile.

Therefore appropriate compensation for the value lost, including the modulation of the payment and accounting for full damage is necessary, just like creating sufficient incentives (see the rather unsuccessful scheme for wetland restoration and flood plain management in Scotland; JBA, 2005).

Even a carefully designed reservoir scheme, considering the hydro-dynamics of the basin, may not be implementable if damage compensation is not handled appropriately: in full extent and on time. In the case where compensation is paid periodically in relation to storage use, delayed compensation generates additional costs to landowners, like interest on bridging loans, which translates into incomplete compensation. The structure of the EPI (ranging from damage assessment and compensation to flat annual payments without respect to the damage of the given year and single payment for the right to flood the landowners land) is important for

[14] However, small scale agriculture may be associated with a "way of life" that a policy maker should respect at least as much as other values.

the landowners and the state alike. Payments under different EPI designs may carry the same present value, while landowners and farmers (or the government) may still have a preference for a given EPI. Payment schemes offering an incentive to shift from intensive agriculture to land uses generating ecological benefits may be mutually beneficial, although some farmers might not have the flexibility to alter their land use in the short run. Higher frequency of flooding calls for EPI designs that provide an incentive to farmers to reduce their damage profile through changing their land use, especially if they are unaffected before the storage is built. There are other cases as well (for example in UK) where these lands are usually low quality or already affected to some degree when weaker incentive of the EPI could be enough to induce change as well. Consequently there is a need to tailor the EPI to the local circumstances.

3. *Flexibility in renegotiating the scheme in light of better (more precise) information about the changing risk*. Different EPI designs have a different demand for renegotiation in case better information is generated about the risk of flooding as the years pass. In case of predominantly damage-compensation based schemes and increased frequency of flooding renegotiation is more important for the government as it seeks to reduce the value of compensation that it pays. A fixed annual payment (or fixed up-front single payment) scheme coupled with more flooding, on the other hand, makes renegotiation more critical for the land user. This was a point of interest in the UK where the Environment Agency said that if their calculations were really wrong then adjustment would be made to preserve relations. They, however, felt it was unlikely the landowner would be ready to accept if compensation should be reduced because of less frequent flooding than predicted.

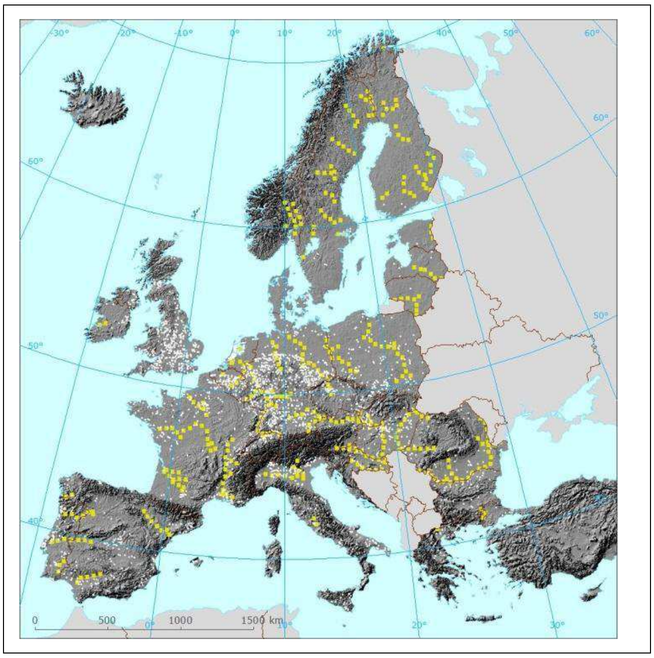
Uncertainty preventing a precise estimation of the flood peak value and flood routing; operation of the flood reservoirs. In case of storage facilities (flood reservoirs) where the inundation depends on a decision (for example opening of the flood gates) an additional uncertainty emerges. Even if hydrological models behind the decision making process on reservoir operation were perfect, reservoir use would not be optimal if decision makers try to shield themselves from charges of mismanagement or possibly even the risk of litigation. Not opening a reservoir or opening it too late is definitely a risk for the operator: if flood damages take place, stakeholders will perceive that flood damages are due to inaction or late action. On the other hand, opening the reservoirs too early may be suboptimal from the perspective of flood defence, but easily provides indemnity to the decision maker.

2.6.5 Outcomes

Stella Consulting (2012) and Burek et al (2012) have explored the applicability of natural water retention measures (NWRM) in the European context. Out of 16 simulated measures[15], two are directly relevant for the TSF: *buffer ponds* in headwater

[15] Stella Consulting (2012) has classified natural water retention measures into 4 categories: forest measures, agricultural measures, urban measures, and water storage measures (storage

areas and *flood retention polders* along rivers. Buffer ponds are natural retention ponds, each 2 ha large with storage volume of 86400 m3, introduced in the upstream areas (50-10 sq.km). Other design criteria included elevation (20-100 meters a.s.l.), urban settlement between 10 and 40 per cent of the area, where soil texture was not sand, where there was no natural lake (Burek et al., 2012). Polders were introduced in locations close to water courses in catchments larger than 10.000 sq.km. Each polder is 200 ha large and has volume of 4 million m3. The distance between polders was designed around 50 km. In total, 569 polders and 2.273 headwater ponds (Map 1) were simulated with a total storage capacity of ca. 2.473 million m3.



Map 1 Location of headwater ponds and polders simulated by Burek et al. (2012);

These areas are *potential* sites for TSF. The room for the spread of TSF is further illustrated by already existing schemes in a number of European countries. In Ungvári et al. (2013), McCarthy (2013), Amadio et al. (2013) and Mysiak et al. (2013) the scope of the TSF has been explored on examples from the Tisza river basin, the United Kingdom and Northern Italy. The situation in other EU countries has also been explored but in less detail. As described below, TSF has been implemented with various physical and policy features in different countries.

In the UK TSF is an established and successfully implemented flood risk management approach facilitated by economic incentives supported by a strong landownership regime. The research reviewed current economic practices in England and Wales via 14 selected interviews in early 2013 drawing on the experience and reflections of representatives of key organisations in England and Wales involved in

in the landscape). The last category, water storage, includes besides buffer ponds and flood retention polders also re-meandering of water courses, wetland restoration and creation, floodplain restoration, and restoration of lakes.

implementing the approach. The key incentive for land owners' involvement in TSF was found to be financial. The dominant approach is appropriation of the land that is used only as the footprint (base area) for the construction of assets associated with the storage (embankments and gates) and maintenance access to such assets. Flowage easement is purchased for the inundated land when the storage is operating. In the majority of the cases reported only a proportion of the land was required. Negotiations are undertaken with landowners or their agents in order to reach an agreement for the purchase price of the land footprint and single compensation for the easement in perpetuity. The negotiating authority, in this case the Environment Agency, endeavours not to pay more than the market value for the land at the time of the negotiation for either the footprint or the easement. The easement value is calculated based on the predicted inundation frequency supported by hydrological modelling. Negotiations are undertaken across many schemes by a single team in order to provide consistency and develop expertise within the organisation. An option of compulsory purchase is available to the authority should the land purchase or easement negotiations fail. However, in the UK it was commented that this is avoided by the authority which does not wish to have the additional gains in capital assets and the associated maintenance costs. On-going expenditure is restricted to maintenance of the assets and so annual compensatory payments or event compensation are not viewed as attractive to the authority.

Where complete control of the inundated land is a prerequisite such as where ecological maintenance or change is required, land purchase of both the footprint and inundated land is more often required. However, flowage easement negotiations may support some ecological aspects. Again the organisation, in this case the Royal Society for the Protection of Birds (RSPB), employs a dedicated negotiation team and is driven more by habitat creation than flood risk management. TSF is viewed more as opportunity than a goal. A different approach in the form of diffuse TSF has been trialled in England. This is where small separate off line approaches are independently developed by landowners supported by farming subsidy payments (Entry and Higher Level Stewardship schemes). It was reported that the problem with this approach is that the final storage volume is unpredictable depending on landowner participation and efforts. The maintenance of the structures and approaches are vulnerable to the continuation of agricultural payments.

TSF in the UK is employed in combination with other flood risk management approaches which spread the risk and create a synergy by reducing the impact of building single approaches on landowners land and to at-risk communities. Success of implementation is highly dependent on local knowledge and a keen understanding of local circumstances and relationships. Transparency in the form of a limited negotiation time and the threat of compulsory purchase as a position of last resort is set against strong legal land ownership laws and a functioning land market all contributes to the EPI success.

In Hungary, the concept for a system consisting of six flood water storage reservoirs was adopted following the 1999-2002 series of record-breaking floods on the river

Tisza. The reservoir concept clearly makes economic sense: in the long run building and utilising reservoirs is substantially cheaper than further elevating the height of dykes. The operation of the scheme, however, is surrounded with avoidable economic damages. In four of the six reservoirs most land is used for crop production via intensive agriculture, while in the other two reservoirs there is a mix of crop production and grazing. In case flood water is released into a reservoir, the yield is damaged, generating direct and indirect costs to farmers. Currently these damages are assessed and a compensation is paid by the state. This process, however, is slow. The obligation to pay a compensation is not planned for within the annual budget of the government, leaving a desire for smoother and more foreseeable payments on the part of the government. As a solution, an EPI has been proposed, following intensive consultations with stakeholders and modelling the outcomes for both parties in relation of changing probability of inundations. The EPI would move away from the pure event based damage compensation to a mix of annual payments and a pre-set amount of damage compensation at an event bases in exchange for the availability of the flood storage service. But this scheme is rational from the state point of view if the annual payments could be bundled with some ecosystem services. A prerequisite of implementation is the homogeneity of the farmers' practice and income level provided by the area. Rather than a perpetual scheme it would be used as a fixed term contract to finance a step-by step adaptation what the shifts in the general circumstances (changes in the CAP, or the payment for agricultural water services) enforce.

In Italy, Amadio et al. (2013) and Mysiak et al. (2013) described the reality of TSF in the context of the relatively small catchments and storage areas, and densely developed floodplains in the Po plain in the Northern Italy. In the first case, the damage reducing potential of a flood reservoir with active control of the inflow has been confirmed but critical limitation revealed, affecting the practical operation of the schemes. The volume of the storage (~ 4 million m3 over an area of 100 ha) is substantially lower than the storage capacity of the Tisza reservoirs (Ungvári, Kis et al, 2013) and the performance of the reservoir is very sensitive to the forecasted flood peak; an error in the order of magnitude of several hours greatly reduces the potential of the TSF to mitigate the damage. The imposed flowage servitude has been encountered with scepticism from the side of the land owners preferring land expropriation. The one-off compensation of average 60.000 Euro per ha (compared to on average 120.000 Euro in case of expropriation) includes both land value loss (up to 40 per cent of the market value) and the present value of the expected damage. The latter is based on the expected flood damage taking into account the intra-annual hazard probability and damage, and relatively low (0.01) interest rate. The litigation costs have contributed to large transaction costs. In the second case, the TSF takes a form of a somehow controlled flood but not in predefined and suitably equipped flood reservoirs. Rather, the TSF is a public-private partnership aiming at reducing flood risk in the area protected by extensive drainage systems temporarily inactive in the aftermath of the devastating earthquake in May 2012. Essentially, it is a voluntary (although it could be made compulsory from the onset) agreement between the land owners (primarily but not only farmers) and public authorities (river basin authority and civil protection agency), operated by the land reclamation boards. The actual compensation is not a part of the agreement but is expected to be based on actual damage in case of necessity.

In other countries, the TSF offers a wide range of experiences. Widely accepted policies are applied only for the most clear cut situations, while the more complex issues are handled on a case by case basis in all of the reviewed countries. Another straightforward method with most of the reported cases from Austria is the (re)transformation of agricultural land in the critical river sections with the biggest flood mitigation effect to natural areas. In these cases the land in question is usually purchased or exchanged for another nearby area. The funding can come from various sources, but the central and regional public bodies are the key sources.

In other cases maintaining the agricultural activity in the area of surplus hazard involves the combination of event-by-event and annual payments. There is no univocal solution that could be described as a national practice. In France the so called "Flood Prevention Action Programs", or PAPIs based on the French acronym are organized on the water-basin level (Erdlenbruch, 2009). These programs are part of a wider legal setup of national damage prevention and risk-sharing insurance policies. The programs are run by locally organized water management institutions. They broker the agreements that are necessary to introduce over-flooding policies in order to shift hazard from high to low vulnerability areas. To finance the agreements water management institutions have the right to impose fees on the beneficiaries. In the long term this poses a threat to the financial sustainability of the schemes from the locally generated sources.

On the German section of the Rhine and on the Maas in the Netherlands the dry polder/off-line flood storage areas that are under construction for cutting flood peaks are based on the same logic as the reservoirs along the Tisza. One difference is the control of flood water flows. In the German and Dutch cases at extreme high water levels the water spills over a lowered section of the dike, there are no floodgates to open. Also as a unique solution in their own flood protection strategy there are no widely developed working schemes settled for the compensation of the additional damage on the agricultural activity.

2.6.6 Comparison

Transient storage of floodwater (TSF) in suitable places is a practical piece of a portfolio of flood risk reduction measures, generating a range of beneficial side effects. While suitable to reduce and delay peak flows to prevent flood catastrophes, its real effect depends on the relationship of discharge and storage volume. TSF should be preferably implemented as part of a wider strategy that includes renaturalisation of streams, floodplain protection and restoration, and other land-use based flood attenuation measures upstream.

Various examples in Europe may help stakeholders engage in the process to select the best approach that fits their needs and a *fair*, *effective* and *prompt* compensation. Fair means that the damage and/or decline of the value of land should be adequately compensated. The upper bound of the compensation is the actual market value of the property, excluding sentimental and subjective value of the property to the owner. Effective on the one hand means that flood risk reduction should be bundled with the provision of other ecosystem services to pay for, on the other hand the compensation should not encourage the taking of excess risk (the phenomenon of moral hazard). Finally, prompt implies that event by event compensation has to be paid as early as possible to avoid the generation of additional costs to the farmers and land owners due to the shock to their revenue stream originating from lost produce.

EPIs are instruments that can promote voluntary agreements on the adaptation of land use in the TSF areas. Of the larger set of instruments (including expropriation, forced change of land use etc.) it has the highest potential to mobilise local knowledge for the most efficient use of the areas' geographical and ecological conditions.

Physical conditions regarding the ideal location for flood storage (see prerequisite and barriers) may influence the potential of the EPI and its design.

Some of the critical aspects that matter in EPI design are the following.

- Transaction costs, as the cost of reaching an agreement is directly related to the utilised human resource which can escalate as the number of involved stakeholders increases.
- The careful consideration of the attitude of the local community toward outside influence of traditionally local decisions on land and farming.
- With regard to the form of EPI payment (e.g. fixed annual, damage based, different combinations, incentives for land use change) there are no universal best solutions and their efficiency critically depends on local and national conditions.

Among the recent circumstances influenced chiefly by the subsidy schemes of the CAP, EPIs have a limited role to play to induce land use change in connection with the public requisitioning of low damage areas for flood risk reduction. The income differences among the different agricultural land uses are so wide that voluntary agreements on ecosystem services can barely cover the financial effect of a conversion from intensive crop production. Therefore compensation, land taking and easements are the dominant forms, with the choice depending on the probability of inundations.

EPIs could contribute to the shift to a more sustainable land use and cost effective flood protection if the polluter/user/beneficiary pays principle (recovery of costs) was applied in the context of flood risk management and the water services that provide ground for intensive agriculture (like drainage). In this case the upstream beneficiaries of the export of flood risk would generate demand for places of mitigation and the decline of the income advantage of the intensive production would generate supply of territories with robust land uses.

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3 Part II: Horizontal issues

3.1 Choosing an EPI? Governance and the choice of a policy mix

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3.1.1 Introduction

What is an EPI?

The key aspect on an EPI is that change is voluntary; it involves channelling private interest to the collective good. That is, the private incentives provided will result in some individuals choosing to change in a way which delivers a collective gain Because it is voluntary, this means that only some people will change. In turn, this implies that there are some differences between people that can be exploited for the collective good. For example, if some people can make a change at a lower cost than others then it makes sense to induce those people to change rather everyone changing – provided that the amount of change that results is sufficient to achieve the objective. Secondly, EPIs rely upon the use of private financial incentives to induce some people to make the change.

EPIs are then different means of exploiting those differences for the collective good. The three main forms of EPIs are then:

– The use of prices or subsidies; generally, these are targeted at reducing some use of water,
– The creation of market in which some thing can be exchanged; commonly, this mechanism is adopted to redistribute usage between users,
– A voluntary agreement between people to make some change; most usually, adopted to reduce usage of some kind.

But innovative instruments may either have a different basic purpose or combine purposes. In summary, the two starting points when considering the adoption of an EPI are:

1. There have to be differences within the population being considered;
2. It has to be possible some of the population to change in a way that achieves the intended objective (e.g. 'good' ecological quality). That they haven't already made this change implies that their present behaviour maximises their private interest but not the collective interest, or that there are other barriers or disincentives for them to make a change. Incentives therefore have to be introduced which are sufficient to overcome these barriers.

The value of each possible EPI is then the extent to which it either overcomes the barriers that exist within the population to making a change (e.g. trading) or provides an incentive to make the change (e.g. a charge/subsidy) sufficient to overcome the barriers and disincentives to making the change.

What is the problem space?

Both as a substance and as a management problem, water has characteristics which make it unique. It is the only commodity which is entirely recycled, through the natural hydrological cycle. Its chemical and physical properties (Ball 2000) mean that only in a few uses are there any substitutes and it cannot be manufactured but the varying naturally occurring supply must instead be allocated between competing uses. It is heavy and incompressible so that transfer between areas except through gravity is exceptionally expensive and generally uneconomic. Plants require water for cooling and other uses in enormous quantities; the figure of 1000 tonnes of water being required to grow one tonne of wheat is the usual rule of thumb. Agriculture is consequently the dominant use of water. Thus, water has to be a cheap, bulk commodity and this in turn means that transaction costs rapidly become a major determinant in the cost of water services and relative transaction costs influence what is the most efficient way of intervening.

Water management involves both natural systems (catchments, coastal zones, aquifers) and artificial systems (storage, treatment and transfer; collection, treatment and discharge). Each system, whilst presenting its own problems, is more or less closely coupled to the others (the obvious exception being unconnected aquifers). The primary characteristic of a system is that there are interdependencies and interactions between the components of the system. In systems, externalities are the inevitable consequence of action at any point; intervention at one place will have effects downstream or upstream and at other times. Externalities are not simply unfortunate but necessary consequences of action in the case of catchments.

That we are dealing with highly dynamic and closely coupled systems is the basis for the adoption of the Integrated Water Resource Management approach (GWP 2000). The nature of water management may be argued to make the management problem more difficult than, for example, air pollution management because of the richness of the interdependencies in and between the systems. The consequences of any intervention have to be considered across the systems as a whole. So, it is necessary to look both for potential synergies between interventions and also antagonisms. The highly connected nature of water management means that both are likely to be experienced.

There are wider characteristics which are critical to water management. Firstly, water availability is a function of meteorology, precipitation and evapotranspiration, times land form, which influences runoff. The runoff resulting from a particular land form is, to a degree, influenced by land use (Calder 2000). Given the temporal and spatial variability of meteorology, water management is consequently about coping with variability, notably the extremes of floods and droughts. Coping with this variability involves providing storage and this is one of the major reasons why water management is so capital intensive. The replacement cost of the water supply and wastewater systems in England and Wales, countries in which rainfall variability is low and where irrigation is not required, is equivalent to about 1/3 of national income. The centrality in water management of coping with variability in meteorology is the reason why climate change has such alarming implications for water management.

Consequently, water management has been central problem throughout human history and consequently produced innovations both in technology and in forms of governance (Fagan 2011; Mithen 2012). The challenge of the future is perhaps even greater.

What is the choice ?

