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Out of sight, not out of mind: developments in economic models of groundwater management

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1. Introduction

Access to water is essential for human civilizations. Thus groundwater, albeit hidden from view in most places, has been actively sought out since the earliest human settlements, especially in locations where surface water was insufficient or highly variable. There are historical records of water wells, built to reach groundwater, since 8600 BCE. Wells appeared independently in different continents as part of the primitive built water infrastructure, which also included ditches, canals, dams and embankments (Hassan, 2011). Water technology evolved along with other scientific advances and there have been significant developments in hydraulic engineering in the last two centuries which, combined with a growing population and increasing levels of economic activity, led to ever more groundwater being harnessed for human uses that encompass irrigation, households and industry.

Notwithstanding the difficulties in assessing the global availability of groundwater, it is estimated that its volume dwarfs all other freshwater sources (Gleeson, Befus, Jasechko, Luijendijk, & Cardenas, 2016) whereas its abstraction accounts for around a quarter of total yearly water use (van der Gun, 2012). Yet it has also become clear that increasing pressures on this valuable resource imperil its sustainable use in many areas. Although numerous aquifers are renewable resources, recharge rates are often insufficient to replenish extractions (Wada et al., 2010), leading to a drop in water tables as aquifers get depleted, with direct consequences in terms of pumping costs as well as longer-term impacts that may be irreversible, such as land compaction. Additional problems emerge in the domain of water quality. While some contamination occurs naturally, a wide range of problems are caused by human activities, including salinity (through water intrusion in coastal aquifers or irrigation-induced salt accumulation in the soil), chemical pollution and biological contamination (Giordano, 2009; Warner et al., 2016). Moreover, the quantity and quality of aquifers can be

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interdependent, as depleted aquifers can suffer from higher pollutant concentrations (Roseta-Palma, 2002).

As more attention has been directed to the “invisible” resource, it has become clear that groundwater cannot be managed in isolation. Physically, aquifers are a piece of the broader hydrological cycle, absorbing percolation from precipitation, runoff and surface storage and in turn naturally feeding springs or wetlands, thereby contributing critically to many ecosystems. Aquifer use and degradation are also linked to social, economic and institutional systems, so that groundwater can be said to be “entrenched in a web of interdependencies” (van der Gun, 2012, p. 3). Methods to understand and handle groundwater in all its complexity are therefore essential. A recent open-access book (Jakeman, Barreteau, Hunt, Rinaudo, & Ross, 2016) provides an excellent overview of the concepts and tools associated with integrated groundwater management, defined as “a structured process that promotes the coordinated management of groundwater and related resources (including conjunctive management with surface water), taking into account non-groundwater policy interactions, in order to achieve balanced economic, social welfare and ecosystem outcomes over space and time” (Jakeman et al. 2016, p.6). Nonetheless, each aspect must be understood properly before it is brought into a multifaceted model.

This survey summarizes the developments in economic models relevant to groundwater management, advancing the understanding of key theoretical concepts and policy options. Economists of all hues deal with allocation in conditions of scarcity, and dynamic models of natural resource management have been applied to groundwater for decades (Burt 1964 and 1967; Brown & Deacon 1972). Two aspects can be highlighted straight away: first, since groundwater stocks are carried over to future periods, dynamic analysis is essential and any costs and benefits included in the analysis will require discounting; second, the positive and normative aspects of management must be clarified at the outset, that is, models that are meant to be descriptive of actual situations are in general different from models that select a specific goal and use it to measure the success or desirability of outcomes. The difference is fundamental even if the results of the two model types sometimes turn out to be fairly close, as happens in Gisser & Sanchez (1980). This paper launched a whole strand of literature preoccupied with the question of whether policy interventions at least have the potential of improving groundwater management in a meaningful sense (see Koundouri 2004b for a survey). It is now apparent that the Gisser-Sanchez effect was linked to the simplified characteristics used in the model specification (Tomini, 2014). Moreover, given the well-documented parlous state of many aquifers around the world today, the focus has mostly shifted from debating whether or not intervention is worthwhile to identifying the relevant features of complex groundwater systems, designing better policies and facilitating their successful implementation.

The goal of economic analysis in groundwater is hence threefold: i) to understand the main drivers of user behavior, including demand and costs, within a given institutional framework; ii) to seek the best possible solution, in light of societal goals, given available knowledge and expected developments; and iii) to design policies whose implementation can lead to better outcomes, that is, policies which might lead situations described in i) closer to the ideal prescribed in ii).

In the following section we provide a brief overview of the most basic aquifer model, stressing the need to be clear on the decision-making framework within which the model's purpose is defined. We also indicate some directions in which the model has been extended that seek to increase its descriptive power, such as the links between water sources, benefit and cost functional forms, and interaction among users. In section three we discuss how uncertainty has been depicted in various groundwater models as well as alternatives to expected-utility theory that could be more commonly used. In section four we focus on the different methods applied to estimate the total economic value of the parameters used in groundwater models (direct and indirect values, option values, passive values) by categorizing these methods in revealed preference methods, stated preference methods, benefit transfers and laboratory experiments. In section 5 we take up the issue of real-world (as opposed to theoretical) policy choices and in section 6 we highlight the way forward.

2. The building blocks of economic models for groundwater management

At least two components must be present in all groundwater management models: a description of the hydrological aspects, whence the water supply will arise, and a module that either summarizes existing demand drivers (positive focus) or proposes goals that ought to be attained (normative focus). The simplest version of the latter is to establish that predetermined amounts of water must be supplied at least cost, in which case the allocation is said to be cost-effective. Achieving cost-effectiveness is a prerequisite for all types of allocative efficiency criteria, but it is by no means enough (for a thoughtful explanation of the efficiency concepts used in economics, see Griffin 2016, chap.2).

