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An integrated assessment framework for the analysis of multiple stressors in aquatic ecosystems and the appraisal of management options

A.Pistocchi, A.Udias, B.Grizzetti, E.Gelati, P.Koundouri, R.Ludwig, A. Papandreou, I.Souliotis

Abstract

The contribution illustrates an integrated assessment framework aimed at evaluating the relationships among multiple stressors and water body status for the purposes of river basin management.

The framework includes the following steps. (1) understanding how the different stressors, hence pressures, affect the status of water bodies. This entails the characterization of biophysical state variables and the definition of a causal relationship between pressures and status. Therefore this step involves interaction between experts bearing ecological understanding and experts providing models to represent the effect of pressures. (2) Identifying the relevant pressures to be addressed through appropriate measures in order to improve the status of water bodies. Although in principle pressures should be addressed altogether in order to seize the benefits of synergies, we propose to consider their reduction one category at a time. (3) Evaluating reduction targets for the relevant pressures identified in a river basin, on the basis of a proposed, simple multicriteria optimization approach requiring a qualitative weighting of the effort associated to reducing individual sources of pressure and the assessment of potential benefits in terms of water body status. This method produces frontiers of optimal trade-offs between effort spent on measures and achievements. (4) Designing management measures through a creative process and political discussion of alternative options balancing costs, benefits and effectiveness based on engineering and economic analysis. (5) Simulating scenarios of implementation of a programme of measures in order to check their effectiveness and robustness against climate and land use change.

We discuss the different steps by highlighting how the interaction between science and policy unrolls at the different steps, we review the assessment tools required at each step and we propose, for step 3, a simplified approach to setting optimal pressure reduction targets based on semi-quantitative assessment of the entailed efforts.

Keywords

Multiple stressors, water bodies' status, Water Framework Directive, Decision support, river basin management measures, optimization

Introduction

Aquatic ecosystems are threatened by a variety of pressures, such as water abstraction, organic and inorganic pollution, invasive species, pathogens, geomorphological alterations, land use and climate variabil- ity and change (Vörösmarty et al., 2010). Pressures of such diverse na- ture have adverse effects on organisms and ecosystems and ultimately threaten water quality and biological diversity. While pressures can op- erate on very different spatial and temporal scales (Brown et al., 2013), their impact strongly depends on intensity, timing and duration of ex- posure (Stevenson and Sabater, 2010). These highly complex multi- ple-stress situations can cause synergistic or antagonistic net effects (Coors and De Meester, 2008; Ormerod et al., 2010; Jackson et al., 2016). Despite increasing awareness and efforts to target this complex- ity, current scientific knowledge, especially for real world applications, is largely limited to the impacts of single pressures on the ecological sta- tus of water bodies and ecosystem functioning. Deficits remain in the capacity to deal with large variability in the available data and in disentangling the effects of different stressors to confidently infer which ones cause unacceptable environmental change (Downes, 2010). Thus, while it is possible to determine a hierarchy of pressures in interacting with aquatic biota (Palmer et al., 2010; Woodward et al., 2010), there is still a strong limitation in the capacity to predict the re- sponses of aquatic ecosystems to multiple pressures (Friberg, 2010; Petrovic et al., 2011; Heathwaite, 2010). Further, the combined result of multiple pressures can eventually create new and most likely very pronounced effects of unknown consequence (Navarro-Ortega et al., 2015). Tockner et al. (2010) call for management approaches to be adaptive and embedded within a catchment concept to deal with up- coming pressures originating from global change. The interactions of local (e.g. nutrients) with global stressors (e.g. climate change) must also be considered to identify sites where the management of a local pressure will provide the greatest impact reduction (Brown et al., 2013). However, with lacking knowledge on causalities, management options to conserve, rehabilitate or ameliorate aquatic ecosystems will remain vague (Downes, 2010).

Yet the European Union’s Water Framework Directive (WFD) 60/2000/EC stipulates that water bodies must, by 2015, achieve “good status” corresponding to certain normative definitions, and requires Member States to set out river basin management plans to ensure this. When the status is less than "good", the responsible anthropogenic pressures must be identified, and adequate measures must be taken to restore the water bodies' conditions. The WFD admits less ambitious targets than good status for certain water bodies, if the costs of the required measures are disproportionate to benefits.

Therefore, river basin management planning requires an assessment framework making causal relations explicit between multiple pressures and water bodies' status in order support the design of a programme of measures considering their costs and benefits for different water users and the environment (including the market and non-market value of healthy water resources and aquatic ecosystem services) in a transparent and sound way.

River basin management, as a socio-political endeavor rooted in the activity of planning, inherently addresses *wicked problems* as famously characterized by Rittel and Webber, 1973, and cannot be simply treated as a scientific or technical question with a univocal and optimal answer. On the contrary, it entails an *argumentative process* (Kunz and Rittel, 1970) where the community develops a discourse on how it would like the river basin to be; it brings up, discusses and debates issues to arrive at an agreed-upon strategy; and, in the course of this process, poses questions of fact to experts, discusses the answers and may turn them back into new issues. *"Through this counterplay of questioning and arguing, the participants form and exert their judgments incessantly, developing more structured pictures of the problem and its solutions. It is not possible to separate "understanding the problem" as a phase from "information" or "solution" since every formulation of the problem is also a statement about a potential solution.*"(Kunz and Rittel, 1970, p.2).

Effective scientific support to such an argumentative process requires first of all to clarify which *questions of fact* need to be answered in the assessment framework, while fully acknowledging the cooperative nature of the work to be supported. We propose to organize the assessment in five steps, each requiring to harness specific scientific knowledge (Figure 1). The first step is the analysis of the watershed and the *classification of the status of water bodies*. At a second step, the *most relevant pressures* causing less-than-good status should be identified. A third step addresses the evaluation of *pressure reduction targets*. These targets are the basis for the fourth step, i.e. the *design of appropriate management measures*. Initially, the water bodies are evaluated under the current status. Socioeconomic and climate change scenarios can be considered, in order to test the impact of evolving drivers on the conditions of the river basin and on the performance of the measures. The *simulation of these scenarios* may be seen as an additional step, making the framework iterative, as sketched in Figure 1. In the following, we discuss the different steps highlighted above and the tools that may be used in the assessment. The assessment framework does not necessarily require the development of dedicated supporting software because the choice of the specific modelling tools is relatively marginal in the overall process. As a partial exception, in this contribution we propose and discuss a simple method to optimize pressure reduction targets, treating reduction costs in a semi-quantitative way, specifically designed for use in participatory processes.