EU policy is for the increasing involvement of stakeholders in water management. Who are the stakeholders, the nature of this involvement and the form of the appropriate decision process are then difficult questions (Green and Penning-Rowsell 2010). But the question for this project is:

– Why and when should the stakeholders in a decision consider one form of EPI either instead of other forms of intervention or as a complement to some other forms of intervention such as regulations?

In turn, what are the reasons why the stakeholders might choose to adopt an EPI? The following discussion does not consider the ideological issues sometimes associated with the adoption of EPIs (i.e. that governments should be reduced in scope) but considers only the practical aspects to be considered when choosing whether or not an EPI, and which type of EPI, is a useful policy tool.

Historically, the focus has been on modifying the environment to satisfy immediate human requirements. The shift to sustainable development requires that instead we modify our behaviours to the limits created by the sustainable use of resources. Thus, the problem is one of change, how to induce change.

Two reasons for considering the use of an EPI are:

– The existing mix of policy instruments do not readily address an emerging problem area or the objectives that it is now deemed necessary to achieve.
– The desire to have a 'better' policy in practice; one, for example, that enables resources to be used more efficiently within sustainable limits.

There are seven possible starting conditions. The problem is currently:

1. Uncontrolled in any way (e.g. in many areas, groundwater is treated as an Open Access resource).
2. Other instruments have been tried and failed.
3. Some form of EPIs have been tried and failed in some way. This includes the condition where the existing mix of individual incentives do not induce behaviour which is in the collective interest.
4. Some combination of instruments have been tried and failed in some way.
5. An EPI has been tried and is more or less successful.

6. Other instruments have been tried and have succeeded, or succeeded in some regard.
7. Some combination of instruments have been tried and have succeeded or succeeded in some regard.

In terms of policy urgency, the first two conditions may be expected to be given the highest priority. It is not unreasonable to give these conditions priority over improvements over conditions where there is partly success. What constitutes 'failure' or 'success' is a difficult question but one for the stakeholders to decide. But an earlier EPIWATER report outlined some parameters which the stakeholders might wish to consider (Zetland and Weikard 2011). The current focus in Europe is on 'well-being' as the overall objective (CEC 2009; Stiglitz et al 2009), which defines multiple sets of outputs and inputs, where efficiency is the relationship between some set of outputs to some set of inputs. A problem here is that whilst economic analyses focus upon the efficiency, a restricted concept of what are the relevant outputs is used. Hence, an improvement in the use of resources may be offset by reduction in the creation of some desired outputs which are outside the limits of an economic analysis.

Choices are always comparative: the selection is between the options that have been identified. So one reason for choosing an EPI may be that is simply that the alternatives are impractical for one or another reason. This often a simpler question than that of whether an EPI is superior to the alternative. For example, the western USA and Australia may be said to have had dysfunctional water allocation systems, ones which were not adaptive to changing conditions, which in the case of the USA was accompanied by very high levels of public subsidy of irrigation water (Wichelns 2010). In neither case was the adoption of pricing of irrigation water at full cost recovery a feasible option, nor was a command and control approach. So, here, water trading has been a feasible and successful means of promoting the reallocation of water to more productive uses, notably in Australia (Young 2011). However, in the case of Australia, the Federal government has had to buy back water to increase flows in rivers to environmental flows (Connell 2011), even though the water being purchased in this way are the releases from taxpayer funded reservoirs. In these conditions, the adopted EPI may not be ideal but may be the best that can be achieved at present.

The issue in a choice is also usually what to do first. Moreover, this approach is reinforced by the increasing stress on using an adaptive management strategy (Mysiak 2010). This requires adjusting the strategy in the light both of learning and changing conditions. Hence, what is done next is a decision to be left to later and then to be made in light of both the conditions then prevailing, what has been learnt. What is done now should not therefore unnecessarily restrict the possibilities of what can be done later.

3.1.2 Failure

Existing interventions may be judged to have failed; in comparing possible future interventions it is as least as, and possibly more, important to look at potential failure modes as to consider success. There are at least three possible modes of failure:

- Ineffective; the desired change is not induced
- Inefficient; the resource cost of the induced change is excessive
- Perverse consequences; an unintended and undesirable change is induced

A fourth possibility that the induced change has undesirable consequences is not a failure of the intervention option but of the analysis.

EPIs, because they rely upon voluntary action and leave open the individuals' responses to the incentive provided, are perhaps particularly open to the possibility of perverse consequences. An example is the consequences of subsidy programmes intended to improve the efficiency of water use in irrigation; for example, in the USA (Environmental Working Group 2013). Subsidy programmes, therefore, perhaps need particularly careful design, monitoring and review in terms of potentially perverse consequences. Subsidies also can create path dependency; once started, they can be difficult to remove. Notably, in the World Trade Organisation negotiations, it has been argued that existing agricultural subsidies can cease to have a distorting effect on trade because the expectation of continuing subsidy has been capitalised into the value of the land. Perverse consequences are possible for all interventions; the issues are how pervasive and how great will be those perverse consequences?

3.1.3 Conditionalities

The shift to sustainable development requires that we modify our behaviours to within the limits of the sustainable use of resources. Thus, the question is:

- How to change the behaviour of the relevant target populations in the desired direction?

The preconditions for success in doing this are:

  - Understand the nature of water and water management.
  - Understanding the barriers facing the target population in making the desired change, and to tailor the appropriate incentives to overcome those barriers.
  - Understand the purposes of governance and how governance works since as to how to intervene are made through governance.

The failure either in adoption or in practice of some EPIs in the past may be argued to be the product of a failure of understanding in one or more of these three areas. For the reasons summarised in the annex, theoretical economic analyses of EPIs have often failed in adoption or practice as a result. But failure in any form of intervention will occur if there is a failure in understanding in any of these three areas; the problem is not limited to the use of EPIs but is common to all forms of interventions.

Looking for differences

If everyone is the same, then there is no practical gain from adopting an EPI rather than intervening in a different way unless the transaction costs are lower for the EPI than from the alternative approach. Conversely, the greater the differences within users of a particular catchment or sub-catchment, the greater the potential to exploit those differences for the collective interest. So the initial questions to ask concern the population who might change and the area in which a change is desired. The area is likely to be a catchment or sub-catchment or the land above an aquifer. Given the predominant use of land and of water, the relevant population is often farmers. But it may be either the total population making some type of use of water (e.g. abstracting, discharging runoff, discharging wastewater) or some sub-population. The requirement is to identify the target population. Then within that target population there must a difference that can be exploited. That difference is in the costs of making a change either directly in making the change (e.g. investing in more efficient irrigation techniques) or in consequence of making the change (e.g. changing to a lower value crop but one which requires less water). What this implies is that the greater the number in the population, the more likely it is that there are differences that could be exploited because of an increasing heterogeneity of usage. A sub-catchment in which there are only ten farmers, all growing potatoes, is not a promising candidate for the use of either a charge or a trading system. In that instance, promoting the establishment of a Water User Association (Salman 1997) is probably a more promising means of dealing with the problem in that sub-catchment.

Assuming that there are differences within the population, it has to be appropriate to seek to take advantage of those differences. Firstly, it has to be physically appropriate. Because an EPI promotes voluntary change, the implication is that it is irrelevant as to:

- who changes,
- how they change,
- where the change occurs and
- when it occurs.

If any of these conditions make a significant difference to the achievement of the collective good then an EPI is less likely to be appropriate. For example, the adoption of Sustainable Drainage Systems (SuDS) which reduce and/or delay surface water runoff is increasingly being promoted to reduce the risks of flooding. But a catchment is a dynamic system where part of flood risk management is to reduce the risk that the flood peaks from different tributaries coincide at some downstream point. Hence, if the adoption of SuDS is focused in some particular parts of a catchment as a result of urban redevelopment, it might make the flood problem worse. When any of four conditions matter, any intervention strategy must be appropriately tailored to the specific circumstances.

A secondary issue is whether it is more efficient for the target population to act individually or collectively. There are frequently strong physical economies of scale in water management. So, in the above example, if a community bands together to build a retention or detention basin, this may be a more efficient use of resource than if each land owner installs, say, permeable paving and green roofs.

A specific problem with applying EPIs in water management is that catchment are systems and the innate feature of systems that making a change in one location will result in changes elsewhere in the system. For example, a failure in traditional approaches to flood risk management is that they have often simply moved the flood around. Thus, interventions on the Rhine for flood risk management and other purposes have both increased the peak flow that reaches the Netherlands and reduced the time taken for the flood peak to reach the Netherlands (Bosenius and Rechenberg 1996). What therefore results in a local improvement may then simply degrade the overall performance of the system. Any intervention, including an EPI, must therefore be tailored so as to ensure that the performance of the system is improved. Lipsey and Lancaster (1956-57) referred to this as the problem of the 'second best': when there are a large number of imperfections in a system, addressing one local problem may have the effective of degrading the performance of the whole. This is perhaps a particular problem in urban areas where land values are influenced by the positive and negative externalities created by other land uses. Hence the performance of the system as a whole needs to be considered and it is probable that a policy mix will be appropriate and one which recognises market segmentation.

Secondly, it has to be socially appropriate to exploit those differences. For instance, a number of socio-economic and demographic factors influence household water consumption (e.g. the number of people in a household, the age structure of the household, the health status of its members, income and so forth). It may be inappropriate to tailor the EPI to exploit all of these differences and appropriate to tailor it so that households with some differences are not adversely affected by the EPI. Thus, in Australia, special free allowances of water are made to households where a member is receiving home kidney dialysis and hence has a very high demand for water (Kidney Health in Australia/ The Home Network nd). Again, in metered households in England, low income households where a member has one of a specified list of health conditions is entitled to a rebate on their water services bill (Consumer Council for Water nd). Most obviously, to use prices requires that it is appropriate to exploit differences in ability to pay coupled to desire for water.

Change

Power is the capacity to induce change (or conversely to resist change) (Lukes 1974). Hence, anything that has one of these two effects is a form of power, and the questions are:

– How does power work?
– What forms of power are most appropriate in which contexts?

For power to work, it may hypothesised that there are three necessary elements (Green 2010):

– A signal as to what is the desired change

- An incentive which is sufficient to overcome the barriers to changing
- A means of testing compliance if the incentive is not sufficiently greater for all so as to overcome the barriers and also constrain the changes to those that are desired. For example, metering provides an incentive to making illegal connections, to falsify readings, or bribe meter readers, all of which occur, as well as to reduce consumption. If the barriers to making illegal connections or falsifying readings are less than to reducing consumption, then it might be expected that the result of metering will be the former rather than the latter.

The signal has to be such as to attract attention where human beings have limits to their capacity to recognise and process signals (Klingberg 2000). It does not seem to be as simple as that the size of the incentive is sufficient for a signal to be selected and recognised.

In seeking to make change, it is necessary to consider why change has not yet taken place. For a change to occur, first there have to be some within the target population who have the capacity to change in the desired way. Secondly, in order to achieve that change, the incentives for them to make the change have to be sufficient to overcome the barriers to making that change. Currently, there may be no incentives to change or they may be insufficient to overcome those barriers. Whilst part of the attraction of EPIs is that they leave open the scope for innovation, the invention and adoption of more efficient or effective means of making the change, it is sensible to start by identifying a practical means whereby some of the population could make a change which has the desired effect. If such a practical means cannot be identified then the EPI would be being introduced in hope rather than expectation. Experimentation would seem to be appropriate in such circumstances.

There are a number of barriers to be considered. The simplest case is when the change could be made through a change in behaviour (e.g. turning off the tap when brushing teeth). But often changes require both an available technology and investment. For example, drip irrigation is arguably the most water efficient means of delivering irrigation water to plants but small scale farmers were prevented from adopting this technology until low cost drip-irrigation technologies were developed (International Development Enterprises nd). There are institutional barriers as well; for example, those renting properties do not have the incentives to make permanent changes to the fabric of the property and may be inhibited by the rental agreement from doing so. In the case of furnished properties, those restrictions apply to water appliances (e.g. washing machines) as well.

There may be physical issues as well. For example, it is generally found that the crops providing the highest economic return to water are vegetables and fruits (Molder 2007). But equally often, low value crops such as grass are also found to be being irrigated and the obvious question is: why is the farmer growing grass rather than fruit or vegetables? There are many reasons but one is simply that crops, like all plants, fit into ecological niches (Grigg 1991) and areas in grass may simply be unsuitable for growing fruit or vegetables. Hence, charging for water may not result in a switch from grazing to horticulture but only increases input costs to farmers.

A clear understanding of what are these barriers and what is necessary to overcome them, and consequently what is required of the EPI or other instrument is required. One of the ways of gaining that understanding is to listen to and discuss with the target group what is desired and what could be done to achieve those ends. Those it is desired to change are likely to have a better understanding of the problems they would face in making a change than an analyst. Analysis is then required to determine what would be the consequence of making the change. In general, for this and in order to build up trust and agreement as to proposed measures, a central message is that a lengthy period of discussion with the stakeholders is required prior to finalising the design of the EPI and implementing it. This is a common message from the ex-ante case studies in France, Hungary and Spain.

Making a change has two cost aspects: the consequences of the change (e.g. changing cropping patterns, adopting demand management techniques, constructing local wastewater reuse); and the costs of change itself. The net consequential gains from the change have to be sufficient both for individual making the change and to the collective if the change is to occur.

Making the change itself involves costs to the individual sometimes but not always these include money and resource costs. For example, since Simon (1957), it has been recognised that neither organisations nor individuals seek to or do make optimal decisions; they try to do the best they can within the constraints. Hence, Rees (1969) concluded that the reason why firms use more water than would maximise profitability is because water costs are too low a fraction of input costs. Firms therefore focus their attention on the issues which critically affect firm survival but to do so means neglecting other issues. Individuals work the same way: prioritising the apparently important but consequently neglecting what is deemed to less important. Confronted with a crying baby and a note upon how to reduce water usage, a parent can be expected to worry about the baby. Thus the one true scarcity is attention.

In turn, as Stiglitz (2008) pointed out, acquiring information has costs including attention costs. Human beings also have cognitive limits: again, we deal with these limits by excluding some things from consideration. Hence, even if were possible to have perfect information about everything, we could only use some of it, that which we selected as being the most important. The two ways of attracting attention to an issue are therefore to make the issue more important (e.g. by increasing the relative price) or to cut the costs of attention. For example, whilst tradable licences were initially introduced in Australia, the volume of trading was low; the volume of trading only accelerated when there was a drought (Kaczan et al 2011). Suddenly water became more important; in addition, a lot of effort went into reducing the costs of attention. Conversely, it might be suggested that if real energy prices are rising rapidly, and energy costs are a higher proportion of input costs or household budgets, then this is not likely to be an effective moment to introduce water demand management. However, conversely, if water demand can be linked to energy demand and energy costs (Energy Saving Trust 2013), this may be an effective strategy. Conversely, water management actions that involved increased energy requirements are less attractive.

At the same time, to provide a collective gain, the gains from the change must exceed all the costs of making the change and inducing the change. For example, in England, the costs of introducing water metering to domestic properties is estimated to add £30 to the average domestic water supply bill (Walker 2009); a 16% increase as compared to the costs of what is in effect a property tax. This transaction cost has to be justified by reductions in the cost of supplying those water services. In consequence, in many countries, individual apartments are not metered (e.g. Paris, Copenhagen, Germany) because the additional cost of individual metering cannot be justified against the savings of water services that may result from metering. In general, these 'transaction costs' are often a significant issue in water management and must be considered in choosing between options. For example, separately charging in urban areas for surface water drainage only really became feasible when GIS systems were available. Previously, the cost of calculating the charge for each individual property would have been a significant fraction of the resource costs of dealing with the load.

3.1.4 Power

If it changes behaviour, it is power. There are thus many potential forms of power (Green 2010), the question is: what is most appropriate in the circumstances? But broadly, power may be argued to work either by influencing what the individual wants to do or what they think that they ought to do. This implies that influencing both may be more effective than addressing only one side of the equation. Equally, that it is undesirable to induce a conflict between the two. The force of social norms in particular should not be underestimated; in Social Dilemmas Theory studies of games, it is commonly found that those breach the group norm are punished by other players even when there is a penalty of so acting to those other individuals (Nowak and Highfield 2011). What Nowak also found experimentally is that rewards for acting in the collective interest is more effective than punishments in promoting acting in the collective interest (Nowak and Highfield 2011). This may suggest that subsidies are more likely to be effective than charges or prices in inducing desired behaviours.

3.1.5 What is governance?

There are many suggested definitions of governance but one that captures what governance has to do is that given by the UNDP (1997) :"*Governance comprises the complex mechanisms, processes, and institutions through which citizens and groups articulate their interests, mediate their differences, and exercise their legal rights and obligations.*" Governance is then done by different agencies, groups and organizations; critically, governance is done by people interacting and it is who interacts and how they interact that determine the success of governance. Governance is not only social relationships in action but the resulting actions either reflect or create social relationships. A central question is therefore: what are or what ought to be social relationships as expressed in the courses of action adopted?.

Governance arrangements can analysed in several different ways (e.g. Ostrom 1995) but central to those analyses are the interplay of actors, rules and power. One purpose of rules, which govern who interact and how they interact, is to set boundaries to power, those boundaries being functional as well as geographical (Green 2010). The nature of these boundaries and the fit of these boundaries the problem (Young et al 1999) is then critical in feasibility of a particular intervention. An example of a functional boundary is where a water management agency has the power to set charges but only sufficiently to cover its administrative costs. Following from the WFD, it might then be given the legislative power to set charges sufficient to achieve full cost recovery as defined in some way.

The practical questions to consider in relation the possible adoption of any policy instrument are:

- Who has the power to implement the proposed instrument?
- Does the instrument fit the problem?
- Is this the best means of achieving the desired result?

To illustrate the first two questions, consider a tradable instrument for water abstraction. Could this be adopted by the competent authority for a catchment using its existing legislative powers or would it require new legislation at the provincial level, at the national level at provincial level, or even European legislation? Secondly, the administrative boundaries of the body with powers may not fit the problem domain; the catchment, aquifer or coastal region (Young et al 1999). Notably, on international catchments, international agreement and national legislation in each of the relevant countries would be necessary to enable water trading across national borders. The greatest proportion of catchments in terms of area in Europe are transnational in nature.

The final question is the core of this report. Is, and where is, the use of an EPI the most effective and efficient form of power to use?

3.1.6 General conditions

In considering using an EPI, there are both general conditions to be considered and those specific to the particular EPI being considered.

- The good or resource being considered has to be clearly definable. For example, a problem experienced in California when trading was introduced was that of 'paper water' (California Department of Water Resources 1993). 'Paper water' exists when return flows are not considered when trading withdrawals; for example, lining an irrigation canal will not yield a net gain in water availability if the leakage water at present simply accumulates in an aquifer from which is then abstracted.

- The larger the target population and the greater the differences within that population, the more likely is an EPI to have an effect. It is also necessary look for and take account of market segmentation.
- A highly desirable potential consequence of an EPI or any form of intervention is that it induces innovation or economies of scale in the production of an existing technical option. However, unless a viable form of change can be identified before the introduction of the EPI, this is to substitute hope for expectation.
- If one policy area is highly coupled to other areas, then the consequences of change in the first area may be difficult to predict. It may not be desirable if, for example, a reduction in water pollution simply means that those pollutants are discharged to the atmosphere or accumulate in the soil. Conversely, if a single intervention has clear benefits across different problems, then this is a major advantage. For example, the adoption of green roofs is considered not only to reduce surface runoff and water pollution but also to ameliorate the heat island effect in urban areas and reduce air pollution (Banting et al 2010). This is to say that how people respond to a particular EPI may determine the desirability of introducing that EPI.
- Any higher transaction costs as compared to the alternative have to be smaller than the reductions in comparative resource costs.
- Economists commonly assert that clear 'property rights' have to be established before any form of trading can be established. The danger here is to wish away a difficult question. However, contemporary discussions in the law (Davies 2007; Worthington 2003) define property rights as relationships between people articulated through access to some 'thing'. The nature of those personal relationships are articulated through property in different ways (Schlager and Ostrom 1992) and in different cultures (Meinzen-Dick and Nkonya 2005). The obvious question is, therefore, what should be those inter-personal relationships? This is not consequently solely a question of what set of relationships will result in the most efficient, in some sense, use of resources.
- Economic analyses typically assume that the resulting changes will be small in effect. But it is sensible to examine the likely nature and extent of changes first e.g. the shift to maize for biofuel production is seen as one of the factors that resulted in a sudden non-marginal upward shift in global grain prices (Hélaine et al 2013); an example of perverse consequences.