The hydrological element in general does not depend on the model's focus. The description of the groundwater resource comprises at least one water balance equation incorporating water stocks and flows, which in dynamic terms is commonly expressed through changes in either stock size or water-table height. For a single-cell unconfined aquifer[4], also known as a "bathtub", this could be:

$$\dot{h} = \frac{R_t - (1-\alpha) w_t}{AS} \qquad \text{with an initial value } h_0 \qquad (1)$$

where $\dot{h}$ is the evolution of the water-table height in continuous time. Height increases with recharge R_t and decreases with water extraction w_t, although part α of this extraction percolates back to the aquifer. A is the aquifer area and S is the storativity coefficient. Note that equation (1) assumes that changes occur instantaneously and uniformly throughout the aquifer, which is not a faithful portrayal of actual aquifer dynamics. Additionally, in this simple model there is no explicit consideration of linkages between the groundwater stock and other water sources. Full-fledged hydroeconomic models include more realistic depictions of the hydrological component, through simulation models such as the US Geological Service's

[4]For an introduction to the terminology used to characterize the physical aspects of groundwater models, see Shaw (2005, chap.7). A more detailed summary of the main natural attributes of hydrogeological systems is included in Pierce et al. (2016, p.643).

MODFLOW (McDonald & Harbaugh, 2003) or the European MIKE SHE (Abbott, Bathurst, Cunge, O'Connell, & Rasmussen, 1986a, 1986b). Simulation models can accommodate aquifer heterogeneity and the conjunctive use of groundwater and surface sources: they may take into account the network of natural or artificial links between water sources, boundary conditions for each stock and flow, uncertainty in recharge, discretization of the spatial domain, water quality modelling and significant impacts on related ecosystems (Harou et al., 2009; Pulido-Velazquez, Marques, Harou, & Lund, 2016). Although a full review of such aspects is out of the scope of this work, some recent examples, mostly depicting irrigation since it is by far the largest consumptive water use[5], do illustrate the vast impact of hydrological assumptions on economic model results. The importance of spatial analysis, with differentiated well yields, is underscored by several authors, who note that crop choices, pumping decisions and income distribution may vary significantly within the same aquifer area in ways that are not captured in single-cell models (Brozović, Sunding, & Zilberman, 2010; Edwards, 2016; Guilfoos, Pape, Khanna, & Salvage, 2013; Kovacs, Popp, Brye, & West, 2015; Pfeiffer & Lin Lawell, 2012). Bulatewicz et al. (2010) recognize the importance of building multidisciplinary tools and propose an Open Modeling Interface (Open MI) to improve links between hydrological, economic and agricultural models.

In terms of economic features, it is expected that users who pump water are doing so in a way that maximizes their net gains– indeed, all positive models of groundwater extraction use some version of this setup. Individual gains will include both the benefit from water use, calculated from consumer surplus for household use or production revenues for farmers, utilities or industrial firms, and the cost, which will generally depend also on the aquifer size, since lower height increases the energy necessary to pump water to the surface. That is, at each point in time every individual user is considering a problem such as:

$$\text{Max}\ \pi(w,h) = b(w) - c(w,h) \qquad (2)$$

where both benefits and costs are measured in monetary terms. Individual users may be myopic, so that they are not considering the future consequences of decisions taken in a specific moment and essentially solve the static problem (2) at each moment, or they may take into account, albeit partially, the aquifer dynamics. If decisions are taken in a dynamic model, instantaneous net benefits will be aggregated across time periods considering the relevant private discount rate. The most common functional form for benefits is quadratic, implying that marginal benefits are linear, although for a good model fit demand should be estimated based on available data for each case study (Olmstead, 2010; Scheierling, Loomis, & Young, 2006). As for costs, it is often assumed that the stock effect is multiplicatively separable from the quantity effect:

$$c(w,h) = c(h)w \qquad (3)$$

Krishnamurthy (2016) points out that this specification has two important properties: first, the marginal cost of pumping does not depend on the level of extraction; second, the stock effect

[5] Globally, agricultural water withdrawal accounts for 69% of total withdrawals (detailed data can be found in the FAO AQUASTAT website, http://www.fao.org/nr/water/aquastat/main/index.stm)

does not change within each period. The latter might be valid for applications where time is measured in small increments (in which case continuous-time models are better), but probably will not hold if time periods are entire growing seasons. Moreover, the author stresses that the separability assumption "has little theoretical or empirical support" (Krishnamurthy, 2016, p. 7). He therefore proposes alternative specifications for discrete-time models that take into account both the lack of separability and the stock changes within a given season; moreover, the suggested cost functions can also allow for the cones of depression around wells, a relevant issue especially in confined aquifers. Foster et al. (2014) describe how more realistic depictions of the agricultural production function and the irrigation process, considering well yields and intraseasonality, can have dramatic impacts on pumping behavior.

In the presence of multiple users, the most typical framework to study equilibrium outcomes is non-cooperative game theory, where each user individually tries to achieve her best result while taking into account other player's actions. The relevance of applying game theory to common-property resources (CPR) has been noted at least since Hardin's (1968) paper on the tragedy of the commons, which has the structure of a Prisoner's Dilemma since all users end up worse by acting independently. Nevertheless, within non-cooperative games solutions can still vary significantly depending on user strategies and aquifer properties (Madani & Dinar, 2012b; Negri, 1989; Rubio & Casino, 2003; Soubeyran, Tidball, Tomini, & Erdlenbruch, 2014). The sensitivity of user behavior to spatial aquifer characteristics appears even in a laboratory setting (Liu, Suter, Messer, Duke, & Michael, 2014; Suter, Duke, Messer, & Michael, 2012). Moreover, in the simpler models, users are assumed to be homogeneous, although a few authors have considered user heterogeneity and highlighted the different results that strategic interactions can yield (Erdlenbruch, Tidball, & van Soest, 2008; Koundouri & Christou, 2006; Roseta-Palma & Brasão, 2004; Saleh, Gürler, & Berk, 2011).

Dinar & Hogarth (2015) provide a survey of 294 game-theoretic applications related to water resource management, including an overview of historical trends in the use of non-cooperative and cooperative games. The latter assume from the outset coalitions among groups of players, namely the grand coalition of all players. While non-cooperative games model the strategic interactions between players, grand-coalition cooperative solutions seek the maximum aggregate benefit achievable. Such solutions are thus useful to assess the gains from cooperation and to search for alternative distributive rules (Madani & Dinar, 2012a), although it is not clear that they can adequately take into account third-party effects such as environmental damages. Also, simulating cooperative-game solutions alongside non-cooperative ones somewhat obscures the difference between positive and normative work, which we believe should always be clearly stated. Furthermore, Dockner et al. (2000, p.32) point out that the predominant view among game theorists is that "if cooperation emerges, it should be as a Nash equilibrium outcome of a non-cooperative game".