Understanding water bodies' status and pressures

Classifying a water body's status and identifying the most relevant controlling pressures are the first two steps of the assessment. The evaluation of a water body's status is a multi-criteria judgment (see Figure 2). *Criteria* can be defined by law: for instance, Annex V of the WFD relates the ecological status of surface water bodies with the criteria of flow regime, river morphology and water quality. These criteria must be represented by *indicators*, in turn evaluated from appropriate state variables of the water bodies (e.g. suspended sediment concentration; hydrological descriptors of flow regime). Indicators often take the form of a comparison between state variables with reference values, such as environmental flow requirements negotiated using structured procedures (Stalnaker et al., 1995; European Commission, 2015), or environmental quality standards for chemicals and nutrients. Morphological aspects are usually addressed through situational judgment (e.g. Gurnell et al., 2015; Rinaldi et al., 2015), as the complexity of the processes involved is extremely high.

Indicators of the individual criteria must be eventually combined in some form of synthetic indicator. The simplest form of such synthetic indicator is the worst among water quality, morphology and flow regime indicators as representative of the overall status (the so-called "one out – all out" approach). However, sometimes there are interactions among the individual criteria indicators, that can be captured using statistical modelling techniques (e.g. Grizzetti et al., 2016), or composite indicators (e.g. Vörösmarty et al., 2010, addressing threats to biodiversity and human health at the global scale). Irrespective of the method adopted, the explanatory value on status of combinations of multiple indicators can be tested statistically (see e.g. discussion in Chung and Fabbri, 1993).

State variables, such as pollutant concentrations or flow regime descriptors, may be directly related to individual pressures using models of hydrological or biogeochemical processes. An inventory of the pressures controlling the state variables is therefore a necessary prerequisite in order to enable reasoning on the measures to be taken for water bodies' status improvement. Pressures may be known only in part due to data incompleteness, or may correspond to assumed future scenarios (e.g. for land use or climate). Biophysical process models can be used to simulate the change of indicators with the pressures, and to identify which pressures are relevant and should be addressed with measures (Figure 2). However, process models do not need to be mathematical simulation tools at this step, as the key aspect remains to understand the relevance of status-pressures relations at a conceptual level. Therefore, also qualitative evidence of these relations may suffice.

Setting targets for action

Once the relevant pressures have been identified, a decision is needed as to which extent we should reduce those pressure hampering the achievement of water bodies' status objectives.

Although, ideally, we would like to have "good status" in every water body, this may entail costs that the society is not willing to pay. At this step of the assessment, we aim at identifying a frontier of optimal trade-offs (or "Pareto front") between a given level of pressures and the effort required to achieve it. "Effort" is meant as the financial, social, or technical difficulty of acting on a specific pressure. It represents a genuinely economic concept, but cannot be necessarily expressed in monetary terms. For instance, reducing pressures on flow regime due to abstractions by a water-intensive industry in a certain region may be a cost-effective solution to improve water bodies' status, but threats by the company to discontinue production, hence cause unemployment, may bring water managers to choose other solutions. Valuing the effort is in essence a political decision and usually rests on non-monetary as well as monetary values.

In principle, actions on pressures addressing different status criteria (e.g. morphology, flow regime and water quality for rivers) may be synergic: for instance, in a water body suffering from flow regime alteration as well as pollution, high concentrations of contaminants can be exacerbated by reduced dilution due to abstractions. In such a case, treatment of pollutant emissions can be more effective if combined with restriction of abstractions. However, most management measures have effect predominantly on one single pressure. Therefore, as a first approximation we may look at each individual status criterion independent of the others. Consequently, pressure reductions can be optimized separately. In this section, we propose a simple optimization method applicable to many problems of practical interest.

A simple method to optimize pressure reduction targets

Let us consider a numerical water body status indicator for a given criterion. Without loss of generality, let such indicator increase as the status of the water body deteriorates. For instance, this indicator may be chosen as a chemical concentration for the criterion water quality, or the ratio of yearly abstracted volume to available renewable water volume (the "water exploitation index", WEI), for the criterion flow regime alteration. Furthermore, let the indicator have a threshold representing good status (e.g. a standard for the concentration of nutrients).

Within a river basin, with regard to one criterion, we minimize the extent of violations of thresholds for good status. This means minimizing the following objective function:

$$J = \sum_{i \in River\_basin} max\left(0, K_i - \widehat{K}\right) L_i \qquad \text{(eq.1)}$$

K_i representing the indicator for the status criterion under consideration, for the generic i[th] water body in the river basin, L_i a weight reflecting the importance of the water body (e.g. its size or ecological or use value), and $\widehat{K}$ the threshold value of the indicator for good ecological status.

We then need to relate K_i to a set of pressures, {P_1, …, P_n}, that are known to drive the value of the indicator. For instance, if K_i is a chemical's concentration (e.g. in kg/m^3) and $\widehat{K}$ its threshold for good status, {P_1, …, P_n} may represent *n* chemical emission sources. If the relationship between pressures and the indicator is additive, the problem can be formulated in a particularly tractable way. For instance, if we consider as status indicators K_i the concentration of a chemical, under assumption of constant chemical pollution emissions we can write a simplified relationship between pressures and status indicator (see Pistocchi, 2014, chapter 7):

$$K_i = \sum_{j=1}^{n} \frac{P_j \exp(-\gamma t_{ij})}{Q_i} \qquad \text{(eq.2a)}$$

where P_j represents the chemical's emission intensity (e.g. in kg/s) from the jth source, t_{ij} the time of travel of water from emission j to water body i (to be considered ∞ if the source does not affect the water body), Q_i the discharge in the water body and γ the chemical decay constant. If we consider as status criterion the flow regime, and as indicator the WEI, we have:

$$K_i = \sum_{j=1}^{n} \frac{P_j \delta_{ij}}{Q_i} \qquad \text{(eq.2b)}$$

where now P_j is the abstracted discharge at site j and δ_{ij} the Kronecker delta equal to 1 if abstraction j is in the catchment upstream of water body i, or 0 otherwise. For additive pressures, a reduction of each of the pressures by a given factor combines linearly with reductions to the other pressures in the resulting value of the status indicator. For chemical concentrations, eq. 2a becomes:

$$K_i = \sum_{j=1}^{n} (1 - \alpha_j) \frac{P_j \exp(-\gamma t_{ij})}{Q_i} \qquad \text{(eq.2a')}$$

while for WEI, eq. 2b becomes:

$$K_i = \sum_{j=1}^{n} (1 - \alpha_j) \frac{P_j \delta_{ij}}{Q_i} \qquad \text{(eq.2b')}$$

where α_j is the rate of reduction of pressure P_j. Under baseline conditions, apparently, α_j=0.