3.1.7 Specific conditions for

There are some specific conditions for particular EPIs:

- Voluntary agreements require the existence, or the capacity to create, a community of some kind: a group with a shared interest and an identity. These communities can be of many kinds and the nature of the shared interest can vary. Practical examples of such communities include trade associations such as the

Federation House Commitment of members of the UK Food and Drink Federation of manufacturers to reduce water use in manufacture by 20% by 2020 (Food and Drink Federation (nd), local farmers, and groups such as Business Improvement Districts (Land Use Consultants and Green Roof Consultancy 2010).

– The most successful examples of water trading (e.g. Australia, USA) depend upon a taxpayer financed system of storage and infrastructure.

– The most successful examples of using prices commonly involve hypothecating the revenue in order to provide soft loans or grants to further the objective for which the charge is intended (Andersen 1994). A critical purpose of prices is to raise revenue, notably sufficient revenue to cover costs. The revenue raising, behavioural change inducing effects, and the allocation functions of prices may need to be considered separately and no single pricing schedule may be ideal for all three purposes.

– Charging for potable water in particular has undesirable consequences if it is too effective in reducing demand, or demand falls for other reasons, as it can, as in some cities in the eastern part of Germany, result in revenue falling below costs (Hummel and Lux 2007).

– Generally, the effects of pricing in water management as a means of changing behaviour have been disappointing, having only a weak effect at best. This is possibly for the reasons identified by Rees (1969)

3.1.8 Making change in practice

The complexity of water management makes it dangerous to be too specific in setting out a path of adaptation. The essential requirement is to seek understanding of the individual case rather than to follow a fixed set of rules. Part of this understanding is to identify where there may be synergies (and conversely antagonisms) between both intervention options and also problem spaces.

It should go without saying that interventions need to be based upon a recognition of the fundamental nature of water and water management e.g. that nitrogen is an essential crop nutrient which has to be replaced, and to be available to the plant, it has to be inorganic and water soluble. Changing nitrogen loads in runoff and infiltration water on an individual farm consequently currently largely depends upon changing crops.

Making the transition to sustainable water management requires resilience in the face of shocks such as floods and droughts (Green et al 2011), being adaptive to the changes such as climate change and an ageing population (Giannakoursis 2008), and also innovating and learning how to do better (Argyris and Schon 1966). EPIs can be part of that process of innovation and learning but they need to be implemented in a way that allows that innovation and learning process to continue. Thus, part of the failure of the adoption of tradable abstraction licences in Chile was the failure to incorporate methods of learning from the experiment and adapting to conditions (Harris 2011).

EPIs are still an innovative approach and the full scope of the potential for innovation has not yet necessarily been explored. But it is the general nature of innovations that some will fail; hence, we need safe ways and places of making innovations. Ideally, they should be reversible. Alternatively, they should be tried locally first.

The choice is between EPIs as well as between EPIs and other instruments; for example, between adopting full cost recovery or some trading mechanism or voluntary agreement. So the choice is not whether or not to adopt an EPI but also which EPI. Combinations of instruments may yield synergies (or alternatively be antagonistic). In some of the case studies, a mixed policy approach has been adopted; here, it is impossible to separate out the relative significance of the different component instruments.

The two starting conditions where there is the greatest urgency to identify a feasible and effective form of intervention were argued earlier to be when either:

– Nothing is in place at present
– Command and Control, or another form of intervention, has been tried but is not deemed to be successful.

In the first, the logic is to start by a publicity campaign for the individual target groups. This may have some effect in inducing a change in behaviour (for example, voluntary calls for demand reduction during a drought in the short term at least result in a substantial reduction in demand) (USACE 1995) but primarily a publicity campaign is an agenda setting device which raises the problem that has to be addressed.

What comes next depends upon the specific conditions. But a common feature is the need for a significant period of analysis and discussion before adopting a particular strategy. Secondly, the most successful approaches to making a shift to sustainable water management seem to involve a policy mix rather than a reliance upon any single instrument. This approach can take advantage of the relative strengths and weaknesses of the alternative instruments. For example, in the dramatic shift to sustainable urban drainage systems in Germany there can be seen to have involved Federal law, local regulations, charges for water usage, information campaigns and demonstration projects (Green and Anton 2012).

In the second condition, where a Command and Control approach has been tried but is regarded as having failed, the differences in the forms of command and control instruments need to be taken into account. The general feature is that they require all to make the change, either in the form of prohibiting some action or requiring some action. That requirement may either be specific (e.g. some cities in Germany prohibit car washing except in designated areas) or performance related (e.g. specifying that discharged loads may not exceed some limit). 'Failure' may come in several forms and it is the nature of the perceived failure that is important as the problem may be common to EPIs as well. For example, if there is a regulation limiting changes in land use but land development takes place in violation of the existing regulations then the only useful form of EPI would be one which resulted in compliance improving. It has to be recognised that EPIs are depend upon a compliance regime, another set of rules, which limit actions. Thus, whilst Adam Smith famously argued the virtues of market solutions as means of channelling private interest to the public good ("It is not from the benevolence of the butcher, the brewer, or the baker that we expect our dinner, but from their regard to their own interest"), bakers and brewers were amongst the strictest regulated trades in history, with ferocious penalties for selling underweight or adulterated bread. Moreover, Smith argued for even stronger command and control requirements: he asserted that those in the same trade or profession should never be allowed to meet, as this would inevitably result in a conspiracy against the public interest.

What Adam Smith's example illustrates is that there is less of a clear difference between regulations and EPIs than is sometimes assumed. Thus, that a policy mix approach will often be the best strategy; a combination of different forms of power.

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3.1.10 Annex

The approach advocated here is to be pragmatic. In particular, to avoid basing the design upon theoretical economic assumptions. Unfortunately, much of the literature on economic instruments is based upon discredited economic assumptions, ones which have subsequently proved to be false. Notable assumptions that have been made are:

– *The individual or firm is both rational and optimising*. Simon (1957) pointed out that firms do not seek to optimise but to satisfice; Kahneman's work (Kahneman 2002) showed that even if individuals seek to be rational and to optimise, we have limited cognitive capabilities to do so.

- *All participants in a market place have perfect information*. But as Stiglitz (2008) has shown, information both has costs and information disparities affect the outcome. Perfect information is an impossible condition; an example here is that it implies that you have read every book on economics and remember every one.
- *There is no cost incurred in making any exchange*. But Coase (1937) pointed out that firms exist precisely because it is cheaper to perform some functions inhouse rather than negotiate to buy them in the market. Hence, Coase (1991) asserted that the costs of making exchanges, the transaction costs, have to be a central concern of economics because they can determine which form of intervention has the lowest overall costs. For example, if it costs more to install, maintain and read monitors for some use or pollution than the value of the resulting reduction in the demand for that resource or the reduction in pollution, then it is not, from an economic perspective, efficient to adopt monitoring. The relative transaction costs thus can be crucial in deciding either the form of the EPI to adopt or whether to adopt the EPI in preference to another form of intervention. For example, it would be unlikely to be unrealistic to consider an EPI to limit aquifer drawdown if this required that the depression cone of every well to be monitored. Economic analyses which compare an EPI to a command and control intervention and assume the certibus paribus condition, are making the assumption that there is no difference in transaction costs.
- *A perfectly competitive equilibrium can exist and is desirable*. The conditions under which a perfectly competitive equilibrium could exist are highly restrictive (Carrie and Steedman 1990). In turn, Lipsey and Lancaster (1956-57) pointed out than when there are several significant divergences from those that would exist under perfectly competitive conditions, addressing one divergence can simply make other things worse.
- *An equilibrium condition is desirable.* In seeking to make a shift to sustainable development, the desire to induce change. Learning and innovation concern change. Whilst it may be that in the very long run, sustainable development is an equilibrium condition in which no further innovation or learning occur, the near present is one constant becoming. We do not want an equilibrium yet.
- *Marginal costs rise and fixed costs, notably the costs of capital, can be ignored*. Marshall's (1920) classic scissors diagram on which the argument for the adoption of marginal cost pricing rests ignores the question of whether the Producer Surplus is sufficient to cover the fixed costs of production. Ramsey (1927) pointed out that marginal cost pricing only results in sufficient revenue to cover costs if marginal costs are greater than average costs. In water management, in addition, once a capital investment has been made, the marginal costs of increasing production up to the limits of capacity of that capital works are often constant or falling.
- *Whilst there is an economics of competition, the economics of cooperation are less developed* (North 1990). The economies of scale often present mean that the choice

in water management is also often to determine when competition or cooperation is likely to be the best means of inducing efficiency.

Economic analyses in the past have also sometimes failed to recognise the specific characteristics of water and water management. For example, to make the assumption that what works for electricity, and other forms of electromagnetic radiation, or for gas, is equally applicable to water management. The similarities (e.g. capital intensity, economies of scale) must not be allowed to camouflage the differences:

– Catchments and aquifers are systems where the critical feature of a system is the interactions between components. In consequence, change at any point can be expected to have effects elsewhere. Water is also closely coupled to many other systems e.g. agriculture and food production.
– Water is heavy and incompressible.
– Water has to be allocated rather produced.

Since a basic economic assumption has been that individuals are rational and optimising, and that markets also optimise, it has been possible to assume that prices always work. In turn, it has therefore been unnecessary to develop to explain how and when prices, or other incentives, work.

3.2 Institutional design and sequencing

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3.2.1 Introduction

The EPI-WATER project developed and used a multi-dimensional Assessment Framework (AF) to evaluate the past (ex-post) and expected (ex-ante) performance of Economic Policy Instruments (EPIs) on achieving water policy goals. The evaluation process of existing and innovative instruments took place under different environmental, socio-economic, cultural and institutional conditions. The systematic evaluation of EPI performance implemented under this AF paved the way to understand under what conditions EPIs (or their delivery mechanisms) and the policy mixes embedding them work or fail, and these results were used to improve the design of existing EPIs or to develop and implement new EPIs suitable to local conditions (Zetland et al. 2013). This paper focuses on the interaction between policy design and the institutional setup and discusses how different EPIs performed under these conditions. Moreover, it outlines the importance of institutional change and sequencing of policy interventions to make EPIs more efficient while socially acceptable.

EPIs introduce incentives purposely designed with the objective of aligning the decisions of individual water users with the overall objectives of water policy (Gómez et al. 2013). Besides stimulating an efficient and equitable allocation of water resources and thereby minimising the overall economic cost of reaching a pre-defined policy objective, EPIs can deliver various additional objectives: i) to generate revenues to maintain and improve water provision (i.e. cost-recovery, earmarked subsidies); ii) to foster water conservation; and iii) to create a permanent incentive for technological innovation (dynamic efficiency). Strosser et al. (2013) stated that if an EPI is properly designed, then it must result in changes in the use of water in accordance to the provisions defined by the WFD, e.g. reducing water abstraction and water consumption by adapting practices and production processes; reducing the use and discharge of polluting substances into the aquatic environment; decreasing or halting hydro-morphological alterations originating from specific economic and land development activities. The authors concluded that this change in water use would eventually result in a change in the status of aquatic ecosystems, therefore contributing to the achievement of the environmental objectives of the water *acquis communautaire*.

Noteworthy, economic instruments operate within the given administrative and legal contexts. Thus policy performance is to a large extent governed by the fit between the policy design and the institutional set-up; in other words, by the feedback loop from implementation to design and vice versa. Therefore it is of great importance to examine the performance of institutions on different EPIs and to explore institutional factors that may determine the success or failure of economic instruments as part of a complex policy mix.

3.2.2 Institutions: theories and importance for water management

Institutions are the formal rules and informal norms that define and modify the choice sets of individuals and their interactions by affecting the cost of exchange (transaction costs) and production (transformation costs) (North, 1990, Saleth and Dinar, 2004, Zetland et al. 2012). Most institutions are difficult to describe, highly adapted to local conditions, and with a purpose of balancing many competing interests. Institutional constraints vary in strength, according to their permanence (from culture and religion to constitutions, laws, rules and regulations) (Zetland et al. 2013). It is worth mentioning some institutional theories that can allow us to understand how the institutions evolve and, therefore, how they may affect the evaluation and implementation of existing or new EPIs.

Institutional reforms can be evaluated using the 4-level analysis of institutions developed by Williamson (2000) where informal, formal institutional environments and institutions of governance are distinguished. The first (top) level (Level 1) analysis of institutions defines the informal institutional environment that includes customs, traditions and norms, which change very slowly. The next level (Level 2) sets the "formal rules of the game", i.e. the formal institutional environment, which includes the constitution, the legal system, judiciary, polity, and property and contract rights. "The play of the game" (Level 3) is the economic organization of contracts and governance structures, and more generally of managing transaction costs and seeing economic activity through to completion. Level 4 is the level at which neo-classical analysis works e.g. evolution of resource allocation and employment, and changes continuously. For instance, well-defined property rights (Level 2) are a pre-requisite for the implementation of water markets (Level 4) to avoid third-party effects (Strosser et al. 2013). In addition to this, for the institutions to perform properly or the good governance of common-pool resources require the fulfilment of several principles such as the ones developed by Ostrom (1990). These principles are mainly related to the clear definition of boundaries of a resource system, collective action, monitoring, sanctions and conflict adjudication mechanisms. Ostrom (1990) argued that the participants of a resource system make their own rules (collective action agreements) that are imposed and monitored by local users (monitoring). Punishments need to be employed in the event of breaking up the rules (graduated sanctions) that clearly define who is entitled to abstract water from a well-defined resource (clearly defined boundaries). Effectively assigning costs proportionate to benefits (proportional equivalence between benefits and costs) also needs to be taken into consideration.

If all the above principles converge in a resource system or common-pool resource then collective action and monitoring problems can be solved in a reinforcing manner. Following the concepts of North (1990), Williamson (2000) and Ostrom

(1990), Saleth and Dinar (2004 and 2005) provided a more specific definition of institutions related to water, i.e. water institutions. The authors defined water institutions as rules that altogether describe action situations, delineate action sets, provide incentives and determine outcomes both in individual and collective decisions related to water development, allocation, use and management. Within that framework, water institutions are unbundled into three categories: water law, water policy and water organization. The first category includes aspects related to water rights and conflict adjudication. The second category is related to policies regarding use priority, cost recovery and water transfers whereas the third category is associated to the structure of water administration, revenue collection, regulation/accountability and information and technological capacity. An evaluation of the linkages among these unbundled institutions and the institutional channels through which their impacts are conveyed is very valuable for appraising, developing reform design and implementation principles (Saleth and Dinar, 2005).

Finally, a major component of institutions to form and evolve adequately is to successfully address transaction costs. Managing natural resources (especially water) is a very complex societal issue that needs to involve legal, environmental, technological, financial and political considerations that are difficult to co-ordinate in an effective manner and often results in relevant transaction costs. While some transaction costs are strictly necessary and a part of successful processes (e.g., the approval of certain water trades by water authorities may be the only way to internalize environmental costs and prevent third-party effects from happening), others may hinder institutional and economic performance. For example, current decision-making systems have often resulted in political decisions overshadowing and prevailing over other considerations (Martin et al., 2008). In other words, the relevance of transaction costs (especially bargaining costs required to come to an acceptable agreement with all the parties involved) has been often magnified while that of environmental costs has been understated, thus delaying the necessary water policy reform. This follows a basic economic principle: as long as the transaction costs of the water policy reform are perceived to be larger than the opportunity costs of the status quo, the former will not be implemented (Dinar and Saleth, 1999).

3.2.3 Empirical evidence on water institutional performance and EPIs

The role of institutions is of paramount relevance when assessing the actual outcome delivered by EPIs. Institutional failure may hinder the performance of otherwise theoretically sound EPIs in accordance to mainstream neoclassical economics. This is the case for example of water markets in many regions worldwide. As a result, there is some controversy regarding the ability of water markets to encourage a more sustainable water use (i.e., its role as an economic instrument). For example, water trades very often transfer water from technically inefficient to technically efficient water uses with higher financial returns, but little attention (if at all) is paid to physical return flows. This means that water markets may result into higher water consumption, i.e., negative environmental outcomes. Although river basin authorities in some cases may reject water transfers if they reduce environmental flows or damage water ecosystems (USA, Spain), in Spain the rule of positive administrative silence applies (BOE, 2003), which is a major flaw in the Spanish system.

In Spain voluntary transfers of water use rights among existing right holders occurred to allocate water from low to high value uses. Water reforms allowed the introduction of the so-called water right lease contracts and exchange centres or water banks that eased certain transfers of water rights for a given period of time including a pecuniary compensation (Gómez et al., 2011). Water transfers required the approval of the administration either through a formal administrative resolution (time consuming) or through positive silence (time consuming and potentially negative for the environment). In Australia, unbundling of the water licencing regime allowed people to hold water licences without requiring any land ownership . Subsequent reforms defined water licenses as shares and issued them in perpetuity, such as in the Chilean system. Separate bank-like water accounts were then set up to record the amount of water allocated to each shareholder and track use and sales of that water (Young, 2011). Further reforms also addressed the over-allocation problems created by the development of water markets with a purpose of guaranteeing that environmental assets received an appropriate allocation of water for regeneration e.g. government buybacks entitlements for water allocation for the environment (Sharma, 2012). In Colorado, within the Northern Colorado Water Conservation District (NCWCD), water use rights became tradable with no priority. The amounts of transfers were annually collected and then classified by size and nature of seller and buyer (e.g. agricultural to urban or agricultural to agricultural), whereas the approval of water transfers did not have to pass through the water court but rather required only the approval of the NCWCD board (Howe, 2011). In Chile, Individuals were given permanent transferable water-use rights of “national property for public use” but those rights were not specified, secured and registered in a way that facilitated trading (Donoso, 2011).

Further examples on how institutions performed on different EPIs in terms of water quantity and quality issues are provided below illustrating the linkages between government, stakeholders and society to achieve water policy goals. In the irrigation district of Tarabina within the region of Emilia Romagna in Italy, the cooperation between national and regional organizations facilitated the introduction of volumetric pricing in agriculture resulting in a remarkable reduction in tariffs for non-irrigators and therefore in an improved water allocation among farmers and overall water use with respect to the previous flat-rate tariff. Increased scarcity and higher marginal provision costs can show the inefficiency of flat rates. If water expenditure becomes relevant in household and farm budgets, the most efficient users may have the incentive to highlight their responsibility through, for example, the installation of a metering device or by accepting to pay a higher unit price in exchange of being charged for its real consumption rather than by the average consumption of all water users. Driven by equity concerns and individual incentives, the previous financial instrument might only become fairer, but also a real economic incentive with the ability to reduce water demand and improve its allocation in the economy. This voluntary agreement may actually help address equity issues and cost recovery. However, it should be noted that the assertion that higher water prices may reduce per se water use is debatable (Cornish and Perry, 2003; Cornish et al., 2004; Hellegers and Perry, 2006; Molle, 2001; Perry, 2005; Steenbergen et al., 2007).

Regarding urban water tariffs in Italy, the government decision to introduce performance indicators in the calculation of water tariff formula allowed improvements in the efficiency of the water network, however, the abrogative referendum in 2011 has questioned the remuneration of capital investment into water infrastructure. In California, the cooperation between the state and local institutions, private and public, allowed the introduction, evaluation and further recommendations of the Water Budget Rate Structure, to improve the efficiency of water use in new and existing urban irrigated landscapes. Although it is generally regarded as a success, its applicability is heavily burdened by information availability and monitoring costs (Lago et al. 2012).