At any rate, descriptive models of user interaction should attempt to reproduce the institutional arrangements prevalent in each situation. Enduring examples of successful collective action for common-property governance have been uncovered in the literature, which "demonstrate the feasibility (but obviously not the likelihood) of robust, self-governing institutions for managing complex CPR situations" (Ostrom, 1990, p. 103). In her Nobel Prize Lecture, Ostrom summarized Institutional Analysis and Development, a framework to analyze

CPR that is compatible with game theory models of user behavior, and listed the core aspects that have been shown empirically to distinguish successful regimes from cases of failure. These aspects include clear identification of user and resource boundaries, adequate monitoring, collective-choice arrangements, sanctions and conflict-resolution mechanisms, among others (Cox, Arnold, & Villamayor-Tomás, 2010; Ostrom, 2009). Saleth & Dinar (2004) pinpoint relevant aspects of water institutions, organized in three linked categories: water law, policy and administration. Ross (2016) proposes an alternative framework for groundwater governance based on the Earth Systems Governance Project (Biermann et al., 2009), which distinguishes five classes of issues: architecture, access, accountability, adaptation and agency.

So far we have discussed the building blocks of models that intend to simulate the evolution of groundwater resources without a centralized decision-maker, assuming that any optimization that occurs is that performed by individual users who are going about their pumping activities unimpeded. Such decentralized aquifer management will not achieve the most desired solution, the so-called “social-planner solution”, considering a normative measure of aggregate Pareto efficiency in which all the net benefits generated from a given resource are monetized, added and maximized (a much less common alternative is to look for Pareto frontiers using more than one goal, see Siegfried & Kinzelbach 2006; Salazar et al. 2007; Madani & Lund 2011). In terms of water quantity, the optimal centralized solution mandates the equality between the marginal benefit gained from a unit of water and its marginal costs in each period, considering all users, now or in the future, and thus explicitly incorporating the dynamic path for the shadow value of water. Nonetheless, alternative normative goals can be proposed, based for instance on equity considerations or environmental constraints.

There are several well-known reasons for the gap between the gains in centralized and decentralized solutions in common-property use. Essentially, the problem is that users do not fully account for the *in situ* value of groundwater and therefore extract too much. The externalities created by users lead the solution astray from the equality mentioned above. This is evident in the extreme case of myopic non-cooperative users, which completely ignore the future, but externalities are present to some extent in all decentralized situations. Provencher & Burt (1993) identify the pumping-cost externality, since users will face a higher pumping cost as the stock becomes lower, the stock externality, associated with the lost opportunity of using the water in the future, and the strategic externality, when players in a non-cooperative setting pump more in order to limit other users’ extraction. There might also be water-quality externalities, given the links between quantity and quality noted in the Introduction. Additional externalities arise from two other aspects that pertain to the *in situ* value of groundwater: the buffer value of groundwater, stemming from its role as a reservoir, backing up much more variable (and uncertain) surface water sources, and the environmental functions groundwater provides, namely supporting ecosystems, maintaining other water flows and preventing land subsidence and seawater intrusion, among others (van der Gun, 2012). Sections 3 and 4 will develop these two points.

A final point to consider is that in a dynamic setting there may be differences in private and social discount rates, with the former taking higher values due to individual users’ impatience. If social discount rates are lower, future benefits count more in the social planner model, leading to less optimal extraction today. Indeed, it has been said that “the discount rate is the most important parameter in dynamic decision making" (Griffin, 2016, p. 78). The larger the planning period the more significant the present bias – it is no coincidence that the discounting debate has raged most fiercely in climate-change assessments, where time periods of hundreds of years are not unheard of. In such cases uncertainty is rife and it might be best to apply declining discount rates (Groom et al. 2005; Groom et al. 2007; Gollier, Koundouri, Pantelides, 2008; Koundouri, 2009; Hepburn et al. 2009; Arrow et al. 2013; Arrow et al. 2014). For dynamic groundwater management, proper analysis ought at least to take into account a variety of possible discount rates, for example through sensitivity analysis.

3. What if we are not sure? The impact of uncertainty in groundwater management

Given the dynamic structure of natural resource management models, allocation decisions will have to be made based on beliefs about the future, which is by definition unknown. Various types of uncertainty are rife and the literature is accordingly vast, but recent summaries can be found in (Shaw, 2016; Tsur & Zemel, 2014). In the first, the authors begin by distinguishing between situations where decision makers are unsure about the values of natural or economic parameters (ignorance) and those where there are unpredictable fluctuations in resource evolution, for example due to weather variability (exogenous uncertainty)[6]. A basic dynamic model of deterministic natural-resource management, the "canonical" model, is presented, and its results are then compared with those obtained when there is uncertainty, coming from several alternative sources. In all cases, the view is that of a centralized planner wishing to maximize the expected present value of the net benefits from exploiting a resource stock.

When uncertainty relates to the existence of a final period for resource use, for example due to a catastrophic event that makes exploitation unfeasible, the optimal extraction path, if decreasing, will certainly be more prudent than in the canonical model if the problem is ignorance. However, if the event trigger is random, yet the hazard rate that measures the event's probability of occurrence depends negatively on the stock, the overall effect on extraction depends on the relative strength of two distinct effects: the implied higher discount rate means more impatience, hence faster resource consumption (why save if the resource will be lost in the future anyway?), while the endogeneity of the hazard rate encourages conservation.

While catastrophic outcomes are liable to occur in certain aquifers (Tsur & Zemel, 2004), a second type of uncertainty, more widespread in groundwater management given the variability inherent to weather patterns and the hydrological cycle, is that of stochastic resource evolution. This may be introduced through random fluctuations in stock, modelled through a stochastic recharge in equation (1), following the general framework of Pindyck (1984). This approach is brought to bear in Zeitouni (2004), who finds that under certain conditions the optimal strategy is to aim for a certain stock level (pumping nothing if stock is

[6] This is not the traditional division between risk and uncertainty, which is described in (Shaw, 2016) and summarized below.

below this and the whole surplus if it is above). Another option is for recharge to be subject to discrete shocks. For example, in de Frutos Cachorro et al. (2014) there is a shock in recharge at an unknown date, which may be known or unknown beforehand. The authors conclude that, although optimal extraction may be higher in the short-run in the stochastic case, the long-run steady-state stock level is also higher.