The minimization of J is sought through variation of each pressure P_j, under constraints given by:

- A maximum level of the rate of reduction of each pressure, $\hat{\alpha}_{j_i}$, corresponding to what is technically achievable, i.e. $\alpha_j \geq \hat{\alpha}_j$,for $j \in \{1, \dots, n\}$
- A maximum level of the effort affordable for pressure reduction, C.

The effort constraint may be conveniently represented as a weighted combination of the rates of reduction of individual pressures:

$$C = \sum_{j \in \{1,\dots,n\}} w_j \alpha_j \text{ (eq.3)}$$

w_j being a weight interpreted as the relative difficulty of reducing pressure j of a given rate, compared to the other pressures. Weights must reflect the relative difficulty of reducing to the same degree one pressure in comparison with another. They can be elicited through a process of stakeholders' involvement and/or experts' consultation, based on standard multi-criteria analysis techniques used in participatory decision making.

The mathematical optimization problem presented above can be then solved using standard methods, to yield a set of optimal reduction rates α_j for a given value of C. By repeating the optimization for different values of C we may draw the optimal frontier of trade-offs between effort and status. For several practical cases, this approach may be useful as a first approximation, although the effect of pressure indicators may not always be additive, and more sophisticated techniques may be necessary (e.g. through a response surface methodology approach: Box and Wilson, 1951).

One problem with this formulation is that the set of pressures should be reasonably limited in order to keep control on the computational burden. Consequently, it may not be possible to consider individual pressures such as emission points when they become large numbers. This computational restriction corresponds also to a limitation in the design of reduction targets: these must very often be general and not specific, as they apply to whole economic sectors defined by law, within regions or countries. In practice, for chemical concentrations we may consider a relatively small number of emission patterns, i.e. maps of emissions from different categories of sources, and calculate their effect in terms of concentration in virtually every point in the basin through weighted flow accumulation and flow length functions in a GIS (see e.g. Pistocchi et al., 2012, and Pistocchi, 2014, ch. 7). For abstractions, we may consider a limited number of abstraction maps for selected categories and have their effect in terms of WEI everywhere in a catchment, also through weighted flow accumulation.

Example application

A purely illustrative example may help to illustrate the concept. Let us consider one of the several river basins in Europe, the Po river basin in Northern Italy (Figure 3), where the WEI is calculated for each sub-basin as the ratio of cumulative upstream annual abstractions on annual availability. The latter is determined using a simple Budyko-type model (Pistocchi et al., 2015). Abstractions according to de Roo et al., 2012, are considered from the following sectors: agriculture, livestock breeding, industrial use, energy production and households. Both availability and demand are presented in Figure 4. The river basin is administratively divided in regions (Figure 3) with legal power to set specific abstraction limitations. Let us assume that three of these regions (labelled in Figure 3 with their identifier numbers), representing the most important pressures, agree to set targets to reduce abstractions so to minimize the WEI over the whole basin. In this case, for each water body in the river basin, under baseline conditions we can compute the water body indicator for flow regime alteration $K_i = \frac{P_j \delta_{ij}}{Q_i}$ and, if we aim at minimizing alteration per se without a specific threshold considered acceptable, $\widehat{K} = 0$. We may also assume, for simplicity, that all water bodies in the river basin are equally important, hence L_i=1 for any i.

We want to compute optimal abstraction reduction rates α_j, in each of the three regions, for each of the 5 economic sectors. This means we consider a total of 15 pressures {P_1, …, P_{15}} (Table 1). If reducing abstractions is equally difficult in all sectors, the weights are set to a constant value. We set the constraint of a sector-specific maximum reduction rate equal for all three regions (Table 1). With a standard optimization algorithm, we obtain the optimal trade-off frontier represented in Figure 5. Each point of the frontier corresponds to a set of reduction targets $\{\alpha_1, \ldots, \alpha_{15}\}$ for each sector and for each region (Figure 5). If we now repeat the exercise considering that not all sectors have the same social, economic or political priority but they follow a hierarchy represented by the weights in Table 1, we obtain the results summarized in Figure 6.

First of all, it is worth noting that both cases yield relatively similar pressure reduction targets. The logics are constantly to address the most important pressures first, and then marginally less relevant pressures. In this specific problem, the frontier is relatively linear and it is therefore more difficult to identify a discontinuity in the marginal benefits of reducing pressures.

As expected, the higher C, the lower J. For increasing C, larger parts of the river basin have their WEI reduced compared to the baseline scenario (BLS) reflecting unreduced pressures, as can be seen from the maps in the middle and lower right panes of Figure 5 and Figure 6).

For instance, if we spend an effort corresponding to point A, the most efficient solution for equal priorities is achieved with most of the reduction to domestic and industrial abstractions in region n. 212 followed by reduction to energy abstractions in region n. 215 (Figure 5 middle left). If we differentiate priorities, the reduction to the domestic sector in region n. 212 becomes much less important (Figure 6 middle left).

If we increase the effort up to what corresponds to point B, the reduction rates increase more homogeneously across regions and sectors, while still identifying higher reduction targets in region n. 212, having a higher basin-wide influence due to its being upstream (Figure 5 and Figure 6 bottom left).

Once a frontier is available, the planner may decide to invest more effort and to achieve better conditions of the water bodies, or to invest less and be content with worse conditions. This decision does not stem from a mathematical optimization, but depends on the value socially attributed to a given status of the water bodies, requiring an appreciation of the implications of choosing e.g. between the two example points on the frontiers shown in Figure 5 or Figure 6.

The method proposed above allows analyzing the implications of a higher or lower level of effort for the improvement of water bodies' status, taking into account the relative difficulty to tackle individual pressures in combined social, economic and technical terms. It allows understanding in a semi-quantitative way the marginal value of incremental efforts towards good status: usually the frontiers obtained with this approach have a shape as in Figure 5 or Figure 6, indicating high improvements on efforts at low pressure reduction rates, and marginally decreasing improvements on efforts at high rates. By design, it is therefore a method to support the decision on an appropriate level of commitment in the design of river basin management measures. Its implementation does not require complex and detailed calculations on costs and benefits, nor complex scenario simulations, and can be easily done in the form of a web-based service.

Designing programmes of measures

The "Pareto fronts" obtained with the above method represent optimal combinations of pressure reduction targets for a given level of effort, showing the marginal improvements at different levels of commitment on individual pressures. The political decision on which point to choose on the Pareto front selects the corresponding set of pressure reduction levels. The design of a programme of measures is an iterative process of identification of technically feasible measures and evaluation of their costs, benefits and effectiveness. If the measures prove too expensive, the point on the Pareto front may be reconsidered. This process resembles more a professional's "reflective practice" (Schoen, 1983) than a scientific analysis, although it increasingly takes a cooperative and participatory direction with a more systematic involvement of stakeholders supported by experts (as narrated by Kunz and Rittel, 1970).

