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Recycling water, but at what price? The highly political economy of wastewater reuse

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Abstract:

It is generally assumed that the circular economy generates net benefits; and that these benefits can be fairly accurately measured. Based on the case of treated wastewater re-use (TWWR), this article challenges both assumptions. We argue that economic assessments of TWWR are always deeply political, as they cannot be detached from the promotion and prioritization of particular collective goals. We successively explore: the politics of benefit assessment; the politics of cost assessment; and the politics of cost-sharing arrangements. We argue that this inescapable political dimension, amplified by wide-ranging economic uncertainties, justifies setting up participatory arenas for each individual project.

Keywords: circular economy; treated wastewater re-use; political economy; cost-benefits analysis; positive bias

[SDG 6: Clean water and sanitation]

The concept of the circular economy is being increasingly touted as a way to reconcile the economy with ecology. Economists who study it usually share two assumptions. The first is that its benefits, regardless of how they are measured exactly, far outweigh its costs. Keeping components and materials in “closed loops” would produce a much higher payout than letting them go to waste and having to produce them anew (McKinsey Center for Business and Environment and Ellen MacArthur Foundation, 2015). The second assumption is a positivist one. It is considered, implicitly, that the costs and benefits of the circular economy can be measured in an “objective” way, and are therefore potentially consensual. The main challenge for economists would be to measure them as accurately as possible in order to quantify the expected profitability of investing in various circular-economy initiatives.

This paper directly challenges the latter assumption, and, in so doing, also indirectly challenges the first. It shows that the economics of the circular economy is never value free. Rather, it is always dependent on the promotion and prioritization of particular political goals. Politics – that is, the process through which certain collective goals and certain collective interests are prioritized over others¹ - shapes how certain types of costs and benefits are considered while others are ignored, how the former are assigned some particular monetary value, and how costs are distributed among stakeholders. In other words, we argue that it is high time that purely economic approaches to the circular economy, although valuable in their own right, be complemented by a political economy approach.

We set out to do that by analyzing a specific domain of this wide ranging, umbrella category: treated wastewater re-use (TWWR). Also commonly known as water recycling or water reclamation, it refers to the process of reclaiming water from a variety of sources, then treating and reusing it for beneficial purposes such as agriculture and irrigation, potable water supplies, groundwater replenishment, industrial processes, and environmental restoration.

As is the case of the circular economy in general, most analyses of TWWR are based on the assumption that it is economically advantageous. It is seen as a “no-regret measure” (Van Der Hoek et al., 2016), transforming as it does, unproductive “waste” into productive “resources” (UNWWAP, 2017; Winpenny et al., 2010). Through cost-benefit analyses, most economic studies purport to show that, although *direct* benefits may sometimes appear limited, TWWR nevertheless has *indirect* benefits that make it profitable anyway. The task of policy-makers, therefore, should be to raise users’ awareness of these indirect benefits (such as additional nutrient inputs and fertilizer savings, see, for example: Bichai et al. 2018). Beyond these measurements of benefits, however, few contributions deal with concrete modes of financing. Who actually pays for TWWR in concrete cases, to what extent, and why?

By focusing on *treated* wastewater, we are not losing view of the fact that most of the world’s reused water remains today untreated. In 2014, 36 million hectares worldwide (some 12% of all irrigated land) was estimated to receive some amount of urban wastewater, but only around 15% of this was deemed adequately treated. We concentrate on TWWR, however, as it is now growing more rapidly than the reuse of untreated water: the cumulative installed capacity

¹ This definition is rooted in a tradition, well exemplified by the work of political theorists Ernesto

Laclau and Chantal Mouffe, who defines politics as a continuous process of choice and articulation between different, and ultimately incompatible, conceptions of the common good.

for municipal treated wastewater reuse worldwide rose from about 67 million m³/day in 2005 to 169.1 million m³/day in 2020 (IDA, 2022, p. 6–14). Also, because it allows to mitigate health risks, it is TWWR rather than raw sewage that is promoted and financed by governments worldwide, and considered a “critical part of the circular economy” (IPCC, 2022; UNWWAP, 2017, p. 114). However, it is worth bearing in mind that “treated” wastewater is itself an umbrella category that includes varying levels of treatment, from the most basic, primary treatment to the most sophisticated, “tertiary” treatment processes based on ultraviolet radiation.