In Germany, the government of Baden-Württemberg, in cooperation with relevant water stakeholders, initiated a program to monitor groundwater quality. Water utilities developed their own groundwater quality database in order to increase transparency on water quality levels and monitor and assess the impact of the measures taken to improve groundwater quality (Möller-Gulland and Lago, 2011). In France, the implementation of financial compensations for environmental services was facilitated by several institutional reforms. These reforms resulted in putting stakeholders together to think about sustainable land management; reinforcing the partnership between farmers and the water company; and in helping to switch from "ready-to-use" solutions to solutions compatible with the conservation of traditional agriculture based on quality products (DeFrance, 2011). However, in other cases in Europe such as Denmark, the implementation of an EPI, i.e. pesticides tax, was not followed by a supportive institutional setting. Farmers opposed the tax arguing that it would weaken the competitive position of Danish agriculture while some political parties argued that they were against allowing polluters to pay for their actions rather than to ban dangerous pesticides (Pedersen et al. 2011). Finally, as far as the restoration of river ecosystems is concerned, institutional reforms in Switzerland and Italy were also subject to the evolution of the values of society and subsequent debates, particularly around the desirability of a mature technology such as hydropower generation (Garzón, 2012).

3.2.4 Institutional changes for the efficient implementation of EPIs

The analysis of the institutional setup for the efficient implementation of EPIs is of paramount importance at least for three reasons:

- This analysis creates the conditions to assess whether the enabling factors required are already in place or to what extent institutional change must be somehow modified to make EPIs implementable in an equal and efficient manner.

- It helps identify whether, further to institutional change, the design of EPIs itself needs to be refined so as to enhance their efficiency and reduce their implementation costs.
- Both the institutional change and the potential effectiveness of EPIs are also contingent on their social acceptability. This essentially refers to the social perception of whether innovative EPIs proposed for the reform of the policy mix are a real break up with status quo or rather a soft adaptation.

As already mentioned above, water policy reform is only to occur when transaction costs involved are lower than the opportunity cost (i.e., foregone benefits) of sticking to the status quo. The inertia of path dependence can only be broken should all those costs and benefits be evident.

As in Garrick et al. (2013), institutional change is therefore driven by efficiency but too often the process is far from being all of a sudden, not even smooth. Institutional change is about getting over failures and overcoming a number of obstacles such as the vested interests of major water users or political contest based on regional sensitivities or environmental concerns.

The introduction of EPIs is part of a transition in water policy. This may come about under different (and non-excludable) forms:

- Through improving the EPI design, so as to reduce direct transaction costs while guaranteeing its effectiveness.
- Streamlining and sequencing their implementation strategy, in order to minimize other transaction costs linked to the institutional setup.
- Designing better strategies to avoid institutional lock-ins, focusing on social and political acceptability.
- Making the most out of synergies between different incentives and also between EPIs and other policy instruments (information, command and control, etc.)

3.2.5 Sequencing

Appropriate sequencing of policy interventions (reforms or instruments) is critical and may have a considerable bearing on the success of any adjustment program (Montiel and Angor, 1996:500 cited in Esteban and Dinar, 2013). The costs of policy implementation are largely determined by transaction costs and may vary significantly according to the order followed in the whole reform process, which may contribute to pave the way for a smooth transition or alternatively complicate the process. Batie (2008 cited in McCann, 2013) indicates that adaptive management may be helpful to minimize transaction costs during a reform process; a policy is implemented, the results are observed and then adjustments are made. Moreover, the sequential linkages between institutional aspects (e.g. user organization, cost recovery, water rights, accountability and conflict resolution) can be used to enhance the prospects for upstream and downstream institutional changes by exploiting the scale economy benefits and path dependency properties (Saleth and Dinar, 2005).

The reform sequencing can take different forms, from national to regional level or even across groups (e.g. irrigators, urban, industrial).

Several insightful lessons can be learned looking at international experience on reform sequencing In the Murray-Darling River Basin in Australia the introduction of water markets is following a nationwide adaptive management approach comprising sequential steps, from well-defined property rights in the first stage (i.e. unbundling water licence) to securing water for the environment in the more recent policy reforms, e.g. through buyback programmes (Young and McColl, 2002 and 2003; Crase, 2012; Adamson and Loch, 2013).

Another example of reform sequencing in water markets in over-allocated basins can be found in Chile, where the introduction of a ceiling (i.e., cap) to limit water diversions in 1994 was followed by water right reforms to (partially) take into account third-party effects. In Morocco, reform sequencing is perceivable both in the sectoral focus of reforms (since irrigation reforms occurred after the consolidation of urban sector reforms) as well as in the regional focus (since areas under larger-scale public irrigation have undergone reforms far ahead of those privately managed and reliant on groundwater) (Doukkali, 2005 cited in Saleth and Dinar, 2005). Another example of reform sequencing comes from South Africa, where the legal and policy reforms at a national level preceded the organizational reforms, especially those at the regional and local levels (Samad, 2005 cited in Saleth and Dinar, ibid.).

There are also ex-ante studies that assess the impact of sequencing in policy reform. For example, Estevan and Dinar (2013) assess the performance of an alternative quota and tax sequencing in the Western La Mancha Aquifer. In their simulation study water quota was selected during the first 10 years, shifted to the water tax from year 10 to 15 and finally shifted back to the quota in year 16 through the end of the planning horizon. Their conclusions were twofold. First, the sequential package of the examined policy instruments proved to be more efficient compared to the implementation of individual policy instruments. Second, timing is important to make the sequential package happen. The sequential package of quota and tax was the longest to reach a steady state and the least stable along the planning horizon path (Esteban and Dinar, 2013).

Finally, another ex-ante assessment of economic policy instruments sequencing comes from the interconnected Tagus and Segura river basins (Gómez et al., 2013). The authors consider three instruments, namely, pricing for water security, water trading and drought insurance. The implementation of these instruments demands some important changes in the regulations in place, and therefore phasing up the legal reform and sequencing the setting up of these economic policy instruments might be the key to water reform success. As an example, there is no provision in the current financial legislation to put a price on the means chosen in their study to provide water supply security. When security is provided by additional infrastructures cost recovery provisions are enough to finance the system but this is not sufficient to finance alternative security means such as demand management, maintaining excess capacity or promoting the recovery of groundwater bodies. In addition, in particular for water pricing and trading, the reform also needs a stepwise approach in order to gain broad political and social support. For example, some small-scale projects would help demonstrate that expected benefits in avoiding overexploitation are for real and not just another alibi for rent seeking.

Furthermore, starting in some properly chosen locations and gradually extending the system as "learning by doing" can gradually increase the value of the insurance system and its effectiveness and foster network economies to make the option affordable to other crops and areas. In the particular case of drought insurance, experience shows that the best candidates are those areas where farmers have fewer options to change decisions in dry periods, with ligneous rather temporary crops, profits and then willingness to pay (WTP) for water security are higher, as in the many parts of the Segura River basin (SRB), and groundwater sources are already over exploited, so that the demand for outside insurance is higher. Success in this scenario may be the key to demonstrate how the drought insurance may be in the best interest of farmers and might also work to allow the recovery of water sources.

3.2.6 Conclusions

This paper focused on the linkages between policy design and the institutional setup and discussed how different EPIs performed under these conditions. Empirical evidence underlined the important role of institutions on the performance of EPIs to achieve water policy goals, for instance water reforms to facilitate voluntary water trading so that water is allocated to the most high value uses without major environmental third-party effects (e.g. in Colorado). The importance of the institutional change to make EPIs more efficient while socially acceptable was also emphasized, for instance, in the case of over-allocated basins where water rights reforms were needed to deal with significant environmental externalities (e.g. in Chile and Australia). Future institutional changes need to account for non-environmental third-party effects. For example, local Tagus river basin district (Central Spain and Portugal) agro-industry will suffer the consequences of water markets between the Tagus and the Segura (SE Spain). In the 2006 drought in Castile and León (Central and Northern Spain), for example, water restrictions reduced Gross Value Added (GVA) in this industry by more than 4%. Appropriate sequencing of policy instruments matters. It was underlined that the costs of policy implementation, largely determined by transaction costs, may vary significantly according to the order followed in the whole reform process and a wrong timing may even make impossible some reforms. In this case, adaptive management can play an importance role to minimize transaction costs; a policy is implemented, the results are observed and then adjustments are made. Obviously, the costs of this adaptive management approach may considerably change depending on the scope of the reform (e.g., from local adaptive management in Spanish agricultural insurance to nationwide adaptive management in Australian markets). Therefore, our analysis implies that policy makers need to understand the dynamic link between economics and institutions in the design and implementation of an EPI to achieve water policy goals. McCann (2013) stated that ultimately, policy choice and policy design needs to be matched to the specific physical and institutional characteristics of the problem. Thus, EPIs can be more efficient and socially accepted if institutions, institutional changes and sequencing of policy interventions are appropriately taken into account in the whole process. At the end of the day water policy (or even policy itself) is not only about what but also definitely about how.

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3.3 Integration of EPIs with policy instruments adopted in other sectors such as agriculture and energy

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3.3.1 Introduction

Water protection has always been central to the environmental protection efforts of the European Union (EU). Since the early 1970s several legislation have been passed to protect European waters, with the Water Framework Directive (WFD) (EC, 2000) as the latest major piece. However, there is still room for improvement in the quality of aquatic ecosystems as evidence shows. The EEA Water assessment reports and the Commission assessment of the Member States' RBMPs (EC, 2012a) developed under the Water Framework Directive assumes that the WFD objective of "good ecological state" is likely to be achieved in slightly over half (53 %) of EU waters. The "Blueprint to safeguard European waters" (EC, 2012b) therefore concludes that major additional action is needed to preserve and improve EU waters.

The potential of economic instruments to support the achievement of the WFD is mostly underexplored in Member States yet. In order to better understand why this is the case DG Research funded the EU-wide research project EPI-Water (standing for: Evaluating Economic Policy Instruments for Sustainable Water Management in Europe) . Launched in January 2011 for a three-year period, its main aim is to assess the effectiveness and the efficiency of Economic Policy Instruments (EPIs) in achieving water policy goals. Based on its findings policy recommendations to EU and Member States officers for how to design and use outcome-oriented EPIs should be made.

Following a review of the literature, the EPI-Water consortium defines EPIs as incentives designed and implemented with the purpose of adapting individual decisions to collectively agreed goals. EPIs for sustainable water management are consequently designed and implemented both to induce some desired changes in the behaviour of all water users in the economy (being individuals, firms or collective stakeholders) and to make a real contribution to water policy objectives, in particular reaching the environmental objectives of the EU Water Framework Directive, at least cost for society.

Box 1: Definition of what is an EPI (Delacamara, et al, 2013)

The project thereby builds on the work developed in 30 ex-post assessments of EPIs in Europe and around the world, and in-depth ex-ante assessments of the viability and expected outcome of EPIs in five EU areas facing different water management challenges (flood risk and waterlogging in Hungary, water scarcity and drought risk in Spain, biodiversity and ecosystem service provision in France, water scarcity in Greece and water quality in Denmark).

While a wide set of conclusions and recommendations can be made from the project , the specific aim of the paper is to show policy makers what the main barriers and success factors are to use an EPI for policy integration of water management into other sectors such as energy or agriculture.

3.3.2 Policy expectations on EPI´s in regard to integration of water management into other sectors

Environmental Policy integration - What is it about

Environmental policy integration can be seen as a policy process that aims to place environmental considerations at the heart of decision-making in other sectoral policies, such as energy agriculture or industry, rather than leaving them to be pursued separately through purely environmental policy instruments.

Environmental policy integration as a part of sustainable development is commonly understood as balancing environmental interest against economic and social interests and policies in a way that trade-offs (or negative effects) between them are minimised and synergies (or win-win-win opportunities) maximised (Berger and Steuer, 2009). From an environmental perspective integrating common policy aims should result in positive environmental outcomes, economic efficiency and further equity. Also the issue of burden sharing is a relevant one in particular in relation to the polluter pays principle.

Since 1997, environmental policy integration is a requirement under the EC Treaty. Article 6 of the Treaty states that "*environmental protection requirements must be integrated into the definition and implementation of the Community policies [.] in particular with a view to promoting sustainable development*".

Accordingly, EU SD strategy's policy guiding principles (EC, 2006) also emphasise the need to "*promote coherence between all European Union policies and coherence between local, regional, national and global actions in order to enhance their contribution to sustainable development*". Since the integration of policies between different governments should proceed in a cross-sectoral manner, the concepts of horizontal and vertical integration are obviously closely related as shown in the Figure 7.

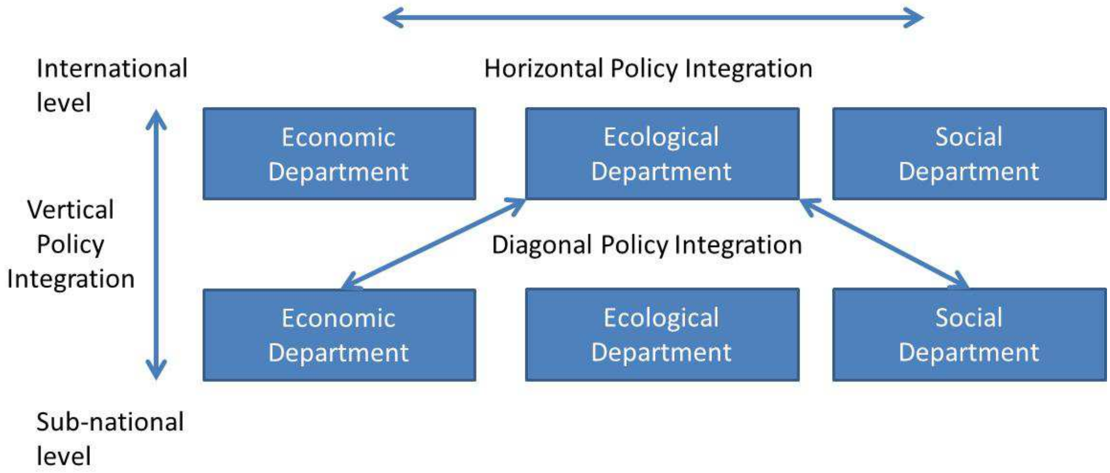

Figure 7: Horizontal, vertical and diagonal policy integration (source Steurer, 2008)

When discussing environmental policy integration it is important to consider two aspects:

– The integration of 'environmental objectives values and norms, ecological capacities, and codes of ecologically good conduct into the political and administrative policy-making process of sectoral agencies and authorities; and,
– Giving 'environmental concerns' specific weight or preference through political decisions at the highest level of authority, and communicating and implementing them into the political and administrative decision-making process of sectoral agencies and authorities".

In Europe the first step is mostly performed (based on the requirement of the treaty) but the second step often lacks in particular on the local level due to lack of local support and commitment as well as local public resistance. In addition the lack of financial or human resources might result in an implementation lack[16]. However this local level is (and in particular in relation to water management as several water management measures have to be implemented on that level) a crucial factor for achieving full integration. Due to their proximity to the citizens and other important stakeholders, local communities as well as regions are able to tailor their action to people's needs, and to strategically link different areas of policy. Local governments are therefore best placed to make sustainable development a practical reality. However this challenge is often not met and integration (in particular when it comes to the spatial development perspective or land-use planning) is not achieved.

For water the main instrument for policy integration is **Integrated Water Resources Management** (**IWRM**), which can be seen as "a process which promotes the coordinated development and management of water, land and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems" (Rahaman and Varis, 2005). As such integrated water management needs to deal with several sector (such

[16] This experience was also made in the WP4 case studies in the context of EPI-Water.

as agriculture, energy, tourism, industry, fishing, transport) activities in order to achieve the goal of sustainability of vital ecosystems.

The role of EPIs as a tool for integration

EPIs can have different targets such as cost recovery, demand management or changing behaviour (Delacamara , et al, 2013). The role as a tool for policy integration was first mentioned at the EU level in the 2000 Communication "Pricing policies for enhancing the sustainability of water resources" of the European Commission (EC, 2000). There in chapter 3.7 "*Water pricing and other policy initiatives of the European Union*" the following is stated: "*Co-ordination and synergy between water pricing and other policy domains of the European Union are key elements for economic and environmental effectiveness. Several policy areas are clearly relevant in this regard.*" The Communication then further refers in particular to the agricultural policy and Structural and Cohesion policies. Energy policies were not of so much concern at this stage.

In March 2007 the first WFD implementation report was released (EC, 2007a). This report addresses three aspects: the transposition into national law, the set-up of administrative structures (Article 3 WFD) and the environmental and economic analysis of river basin districts (Article 5 WFD). This report emplaced Member States by 2009 to "*put in place all the economic instruments required by the Directive (pricing, recovery of costs of water services, environment and resource costs, and the polluter pays principle). Full exploitation of these economic instruments will contribute to truly sustainable water management.*" Considering the definition in Box 1 pricing can be considered as the only EPI. Full cost pricing takes account of the polluter pays principle through the recovery of all economic costs; including financial, resource (or opportunity) and environmental costs related with water services provision.

In 2007 the Communication "addressing the challenge of water scarcity and droughts in the European Union from the Commission" the following aim at national level, by 2010 was set (EC, 2007b): "*Put in place water tariffs based on a consistent economic assessment of water uses and water value, with adequate incentives to use water resources efficiently and an adequate contribution of the different water uses to the recovery of the costs of water services, in compliance with WFD requirements. The 'user pays' principle needs to become the rule, regardless of where the water comes from.*"

In 2007 also the "Green Paper on market-based instruments for environment and related policy purposes" (EC 2007c) was released and the EU Commission considered such instruments as an important tool for implementing environmental concerns across other sectors. For water it is stated: "*How can the Commission most effectively ensure implementation of the water pricing policies set out in the Water Framework Directive? What options could be explored to reinforce the links between investments in national water projects and the introduction of corresponding water pricing to provide incentives for users and avoid distorting competition?*"

Twelve years after the 2000 Communication of the European Commission the issue of integration still remains a pending one as shown in the Blueprint (EC 2012b):

"However, there is a need for better implementation and increased integration of water policy objectives into other policy areas, such as the Common Agriculture Policy (CAP), the Cohesion and Structural Funds and the policies on renewable energy, transport and integrated disaster management. The reasons for the currently insufficient levels of implementation and integration are complex…. They consist of a series of water management problems related to the insufficient use of economic instruments, lack of support for specific measures, poor governance and knowledge gaps. Only in a minority of cases have gaps been identified that would require the completion of the current framework by new action of a legislative/legal nature."

Finally published in April 2013, the EU strategy on climate change adaptation (EC, 2013a) is aimed at encouraging the development and implementation of adaptation action by the MS, with special focus on win-win, low-cost and no-regret options. Under such category of options, the document includes sustainable water management. As one of the actions to be promoted under its adaptation strategy, the Commission included the expansion of insurance and other financial products in the context of natural and man-made disasters. This is expected to enhance the resilience of the European economy in the face of climate change. Lastly, the communication also touches upon the importance of financing adaptation action and notes the availability of funding streams at national and EU level that support drought management implementations.

The Communication was accompanied by a Green Paper on disaster insurance (EC, 2013b). The Green Paper poses a number of questions concerning the adequacy and availability of appropriate disaster insurance (such as flood insurance). The objective is to raise awareness and to assess whether or not action at EU level could be appropriate or warranted to improve the market for disaster insurance in the European Union.

In other words since several years economic instruments are considered as a suitable tool for integrating water management concern into other policy areas. However the question why this has not been achieved still prevails. Lessons learned from a dozen of case studies from EPI-water can be used to better show why this gap still exists and what is needed to achieve a higher level of integration resulting in more sustainable water use and more integrated water management.