Although measures are very specific to the context where they are planned, and therefore always need an *ad hoc* assessment, they may be usually identified in a relatively narrow range of practical options. We propose the classification of measure types provided in Table 2. These can broadly grouped into "Technical measures" (TM) and "Regulatory measures" (RM), and encompass a very broad range of practical solutions likely including most of the concrete measures found in river basin management plans.

TM consist of a specific action at given sites with clear and pre-set design performances. For instance, a wastewater treatment plant is designed for a given target chemical concentration in the effluent or for a target percentage reduction of pollutant loads. Their effect on the stressors may be consequently quantified with appropriate scientific model calculations. This is indeed what is done in detail, at the local scale, in many design processes. TM do not need be limited to traditional civil engineering works, but increasingly include nature-based solutions such as "natural water retention measures" (European Commission, 2012, 2014; see also www.nwrm.eu) and sustainable urban drainage systems (see e.g. Pistocchi, 2015). Direct costs of technical measures (net of possible savings, e.g. due to higher energy efficiency or reduced maintenance demand) can be estimated with relatively high accuracy, with levels of detail obviously depending on the scale of application. For instance, a European-scale study (STELLA, 2012) presented reference investment and operation costs for selected types of natural water retention measures, to be used for comparative screening of strategic options. Tools such as the FEASIBLE model (OECD, 2004) can be used for this purpose. On top of these costs, business and consumers may be affected. For instance, the reconnection of a floodplain may entail relocation of economic activities or the development of a wastewater treatment plant may lead to an increase in local water tariffs.

RM are provisions for the limitation of water use, emission of pollutants, adoption of good practices, restriction of land use in a given catchment, economic instruments or incentivizing conditions for voluntary approaches (see also Grizzetti et al., 2015, Table A5.2, p.94, Koundouri, 2006). Unlike TM that are completely defined by their design specifications, the cogency and enforceability of provisions and the detail of the aspects addressed make the effectiveness of RM and the quantification of their potential outcomes much more uncertain. For instance, a provision such as "increase irrigation efficiency from 70 to 90%" could be mandatory by law or it could be a prerequisite for the receipt of subsidies; in the former case, we can assume that 100% of the water users will conform to it, while in the latter we remain uncertain about the extent of take up. The direct costs of RM include additional administrative, management and monitoring activities, as well as business and social impacts of implementing the provisions: for instance, measures requiring specific investments by the industry may increase production costs, hence competitiveness; limitations imposed to pesticide or fertilizer application in agriculture may impact crop productivity or labor intensity of agriculture; limitations on water abstractions may cause consumer welfare loss that may be evaluated through water demand functions (see Reynaud, 2015); water pricing measures may negatively impact low-income households; etc. It should be stressed that a negative impact in one single sector may be compensated by a positive one in another sector, and a difficulty in the assessment lies exactly in the integration of all the relevant sectors affected.

For a given cost including the above components, measures are expected to deliver socioeconomic benefits due to improvements of water bodies' status. The total economic value of benefits stemming from status improvements should reflect its use and non-use value components (see for example, Pearce et al., 2006). Use values can be broken down to direct and indirect use values, whereas non-use values can be decomposed to option, existence and bequest values (Koundouri, 2009). Direct and indirect values relate to the direct or indirect consumption of a good and could be, for instance, related to income that fishermen gain through using resources of a river or health benefits that are associated with water quality. Non-use values include the "option" value (the value that individuals place on sustaining a river for the possibility of using it in the future, or the opportunity cost between current and prospective uses of the river), the "existence" value (e.g. the satisfaction that individuals obtain from knowing that a river exists and that its ecological status assures its sustainability), and the "bequest" value (e.g. the utility that individuals place on their children having access to the river in good ecological status). Brouwer (2008) used economic valuation to estimate benefits that were input into a cost benefit analysis for implementing measures for the enhancement of chemical and ecological quality in Netherlands. 60- 70% of the total investment cost regarded improvement in the chemical status of the water, whereas the rest concerned restoration projects. In the stated preference survey, the maximum implementation scenario concerned 100% achievement of the WFD objectives, while the medium and low implementation scenario, implied less than 100% achievement of the WFD objectives. He finds that households were inclined to pay approximately 22% more of what they already paid for improved water management and that the discounted benefits were as high as the cost of the maximum implementation scenario.

The economic valuation techniques used to estimate the above values can be classified in two broad categories, depending on whether the information they use to. Revealed preference techniques are restricted to estimating use type values, since their application requires market data (for a more thorough discussion see Freeman et al., 2014). In contrast, stated preference techniques use data drawn from surveys, which are used to describe hypothetical situations, aiming to estimate the economic impact ex ante and ex post, and can capture both use and non-use values. The results of studies based on revealed and stated preference techniques (primary studies) can be utilized by value transfer techniques (e.g. Koundouri et al., 2016, estimating the impact of changes in the provision of ecosystem services in the Anglian river basins in the UK, drawing on results from studies in other regions).

Birol et al., 2006, undertook a cost benefit analysis of suggested measures, to determine the best economic outcome for the enhancement of the environmental quality of Cheimaditida wetland. Some of the considered choice were: adoption of water-saving irrigation technologies and construction of a dyke, construction of a waste water treatment plant to improve water quality, educational seminar for farmers on theory and practice of sustainable agriculture and switching to non-irrigated crops. Besides estimating the costs of each action, the authors used a choice experiment (a survey based method that accounts for the different characteristics of each action that asks respondents to state their willingness to pay for different management options). They found that investing for higher improvements maximized welfare gains, which were far greater than the cost of measures.

Ramajo-Hernández and del Saz-Salazar, 2012, used another stated preference technique, contingent valuation (a survey-based approach that presents different whole management scenarios and not as a bundle of their characteristics that is case of choice experiments and asks respondents to state the price they would be willing to pay for the scenarios to be implemented– Bateman et al., 2002), to estimate the value of achieving improvements in the water quality of the Guadiana river basin in Spain. The results indicated that the respondents were willing to accept a price increase in their water bill, if the water quality of the river would be improved.

Koundouri et al., 2014 estimated the socioeconomic benefits arising from mitigating industrial production in the Asopos river basin using the value transfer approach. Using the results of previous studies and adjusting for the year of data collection and the currency (using the Consumer Price Index) for the policy site country, they found that the monetized benefits from moving from “bad” to “good” water ecology were

€116.94 per household per year. Drichoutis et al., 2014, used laboratory experiments to assess whether respondents were willing to pay a price premium to consume products from regions with water of better quality to avoid health risks. The experiment was implemented in a controlled environment, where a sample of consumers was selected to participate in hypothetical (respondents bid for a good hypothetically produced in the study area) and real auction rounds (respondents bid for a real good). The study found that respondents would prefer to pay more to consume agricultural products from less polluted areas than areas in bad ecological status.