Besides, it should be said that the majority of water reuse worldwide, regardless of treatment level, takes place both informally –outside any recognized, official institutional framework- and indirectly, with wastewater typically diluted in other water bodies before being reused at some distance downstream (Ait-Mouheb et al., 2020). Nevertheless, in this paper we limit our focus to TWWR that is both formal and direct, for two reasons. Firstly, this is the type of re-use most promoted and financed by policy-makers globally. Secondly, it is in the case of formal and direct TWWR that the measurement of costs and benefits is most explicitly thought through, debated and fought over by stakeholders, which makes its political economy more apparent.

Our empirical focus is also on TWWR for agriculture and urban (green) spaces, which are the two most widespread uses worldwide. Agriculture, in particular, dominates water reuse capacity in Mediterranean countries, with countries such as Israel, Jordan and Spain being the most extensive practitioners (IDA, 2022). It is also an object of high ambitions for policy-makers. In 2020, for example, the European Commission expressed hope that by 2025, the volume of treated wastewater reused in agriculture in the EU would surge from 1,000 million m³ per day to no less than 6,000 million (IDA, 2022). By contrast, industrial re-use, important though it is for economic efficiency, does not have the same public salience, nor does it pose public health issues of the same magnitude; while reuse for drinking water, although expected to increase worldwide (Francis, 2024) remains, for the time being, numerically marginal.

Finally, our empirical illustrations originate from contrasting Mediterranean countries, including France, Morocco, Tunisia, Jordan and Lebanon. We believe that Mediterranean countries are representative of the great diversity of both resource constraints and regulatory frameworks to which TWWR are subject.

The arguments presented in this article come from a variety of sources. First, they are based on the authors’ firsthand observations, mostly in France and Morocco. Most of them have been made as part of applied research or consultancy projects. As such, not all of them have yet resulted in scientific publications, although a majority of them have and are referenced accordingly. Since we are a multidisciplinary team - including an economist, a political economist and a researcher in process engineering specifically applied to water reuse - these projects had different orientations. Some were focused on pricing, others on finding viable cost-sharing arrangements, still others on optimizing and reducing the costs of treatment processes. They also took place at different stages of project implementation, with some occurring while projects were under discussion, others when infrastructure was being constructed, and still others concerned projects that had been operational for some time, up to 30 years. Thus, we have re-analyzed and revisited our past work, largely conducted separately, in light of our new, joint reflection.

This paper is also based on an extensive literature review of water re-use in the Mediterranean, particularly for agriculture and green spaces. With a special focus on the

countries of North Africa, Lebanon, France and Spain - countries with which we were already most familiar - we looked at the concrete ways, often far removed from the prescriptions of “pure” economic theory, in which costs and benefits were assessed, and costs were shared. Wherever possible, we also drew on the ‘grey’ literature of feasibility and economic studies.

The article is organized in three main parts. We focus on, successively: the politics of benefit assessment; the politics of cost assessment; and the politics of cost-sharing arrangements. For each political-economic issue, we describe the diversity of approaches that are typically pursued, the main lines of the debates and conflicts that surround them, and the types of compromises that are typically reached in diverse contexts, which may be more or less fragile or robust. In conclusion, we argue that this inescapable political dimension, amplified by wide-ranging economic uncertainties, justifies setting up participatory arenas at the level of each individual project.

1. The politics of benefit assessment

For most economists, assessing the benefits of any infrastructure project in general implies assigning monetary values to all its expected benefits. This is inherently difficult, and always vulnerable to criticism. As Rogers et al (1998) highlighted in the case of water infrastructure projects more specifically, their benefits are the sum of their economic and what they termed their “intrinsic” value. While the former are already far from straightforward -as they include many potential indirect effects on poverty alleviation, economic growth and job creation- the latter can also be divided into various, heterogeneous sub-components, such as the intrinsic value for different types of direct and indirect water users, and the value for the maintenance of ecosystems. Thus, monetary valuation becomes increasingly complex, and increasingly difficult to agree upon, the further we move away from strictly financial information towards non-monetary valuation methods (such as contingent valuation). Such complexities and uncertainties are the terrain on which actors will push forwards the assessments that best correspond to their preferred goal. With regard to TWWR, this terrain is particularly fertile for at least three reasons.