Another type of uncertainty identified in Tsur & Zemel (2014) affects the flow of net benefits attained in each period, as happens when groundwater demand is stochastic. Water demand can depend on climate aspects such as temperature levels and precipitation amounts. In conjunctive use systems, in particular, surface water flows that vary with weather fluctuations ought to affect the benefits to be gained from groundwater use since the sources are often substitutes. Tsur & Graham-Tomasi (1991) show that in this case the in situ value of groundwater is higher, because it can be used to compensate shortfalls in variable sources; this is known as the buffer value of groundwater. In essence, the resource stock provides a form of insurance against drought (Pérez Blanco & Gómez, 2014). However, the effects of groundwater availability in case of drought should be assessed both in the short run, where groundwater improves crop yields and decreases drought sensitivity, and in the long run, where more land might be allocated to “thirsty” crops which are more vulnerable to drought damages (for a historical illustration, see Hornbeck & Keskin, 2014). Furthermore, declining well-yields can limit the use of groundwater as an adaptation tool (T. Foster, Brozović, & Butler, 2015).

It is also possible for several types of randomness to affect a particular resource. Brozovic & Schlenker (2011) combine stochastic stock evolution with a threshold for regime shifts, and results are non-monotonic, that is, initial increases in variance lead to more precaution while higher levels imply less (see also Zemel 2012; Leizarowitz & Tsur 2012).

The presentation of uncertainty provided so far covers centralized decision models where a single objective function is maximized assuming risk neutrality, that is, only the expected value of net gains matters. A few authors have explored the consequences of risk-averse behavior in groundwater management (Knapp & Olson, 1996; Krishnamurthy, 2016). Extending the literature in that direction seems a worthwhile endeavor given that many studies point to risk aversion for farmers, who are the main users of groundwater in most areas (Just, 2003; Moschini & Hennessy, 2001; OECD, 2009). However, there is no guarantee that risk attitudes are the same for all individual users, nor is it clear that a social planner ought to exhibit risk aversion to a similar degree. Again, the difference between positive and normative analysis must be underscored.

Shaw (2016) provides a detailed and insightful overview of many issues related to decision making under risk and uncertainty which are widely recognized in economics but are nonetheless overlooked in the resource management papers cited so far. Two conceptual matters that deserve more attention from groundwater researchers are the nature of uncertainty (what don’t we know, exactly?) and the characteristics of choice under uncertainty (how do people choose in such conditions?).

On the extent of our knowledge, the traditional classification distinguishes between risk, when there is reliable information on possible outcomes and associated probability distributions, and uncertainty, which can be understood to encompass a spectrum of situations. Whenever there are doubts about appropriate probability distributions, the word ambiguity can also be applied. In such conditions, agents may form their own subjective probabilities, or just use alternative decision rules such as considering only the worst outcomes or minimizing regret. However, it is difficult to describe exactly what is going on in people's choices, since the same set of empirical observations can be compatible with various models of preferences and expectations. Economists have tended to assume that agents have objective probabilities, thereby using data to infer only the properties of preferences. Yet this approach has led to "a crisis of credibility" (Manski, 2004, p. 1330), especially since several authors, many of them psychologists, have repeatedly exposed the flaws in the expected-utility model. Most famously, Tversky & Kahneman (1992) postulate that people systematically underestimate high probabilities and overestimate low ones, while also treating losses as more relevant than gains. In other work, numerous studies on risk perception note that an individual's behavior in uncertain circumstances will depend on how she perceives risk, and this in turn varies with personal, social and cultural characteristics (Slovic 2000).

Applications of these concepts in water management models are few and far between.[7] Roseta-Palma & Xepapadeas (2004) use the robust control methodology to analyze the emergence of precautionary behavior of a centralized water manager that faces multiple priors in the model specification of precipitation. Woodward & Shaw (2008) also study robust solutions to water management in a case where instream flows affect an endangered aquatic species. Howitt et al. (2005) offer an example of a positive approach, using data from a reservoir in California to estimate the revealed preferences of the reservoir manager. They conclude that there are indications of risk aversion and that a recursive-utility model provides a better fit than the standard model of time-additive separability. Another possibility is to consider one-sided risk measures such as downside risk, increasingly applied in finance, which focuses on variability in only the negative outcomes (Hanemann, Sayre, & Dale, 2016; Roseta-Palma & Saglam, 2016), whether for the centralized problem or for individual decision makers.

As we move forward into the fascinating world of non-expected utility modelling, the essential discussion to be had is, again, the positive vs normative one. That individual choices are not generally compatible with expected-utility predictions is old news.[8] Thus the positive question is fairly settled, albeit still ignored in most empirical studies of water-using decisions. But should societal preferences reflect individual preferences towards risk or is there, as in discounting, an argument for social preferences to deviate from individual ones? Portney (1992) questions what should be done in fictional Happyville if the public perceives a risk of water contamination while scientific assessments disagree. Salanié & Treich (2009) use this scenario to compare the rational approach, which ignores unfounded consumer beliefs, to the strategies of a populist regulator, who decides based on the principle of consumer sovereignty (or her wish for re-election) and therefore may over- or under-regulate depending on consumers being pessimistic or optimistic. The authors also study a paternalistic strategy that takes into account consumer attitudes as well as the difference in beliefs. Interestingly, there

[7] Shaw (2016) reviews examples in other areas of environmental and natural resource economics, namely climate change, wildfires and fisheries.
[8] Recall that over fifteen years ago, Rabin & Thaler (2001) already compared EU models with the dead parrot of Monty Python fame.

may be over-regulation in the latter case whether consumers are optimistic or pessimistic, a result which could provide yet another reason to apply the precautionary principle (see Randall, 2009).

Climate change, due to the accumulation of greenhouse gases in the Earth's atmosphere, will undoubtedly interfere with the hydrological cycle in many ways, some foreseen and others unexpected. This amplifies the need for water resource managers to seriously consider the issues described in this section. However, the increasing complexity of models leaves researchers with the thorny task of communicating uncertain results in a way that provides usable insights to policy makers and to the public. Guillaume et al. (2016) provides valuable guidance by covering various aspects: the model setup ("modelers and stakeholders need to work together to define a problem, in a manner cognizant of the uncertainty involved", pg. 716); the need for wide participation and for iteration; the methods to generate alternative model realizations; advice on the representation of uncertain outcomes, namely on choosing different levels of detail, from single outcomes (for example, averages), to distributions, to bounds (best and worst cases), to scenarios, to spelling out points of ignorance; and finally, communication, reminding us that "a groundwater scientist cannot expect that those needing to use the estimates will understand the academic terms and metrics. Therefore, translation of estimates into formats of direct use" (p.720) is essential.