Simulation of implementation scenarios

Once a programme of measures is designed, it may be useful to use biophysical models to simulate the status indicators under a scenario of its full implementation, to check if it actually achieves the objectives. Moreover, a robustness check of the measures can be obtained under climate and land use change scenarios, also using model simulations. This prompts for iterating the above steps on the basis not just of the baseline conditions of the water bodies, but also their likely projections.

Ralf?

Summary and perspectives

We have proposed an integrated assessment framework for the analysis of multiple stressors in aquatic ecosystems and the appraisal of management options. We advocate that such a framework should help organizing the use of scientific knowledge in support to river basin management planning, that we describe as an argumentative process where the scientific and professional knowledge available must be deployed in a context driven by societal values and political priorities. Designing management measures cannot be framed as a scientific exercise, but requires creativity and social endorsement. There are specific steps at which the process is driven by scientific knowledge: the identification of the relationship between pressures and water bodies' status, the calculation of pressure reduction targets, the questions on facts emerging when considering different management options. Most of the process of river basin management planning, however, is driven by political discussion: about the value of aquatic ecosystem services and willingness to invest in river basins, the relative difficulty of regulating one or another economic sector, the elicitation of preferences for different types of measures, etc.

In early stages of contemporary river basin management, there has been a tendency to emphasize the need for the development of hydrological modelling software tools or operational model setups. Nowadays, in many river basins individual models may already exist and many global or macroregional models are equipping themselves to address more and more detailed problems, e.g. in a hyper-resolution modelling (Bierkens et al., 2015) or multi-scale modelling (Samaniego et al., 2014) perspective. The idea of collectivizing the hydrological modelling enterprise through "community models" has been recently discussed in Weiler and Beven, 2015. At the same time, hydrological modelling has been increasingly integrated with ecological (Zalewski et al., 1997), social (Sivapalan et al., 2012) and economic (Harou et al., 2009) modelling. The development of modelling and decision support software tools has generated impressive potentials to improve our capacity to address scientific questions, which always carry some risk of technocratic *hybris*. In order to fully seize the opportunities from these potentials, adequate attention should be paid to the structuring of decision processes where the scientific knowledge is to be used, by clearly acknowledging the respective role and place of science and political discussion.

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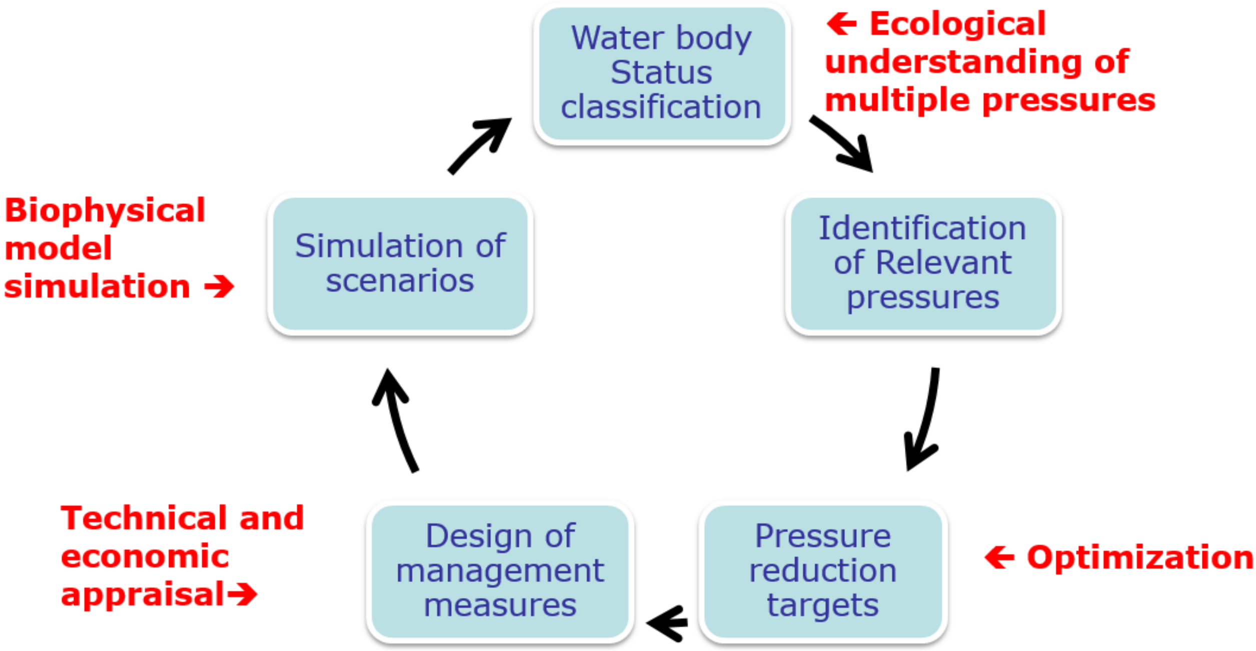

Figure 1 – structure of the proposed assessment framework and knowledge mobilized at the different tiers.