First, *indirect* benefits -that is, positive externalities- are particularly foregrounded in TWWR projects. They range from fertilizer savings to improvements in the quality of receiving environments. By their very nature, however, indirect benefits are more uncertain than direct benefits. They are less easily agreed upon, and therefore more susceptible to be variously assessed according to one’s own political preferences. Second, the taken-for-granted desirability of water re-use, widely perceived as inherently innovative and consensual, can lead to an exaggeration of its *relative* benefits – that is, its benefits relative to other options and scenarios that do not include reusing water directly. It is therefore important to examine the extent to which alternative scenarios have actually been taken into consideration, and if so, how they have been designed. Third, given that wastewater is often a highly coveted resource wherever TWWR projects are put in place, the question of *who*, exactly, will benefit from these projects is politically crucial. To what extent are the needs of these beneficiaries, and the social priority that is *de facto* accorded to them, perceived as socially legitimate? We will look at each of these political dimensions of benefit assessment in turn.

1.1. The challenges of quantifying indirect benefits

“A reuse project’s benefits may be diffuse, difficult to quantify, and spread over time”
(Chesnutt, 2021).

Indirect benefits of wastewater reuse are inherently difficult to quantify. Hussain (2007) points out that “total benefits include direct or primary benefits occurring in local settings; indirect or secondary benefits occurring at the local and broader levels; and induced or tertiary benefits occurring in other sectors at the local and broader levels”. The difficulty of measuring positive externalities is therefore related, in part, to the diversity of scales at which they occur, often in nested form. This is what allow Molinos-Senante et al. (2011) to argue, provocatively, that “only a proportion of projects [are] economically viable [when we only consider direct benefits], while when external benefits are incorporated all projects [are] economically viable”. Even though this assertion may be debatable (since even when taking all benefits into account, TWWR projects may still not be, in fact, economically viable), it highlights the centrality of indirect benefits in *legitimizing* TWWR projects.

Moreover, indirect benefits vary widely according to types of use. Recycling water to clean phosphate, for example, carries very different benefits than re-using it to irrigate golf courses. In the following section, we mostly restrict our discussion to water re-use for agriculture, where three types of indirect benefits are commonly invoked, although some of the following remarks may apply to green spaces as well.

First come *health benefits*. As Hadas and Kislev (2011) remind us, “more often than not wastewater and the ways it is treated or reused affect the health and wellbeing of people not directly involved and markets do not exist to evaluate these health and environmental impacts”. Indirect health benefits spring from the fact that the environmental quality of discharged effluents is improved, or at least that poor-quality effluents are no longer discharged (Molinos-Senante et al., 2011). The quality of recreational water is thus improved, as is the quality of downstream water used for other purposes. Indirect health benefits are often emphasized by project promoters. This has been the case since the very beginning of modern TWWR in California, in the early 20th century (Asano et al., 2007: 6-15). They are rarely quantified however, and tend to be used as a general, unfalsifiable argument. While epidemiological knowledge regarding the risks of TWWR is fairly solid, its economic implications are far from straightforward and have not been much quantified so far.

Second, TWWR provides indirect *environmental benefits* (Molinos-Senante & al., 2011). In cases where other sources of water are substituted, at least partially, with TWWR, this makes it possible to better preserve hitherto overused resources, such as an aquifer, which may justify higher prices for TWWR than for groundwater. By reducing wastewater discharges into vulnerable water bodies and ecosystems, TWWR may also help preserve them, as has been the case in some rural areas in France. Moreover, it can provide ecosystems with much-needed water, thereby helping to regenerate them. This has been part of the rationale of a TWWR project in the city of Tiznit, Morocco, where wastewater was originally intended to be used to irrigate an ancient palm grove under water stress; and in the small town of Manguio, in France, where it served to recreate a reed bed. Naturally, these environmental benefits depend to a great extent on the local context, on the original water resource that was used before TWWR (groundwater, raw water from discharge canals, drinking water for green spaces, etc.), and on the intended use of treated water. They are therefore highly context-specific. For example, environmental benefits of TWWR for irrigation, which depend on the flow at the discharge point of the treatment plant, will vary widely from season to season, especially in tourist areas

with massive seasonal variation in the quantity of wastewater produced, while demand may vary widely depending on growing season conditions.