4. The value of groundwater, seen and unseen

Groundwater resources embody characteristics of a public good whose consumption by an individual does not reduce the amount of resourses available to others, making it a non-rival good, and cannot be excluded from those who cannot afford it, making it non-excludable as well. This leads to non-exclusive water resource property rights which result in the quantification of water *total economic value* (TEV) using non-market valuation methods. The knowledge of this value plays a vital role in policy and decision management towards socio-economic efficiency, in terms of a cost benefit analysis (CBA) that imposes marginal benefits to equal marginal costs, since it allows the most efficient water allocation to prevent the excessive environmental pollution and depletion of the resource.

The TEV of a groundwater resource is categorized to the *use value,* derived from individuals via using the water resource, and the *non-use value* that arises from the water resource even if it is not used. A further classification of the use value refers to the *direct use value*, the *indirect use value* and the *option value.* The direct use value emanates from the direct consumption of the water resource such as drinking water, irrigation and industrial activity. On the other hand the water resource can benefit an individual indirectly, i.e. obtain an indirect use value, through a series of procedures such as flood control, nutrient retention, and income increase. Moreover, the option value comes from the maintenance of a water resource which may provide economic benefits to the humanity in the future.

As regards the non-use value, it consists of *existence value*, *bequest value* and *altruistic value*. Particularly, the existence value is related with the conservation of a water resource which its direct use is not the case for neither a current nor a future generation. Furthermore, the bequest value refers to the opportunity of the future generations to have access to the water resource. Finally, the altruistic value is attributed to the water resource by individuals who may not even intend to use it but believe to its availability against the rest individuals of their generation.

In the existing literature, the TEV of water resources is estimated by a wide variety of methods which can be distinguished in three main categories: (i) *revealed preference methods*, (ii) *state preference methods* and (iii) *benefit transfer methods*. An extensive literature review is provided by Sudevi & Lokesh (2014), Birol et al. (2006) and Koundouri (2004a).

4.1 Revealed preference methods

The revealed preference methods, also referred to as *indirect valuation methods*, are based on data which can be extracted by markets relevant to the natural resource in the sense of the latter being associated with a traded environmental good of a known value over space and time. The information, i.e. 'preferences', encapsulated in the behavior of the related markets is 'revealed' and quantified via various statistical techniques to estimate the willingness to pay (WTP) of an individual that reflects the expected marginal benefit derived from the use of the resource. In what follows we shall present the most common methods employed to estimate the use value of the water resources which are implicitly traded in surrogate markets.

The first method is the *hedonic pricing method* (HPM), according to which the value of a composite (multi-attribute) good is related to the range of characteristics that identify the resource and to the levels these characteristics may take. This method is mostly applied to the real estate market where the price of a house is the reflection of its structural and neighborhood characteristics, i.e. size, number of rooms, number of people who can live in, parking slots, garden, level of crime etc, as well as it is connected with the shadow price of local environmental resources such as air quality, noise levels, aesthetic views, water quality and quantity. On the other hand, the main constraint of HPM is the quantification only of the direct use value of the water resources as it follows implicitly from the consumption of the public good at its market value. However, this method is unable to capture values of several services associated with the natural resource such as flood control, water quality improvement, habit provision of species, and groundwater recharge, which may benefit the individuals beyond its consumption (Boyer et al. 2004). Another limitation of HPM was pointed out by Koundouri & Pashardes (2003) who exhibited sample selection sensitivity of this method to the quality characteristics of a good. In particularly, considering a model of consumer demand for packaged goods, they assumed a land near to seaside which can be used as an input either in agricultural production or in the touristic development. Therefore the salinization of groundwater resources may reduce the productivity due to lower quality but simultaneously may increase the probability of the land usage to the lucrative (profitable) tourist market, leading to a valuation bias of the water salinity effect on agricultural land. In a similar vein Yusuf & Koundouri (2005), making use of empirical data from the Indonesian housing market and taking into consideration the sample selection bias, concluded that the households in urban areas value piped and pumped water access more than well water.

In the same hedonic framework, several recent studies have investigated the relationship between land prices and the accessibility to surface water in terms of both quality and quantity. Mahan et al. (2000) coped with the estimation of proximity effect to wetlands on the property values, obtaining that a decrease in the distance to the nearest wetland by 304.8m from an initial distance of 1 mile causes an increase in property values of 371.6 euros. Colby & Wishart (2002) showed that the sale price of a typical 2,000 square-foot property in the northeast Tucson metropolitan area is reduced by 6% as we move in distance from one-tenth of a mile to 1.5 miles away from Tanque Verde Wash. Using the value of the ambient water quality through data of St. Mary's River watershed in Maryland, Poor et al. (2007) investigated the implicit values of total suspended solids and dissolved inorganic nitrogen which are water quality variables. They showed that a one unit (mg/L) increase in either total suspended solids or dissolved inorganic nitrogen has a negative impact on average housing prices within the watershed of $1086 or $17,642, respectively. Higgins et al. (2009) estimated both the financial value of AU$900,000,000 and river management policies, derived from living close to Maroochy River against to residential property values, by employing an artificial neural networks method in conjunction with a large and complete dataset of properties for the Maroochy region.

The second method is the *travel cost method* (TCM) which is mainly used to get travel information about trips in particular sites or destinations. The WTP of an individual for accessing a site is inferred by the number of trips to the site and their 'travel costs', which capture travel expenses (fuel, hotels, etc), the distance from the final destination, frequency of travelling, and characteristics of the destination that concern among others water resource management. TCM was employed to estimate the recreational value of particular sites where people travel to hunt, fish, hike, swim or watch wildlife. In particularly, several studies tried to measure the welfare effects to changes in water quality such as Carr et al. (2003), McKeen et al. (2005), Shrestha et al. (2007) and Hosking (2011). El-Bekkay et al. (2013) conducted 480 surveys and applied TCM to estimate the consumer surplus per person per visit at $US 65.36 in the RAMSAR recreation site of the estuary of Massa River. On the other hand, it is worth to point out that as in the HPM the travel cost studies can only partially evaluate the total value of a wetland, since they overlook all its public good aspects such as flood control and groundwater recharge or discharge that are unrelated to recreation.