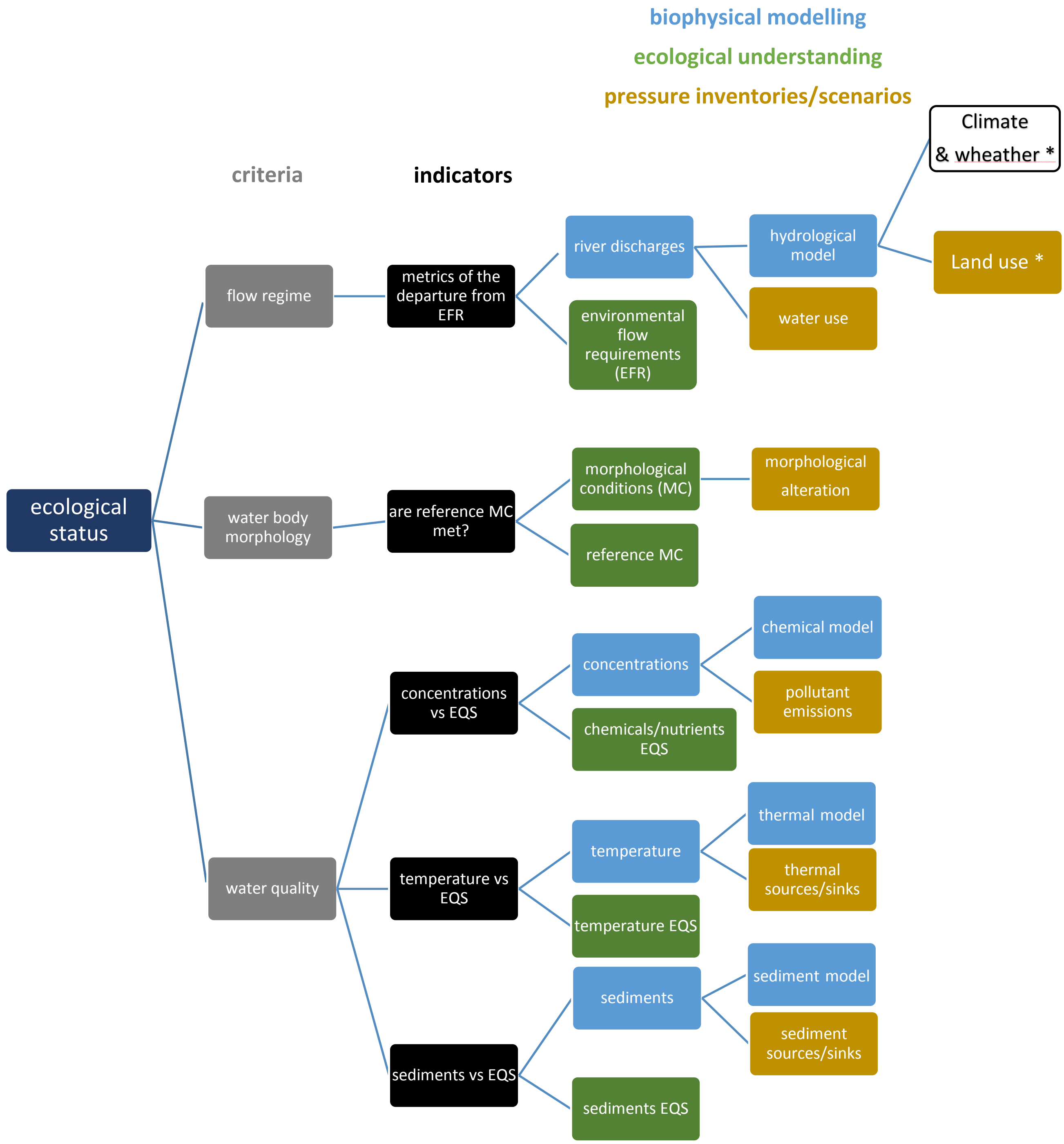

*Figure 2 – Organization of knowledge to classify water bodies' status: example with river ecological status under the WFD, entailing biophysical modelling for selected processes and variables; expert judgment for interpretation; assumption of pressure scenarios; construction of criteria indicators and their combination (*scenarios themselves may be a result of modeling, particularly for future climate forcing). The scheme corresponds to the classification criteria of the WFD (Annex V).*

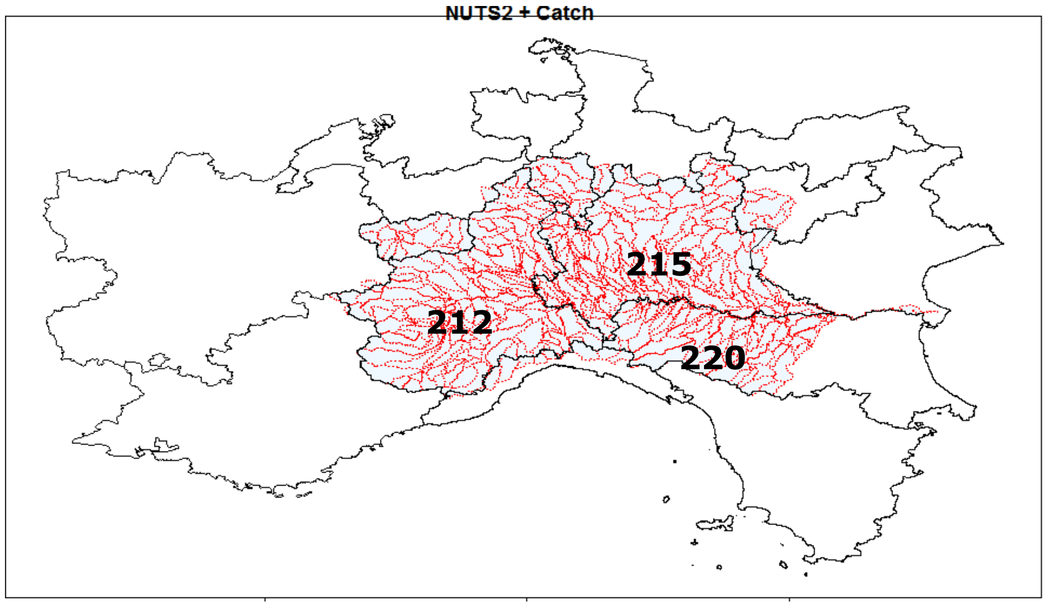

Figure 3 – Example river basin considered for the optimization. The example is merely illustrative and does not correspond to an assessment of any specific policy.