3.3.3 What is the potential for EPI in terms of integration

Water policy is ultimately about making economic development and social welfare enhancement compatible with the improvement and protection of water resources. Water and aquatic ecosystems provide the economy with flows of water services or primary goods for the production of a very wide array of valuable goods and services such as drinking water, biomass (either for food or energy production), power, manufactured goods, recreational services, etc. The quantity, the quality of all these water services as well as its stable delivery actually depend on the state of conservation of all those ecosystems.

None of these water services, though, can be provided without a pernicious effect on these ecosystems. Water abstractions, impoundments, diversions, etc. are a binding requirement in most cases. Besides providing water services for the production of goods and services for the economy, water-related ecosystems provide a myriad set of critical environmental services, which are essential for human welfare and for the ceaseless functioning of the economy.

Contemporary water policy objectives are therefore defined in terms of a desired status of conservation of these water-related ecosystems. The choice of the appropriate policy instruments is thus based upon their ability to adapt the functioning of the economy to these goals.

Economic instruments are just but one kind of the different alternative means available to the ends of water policy. The essential characteristic of an EPI is that it is an incentive deliberately designed and implemented in order to make individual economic decisions compatible with some policy goal. **Economic instruments for sustainable water management, as considered in EPI-WATER, if consequently designed and implemented both to induce some desired changes in the behaviour of all sectoral water users** (being them individuals, firms or collective stakeholders) **and to make a real contribution to the implementation and achievement of collectively agreed water policy objectives**.

More concretely economic policy instruments can play different functions in the process of integration. Based on the work in the EPI-WATER the following functions have been identified:

- To implement the polluter/user pays principle. In other words if economic costs reflect real resource costs the polluter/user competitiveness is increasing if these costs (and therewith the use of water) are kept low. For example the large majority of EU domestic / manufacturing water facilities nowadays is facing waster pricing which has the aim of rationalizing water uses and allowing for the application of the polluter / user pays principle. However it is rarely used in the agricultural sector. Application of pricing at national and local level can be found in e.g. Hungary, Netherlands, UK, Italy, or Cyprus.
- To implement full cost recovery. For instance, in seeking to incorporate the environmental and resource cost requirement to rural development policy, the Commission has proposed water pricing and cost-recovery to be a pre-requisite to the allocation of funds from such plans (EC, 2012b).
- Be part of risk management: 1) Agricultural insurance is seen as an important instrument to help farmers manage risks associated with production. Given the uncertainties concerning the incidence of droughts; residual production and income risks would still persist despite the application of water markets or smart pricing (Volaro et al., 2013). Ultimately, without compensation mechanisms that cover total income losses faced by farmers due to adverse weather conditions, the real insurance system for income stabilization is often found in alternative water resources. Illegal water abstractions, especially from aquifers (as an example in Spain), offer a source of income coverage from the production losses that are not

compensated for during a period of droughts. As a criticism to the current regulatory framework in Spain these sources of water are scarcely controlled and rarely pursued and punished (Gómez and Pérez, 2012).

2) Payments have been widely used in the UK, Germany and the Netherlands notably for temporary flood storage on agricultural land. Lump-sum and annual payments for creating flood storage are becoming more frequent across Europe, usually on a project basis but also through more established programmes such as payments for natural flood management via the Scottish RDP.

– Triggering innovation. For example the efforts made under the European Innovation Partnership (EIP) on Water will also be crosscutting and will aim for integrated solutions that can address water scarcity issues in urban and rural areas. In this line the pricing mechanism proposed in the WFD could be expanded into a *smart pricing* system that factors issues like current climatic conditions, geographic location and productivity of water use types into the price of water services.
– Increase cooperation across sectors. For example payments for ecosystem services (PES) schemes are cooperative agreements based on voluntary transactions between at least two social actors with the aim of securing the provision of ecosystem services (ES) (e.g. clean water supply, flood risk mitigation, etc). Most PES schemes involve the buying of an ecosystem service (by e.g. a drinking water company) through maintaining a specific land use (by e.g. the agricultural sector) or securing a land use change that will produce that service.

3.3.4 *Main barriers and success factors for integration*

Across case studies within the EU, synergies were found between the EPI and the Water Framework Directive. The impact of Energy related EU policies on the EPI implementation needs to be viewed in a more differentiated manner. Generally it can be said that EU policies, such as the Renewable Energy Directive, showed synergies with EPIs addressing Hydropower (e.g. Mattheiß, 2011; Mysiak et al., 2011b). Contrary, barriers between the EU Renewable Energy Directive and the German Atomic Energy Act were identified for the implementation of EPIs addressing diffuse pollution from agriculture. These directives, by increasing demand for biofuels and thus changing market incentives, overrode the incentives provided by the "*Marktentlastungs- und Kulturlandschaftsausgleich*" (MEKA) and "*Schutzgebiets- und Ausgleichsverordnung*" (SchALVO) programs in Baden-Württemberg which were intended to reduce intense agricultural practices (Möller-Gulland and Lago, 2011). Additionally, incentives provided by the EU Common Agricultural Policy presented barriers to the successful implementation of EPIs targeting diffuse agricultural pollution. EU policies which target effluent quality from point sources, such as the IPPC and Urban Wastewater Directive, created synergies with the EPIs addressing point source pollution, such as the effluent tax in Germany (Möller-Gulland et al, 2011b). Based on this assessment of case studies and additional work in the WP 4 case studies the following key success and barriers for using an EPI as a tool of integration have been identified:

Barriers

Integrated Water Resources Management (IWRM) considers multiple viewpoints and dimensions of how water should be used in the sector, but also in other sectors. While for IWRM the sustainable use of water is a key issue other sectors have different viewpoints or conflicting objectives. When formulating these sector objectives there is often a negative trade off to the water sector challenging the concept of IWRM. For example the aim to increase the share of renewable sources within the energy mix indirectly implies a higher water use in agriculture (due to irrigation and increased pollution) or morphological destruction due to hydropower generation.

Barriers between the EU Renewable Energy Directive and the German Atomic Energy Act were identified for the implementation of EPIs addressing diffuse pollution from agriculture in Germany. These pieces of legislation, by increasing demand for biofuels and thus changing market incentives, overrode the incentives provided by the MEKA and SchALVO programs in Baden-Württemberg which were intended to reduce intense agricultural practices.

Box 2: Failure of integration: EU Renewable Energy Directive and the German Atomic Energy Act forcing intense agriculture (Möller-Gulland et al, 2011)

As the example shows some policy interventions (based on conflicting objectives) are **overriding the incentives provided by the water EPI[17].**

In addition several EPIs also do not have a clear objective and therefore integration is difficult to obtain. For example, the OECD (2010) provides a detailed discussion of potential tensions between four sets of objectives (economic, social, financial, environmental efficiency) in the case of tariffs for water supply and sanitation services. They are designed to suit multiple purposes (e.g. fixing budgets and mitigating negative impacts of water use) which is not always possible. In the context of the EPI-WATER research project, it was considered that environmental objectives were the priority since they have been placed as an overarching and cross sectoral policy goal by the WFD, while financial and development objectives remain instrumental. However as the example below shows this assumption is not always reality.

The shared perception of an EPIs cannot be easily identified. Clear instruments without any identifiable purpose (at least in what concerns water policy) are nothing more than a rarity. Some EPIs, for example, have been able to survive long after the obsolescence of the original objectives for which they were conceived. For example, the water load and the water resource fee in Hungary, which were already in place before Hungary's accession to the EU, and even to the economic downturn that came along the evolution from a centrally planned towards a "free" market economy. The survival of these instruments owes more to their convenience to raise public revenue rather than to the social and political commitment to improve water

[17] It should be noted that also the Common Agricultural Policy has been a long time over riding incentive. The decoupling of payments introduced in 2005 by the Common Agriculture Policy reform (CAP) likely helped in the reduction of the quantity of water used (see Mysiak et all, 2011a).

governance and preserve the environment

Box 3: Example that environmental objectives are not always the main focus of an EPI (Központ et al, 2011a;b)

There is no doubt that with the implementation of the WFD the conflict of policy objectives has become obvious (EC, 2012c) and the WFD mechanisms (e.g. Art 4.7) to balance them are widely used (e.g. new hydropower is built without using EPI´s to let the polluter pay for the environmental damage).

Another barrier for the limiting integrating impacts of EPI´s the enforcement and control of the EPI is often weak. Pressure from other sectors to not enforce and implement such mechanisms limit the effectiveness of the EPI.

In the Pinios RB in the last decade a copiousness of Regulations, Decisions, Laws, Circulars, Common Ministerial Decisions, etc. have been edited concerning the implementation of agricultural activities in the framework of the CAP, WFD and National Agricultural Policy. Yet, this legislative framework was not satisfactory implemented in the Pinios RB. All of the analysed interventions have a common pre-condition: controlling the illegal abstractions and applying a robust institutional setting. The governance framework is currently loose and many actors are involved in the water and irrigation management, resulting in an inability to control, monitor and enforce policy. In general the successful implementation of all the studied interventions and EPIs requires a robust early design which adequately considers all the necessary, preconditions, a targeted consultation with the stakeholders, a strong implementation strategy, built around a solid institutional setting and a continuous monitoring in order to be able to evaluate the actual performance of the policy implemented and redesign it accordingly.

Box 4: Weak implementation of existing safeguarding mechanisms (Kossida et al, 2013)

Economic instruments, as with any other policy instrument, are not without cost. In some cases, the transaction costs beard by other sectors may outweigh the benefit of the transaction, in which case the transaction may not occur, and the benefits of the economic instruments will not be achieved.

Transportation fees (which are considered as transaction costs) in the Tagus-Segura Water Transfer are 0.1 €/m3, while transportation losses are estimated at 10%. Bearing in mind just these two cost categories, the potential for water trading is reduced by 30 hm3 (10% reduction), along a price increase of 16%. The average technical efficiency in the irrigated areas of the TRBD connected to the Water Transfer is estimated at 39.9%, meaning that 60.1% of water is actually "lost" and either returns to the watercourse or evaporates. Return flows are estimated at 19%. Considering that ratio, the potential for water trading would be reduced by 19.6% (from 240 to 193 hm3 per annum), while prices would be 3.7% higher. Under precautionary principles (return flows at 60%), water-trading potential would fall by 65% and water prices would increase over 40%.

Box 5: Transaction costs as a barrier in intra-basin water trading (Gomez et al, 2013)

Finally another barrier identified is related to social acceptance and perception of EPI´s outside the water sector. Factors which create the perception of risk in the minds of those who develop an EPI or are political responsible for its implementation also produce scepticism and uncertainty at the stakeholder level.

For example an important political barrier to implement water markets is the fear that such markets will lead to the commodification of water (and not only the privatization of water use rights), making it accessible to whoever can actually pay for it, including through importation, irrespective of other social and environmental goals. As a matter of fact, as shown by the case study in the Tagus and Segura interconnected river basins (Spain), as long as some provisions to account for third-party effects are adopted, the potential of inter-basin water trades significantly decreases. Markets may then remain relevant at a local level.

Water prices and taxes raise the cost to industry and agriculture. Thus prompt fears of decreasing competitiveness even if water is only a small fraction of the budget (energy costs for pumping might be much higher).

Box 6: social acceptance and perception outside the water sector as a barrier (Gomez et al 2013)

All the issues mentioned above are often combined and reflect an insufficient design of an EPI.

Success factor

The main success factor for using an EPI as a tool for policy integration is to avoid or overcome the above mentioned barriers by designing the EPI right from the beginning. Guidance to do so is given in the EPI- Guidance document (Delacamara, et al, 2013). In addition some more success factors hen identified have been identified.

Earmarking (and labelling) are important instruments in selling water policies to other sectors and generating revenues

All discharges of effluent require a permit. This permit is issued only if the effluent to be discharged is kept as low as possible for the required process and with the best available technology. The effluent tax is based on these permits, rather than on actual measurements. The tax rate is based on damage units, which are calculated as the equivalents of pollutants in the discharged effluent.

The revenue of the effluent tax is earmarked for investments in water quality programs by the Länder, such as the construction of municipal sewage treatment and the administration of water quality programmes tax's incentive effect in improving water quality and implementing the Polluter Pays Principle.

Industries, such as the paper industry, developed production processes which required less wastewater development. Others, like the chemical industry, invested in effluent abatement measures and considerably reduced the discharge of pollutants. In other words the EPI has integrated water management into the sector of industry.

Box 7: Successful integration: "The effluent tax in Germany" by earmarking the revenues (Möller-Gulland et al, 2011b)

In Switzerland, the green hydropower standard was set in the late 90s. The Association to Promote Environmentally Friendly Electricity (VUE) was founded to develop a broadly accepted standard of quality for green electricity. In the summer of 2000, the label "naturemade" was publicly launched. The concept of Green labelling scheme has two main objectives: 1) Economic objective: to have a reliable and objective certification scheme that is trustfully accepted by the consumers and ensures fair competition on the market; and 2) Ecological objective: the improvement of local river conditions by setting an incentive to develop sustainable hydropower. The instrument contains two delivery mechanisms. The first one is a standard covering 45 scientifically defined criteria (some criteria include minimum flow regulations, hydro-peaking, reservoir management, bedload management, power plant design, etc.). The scheme allows a supra-regional comparable certification of different power plants, regardless of their age, size, or how they are built or operated. The second delivery mechanism consists of eco-investments defined as a fixed mark-up on every kilowatt-hour sold as green hydropower. On an annual basis, this surcharge must be re-invested in the river system in which the plant is located in the form of river restoration measures adapted to the demands of the individual river system

Box 8: Successful integration: "Green hydropower" by earmarking the revenues (Dworak, 2011)

Another relevant success factor is the issue of sequencing. According to the WP 4 case study in Tagus and Segura basins phasing up the legal reform and sequencing the setting up of the EPI might be the key to water reform success. Promoting good practices and substitution of water sources also needs a stepwise approach in order to gain broad political and social support. Sequencing the implementation also minimizes institutional transaction costs.

An important insight from the analysis is that EPIs are but one piece of the institutional change required in current water management practice. According to Ostrom (1992) the water governance challenge consists in finding a suitable non-coercive mechanism that motivates collective action. Finding the right policy mix (different EPI´s for water management, EPIs for different policies, command-and-control instruments, information) is crucial.

The German effluent tax is one piece of a policy mix, which also consists of discharge permits, pollution limits and mandatory technological standards. The policy mix has been mostly successful in obtaining its objectives but the real contribution of the effluent tax is impossible to single out. The tax is also based on permitted effluents both in volume and composition in such a way that incentives for further pollution reduction without technological change are missing. However, at least three complementary instruments may have played a significant role in reducing pollution and increasing the dependability of water quality targets. First, monitoring systems help verify that pollution limits are not surpassed and to set non-compliance fines that provide an incentive to stay within limit alues. Second, along the implementation process three-quarters of private enterprises and two-thirds of municipalities had increased, accelerated, or modified their abatement measures for water pollution in anticipation of the charge. Finally, although the role of the effluent charge to reduce pollution substantially faded once the prescribed limits were obtained, firms still have the option to prove they are below these limits and are subsequently eligible for a tax rebate

Box 9: EPI´s as part of a policy mix can ensure successful integration (Möller-Gulland et al, 2011b)

3.3.5 Recommendations for policy makers to make better use of EPI´s in the context of policy making

Based on the assessments made and the experiences gained in the EPI-water project the following recommendations to policy makers can be made:

- **Clearly express policy integration aspects as part of the objective of the EPI**. The EPI should have clear (environmental) objectives (and frameworks for implementation) and a clear focus on the sectors it addresses. In other words it should be made clear what the EPI is aiming for (e.g. reduced nitrate) and who will be addressed by the EPI (e.g. the agricultural sector only). Making this clear is important in the screening phase (see Delacamara, et al, 2013) to select the right EPI.
- **Integration in policy formulation and the design of the detailed mechanism**. This does not only referred to the EPI itself, but also to the surrounding policy mix which has been mentioned before. An example of a comprehensive framing is the checklist for "improving policy coherence and integration for sustainable development" developed by OECD (2002)[18] which can also be used in this context as well as the EPI-water Guidance document (Delacamara, et al, 2013). In particular attention should be paid to:
    - clear commitments and leadership. If this does not exist within the water sector, other sectors might stop the process or claim a design which reduces the environmental outcomes.
    - a high level of transparency as this often triggers discussions with other sectors building the foundation for better integration of water issues in other sectors. It also triggers a common understanding of the EPI across the affected sector
    - encourage stakeholder involvement in decision-making
    - the key barriers identified above are avoided in the design and the key success factors are applied.
    - that from the design the economic, social and environmental benefits justify the costs, the distributional effects are considered and the net benefits are maximised.
    - the use of adequate assessment methods when designing EPIs which evaluate their performance under different scenarios within different sectors. Thereby it needs to be ensured that the diversity of sectorial knowledge and the scientific input is adequately managed;
    - setting safeguards in such a way that would not impair the achievement of the environmental objectives
- **Ensure implementation and enforcement by government agencies and other actors in a cross policy way.** This should not only cover the fact that other sectors

[18] OECD (2002): Improving policy coherence and integration for sustainable development: A checklist, Paris: OECD.

consider/built on the EPI when developing their own policies, this also could cover aspects of shared implementation and responsibilities (e.g. the Nitrate Directive is often implemented by the agricultural ministries in EU- Member States and not by the environmental ministry), data and information sharing

– **Sequence the implementation of the EPI.** Sequencing can also reduce initial costs, gain political and market acceptance, and build trust through learning by doing. For example, sequencing the introduction of drought insurance may involve starting with the inclusion of permanent crops where exposure to risk is easier to control, and extending coverage to new crops and areas. A proper sequencing will reduce insurance firms' incentives to engage in rent seeking and regulatory capture and will link the development of the market to its own performance.

Allow exemptions or extensions of deadlines for sectors into which integration should take place in order to allow the sector to cope with the changes. However this may impede the functioning of the EPI and thus the achievement of the desired results

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3.4 Pricing Water Security

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3.4.1 Introduction

A water scarce river basin (or economy) is better defined as one that is about or has already exhausted the potential to harness available water sources to provide services for the different economic activities in which water is an essential input, such as households, the environment, irrigated agriculture, hydropower, the manufacturing and the services industries.

In Mediterranean countries in Southern Europe, but also in China, Australia, South Africa and some less developed countries, evidence shows that trends towards increased water scarcity come along with the obvious add-on of higher exposure to drought risk (OECD, 2013). These unsustainable trends may go on throughout time even when the overall amount of water used exceeds long-term average renewable resources available at the expense of further depletion of groundwater sources and the resultant deterioration of the water environment (Blanco-Gutiérrez et al., 2011; Pfeiffer and Lin, 2012).

This definition of water scarcity calls for a transition in the way water is governed that might take place once it is evident that relying on mobilizing additional freshwater to provide water security in the long term and resilience in the short and the medium term is quite unlikely.

The clear outcome for water policy is that in a water scarce region, economic development can only proceed and be made sustainable by means of managing resources already available. This should be done in such a way that progress in one activity or one area (let us say irrigation, urban development, tourism, etc.) needs to be offset by water savings in other areas (e.g. through reductions in water demand, higher water efficiency, water reallocation, replacement of conventional by desalinated or recycled water sources, etc.), rather than allowing for an increased supply by adding new freshwater resources (EEA, 2009).

Thus, in a water scarce economy, the role of water pricing in order to provide water security needs to be redefined. So far, in advanced societies, prices are expected to play an important although still limited role in recovering the costs of building and operating water supply infrastructures as well as the operational costs of delivering water services and wastewater collection and disposal services to households and firms (EEA, 2013).

These financial arrangements are of a limited use once the change in the means to build water security is recognised. In a water-stressed economy water security is conceived as the result of reducing water demand down to a level that can be sustainably covered with the existing resources, promoting savings and technical efficiency so that new economic activities can take place without compromising long-term sustainability and building the collective means to curb current scarcity trends and to develop the institutions and the assets that may provide resilience in dry periods.