The third method is the *replacement cost method* (RCM), which essentially values the benefits incured from the use of an environmental good via quantifying the 'cost of replacing' this good with an alternative that provides the same benefits and is at most of same value than the former. Due to this method, Acuña et al. (2013) performed a CBA on water purification by estimating the expenditures arising from replacing ecosystem services with artificial technologies. A drawback of this method is that each time applies only to a certain use of an environmental good, exhibiting limitations to capture the TEV of the good as a whole.

Finally, we have also the *avertive expenditure method* (ABM) which in the water resource framework focuses on the WTP of a household for drinking water safety. This is evaluated from the 'expenditures' of measures (e.g., buying bottled water, boiling water for cooking and drinking) undertaken by the household as a result of a developed 'averting' or defensive behavior towards increased degradation of the available water resources due to pollution. A general limitation of this method is that the averting expenses deduce only the minimum value one can attribute to increased pollution, since it is not possible to reflect all the costs related to pollution. Introducing a perception measure to the conventional ABM, Um et al. (2002) justified efficiently the inconsistencies between perceived risk beliefs and objective risk measures regarding water resource contamination in Pusan, Korea. Pattanayak et al. (2005) made use of a unique data set from a survey of 1500 randomly selected households in Kathmandu, Nepal, and employed a utility maximization household production model to assess coping costs and WTP as two complimentary components of households demand for improved water services.

4.2 Stated preference methods

The stated preference methods, also called *direct valuation methods*, aim to assess either the use or the non-use value of environmental resources for which no related market data are documented in any trading market. These methods employ structured surveys in order to collect the market data needed, as 'stated preferences' to a questionnaire, to estimate the WTP over different quantity and quality levels of particular aspects of a natural resource, and in turn the associated demand curves. The main categories of these methods are overviewed in what follows.

The most popular state preference method is the *contingent valuation method* (CVM) which conducts surveys using questionnaires to elicit individuals' preferences via their WTP for a specific change in the levels of quantity or quality of an environmental resource. This valuation technique is 'contingent' upon a hypothetical scenario which provides an explicit description of the natural resource and how it is becomes available in a hypothetical market, and it is tested on a sample of the total population via interviews. The questionnaires introduce the hypothetical scenario and include several types of questions regarding the levels of understanding of this scenario from the respondents, the attitude of the respondents to the natural resource of interest, the proper payment vehicle for capturing WTP (e.g. tax increase, surcharges, single payments), and finally the socioeconomic characteristics of the respondents (e.g. gender, age and income). Arranging consultation meetings with focused stakeholder groups of interest and relevant scientific experts is as crucial to the suitable design of the questionnaire as the pre-testing stage and the manner that the interviews will carry out (in person, via mail, e-mail or telephone surveys) is to its implementation. Eventually, the gathered sample data of TEV can be used as dependent variables against the aforementioned socioeconomic factors to assess the corresponding demand curves of the natural resource, as well as provide a mean value for its aggregate TEV in the total population.

In CVM the survey designer is in total control of all the information and choices that are provided to the respondents, so that the results of the variables under investigation can be isolated from the effects of other factors and can be utilized to estimate use and non-use values of a natural resource. Nevertheless, the main drawback of this method is the hypothetical nature of the designed scenario and refers to how different the prices extracted from such surveys are in comparison with those coming from a real market. What is more, the approximating value of WTP is strongly criticized for its lack of validity and reliability since several types of bias may appear in different stages of the survey, including information bias, design-starting point bias, vehicle bias, Yea-saying bias, hypothetical bias, selection bias, protest bias, sequencing bias, elicitation bias, anchoring bias and embedding effects (see Birol et al. 2006, Sudevi & Lokesh 2014 and references therein).

In case of water resources, Birol et al. (2008) showed that the respondents would be willing to pay different values according to whether they are users or nonusers in order to ensure the sustainable management of the Acrotiri wetland in Cyprus. In the same spirit, several researchers have recently conducted contingent valuation surveys in order to investigate people's WTP for the improvement of a river water quality (see Alam 2006; Imandoust & Gadam 2007; Monarchova & Gudas 2009; Nallathiga & Paravasthu 2010; Tu 2013) and coastal water quality (see Hokby & Soderqvist 2003; Hanley et al 2003; and Zhai & Suzuki 2009).

A stated preferences method more sophisticated than CVM is the *choice experiment method* (CEM), which now exposes the survey participants to several hypothetical scenarios of 'choice' instead of one. All these alternative senarios cover a wide range of combinations between different states of the environmental resource at stake and different levels of prices of its investigated characteristics, formulating corresponding choice cards in which the marginal WTP of each isolated characteristic can be estimated through a random utility econometric model and all these shadow prices can be integrated to give the total WTP of the whole resource. This is realized with the use of particular 'experiment' design methods which ensure that all the selected characteristics of the resource are statistically independent to each other, so that colinearity leading to false estimates is avoided. These alternative scenarios should be easily comprehensive and produce choice cards that represent possible and realistic future events subject to uncertainty (e.g. climate change conditions), so as to get as many realistic responses as possible. Finally, the respondents state their preferences by either selecting a choice card or by ranking them in order of favorability (Bateman et al., 2003). An advantage of CEM is the fact that it can solve some of the bias problems that are presented in CVM, i.e. the strategic bias and yea-saying bias (cf. Birol et al. 2006), while the fact that it requires a more complex experimental design process is considered to be a limitation.

In the recent water related literature there are quite several applications of CEM, especially to value the improvement in groundwater water quality. As regards wetland quality, Koundouri et al. (2012) applied a CEM to estimate the economic value coming from the water improvement in Rokua wetland in Northern Finland with respect to water quantity, recreation, land income opportunities and investment on research, whose marginal WTP was in the ranges of 9.71-11.95 for recreation to 33.5-36.92 for research opportunities; see also Birol et al. 2007 and 2009. Studying decisions taken by couples regarding which beach to visit while on vacation in the Caribbean island of Tobago over different available scenarios of a CEM relevant to coastal water and beach quality, Borg et al. (2009) concluded that the individual preferences are rather different than the joint couple choice which usually coincides with the woman's preferences. As regards river basin quality, Davila et al. (2017) used a CEM to rank and evaluate a set of proposed improvements in Asopos River Basin of Greece, towards sustainable river basin management, as per the prescriptions of the European Union-Water Framework Directive (2000); on river basin sustainable development see also Birol & Das (2010) and Tait et al. (2012). Brouwer et al. (2010) used a CEM to demonstrate the spatial preference heterogeneity effect of deriving different marginal values of WTP for water quality improvement across different parts of the Guadalquivir River basin in the south of Spain, where inhabitants in one subbasin also hold WTP values for other subbasins but attribute the higher one for their own subbasin with respect to everybody else's. On spatial preferences heterogeneity in the ecological status of Asopos River Basin of Greece see also Koundouri et al. (2014).