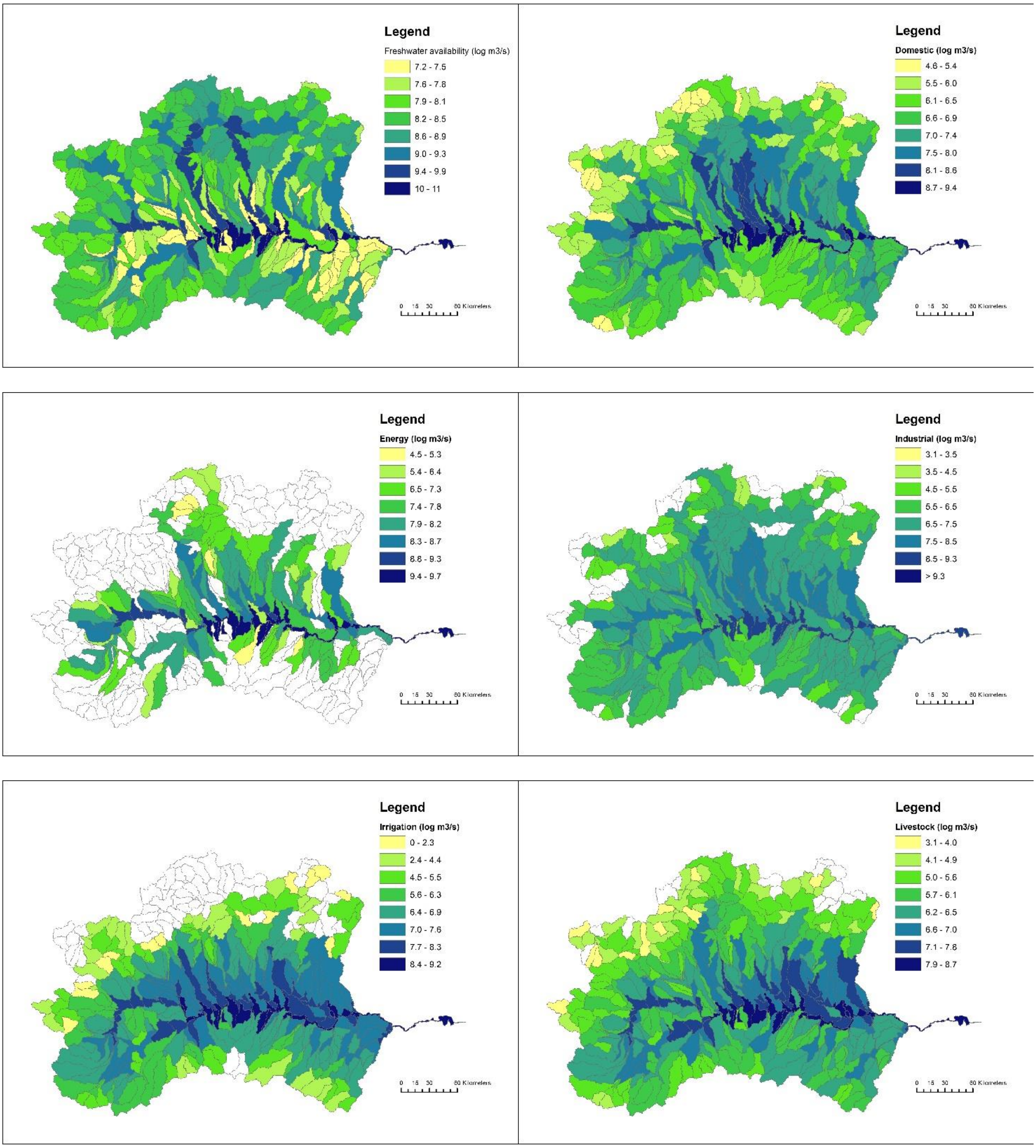

Figure 4 – annual average availability and demand for the example catchment

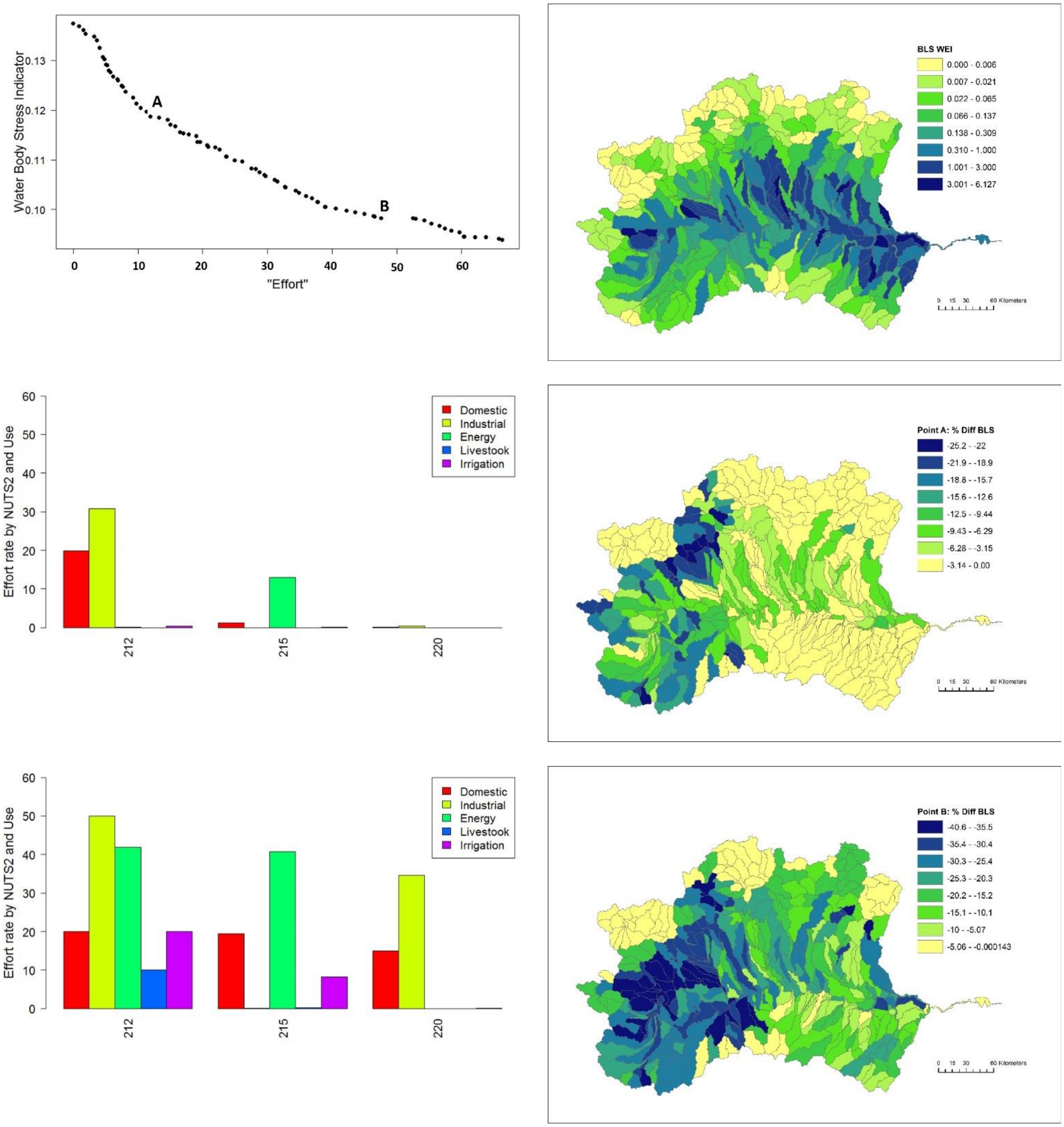

Figure 5 – example WEI minimization when all abstractions in all sectors are equally difficult to reduce. Left column, from top down: optimized frontier and optimal pressure reduction targets for the different regions and sectors at point A and B of the frontier, respectively. Right column, from top down: WEI under baseline scenario (BLS) and % variation of WEI in the different parts of the river basin, for point A and point B of the frontier respectively.