3.4.2 Policy challenge

Identifying water challenges in a water scarce economy is at the same time a way to define the main priorities for water policy, by sorting out the minimum set of changes required to restrain water scarcity and to reduce drought risk, and a method to define the particular effectiveness criteria which may guide the design of the instruments intended for that purpose: pricing water security in our case.

This section is organised around answering very basic questions: what is the set of minimum changes required in water governance to reduce water scarcity and increase water security in the long term? According to our research these changes are as follows:

1. **Closing the river basin district**. Closing a river basin is not equivalent to closing water transfers to or from other basins. It means accepting that any further advance in water supply cannot be met with additional water resources from the river basin (Molle et al., 2010). It also means defining use rights that are available anytime, that can be contingent on rainfall and runoff, and putting in place the institutional arrangements to enforce these property rights (Falkenmark and Molden, 2008). By definition, closing the river basin implies that any new water use needs to be accommodated within available resources, which is equivalent to saying that increases in water use in one activity or place need to be outweighed by savings in the same or other activities or places or alternatively through additional resources from other (non-water scarce) basins or non-conventional resources. A river basin closure will convert available resources into a horizontal constraint for many relevant economic activities including urban development, irrigation, hydropower, manufacturing and services that need to be coordinated as part of water policy.
2. **Controlling scarcity trends**. Although it seems a natural consequence of closing the river basin, some transitional measures will be required to reduce the excessive use of water and come to terms with available renewable resources. This implies hindering groundwater over-drafting and even allowing for the replenishment of depleted aquifers by natural or artificial means and the recovery of minimum environmental flows (Balali et al., 2011; Batalla and Vericat, 2009; Truong, 2012; Tsur and Dinar, 1997).
3. **Building up collective water security**. In close connection with the two previous requirements some *ad-hoc* provisions need to be made in order to increase drought resilience. These include a clear strategy to identify the resources that are available during normal periods as well as those who will play the role of buffer

stocks in dry periods. Allocation of water resources also needs to be contingent on natural supply variability and the flexibility mechanisms required are to be in place in such a way that potential market and environmental damages are minimized. Unlike water abundant areas, in places under water stress leaving the response to droughts to emergency responses from the public sector and spontaneous and uncoordinated responses from water users is a way to increase water scarcity and drought vulnerability in the following periods (Gómez and Pérez-Blanco, 2012). Rather, security needs to be built in an anticipated and coordinated manner and the institutional framework will need to be adapted to allow for flexible decisions (EC, 2008b).

3.4.3 Opportunities

Putting a price on water, not even the right one, has always been a challenge. Nevertheless as water becomes scarcer and water users understand they cannot take for granted a reliable supply of the resource anymore they also become aware of the importance of having enough water at the right moment. In other words, scarcity comes along with increasing opportunities to price water in accordance to its increased economic value (EC, 2000; EEA, 2013). These are the main opportunities to price water security in a water scarce economy:

1. **Willingness to pay for water security**. This is particularly evident in regions where, as it is usual in the European Union, there is a hierarchy of uses that distributes the risks of any water shortage. In that case priority uses, such as household consumption, are relatively protected from droughts, while users considered as less important, such as farming, bear a more than proportional share of drought risk (EC, 2008b). In water scarce areas those who assume the risk will also be willing to engage in saving water, using more technically efficient devices to make their activities less vulnerable (Ward and Pulido-Velazquez, 2008). They are also willing to accept metering, as a means to signal that they are not responsible for using water irresponsibly and to press in favour of marginal pricing instead of cost-sharing schemes that are increasingly considered as unfair (Molle, 2009). Probably the best evidence of the increasing willingness to pay for water security can be found in the downward trend observed in water tables in many Southern Mediterranean river basins in Europe (Blanco-Gutiérrez et al., 2011).

2. **Managing the water portfolio.** When water is scarce the supply of water derives from a mix of different sources (runoff, inter-basin water transfers, groundwater of different qualities and accessible at different costs, recycled water of different qualities suitable for different uses, brackish and seawater, etc.). These resources differ in many respects but particularly in their financial costs and their reliability. Surface water is "financially" cheap but unreliable, groundwater is increasingly expensive, as water tables fall, and only reliable in the short term, and desalinated water in turn might be abundant, depending on the installed capacity in place, and reliable but "financially" expensive. Managing this water portfolio is one of

the most important opportunities to respond to the challenges of water policy. Assigning a role to any water source, in terms of provision of water and security, is a critical decision that cannot be simply left to customary decisions or to individual users. Individual water users will always prefer cheaper resources once they are available and will only accept expensive ones in case of need (OECD, 2013). Yet these rather obvious preferences can lead to important inconsistencies in the long term. Cheap resources will be overexploited in the short term and alternative sources will only be developed once other alternatives have become expensive enough (that is to say when freshwater sources have been sufficiently degraded). Managing the water portfolio implies not waiting for this scarcity trend to take place and advancing the building up of a sustainable water supply making the provision of water security cheaper in the short and medium term.

3. **Bridging the efficiency gap.** Even in water scarce regions important amounts of water are devoted to low productivity activities. This happens when institutions in place do not allow other alternative uses different than the prevalent ones, following the 'use it or lose it' principle (Rey et al., 2011). Under these conditions there might still be important gains to be drawn from enhancing the efficiency with which water is used in any place and activity as well as from allocating water to its more productive uses. The efficiency gap is a clear opportunity that can be used to obtain a combination of economic and environmental benefits at the same time by producing more without further environmental degradation or to obtain the same market values with less pressures over water ecosystems.

3.4.3 Design

Further to its contribution to cost recovery, the proposed reform in water pricing is meant to make pricing an actual mechanism to match water supply and demand (contributing to the river basin closure), and assigning each water source a price depending on its role in terms of the supplied quantity and its weight for water security in the short and the longer term.

The pricing of water security is introduced as an economic policy instrument (EPI) for water management through the use of a market mechanism to guarantee the existence of buffer stocks and to allow for the recovery of depleted aquifers as well as to reduce water demand on a current basis.

One essential ingredient in price design is the consideration of water security as a public good that must be paid for collectively. In other words, the value of well-preserved aquifers or of water recycling facilities able to provide the buffer stocks to smooth water supply during dry periods must be shared with all the users enjoying a contingent access to this resource, even if these alternatives are not used on a regular basis.

The pool is formed by water utilities (on behalf of households and other urban consumers) that in spite of having the use of water guaranteed by the hierarchy of uses in place, are also interested in paying for the cost of their own water security as a way to reduce conflicts with farmers and other (formally) low priority users. In addition, pricing water security may help mitigate uncertainty, long bargaining processes and transaction costs that characterize the achievement of urgent solutions to water shortages in dry periods (McCann, 2013).

All those water users willing to take part in this kind of collective insurance system must pay on a regular basis for the capital costs of maintaining current desalination plants in operation so that they can cover the deficits in due time. The first basic question is then by how much should the water price be increased in order to raise enough revenues to fund the capital cost of these desalination plants. As a first approach this analysis can be performed in the drinking water sector taking account of the main drivers behind water demand. That is to say, the expected effect of changes in prices over the amount of water demand, the positive effect of increases in income, the evolution of the scale of water consumption due to population change and to the expansion of other activities such as tourism, and so on.

Sharing the cost of water security is basically a political question that can be agreed between stakeholders. Those who contribute will receive in exchange privileged access in dry periods. The suggested design is meant to address a crucial issue: how much current water prices would need to be increased so as to guarantee water provision in dry periods? The suggested scheme works as a cost-sharing mechanism among those interested in having a secure water supply and there are many opportunities depending on the number and the variety of users joining the risk pool. All these alternatives can be assessed on the basis of prospective water demand models.

Box: Basic design of pricing security in the Segura River Basin

In the exercise performed in the Segura River Basin District (SRBD), charging households for the capital costs of desalination plants would result in an annual price increase rate of 0.72% during a cost-recovery period of 30 years and would have a negligible effect on household water demand (< 0.7% decrease). Urban water security increases in turn water availability and security in agriculture, which results in less income variability, stable employment and positive forward linkages to other economic sectors (i.e. agro-industry). This provides the rationale for sharing costs. Yet, while household demand is often inelastic, irrigated agriculture is more likely to suffer negative impacts from higher water prices. According to the revealed-preference model (RPM) built for the purposes of this analysis, this is not the case in the SRBD, where farmers show an inelastic demand curve up to a price of 0.4 €/m^3. Below this price, the impact of higher prices is absorbed by the gross margin with no negative effect on employment. Caveats apply, though, regarding spatial heterogeneity and equity concerns. It is also important to note that the replacement of overexploited groundwater resources with desalinated water would not be feasible in some areas, which makes the case for other EPIs to be implemented. Capital costs represent *circa* 20% of the production costs of desalination. Nevertheless, high variable costs are still a hindrance.

3.4.4 Potential

The main purpose of the pricing scheme is to progress towards a sound management of the water portfolio along the required transition towards a sustainable economy. This transition requires to have in mind a clear picture of what the future water portfolio will be as well as identifying the adjustments that would need to be made in current water pricing practices in order to ease the transformation of the current one.

In addition to the role of BAU (business-as-usual) pricing practices as cost-recovery mechanisms, the new (smarter) pricing system is called to play a relevant role through its contribution to the following intermediate objectives:

- Being an instrument to manage the basin closure. It must be designed in such a way as to shift the roles currently played by water supply sources. For example, excess capacity of desalinated water might be mobilized for second-priority uses while remaining available for first-priority ones during extreme dry periods. If partially replaced with alternative resources, groundwater sources might be restored in the short and medium terms so as to be able to play the role of a buffer stock in the long term (not excluding the sustainable use of the aquifers still in good status). Hence, the existence of both sources is guaranteed: overexploited aquifers are protected and desalination becomes financially sustainable.
- Bringing water security to the forefront of water policy. Security is important for the entire economy and for water users themselves (who, as explained above, are willing to pay for that). The pricing system must progress towards the internalization of the resource costs into water prices. In water-stressed economies this water cost is mainly reflected in the value of water security.
- Evolving in the recovery of water supply costs in order to guarantee the financial sustainability of all the sources in the water portfolio but also to manage demand and balance water demand and supply through the recovery of resource costs.
- Reducing water demand, when possible, via a sensible design of the pricing system.

This is to be achieved by setting a price on water security (in exchange of having water in dry seasons). Users are willing to pay a mark-up on price that guarantees steady water supply at a reasonable (known) and stable price. Initially this security would be provided via desalination.

In normal times there will be excess supply of desalinated water. Yet, as the fixed costs are covered by the excess price, this water might become available at an affordable price (equivalent to operation and maintenance costs). Among the different alternatives there is the possibility to promote the use of this water source as a substitute for water abstraction in overexploited aquifers.

It may also be considered that any increase in the security of water supply for urban uses also means an increase in the security of supply for irrigated agriculture. The latter would benefit from the decision of the former of using desalinated water instead of further reducing water supply to the irrigated sector as permitted by the legal hierarchy of uses in place. This would allow the introduction of a security mark-up in agriculture. Given the elasticity of demand this would not reduce water use in the irrigated sector and would rather create additional revenues that could be used to reduce the burden to the urban sector.

Increased water availability and water security in agriculture will likely result in reduced income variability, stable employment and positive forward linkages with other economic sectors (e.g., agro-industry). Therefore, it seems reasonable to split the cost of water security among users that benefit from it.

3.4.5 Prerequisite

As above-mentioned pricing water security is not a 'silver bullet' and its objectives can only be reached when this alternative is made part of an integral water policy reform. Public commitment to restore the sustainable use of water and transparency regarding the pricing scheme is a pre-requisite for the legitimacy and the political acceptance of higher water prices that will only come about if associated to the perception that users are receiving something important in exchange.

Providing water security in the short term must be coordinated with decreasing scarcity in the medium term and enhancing security in the long term (OECD, 2013). Along these lines, additional prices need to be connected with perceptible benefits (e.g. paying a risk premium gives you access to water in dry periods), but it is even more important to avoid the temptation to use the pricing scheme as a means to increase water supply in normal times. That is, not using the now cheaper resources to complement the already depleted groundwater sources and providing instead subsidies in exchange of steady and publicly observable improvements in phreatic strata. This requires setting up proper monitoring and administrative systems, apart from the infrastructures required to deliver the water to the target irrigation districts.

3.4.6 References

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3.5 Environmental taxation

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3.5.1 Introduction

Nearly a century has passed since the idea of introducing taxes with a tax base related to emissions and other environmental burdens was first proposed by Pigou (1920). There was for several decades resentment to his innovative idea, widely perceived as impractical, and it remained a footnote in economics textbooks for a long time. One of his former students, Ronald Coase (1960, 1988) even developed a formal theorem to explain why the idea would not be appropriate. Still, numerous economists particularly of US origin began from 1960 and onwards to explore theoretically principles of environmental taxation.

With challenges of industrialisation and increases in the associated pollution and material flows gaining momentum in the aftermath of World War II, the idea attracted more serious attention among decision-makers. In France new framework water legislation, *Loi sur l'eau*, was presented in 1959, and it included the introduction of effluent charges to raise funding for pollution abatement purposes (Barraqué, 2000). *Loi sur l'eau* was agreed in 1964, but then postponed, so in the end France did not go alone in launching a tax on water pollution. When implemented in 1971, similar systems of effluent charges were agreed and adopted also in the Netherlands (1971) and Germany (agreed 1971 and implemented from 1976). It appears that local Dutch waterboards, with prerogatives for taxing water users, had individually pioneered effluent charges from the 1950's, hence first implementing the idea in practice. At the end of the day use of effluent charges was in fact a rather pragmatic approach to settle the costs of abatement and overcome the inertia of action.

More ambitious environmental legislation was at this time being introduced in United States and Japan (Vogel, 1993). In Japan, where air pollution was more in focus, a significant levy on SO2 was introduced in 1974 to finance compensation payments for pollution victims suffering from respiratory diseases and poisoning (Committee, 1997; Matsuno and Ueta, 2000). This scheme was a solution for finding an appropriate way to issue a health bill to polluters, rather than to the general tax payers. So was also the levy on chemicals instituted under the US Superfund scheme in 1985 to finance clean-up of hazardous waste sites, as the first tax on municipal landfill waste introduced by Denmark in 1986 to sponsor measures for recycling and cleaner technology (Rahm, 1998). Both the European water effluent charges and Japan's air pollution levy proved nevertheless reasonably effective in curbing emissions, which helped stimulate further reflections and proposals for making use of economic instruments around the world, and also the waste related measures were significant in their own way.

From about 1990 attention shifted from viewing such taxes as being useful instruments for environmental policy purposes, towards their potential under the consideration of more fundamental changes in taxation policies, whereby other taxes could be substituted with environmentally-related taxes. In this process far more complex public finance questions appeared. Also a broader set of taxes became of relevance, as tax bases relating to energy, transport and natural resources were most significant in their potential for raising revenues. Under this new discourse it became apparent that long-established excises and taxes, for instance on petrol, and dating back to the days of Pigou, would be equally relevant for consideration, even if questions over the exact match between their tax base and Pigouvian principles of internalising externalities can be raised. Supportive to this new broader perspective on environmentally-related taxes were the concerns with carbon and the greenhouse effect, which following the first IPCC report, moved centre stage in global environmental policy deliberations. The idea of introducing a price on carbon fuelled a new wave of interest in how to align the broad set of excises and taxes on energy and transport under these new perspectives. The opportunity to reform tax systems using this kind of taxes to replace other distorting taxes immediately began to catch the imagination of academics, tax administrators and policy makers and transform into policy-making.

3.2.7 Theoretical and disciplinary orientations

Theoretical frameworks within economics, law and behavioural sciences have allowed researchers to deduce findings pertaining to environmental taxation, while the richness of empirical material increasingly available allows for testing of normative propositions against positive observations, in turn generating new ideas and theories on the tax system and its possible 'greening'. While the very first literature on environmental taxation was inherently economic in the neoclassical tradition, scholars coming from the law and economics tradition challenged and expanded that framework for research. With environmental taxation transcending from the world of ideas to one of legislative action, the issues and challenges with administrating the law and examining specific cases in fact opened an entirely new area of research to legal scholars. A separate behavioural approach rooted in policy sciences has emerged too, addressing broader questions going beyond the purely economic efficiency criterion, in particular about equity implications and ultimate policy effectiveness. But also scholars traditionally preoccupied with concerns about optimal taxation design, in particular from public finance, have subjected environmental taxation principles to investigations. Policy questions spurred by concerns about optimality in turn have led to the involvement of environmental engineers with tools that enable estimates of what might be suitable rates of environmental taxes reflecting external costs.

The study of environmental taxation is closely linked with studies of other policy instruments applied in the area of environment, energy and climate. It belongs to the family of market-based policy instruments, which seek to take advantage of the dynamics of signals that originate in the market place, but which can be reinforced or adjusted with appropriate mechanisms. The inverse of an environmental tax is an environmental subsidy which places a premium on specific actions with presumably favourable implications. Subsidies can be granted directly with transfers and favourable loans, or indirectly with tax expenditures that allow polluters to offset their costs against tax liabilities. While both are price based policy instruments, their shared contrast is with quantity-based policy instruments, that place a cap over emissions and require markets with exchange of pollution rights to be established. Where quantity-based allowances are handed out for free, they imply a subsidy to polluters, whereas allowances that become available only upon auctioning share the features of an environmental tax. The entire basket of market-based policy instruments, price or quantity related, is different from the more conventional basket of 'command-and-control' policy instruments that do not speculate in provision of incentives, but merely prescribe what behaviour is required by regulators under the law. 'Command-and-control' imposes standards, prescribes bans and requires procedures to be adhered to under a juridical perspective where compliance is an issue that will be resolved ultimately in the court. A third category consists of informal voluntary arrangements, based on more or less negotiated understandings between regulators and polluters. Obviously much of the research relating to environmental taxation addresses differences and similarities with these other types of policy instruments. As such the environmental tax related research on policy instruments is very much part of that wider research agenda.

3.2.7 *Defining environmental tax*

Whereas much of the early literature simply referred to emissions or effluent charges, Ralph Johnson and Gardner Brown introduced a useful distinction between what they termed user charges and effluent charges. While payment for a service rendered, such as wastewater treatment, involves a user charge, the term effluent charge refers to the case where a financial payment is involved for a discharge or emission regardless of any public abatement effort and simply for making use of the environment as a sink. The latter case is the one that usually is understood in the economics literature to represent the principle of Pigouvian taxation.

Europeans on the continent tend to use the term 'levies' to denote the economic instruments used in water management (in Dutch, 'heffingen'; in German, 'abgaben'; in French, 'redevances'). European countries, notably those with schisms between upstream and downstream polluters, had in the post-World War II period prepared payment schemes for emissions that were both different from conventional user charges and implied financial penalties for using the environment as sink for emissions. Administrated by regional and often task-specific authorities, they provided a financial basis for pollution control activities. While these schemes have been subject to much attention in the literature, they are not conventional taxes. Some public services are in fact provided in return for the earmarked payments, as revenues are recycled for pollution abatement.

As the leader in globally oriented assessments of economic policies, the OECD became a key arbiter in the definitional arena of economic instruments for the environment. A 1984 conference was its first initiative. In line with the existing academic literature its early surveys tended to refer to charges, and to categorize the charges according to their design characteristics, such as emissions charges, user charges and product charges[19]. At the same time it grappled with the question of the definitional relevance of both the intent behind the tax or charge—whether it was intended to operate as a behavioural incentive or to raise revenue—and the actual environmental effect of the tax, but due to the mixed purposes of decision-makers no clear cut categories emerged.

In 1997 Eurostat with the European Commission's Tax Directorate, DG TAXUD, joined with OECD and IEA (the International Energy Agency) and in an attempt to solve the dilemma, chose to focus on the nature of the tax base—what was being taxed—rather than the environmental intent or effect of the tax. They defined an environmental tax as;

"*A tax whose tax base is a physical unit (or a proxy of it) of something that has a proven specific negative effect on the environment*"

On basis of this definition Eurostat published a detailed guideline for the operational accounting for revenues from such taxes, dividing them into four basic categories; energy (including carbon), transport, pollution and natural resources. The United Nations subsequently integrated this definition into as well the System of National Accounts as the SEEA, the System of Environmental-Economic Accounting (Statistics Sweden, 2003).