4.3 Benefit transfer methods and laboratory experiments

The benefit transfer method (BTM) is the process of transfering existing data of environmental valuation for a given problem from an area with specific characteristics to a similar one. This method is a standard practice when it is not feasible to carry out primary research due to restrictions on the cost and time of implementation. The transfered value expresses the users preferences in the study area, subject to proper adjustments regarding different socioeconomic characteristics (income, prices, currency, etc.) between areas. Koundouri et al. (2016) used BTM to assess the value of four ecosystem services of the Anglian river basin in the UK, considering several other studies of different methods such as hedonic pricing and choice experiments. Another study by Koundouri et al. (2014) used this approach to estimate the benefits of mitigating industrial pollution, where the value of the change in water quality from "bad" to "very good", as set by the Directive 2000/60/EC, was found to vary between 88.28- 116.94 euros. In relation to this approach, several studies have combined their methodology with GIS (Geographical Information System) data to assess the economic value of conservation and restoration water related projects (e.g. Jenkins et al. 2010; Naidoo & Ricketts 2006), to estimate the value of ecosystem services (Plummer, 2009) and to aggregate benefits from non-market environmental goods (Bateman et al. 2006) among others.

Meta-analysis is an implementation technique of BTM which includes statistical analysis of combined results of previous studies. For example, Van Houtven et al. (2006) identified 300 studies that are related to water quality improvements, most of which were stated preference studies, and Johnston (2005) assessed the relationships between the nonuse components of WTP for water quality improvements and resource attributes associated with use values.

Furthermore, as laboratory experiment we refer to a technique according to which participants make choices following a well-structured scenario in a controlled environment (laboratory). Drichoutis et al. (2014) implemented this technique by engaging respondents in a 6 auction rounds (three of them were hypothetical and three real), where they had to choose whether they would exchange their endowment with an amount of a good from a river basin with good ecological status and a river basin with bad ecological status that could potential raise health concerns. The results indicated that people would bid higher for the goods produced in the region of good ecological status, showing aversion to potential health issues stemming from heavily polluted water. Another study by Carson et al. (2011) assessed the economic consequences of the effects of arsenic contamination on labor supply in Bangladesh. Hence, a labor supply model was estimated that used labor data from local households, which was matched with data on arsenic contamination. The results indicated that labor hours are lost, due to the fact that individuals try to hedge against contamination dangers.

5. What can be done? Policy choices

Once we move into the policy realm, the first step ought to be the establishment of policy goals. Decision makers should define their aims in a way that is transparent and clearly understood. This is a normative choice and several options might be considered, notwithstanding the widespread agreement on the need for an integrated framework of analysis independently of the chosen goals. In the European Union, for instance, the focus of recent water policy has been on the environmental objectives, albeit complemented by economic criteria: the Water Framework Directive (EU, 2000) aims for good ecological status for surface water and good quantitative and chemical status for groundwater, with goals for the latter further developed in the Groundwater Directive (EU, 2006). Yet there is generally more than one way to achieve goals, so the WFD also emphasizes the role of water pricing and the need for cost effectiveness in policy choices.

Globally, the United Nations have strived for a comprehensive vision of water management. A recent resolution of the UN General Assembly (United Nations, 2016) declares an International Decade for Action, “Water for Sustainable Development”, 2018-2028, to focus on “the sustainable development and integrated management of water resources for the achievement of social, economic and environmental objectives […] in order to help to achieve internationally agreed water-related goals and targets, including those contained in the 2030 Agenda for Sustainable Development”. Two of the UN’s Sustainable Development Goals are specifically dedicated to water issues: clean water and sanitation (nr. 6), and life below water (nr. 14). Here, too, there is a mention of the “importance of promoting efficient water usage at all levels”, extending the scope to the “water, food, energy, environment nexus”.

At a practical level, another important aspect in setting the stage for policy intervention, whether for water extractions or pollution discharges, is to have a clear understanding of the legal regime for groundwater in each jurisdiction. As Nelson & Quevauviller (2016) note, the definition of groundwater itself can vary significantly across legal systems, from very narrow views of “percolating” groundwater or “underground streams” that have no scientific meaning but are used in some of the Western US states, to the far-reaching interpretation in Australia’s federal legislation, which encompasses all aspects that contribute to the environmental value of water resources. In terms of property rights, most jurisdictions have evolved to recognize that landowners can use the water beneath their land but do not own it outright, although there are a few exceptions. Such recognition allows for the implementation of public licenses or permits as well as the possibility of legal water trading, an approach that has been tried in several locations, such as Chile (Hearne, Donoso, & Dinar, 2014), Spain (Rey, Garrido, & Calatrava, 2014), the United States (Brozović & Young, 2014) and Australia (Wheeler, Schoengold, & Bjornlund, 2016). In principle trading can occur among different water sources, although specific regulation, monitoring and enforcement schemes must be in place for groundwater trading to be possible (Brozović & Young, 2014; Wheeler et al., 2016). Moreover, to guard against unintended effects and ensure efficient use, the links between surface and groundwater must not be overlooked.

Water trading falls within the category of economic instruments, given its potential to increase allocative efficiency by conveying water to where its value is highest. Price signals of all hues work by providing users with an incentive to adjust their resource consumption. This is, indeed, the motivation behind the WFD water-pricing requirements (art. 9). Higher volumetric prices lead to lower use, whether the price signal reaches agents through trading, tariffs, incurred costs, taxes or other means. Although most groundwater users can rightfully claim that the cost of pumping is already incorporated into their decisions, none of the external effects described in section 2 nor the ecosystem values detailed in section 4 are included in that pumping cost. Therefore direct price regulation to internalize such costs, for example through water abstraction taxes, can be envisioned.