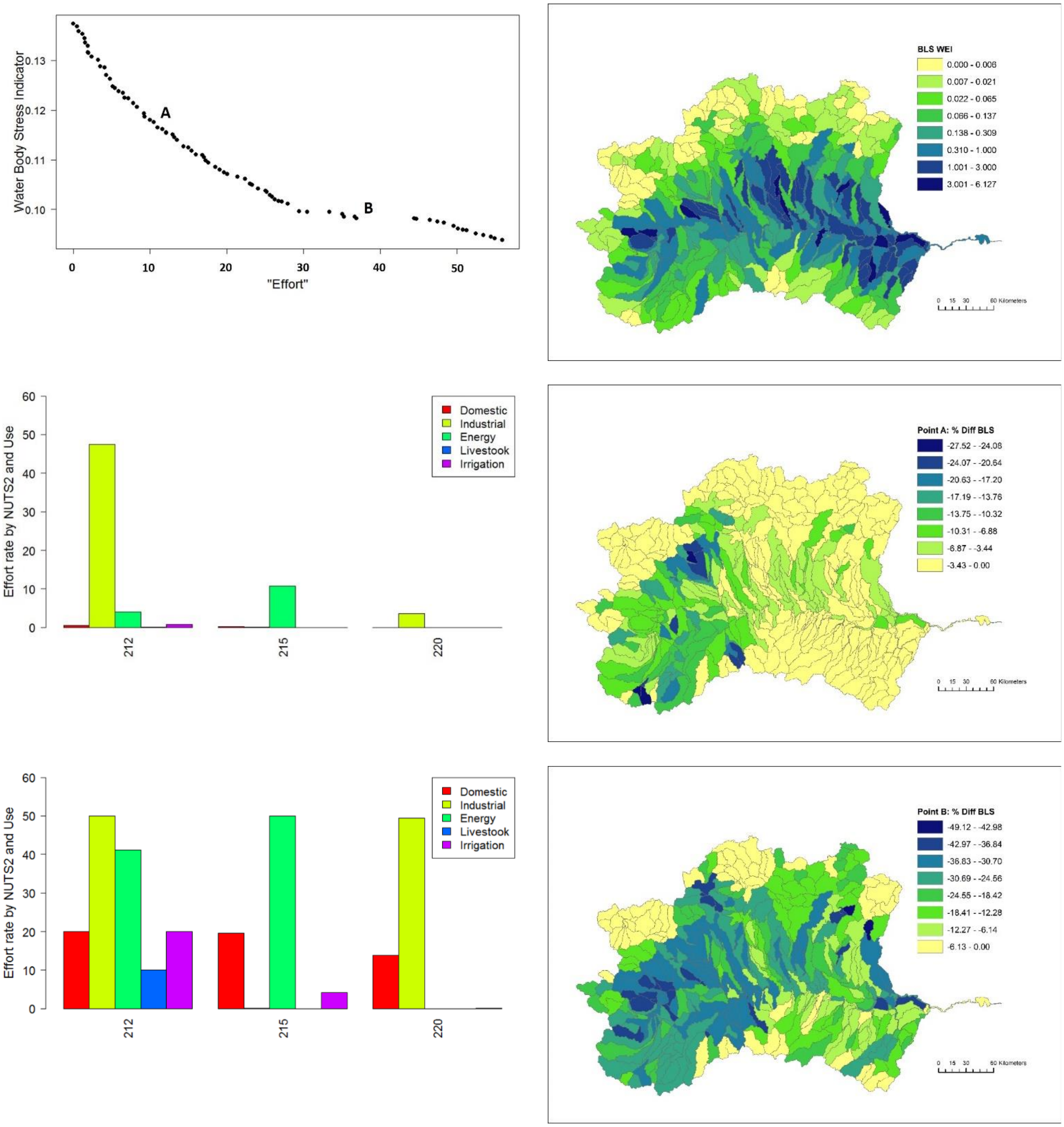

Figure 6– example WEI minimization when abstractions in different sectors have different social, economic or political priorities. Left column, from top down: optimized frontier and optimal pressure reduction targets for the different regions and sectors at point A and B of the frontier, respectively. Right column, from top down: WEI under baseline scenario (BLS) and % variation of WEI in the different parts of the river basin, for point A and point B of the frontier respectively.

Pressure	Region	Sectors	Max reduction Rate (%)	Weight, scenario with different priorities	Weight, scenario with equal priorities
P_1	212	Domestic	20	0.25	0.2
P_2	212	Industrial	50	0.15	0.2
P_3	212	Energy	50	0.3	0.2
P_4	212	Livestoock	10	0.05	0.2
P_5	212	Irrigation	20	0.25	0.2
P_6	215	Domestic	20	0.25	0.2
P_7	215	Industrial	50	0.15	0.2
P_8	215	Energy	50	0.3	0.2
P_9	215	Livestoock	10	0.05	0.2
P_{10}	215	Irrigation	20	0.25	0.2
P_{11}	220	Domestic	20	0.25	0.2
P_{12}	220	Industrial	50	0.15	0.2
P_{13}	220	Energy	50	0.3	0.2
P_{14}	220	Livestoock	10	0.05	0.2
P_{15}	220	Irrigation	20	0.25	0.2

Table 1 – weights adopted for the Po river basin optimization example.

Quality elements	Technical measures	Regulatory measures
Flow regime	Rainwater and runoff harvesting, by storage of water during high flows for its use during low flows, in order to avoid abstractions from water bodies under hydrological stress; Plants for the use of alternative water sources in order to reduce abstractions from water bodies under stress; particularly, a source with virtually zero impact on freshwater bodies is desalinated seawater.	Use of water pricing as an incentive to water use reduction (taxes and subsidies); Prescription of water efficiency and water reuse standards, limitation to abstractions; Limitations to catchment land use (soil sealing, artificial drainage, forests, etc.) Voluntary agreements between polluters and the government.
Water quality	Treatment of wastewater (e.g. urban, industrial, combined sewer overflows) through appropriate wastewater treatment plants, their upgrade and/or combination with constructed wetlands or other natural systems; Cleanup or reclamation of contaminated sites.	Restriction of fertilizer and pesticide use Catchment land use (set-aside of riparian buffers, prescription of stormwater detention areas, drinking water protection zones etc.) Voluntary agreements between polluters and the government. Liability for Damage, which is a mechanism that can control for damage that has already been realized, although it can preventively as well.
Morphology	Design of river and lake restoration, including removal of dams and flood protections, re-meandering of rivers, floodplain reconnection, construction of fish passes, etc.	Catchment land use (restriction of urban and infrastructure development in floodplains and other morphologically active landscapes).

Table 2 – summary of the possible types of regulatory and technical measures