OECD (2001), however, favours a somewhat different terminology and has adopted the term 'environmentally related taxes'. It extends the general definition of a tax and defines an environmentally related tax as:

"*any compulsory, unrequited payment to general government levied on tax-bases deemed to be of particular environmental relevance*"

Taxes are unrequited in the sense that benefits provided by government to taxpayers are not normally in proportion to their payments[20].

By focusing on unrequited payments, the definition excludes charges and fees paid to government for services, such as waste removal and treatment fees. Nevertheless a charge or levy that is an unrequited payment may, despite its name, qualify as an 'environmentally related tax.' As a general rule, a payment should be classified as a tax either if no link between the payment and the service rendered can be established, or if the revenue is much larger than the costs of the service provision in

[19] The European Environment Agency, however, took a different approach, preferring in 1996 to use the overarching term environmental taxes rather than charges, which covered cost-covering charges, incentive taxes and revenue-driven fiscal environmental taxes (EEA, 1996).

[20] Earlier formulations of the definition of "environmental taxes" can be found in OECD, 1997a and 1999

question. There can be a faint line between taxes and other levies, depending upon how one interprets 'unrequited', cf. OECD, 2008[21].

3.2.7 Categories of environmental taxes

There are various ways to categorize environmentally related taxes, depending on whether one focuses on the design of the tax or its purpose. When focusing on design, the taxes are often classified as (OECD, 2003):

– emissions and effluent taxes;
– product taxes; and
– natural resource taxes.

The distinction between the first and the second is that the first taxes the emissions directly, while the second taxes products that are likely to generate environmental damage in their manufacture or use. Although some studies include natural resource taxes in the category of product taxes (see OECD, 1997b), the difference in the rationales for the two types of taxes would seem to warrant separate categories.

Product taxes attach a price to pollution while natural resource taxes place a price on the use of scarce natural resources. Each presents a different social value, although both will serve the ultimate end of environmental protection.

Alternatively, one can classify environmentally related taxes according to their relative environmental and fiscal function (cf. Määttä, 2006), such as:

– incentive environmental taxes, which can also be called regulatory taxes;
– financing environmental taxes; and
– fiscal environmental taxes[22].

The first is driven by its environmental impact, the second by its ability to finance an environmental measure, and the third primarily by the demand for revenue. One can also classify taxes as either independent or complementary by evaluating their role relative to other instruments (Määttä, 2006).

Classifications such as these are useful for drawing attention to different features and functions of the taxes that fall within the OECD's definition of environmentally related taxes. The distinction between fiscal taxes and financing taxes is one that preoccupies public finance experts. Conventional taxes are understood as revenue-raising fiscal taxes, but the innovative feature of environmentally-related taxes is that they are supposed to alter behaviour besides raising revenues. Here the waters begin to divide between different disciplines. Some tax lawyers have been concerned whether basic legal principles of equal standing to the law could be violated when

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[21] The Merriam Webster Online Dictionary defines unrequited as '*not reciprocated or returned in kind*'.

[22] Stephen Smith (1997) has taken a hybrid of the two approaches above, using '*measured emissions taxes*', '*the use of other taxes to approximate a tax on emissions*' and '*non-incentive taxes*'.

economic instruments are used for instance (see Kirchhof, 1993). Economists on the other hand expect any tax to influence behavior, whatever purpose it comes with.

The OECD's definition and these classifications illustrate how current research in the field extends far beyond conventional fiscal taxes. The Pigouvian tax reflecting marginal social cost was meant to internalize environmental burdens in the economy by providing incentives. Also a Baumol and Oates' standard-driven, second-best tax would qualify as an incentive (regulatory) emissions tax. Still numerous environmental taxes have been introduced around the world simply to finance specific schemes to provide relief to environmental burdens of water, air and waste pollution. These taxes fall neatly between the two main categories of fiscal and incentive taxes, constituting financing environmental taxation – sometimes even hypothecated for predefined purposes. They are not user charges, because the revenues contribute towards a common good of pollution abatement, rather than individual service provision of garbage collection or waste water treatment.

To further complicate the vocabulary of environmental taxation, environmental taxes are often known by other names in political discourse and research, such as green taxes, green fees, eco-taxes, charges and levies. Under the internationally agreed approach, however, the nature of the tax base should prevail over the name when deciding whether to admit a tax, fee or charge into the category of environmentally related taxes. The remaining differences in what Eurostat and OECD will classify as an environmentally-related tax underlines the need for further attention and research on terminology issues[23].

3.2.7 EPI-WATER research on environmental taxes

The EPI-WATER project team has analysed, among other, different *product taxes* (mineral fertilisers and pesticides) in Denmark and France (Branth Pedersen et al., 2011; Defrance et al., 2013; Skou Andersen et al., 2013), *resource tax* (groundwater) in the Netherlands (Schuerhoff et al., 2011), and *emission taxes* (wastewater effluent) in Germany and Hungary (Möller-Gulland et al., 2011; Rákosi et al., 2011).

The *Dutch groundwater tax* (GWT) (Schuerhoff et al., 2011), entered in force in 1995 and repealed in December 2011, had been levied on all water abstractions with pumping capacity exceeding 10m3/hour with numerous exceptions (including the irrigation). In addition to the tax, water abstraction fees at lower administrative levels were in force but their aggregate revenues are much lower that the revenue collected by GWT (ca. 10 per cent). The GWT was applied to the volume of extracted water and differentiated according to the provisions regarding the return flow and/or the infiltration of rainwater. The estimated revenue (ca. 170 mil Euro) was paid by water utilities (by 80 per cent); industrial plants (by 12 per cent), until 2001 taxed at lower rate; and only to a small extent by farmers (2 per cent). As a consequence of the tax,

[23] For instance, the German waste water tax (*Abwasserabgabe*) is not by Eurostat accounted for as an environmental tax, whereas OECD does so. Its revenues are hypothecated for Länder-level (regional) EPA's.

the domestic water tariffs increased on average by 10 per cent (ranging from 2 to 19 per cent), contributing potentially to reduced consumption, but the effects of GWT are difficult to disentangle from the tap water tax levied on the quantity of water delivered to households. The water related costs of the industry were far below 1 per cent of the total costs. The GWT was introduced as an environmental tax reducing the fiscal burden on income but effectively filling the fiscal gap after an unsuccessful attempt to increase fuel tax. The impacts on the GW levels are difficult to determine without prior measurements but it should be noted that lower water tables are not a concern throughout the country, in some places the high water levels cause damage. Whereas the GWT was repealed, the tap water tax standard rate from 2014 on is set to be doubled, with lower and differentiated rates for industries.

The *Danish pesticides tax* (PT) (Branth Pedersen et al., 2011) was implemented around the same time as the Dutch GWT with the aim to reduce consumption of pesticides by 5-10 per cent, later slightly revised upwards. The pre-existing general pesticide tax was too low (ca. 3 per cent of wholesale price) for the intended purpose. The PT was set to contribute to meet the objectives of the Danish pesticide policy set to reduce application of pesticides to the treatment frequency index (TFI) level of 1.7. The PT, along with the mineral fertiliser tax (section 2.3), were intended to protect the surface and groundwater bodies, the latter being source of drinking water provision usually without treatment. As a product tax, the PT is levied on the sales prices of different pesticides and differentiated according to the categories of use, rather than the toxicity levels. It is expected though that the tax is redesigned so as to reflect the environmental harm of the chemical compounds. The initial rates were increased twofold after the 1998 review and eventually led to doubling the pesticide retail prices. The collected revenues are reimbursed to farmers through lower land taxes and subsidies for organic and environmentally friendly farming. The observed pesticide application rates are a result of a number of factors including the agricultural product prices, farm structural indicators, as well as weather conditions during the assessment period. As a result of the low elasticity of demand, it is believed that the predetermined environmental goals could be achieved only by a substantial (up to tenfold) increase of the tax rates in the case of insecticides (ibid).

The *water emission taxes* (ET) essentially implement the polluter-paysprinciple (PPP) and offer to some extent incentives to lower water consumption where wastewater generated is approximated by volumes of water consumed. The taxes (and service charges) complement extensive European environmental legislation such as the Urban Waste Water Treatment Directive (UWWTD) of 1991 and the Industrial Emission Directive (IED) of 2010. Hence the taxes may provide incentive either to meet the regulatory limits earlier than expected by the existing regulations, or to reduce the pollution load beyond the regulatory limits if the tax levels are set appropriately to this end. The collected revenues may be recycled in a way to stimulate investments, as in the cases reviewed by the EPI-WATER team.

In Germany, the wastewater discharges are subject to environmental permits with strict requirements to be met since 1957. The '*Abwasser-abgabe*' or *effluent tax* (ET) was introduced in 1976 while recognising the limited pollution control achieved so far. The ET is calculated based on the permitted damage units, calculated as the equivalents of ten pollutants. The revenues are earmarked for further investments in pollution control completed by the *Länder* authorities and for tax incentives aimed at the polluters themselves. The nominal annual values of ET rates increased from ca. 6 Euro to over 35 Euro per damage unit in 1997. The investments into pollution reductions were deducible by up to 75 percent of the ET until 1998 and by 50 per cent ever since. The municipal waste water treatment plants (WWTP) were eligible to 3-year long exemption from the tax to offset the investments. In the case of exceedance of permitted pollution load, additional charges and fines are applicable.

The *Hungarian water load fee* (WFL) was introduced incrementally from 2003 until 2008 as a part of a comprehensive environmental tax package addressing air, water and soil pollution. The tax is determined by nine contaminants contained in the discharged wastewater but takes into account the environmental sensitivity of the receiving environment and the way the sludge is eventually disposed. Until 2011, the polluters' investments in pollution reduction technology and measurement/ monitoring were deducible from the WFL due otherwise by 50 and 80 per cent respectively. To a large extent (up to 90 per cent) the tax revenue is collected from WWTP and only by 10 per cent by other polluters. It should be noted that the WFL overlapped with the implementation of the UWWTD in Hungary and the time from which onwards the full rates of WLF were applied coincided with the deadline to meet the UWWTD requirements in urban agglomerations discharging more than 10.000 population equivalent (PE) load units. The prescribed treatment technology in the agglomerations with discharges between 2.000 and 15.000 PE has to be put in practices by 2015. In 2010, the WFL accounted up to 11 per cent of the total costs of the WWTP.

3.2.8 Environmental tax reform in Europe

Environmental tax reform (ETR) is a part of the Europe 2020 *resource efficiency* initiative (EC, 2011), aiming at among others limiting the environmental impacts of resource use while at the same time improving economic performance, hence decoupling environmental pressures from economic growth. The principle of ETR is a transformation of the tax burden away from where it may cause adverse impact on economic competitiveness such as labour and capital taxation to areas where such impact is lower and to activities with provable negative environmental impacts (Ekins, 2009). The need for a new model was set as early as in 1993 in the White Paper on Growth, competitiveness, employment (EC, 1993) and further reiterated in a number of occasions (EC, 2005a, 2005b), including the Green paper on market-based instruments for environment and related policy purposes (EC, 2007). The stated goal of the 2020 resource efficiency roadmap (EC, 2011) include '*a major shift*' away from labour to environmental taxation and '*a substantial increase*' of the share of public revenues from environmental taxes.

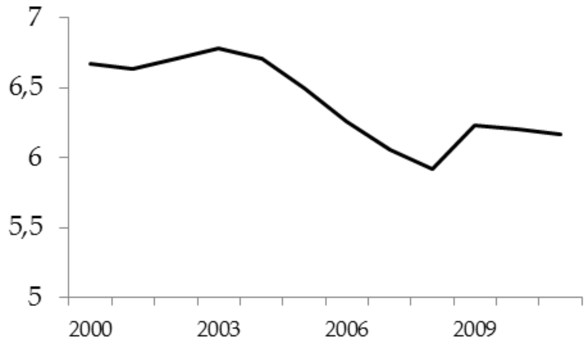

Figure 8: Evolution of the environmentally related taxes in EU-27 as a percentage of total revenues from taxes and social contributions (y axis) over the period 2000-2011. Data: Eurostat ten00064 indicator, own elaboration.

Environmentally related taxes (ERT) addressing water pollution and use vary substantially across the EU Member States (MS) and the data collection is difficult due to highly heterogeneous and complex governance regimes. Overall, they play a small role in the ERT revenues dominated by transport fuel and energy taxes which in some countries such as Denmark and Ireland account for between 35 and 44 per cent of ERT (Eurostat, 2013). According to the latest available data (2011), in the most EU MS the ERT account for between 2 and 3 per cent of the *gross domestic product* (GDP), with 5 MS (including Spain, France, Lithuania, Romania and Slovakia) collecting less than 2 per cent of GDP from ERT, and 2 countries (Denmark and the Netherlands) where the revenues from ERT exceed 3, per cent of GDP (ibid.) On average, ERT account for 1/16 (ca. 6 per cent) of the overall tax revenues. These figures should be interpreted with care as they do not generally indicate the importance of environmental protection in the countries' fiscal policy. Because they reflect both tax rates and tax base, high level of ERT in relation to GDP may indicate inefficient use of resources (Eurostat, 2013). Figure 8 shows the evolution of ERT during the period 2000-2011. It indicates a downwards tendency between 2003 and 2008 mainly as a result of the lack of indexation of tax rates, but also due to a dieselisation of the vehicle fleet across Europe. ,whereby a revenue loss from higher taxed petrol occurred.

Environmental fiscal reform is often discussed as a means of bringing about a so called 'tax shift' in which a progressive increase in the revenues generated through environmental taxes provides a rationale for reducing taxes derived from other sources, such as income, profits and employment, the taxation of which is less desirable. The rationale for using an increase in revenues from environmental taxes in this manner is entirely sound where the fiscal position in the country concerned is relatively healthy.

However, where budgets are out of balance, and in particular, where deficits are leading to increasing indebtedness (leading, potentially, to increased costs of borrowing, and perceived risks of sovereign default, where no action is taken to address such deficits), the more immediate concern may be to reduce the gap between expenditure and revenue generation. Evidently, improved efficiency in public services, coupled with some retrenchment, will reduce public spending, but the exchequer may need to act to increase revenue take to completely close the gap between income and expenditure. Generating additional revenues from taxation may also limit the extent to which austerity has to bear the brunt of adjustment required to bring the fiscal position back into balance. In such situations, the question becomes one of which taxes to deploy to help reduce budgetary deficits.

To the extent that environmental taxes may have a role to play in this regard, their use as a means to reduce budget deficits is not so different to their deployment in the context of environmental tax reform: in both cases, it could be argued that the counterfactual situation (to that where additional environmental tax revenues are generated) is one where other forms of tax have to be used to generate the equivalent revenue. In both cases, it can be argued that the additional tax revenue is generated as an alternative to (further) cuts in spending to achieve a similar (net) budgetary outcome. On this background a shifting of the tax burden, with more emphasis on environment related tax bases, is being considered as part of the European Semester for individual EU Member States.

3.2.7 References

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3.6 Tradable environmental permits

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3.6.1 Introduction

The tradable environmental permits (TPs) are increasingly gaining on popularity in a number of environmental policy fields (OECD, 2013; Tietenberg, 2006). Oddly, whereas there are only few instances of water related TPs in Europe, while common in the Anglo-Saxon countries, the European Union operates the largest multi-country, multi-sector greenhouse gas emissions trading system in the world – the *Emission Trading System* (ETS).

TPs have also been applied to control non-uniformly mixed air pollutants such as sulphur and nitrogen oxides, and to support production of renewable energy sources (RES) (Jensen, 2003). From among the cases analysed in the EPI-WATER, Mysiak et all (2011a) addressed the green energy certificates scheme in connection to the overexploitation of water resources in the Northern Italy. A non-exhaustive list of TPs in Europe include waste management (Bailey, 2003), biodiversity conservation (Drechsler and Hartig, 2011; Wissel and Wätzold, 2010), and fisheries (Grainger and Costello, 2011). Since the 2003 reform of the EU Common Agricultural Policy, the entitlements to subsidy under the pillar I are also tradable and are capitalised in the land prices (Ciaian et al., 2010).

The *tradable (land) development rights* (TDRs), common in Anglo-Saxon countries and experimentally tested in some EU Member States, qualify as environmental TPs when connected to soil and biodiversity protection. TDRs hold promise for spatial planning as a complementary instrument to zoning and compensation of land value losses of downzoned properties. Recently, TDRs have been proposed for flood risk reduction (Chang, 2008; Ward, 2013).

TPs are also called right-based EPIs (Beder, 2001) but this categorisation disregards the differences in the underlying property regimes and may imply, wrongly, the privatisation of the resource (Tietenberg, 2002). Contrary to price instruments (see section 2.1, 2.3, 2.4 and 3.5) that spur individual responses, the TPs entail transactions between authorised users and enable a shift of resource to higher-value use, and/or a minimisation of compliance costs, and/or a re-distribution of scarcity rent. For these reasons, the neoclassical economic theory consider environmental TPs superior to other economic and regulatory instruments.

In this essay we synthetize the results of the six water related TPs analysed in the EPI-WATER project (Ancev, 2011; Donoso Harris, 2011; Howe, 2011; Kieser and McCarthy, 2011; Yates, 2011; Young, 2011). Unlike the sections 2.2 and 2.5 in which the water trading and water emission trading are explored in depth individually, here we focus only on the property regimes under which the TPs operate, and their environmental and economic performance. Furthermore, we review the features and performance of the ETS, as the rich-experience from its implementation may inspire TPs in other environmental fields, especially water emission trading. Furthermore yet, we extend the focus on the potential interconnection of TDRs and water management.

3.6.2 *Meaning of property rights*

Initiated by Coase (1960) and Demsetz (1967), the neoclassical economics research considered the turn to privately held property rights as an evolutionary response to the increasing scarcity of natural resources, a step towards economic efficiency. Under this view, all scarce resources should be owned by someone, property rights should be exclusive rights, and resources should be allocated from low to high yield uses (Mahoney, 2005).

Property rights are *social institutions that define or delimit the range of privileges granted to individuals of specific resources* (Mahoney, 2005). The *right* held by someone is mirrored by correlative *duties* of non-inferences by somebody else (Hohfeld, 1913), including the state authorities. The exercise of rights is qualified by *rules* that authorise or require specific actions to protect the resource and the interests of downstream right holders (Schlager and Ostrom, 1992). Contrary to a layman's view, property rights do not constitute *ownership* of a '*resource system*' (e.g. water body) but privileged *access* (Tietenberg, 2002) to a resource, or better, '*flows of resource units*' (Ostrom and Hess, 2008). *Water rights* are hence shares of water flows and/or volumes of water that may be withdrawn for specific use. *Emission rights* are appropriated shares of the assimilative capacity of the environment, that is amounts of potentially harmful substances that are naturally removed or diluted without causing harm to ecosystems and humans. *Development rights*, a part of the bundle of land related rights, entail authorisations to create impervious surface by soil sealing and/or changing land-cover/use with substantial effects on biodiversity and surface run-off. As in the case of water resource systems, the land property rights do not imply ownership of the land itself but '*the rights and duties over it*' (EC, 2004a).

The definition of *property* in general is concomitant to a number of different rights (Honoré, 1961) of which five are typical for management of *common-pool resources*: *access*, *withdrawal*, *management*, *exclusion*, *alienation* (Schlager and Ostrom, 1992). TPs are enabled by unbundling *alienation* from the pool of other rights and impose limits on the exercise of rights through a collective-choice action (ibid). With other words, what distinguishes TPs from other forms of authorised access is the discretion of the TP holder to alienate, that is sell or lease other authorised privileges.