However, in practice there may be several obstacles in the implementation of full-fledged groundwater pricing. Montginoul et al. (2016) point out that lack of information is a major issue in several groundwater areas. In particular, the most complete schemes require full metering ("who uses how much water in which place at different periods of the year", pg. 554), not to mention a thorough knowledge of interactions between groundwater and surface water and of the impact of extraction on groundwater-dependent ecosystems. Both Mulligan et al. (2014) and Guilfoos et al. (2016) warn that simpler policies, such as uniform water taxes or quotas, which are more likely to be implemented by policy makers, can underperform in real-world conditions where heterogeneity of both agents and aquifers is present. The latter concludes that local management areas might be more effective and can better take into account distributive effects on farmer's incomes. The consequences of policy implementation also depend on when it occurs, namely, on the aquifer's conditions. It is possible to have "windows of opportunity" where policy is beneficial (T. Foster, Brozović, & Butler, 2017). Taher et al. (2016) include climate change and policy scenarios and note the possibility of disproportionate costs on certain users.

Users who are harmed by policies can be expected to resist their implementation, but opposition may also arise from prevailing social norms. Stakeholders may reject an approach based solely on centrally-defined economic instruments and favor solutions that entail user cooperation. Figureau et al. (2015) find that in France there appears to be a clear preference for decentralized solutions where economic instruments are combined with policies to promote social norms, such as solidarity, reciprocity and trust among users. Nøstbakken (2013) provides a theoretical model of quota enforcement in a common-property resource, concluding that higher fines for non-compliant agents can strengthen a norm of compliance, leading to less free riding and less overexploitation. A more complex framework for analysis, using Ostrom's IAD methodology, is proposed in Rahman et al. (2012), although they present an application in fisheries, not groundwater. Dinar & Jammalamadaka (2013) address the role of institutions and social norms, in irrigated agriculture, in the context of climate change adaptation.

De Stefano & Lopez-Gunn (2012) supply a useful overview of the issues surrounding unauthorized groundwater use, which is a major problem in many areas, even in developed countries. However, when aquifers are very large, with thousands of users, and when there are environmental externalities to boot (Esteban & Dinar, 2013), cooperation will be much harder to achieve.[9] Additional complications are associated with the management of transboundary aquifers, of which the UN estimates there are approximately 300 (UN-Water, 2008). Game theory has been, again, a common framework of analysis in transboundary resources since at least Rogers (1969). In these cases it is necessary to establish an agreement, among the sovereign parties of different jurisdictions, on the goals of management, while the specific policies implemented by each party to reach such goals are established separately.[10]

A final point concerns policies to improve technical water-use efficiency, which means, in broad terms, to reduce the amount of water that is necessary to attain a given outcome. In particular, subsidies to improve irrigation efficiency, for example through more modern irrigation infrastructure, have become increasingly popular (OECD, 2008). Although efficiency gains always sound like a good idea, several authors have warned that they may not lead to water conservation. Gómez & Pérez-Blanco (2014) present an analytical decomposition of efficiency impacts on water demand, highlighting that the direct technical effect of improved irrigation equipment might be overtaken by a rebound productivity effect, associated with the higher productivity of water, which leads farmers towards thirstier crops and/or an increase in irrigated area.[11] The final outcome might well be more demand for water. An empirical illustration can be found in Peterson & Ding (2005). A recent survey by Lin Lawell (2016) covers the reasons that lead farmers to over extract groundwater, including externalities, as described earlier, but also the role of institutional incentives and irrigation-efficiency advances.

6. **The way forward**.

A recent leader in The Economist asserted that "water is scarce because it is badly managed" (The Economist, 2016). In many places this is clearly true, with water being underpriced, overexploited and even squandered. For groundwater, this survey has summarized typical complications ranging from a lack of clear information on the stock size, quality, evolution and linkages, to its features as a quintessential common-property resource. Given the importance of groundwater and the poor state of many aquifers, the critical question is no longer whether it is necessary to apply groundwater management policies, but rather what needs to happen

[9] Interestingly, some papers in the literature have explored the deeper links between water management regimes and democracy. Bentzen et al. (2016) find that historically irrigation has made authoritarian rule more likely. Uncontrolled groundwater extraction can be seen as anarchy (Shah, 2009) or as liberation from the State (Kuper et al., 2016).

[10] For an in-depth discussion of the issues related to transboundary water management, see (Dinar, Dinar, McCaffrey, & McKinney, 2007; Dombrowsky, 2007).

[11] Such rebound effects have been widely discussed for energy services, see for example Sorrell & Dimitropoulos (2008).

for management to improve. We have pointed out the importance of starting with a proper normative discussion of the goals to be achieved and a good understanding of the institutional framework, since both aspects will affect the choice of specific instruments to apply. Technical knowledge, monitoring and enforcement capabilities must also be given due consideration. It has become apparent that the textbook presentation of policy instruments as substitutes is lacking – often a combination of instruments is the better bet (Montginoul et al. 2016). Even if the formidable power of economic incentives, such as pricing or water trading, to affect water consumption levels at a large scale is recognized, they should not be expected to solve water scarcity single-handed.

A particular complication emphasized in this survey is the dynamic and uncertain nature of the groundwater resource. Many of the future physical and socio-economic impacts hinge on climate trends, where increasing variability is predicted. This can be modelled in different ways, from relatively straightforward modifications of resource-stock evolution equations to full-blown scenario analysis. The latter strategy is common in climate-change research (IPCC, 2014) and in the emergent field of sustainability science (Swart, Raskin, & Robinson, 2004), but there is scant evidence of its use in groundwater management, where its application could be fruitful.

Phoebe can you add something about the importance of considering the TEV groundwater and its implications in terms of management models and/or policies?

Considering the complexity of the resource and the multifaceted challenges it raises, no single management model is ideal, notwithstanding the significant advances witnessed in hydro-economic modeling. The aims, assumptions and possibilities should be transparent, not only in theoretical research but also in policy proposals. Economists might usefully tone down idealized "optimal solutions" when engaging in real policy discussions, while defending the principles behind economic efficiency criteria. We could also avoid framing debate through oversimplified questions demanding clear-cut answers, such as whether user cooperation is better than pricing or whether intervention always beats no intervention. Pragmatism, multidisciplinary work and communication are essential, and stakeholder participation is highly desirable given the strong links - economic, social and even spiritual - that bind people to their water. There's more to groundwater than meets the eye, and we will still be looking into the unseen resource for many years to come.

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