3.6.3 *Property right regimes of the selected tradable permit schemes*

The EU-ETS was established in 2003 and covers some 11,500 installations over 30 nations, and around 41 per cent of total EU GHG emissions. It is essentially a cap-and-trade scheme, which imposes an emission limit to European factories, power plants and other regulated installations and gives them the possibility to trade allowance surplus. The first pilot trading period run from 2005 to 2007, the second from 2008 to 2012, and the third started just in 2013. Rules and coverage changes substantially among the three phases. During the first phase only CO2 emissions from power generators and energy-intensive industrial sectors were covered. In the second phase, the scope of the system was widened by including additional greenhouse gases (GHG) such as nitrous oxide emissions from the production of nitric acid. The third phase, which brought several changes: new sectors and gases have been included and the system of national plans has been replaced by a single EU-wide cap. The European Commission has identified some further options for structural reforms of the EU ETS, which include: increasing the emission cap, retiring a number of allowances, cutting the annual linear reduction factor, extending the scope of the ETS to other sectors, limiting access to international credits. Both these proposals still need to be approved.

The tradable water rights analysed in the EPI-WATER project exhibit important differences. In the Southern Connected River Murray System, the tradable water (access) *entitlements* are de-jury secure water rights defined as *shares* and issued in perpetuity (Young, 2011). The actual volume of water that can be withdrawn and used is determined through periodic allocations. Both entitlements and allocation are tradable. The existing trading scheme is a result of longstanding water reform that first transformed water licences, then imposed a cap on water abstraction, and finally unbundled the water entitlements so as to allow a separate trading of each individual component.

In Colorado, water rights are embedded in the 'prior appropriation doctrine' (Hodgson, 2006) and their tradability, including their priority or '*seniority*', was established under case law (Howe, 2011). Importantly, the holder of a water right does not own the return flow and is not allowed to change its pattern. This is because the return flow may be appropriated downstream and any change may interfere with the downstream water rights. In the case of water transferred from other '*hydrologically independent*' basins the return flows are owned by those who made the transfer. The water market in the Northern *Colorado Water Conservancy District* (NCWCD) was enabled by a water transfer from the Colorado River headwaters and the NCWCD maintained the ownership of the return flow. Following the full subscription of the shares (each nominally worth one acre-foot or ca. 1233 cubic meters of water), the shares have been turned into freely transferable contracts between the District and the holder, for the beneficial uses within the District (ibid).

In Chile, the customary water rights have been re-established by law in 1981, after land and water entitlements were expropriated in 1967 (Donoso Harris, 2011). The rights entitle consumptive uses for surface and groundwater, specified as volume of water per unit of time and shares of water flow, and non-consumptive uses for surface waters. The right holder is not required to maintain return flow but does not own it. The use of water is discretional to right holders and the rights do not expire in the case of non-usage. The rights are guaranteed by the constitution even in cases they have not yet been formally registered. Two priorities (permanent and contingent) rights are distinguished. *Non-use* tariff is imposed on those who demonstrably don't use the water to which they are entitled.

The tradable water emission permits analysed in Ancev (2011), Kieser and McCarthy (2011), and Yates (2011) display lower security. In neither case the permit constitute a property right but rather individual compliance requirements. In the Great Miami River Watershed of Ohio (Kieser and McCarthy, 2011) the permits had been established as time-limited (10 years) entitlements prior the implementation of stricter environmental protection. In the North Caroline case (Yates, 2011), the emission permits have been specified as a maximum nutrient emission from each point source by the US Environmental Protection Agency (US-EPA). The state statute allowed exchange of permits, and a group (association) - instead of individual – compliance regime. Noncompliance is penalised both by the state authorities and the user association established. In Australia (Ancev, 2011) the individual compliance is determined case-by-case.

Tradable development rights (TDR) represent an EPI for spatial planning with direct links to water management either for water quality concerns or flood risk prevention. The rights related to land ownership are specified in the state statutes and are typically secure. The principles on which TDR rely are simple: zoning policies impose limits to land development associated with positive private benefits and external costs. As a result, the prices of developable land increase while undevelopable land losses on value. The skewed distribution in the benefits of the zoning policy is counteracted by separating the rights to develop land (DRs) from the remaining bundle of property rights, and creating a market for their exchange. Property owners who are zoned out are compensated by selling their DRs to owners of developable land. The latter are obliged to buy the DRs but the price they pay is capitalised in the increased value of the developable land (Ward, 2013). TDR developed in the United States complementary to zoning instruments and as a way to compensate property value loss imposed by downzoning.

TDR have been extended to address the flood risk. Chang (2008) and Chang and Leentvaar (2008) proposed three types of tradable flood mitigation permits: tradable development rights (TDR), tradable flood reduction permits (TFR), tradable risk neutral permits (TRiNe). While TDR works as described above, TFR and TRiNe are similar and represent flood mitigation certificates on the principle 'baseline and credit' (EA, 2007). Mori (2009) and (Ward, 2013) have explored tradable instruments for flood management theoretically and Mori and Perrings (2012) through an empirical assessment in the UK. The TDRs display sizeable potential but require substantial institutional changes (Renard, 2007).

3.6.4 Initial allocation

The efficiency of TPs does not depend on how the permits have been distributed initially, that is the initial allocation is exogenous to the instrument's expected performance. It is widely believed though that the initial allocation influence the acceptance of the TPs. The belief that TPs can depoliticise the management decision making is naïve. The permits can be distributed for free following the historical pattern of entitlements (grandfathering), or distributed by auctions and lotteries (Tietenberg, 2002). Banking of entitlements allows for mitigating short term fluctuations of the permit prices and generally leads to an earlier compliance.

The first two trading periods of the EU-ETS have been characterized by a high level of free allocation of permits, distributed according to historical emissions through National allocation plans (NAPs) defined at domestic level. The use of a certain amount of international credits from the Kyoto Protocol's flexible mechanisms (CDM and JI credits) were allowed. In the second phase, the share of allowances distributed for free decreased to 90 per cent of the total. The penalty for non-compliance was set to 40 Euro per ton/CO2 in the first and 100 Euro per ton/CO2 in the second trading period. Banking of allowances between the second and third phases was allowed. In the third phase, the allowances are allocated through auctions. Manufacturing industries will continue to receive a certain number of free permits, according to a set of benchmarks defined with the objective to protect European industrial sectors from carbon leakage. However, the number of allowances given for free will gradually decrease until to be phased out completely by 2027. In addition, the European Commission is working to define a plan aimed at reducing the surplus of allowances that continues to affect the market. The so-called "backloading plan" consists in delaying the auctioning of 900 million allowances to later in the Phase III of the EU ETS, in order to rebalancing the annual supply and demand and to boost carbon price.

The tradable water entitlements analysed in the EPI-WATER project were initially allocated through grandfathering in all but the Colorado case where they have been sold at nominal prices. In Australia, a distinction is made between high, general and low security 'pools' without further distinction of the seniority of the rights. Where this is possible, the periodic allocations can be carried forward from one year to the next. The initial failure to account properly for the return flow has led to an over-allocation and a costly bay-back of entitlements (Young, 2011). Some estimated 35 to 50 per cent reduction is expected over the next decade (ibid). In the Colorado case, the subscription for shares (allotments) in the NCWCD started in 1939 and was not completed until 1955 (Howe, 2011). Each share refer to ca. 1233 cubic meter of water but historically only 860 cubic meter was delivered. In Chilean case (Donoso Harris, 2011) the customary water rights are specified based on the *foreseeable* use, that is the probable effective water extraction.

The tradable water emission permits have been set based on the current or expected (tighter) regulation.

3.6.5 *Economic and environmental performance*

In theory, the TPs minimise the costs of compliance with environmental standards and maximise the economic efficiency of resource use. The environmental gains are determined by collective-choice action through aggregate limits (on water withdrawal or pollution emissions). Tietenberg (2002) challenges on a number of accounts: First, aggregate limit may not be set independently of the mean of reaching it. Second, the TP can be set with planned reduction of the of the aggregate limit. Third, the policy regime chosen affect the level of monitoring and enforcement.

The installations included in the EU-ETS scheme reduced emissions by 16 per cent in the period 2005-2012. In 2011 the EU Allowance (EUA) trading volumes increased, reaching 7.9 billion tons of CO2e, valued at 148.106 billion Euro. Due to an over-allocation of allowances in the first trading period, and the impossibility to bank them to be used in the next phase, the price of permits quickly fell close to zero in 2007. Since 2008, the effect of the economic and financial crisis led to a decrease of emissions from industrial production. In addition, other factors influenced the emission level in the same phase, such as the release of the energy efficiency directive in 2011 and the increase of energy from renewables. As a consequence, the second phase has been affected by a wide surplus of allowances that flooded the market, and that caused the collapse of carbon price. Since mid-2008 the allowances' price plummeted from a peak of 30 to 5 Euro, the current trading level. At the end of the second phase the surplus was estimated to be well over 1.5 billion allowances, and even as large as 2 billion allowances.

All analysed tradable water entitlements have demonstrated affirmative economic performance. In all but Colorado case, however, the environmental performance suffered settle backs due to initial design faults. In Australia, the estimated increase of Australia's GDP due to water trading amounts to 220 AUD in 2008/09. The value of water entitlements grew on average by 15 percent each year which is higher that return on investment in the share market. Water trading also substantially reduced the economic impacts of the extended (2002/3-2008/9) drought spell in the Southern Connected River Murray System (Young, 2011). However, over the analysed period, both actual water use and the extent of irrigated agricultural land increased substantially. Improved water efficiency, increased capture of overland flows and increased exploitation of groundwater have substantially reduced the (return) flow volumes and patterns with the subsequent impairment of the environmental flow of the river (ibid).

In Chile, over the period 2005 - 2008 some 24,177 transactions were registered and the total value of water entitlements only accounted for 4.8 billion USD, that is 1.2 billion USD per annum. The prices of the entitlements (on average 615,623 USD) are highly variable across the country. An important shortcoming was the initial neglect of ecological flows, leading to a total allocation of some water bodies' flows. In semi-arid and arid regions this led to a deterioration of aquatic ecosystems (Donoso Harris, 2011).

In the Ohio WET scheme , the contracted reductions of pollutants entailed 339 tons of TN and 130 tons through the first six rounds of reverse auctions. Several ancillary environmental benefits were acknowledged, including habitat improvement, stream shading/temperature benefits, stream bank stabilization, flow velocity stabilization, and floodplain preservation, but not analysed in depth. The estimated cost-savings of the scheme amounted to 384.7 million USD (Kieser and McCarthy, 2011).

3.6.6 *Role in Europe*

The EU environmental legislation is rather reticent about the use of TPs for managing water and other environmental resources, with exception of air quality control and fishery. The recently adopted 7th EU *Environmental Action Programme* (EC, 2003) opens up to '*other market measures*', without specifying which, in addition of the ubiquitous call for (better implementation of) water pricing. The 2012 Water Blueprint (EC, 2012a) acknowledges *intrabasin* 'water trading' as potentially useful policy instruments to improve water efficiency and combat water stress, and pledges development of a *Guidance* document for advising the EU Member States (MS) who choose to deploy it. It stops short however of making similar commitment for water emission trading, generally perceived to meet resistance of the environmental groups[24]. Secondary policy analysis documents (Farmer et al., 2012; Strosser et al., 2012) explore the pros and cons of water entitlements trading, many of which have already been discussed in the section 2.2 ad elsewhere in this document. Farmer at al. (2012) note the opposition of the stakeholders to a Community action in the field of water trading, widely believed to be a solely competence of the MS.

In contrary, the 2004 Economic Review embraces the '*evolutionary*' response to increasing scarcity of natural resources by subjecting them to *common-property* regimes first, and eventually turning them into private goods (EC, 2004b). According to this view, the TPs are more flexible and hence able to account for the specific characteristics of individual firms, while offering an incentive to outperform the existing regulatory standards and limits. To this end however the property rights must be defined and assigned, and subsequently made enforceable, through reform of '*institutional and legal framework, and technical exclusion mechanism*' (ibid). Similarly, the inadequately defined and/or enforced property rights are seen as a *potential barrier* to meeting the environmental goals also by Commission's review of the of the WFD River Basin Management Plans (EC, 2012b).

In meantime some EU MS moved on to exploit practically the EPs. UK has engaged in a reform of water abstraction licences, following the *Cave* Report (Cave, 2009) and supporting analysis by the *Water Service Regulation Authority* (OFWAT) and the *Environmental Agency* (EA). Announced in the *Natural Environment White Paper* (HM Government, 2011) and further substantiated in (DEFRA, 2011a, 2011b), the reform entails smooth *transition* to a new regimes, without compensation for the losses incurred, by 2020s. The scope of the reform is to install tradable licensing regime capable to respond to current and future challenges. To this end, Young (2012) made practical recommendations based on the Australian experience with water trading.

The Dutch tradable nutrient quota system (Section 2.5), replaced by the mineral accounting scheme (Section 2.3) which in turn was found incompatible with the EU Nitrate Directive, shed light on the difficulties in implementing the water emission schemes in Europe. The proposed Swedish and Finish schemes (Section 2.5.5), both

[24] EurActiv 05.12.2012, An EU cap-and-trade scheme for water pollution? Greens say no. Retrieved from www.euractiv.com/cap/green-groups-battle-water-pollut-news-515792.

designed so as to contribute to the Baltic Sea Action Plan, might benefit from a legal review of compatibility with the European legislation.

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Annex 1

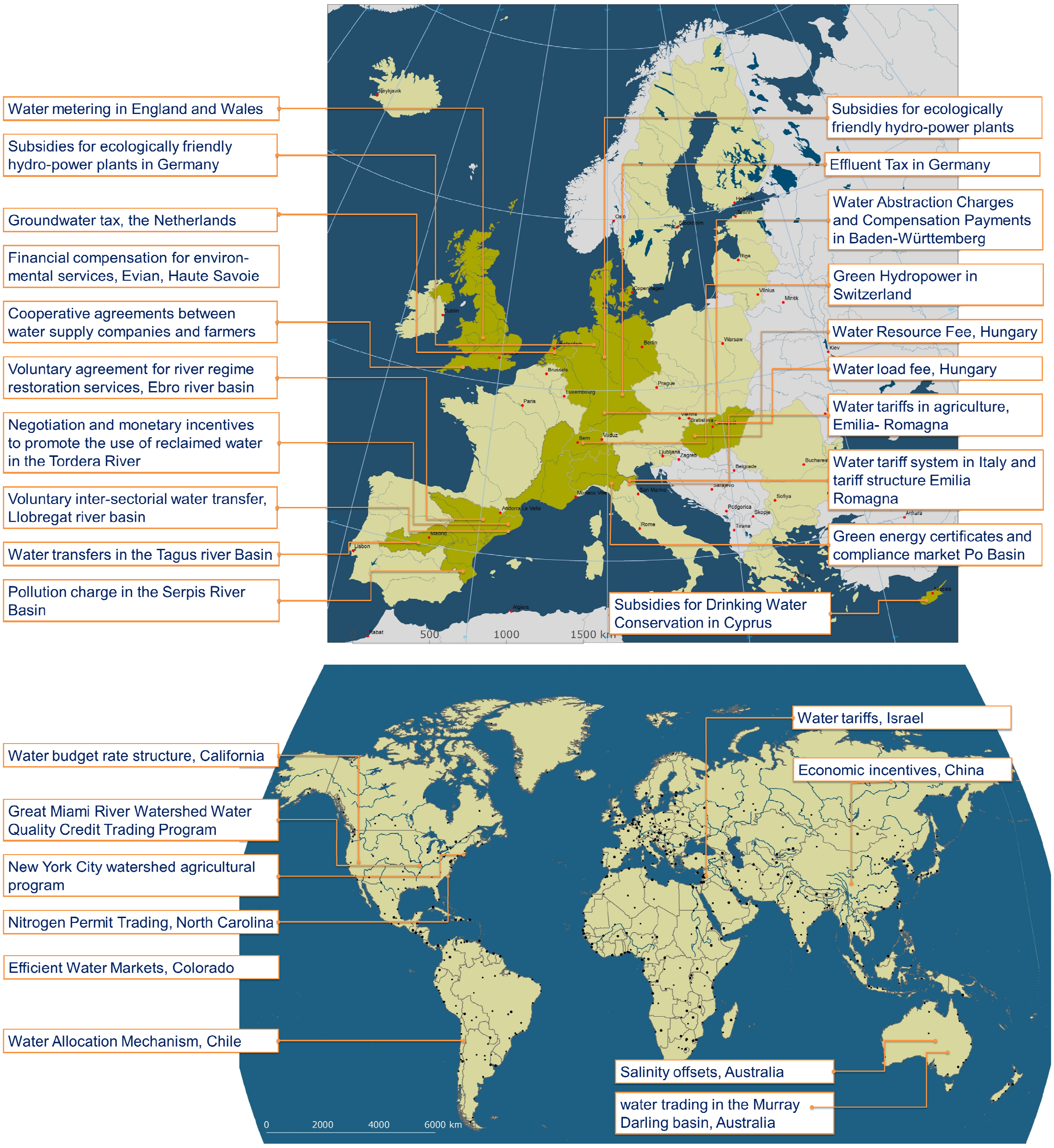

Figure 9: Geographic distribution of the reviewed EPIs (ex-post mode)

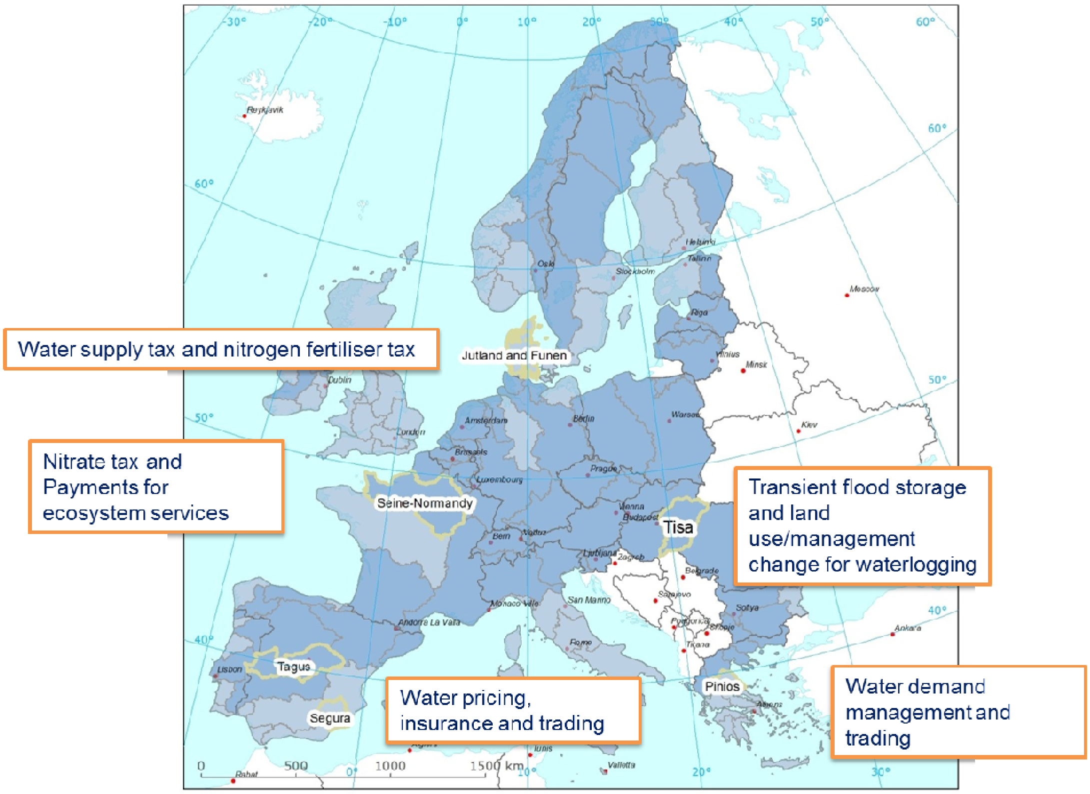

Figure 10: *Geographic distribution of the reviewed EPIs (ex-ante mode)*