Thirdly, TWWR can have *indirect socio-economic benefits*. These include, for example: maintaining (or increasing) local employment opportunities, supporting local, peri-urban agriculture (Omole et al., 2019), and securing the survival of specific value chains. Besides, as TWWR contains nutrients (phosphorus and nitrogen), irrigators need less fertilizer (Molinos-Senante et al., 2011). However, the nutrient content of wastewater varies from one context to another (e.g. the degree to which treatment plants effectively reduce concentrations in nitrogen and phosphorus), making the application of any universal order of magnitude impractical (Asano et al. 2007). For example, Qadir et al. (2008) showed that the nutrient load in each 1,000 m³ of TWW varied from 4 to 24 kg for phosphorus, from 16 to 62 kg for nitrogen, and from 2 to 69 kg for potassium. These marked fluctuations are a reminder of the complex trade-offs between the different possible uses of wastewater. On the one hand, the economic benefits of agricultural reuse are maximized when the nutrient contents of wastewater (particularly nitrogen and phosphorus) are high, even though these benefits can only be realized if farmers take the nutrient content of wastewater into account in their fertilization practices. On the other hand, the same nutrient content can be detrimental for other uses, such as irrigating golf courses and green spaces, or when wastewater is discharged into receiving environments. The same dilemma applies to the circular economy in general. Are the benefits of the circular economy maximized when the recycled resource is recovered in its original state before any human use? Or is it better to reuse some of the properties and components introduced by human use?

Overall, measuring positive social and economic externalities is usually more controversial than measuring health and environmental benefits, as it is subject to even greater uncertainties. For example, it is often difficult to estimate the additional incomes that may be derived from a project as a result of greater water availability or regularity (enhanced water security). Many uncertainties also affect the estimation of (in)direct job creation; and of economies of scale, although the latter are generally postulated and have the *de facto* consequence of favouring large-scale projects. More often than not, the magnitude of these uncertainties is glossed over in official discourses and feasibility studies.

1.2. The fraught issue of relative benefits

In addition to the question of indirect benefits, estimating *relative* benefits induced by TWWR – that is, in comparison with scenarios with no reuse- is also subject to many uncertainties. It is often challenging to differentiate between benefits that are specific to a TWWR project (such as improved water availability, security and quality) and benefits that would likely accrue even in the absence of such a project. Relative benefits also depend on existing regulations, for instance, on whether crops irrigated with wastewater are subject to specific health safety regulations.

Thus, in Morocco, baseline scenarios vary greatly depending on whether one assumes that access to groundwater will remain heavily subsidized through subsidies for butane gas, which powers water pumps (in which case TWWR may remain costly in comparison); or if one assumes that some of these subsidies will eventually be lifted (in which case TWWR might turn out to be a less expensive alternative in the future). The choice of the baseline scenario is therefore of paramount importance when assessing the benefits of TWWR. Here, a widespread tendency is to use "business as usual" scenarios as baseline scenarios, even though this conflation is questionable. As Meunier and Marsden (2009) explain:

“The status quo is not necessarily the optimal reference scenario. It is often possible to improve results without making major investments. When this is the case, it is the improved status quo that should constitute the baseline scenario to compare the project under consideration. In other words, we are not interested in the situation "before" or "after", but in the optimum situation "with" or "without" the project” (Meunier and Marsden, 2009).

Therefore, devising the baseline scenario is actually a crucial political sequence, which can greatly favor the adoption or rejection of any given project.

1.3. Benefits for whom? The social legitimacy of beneficiaries

Equivalent monetary benefits may trigger different social reactions, depending on whether beneficiaries are deemed legitimate and deserving. If they enjoy only a weak social legitimacy, important benefits may heighten perceptions of injustice which, in the event of long-term restrictions on other users, may degenerate into open conflict. In France, water uses that are widely perceived as low-priority have benefited from TWWR. For example, in the Pyrénées Orientales department which was affected by a severe and unprecedented drought between 2022 and 2024, a project of treated wastewater for a golf course has been met with fierce local opposition.

In contrast, in the case of Tiznit in Morocco, the nature of the beneficiaries is the subject of wide local consensus: they are smallholders who practice traditional agriculture in an ancient palm grove, which would be regenerated by the TWWR project. Paradoxically, the only opponents to the project are some of the targeted farmers themselves, who would prefer to open up their periurban plots to urbanization and enjoy the windfall. They resent the fact that the area is now solidly safeguarded for agricultural purposes, being classified as a "food-producing and palm grove area" (*Zone vivrière et palmeraie*). This is, undeniably, an economic distortion imposed on the local land market which has potentially a much higher value for housing or urban amenity than the value of growing a small area of crop.

2. The politics of cost assessment

Cost estimates of TWWR are fraught with the same mirror-image issues as its benefits. Firstly, indirect costs -that is, *negative externalities*- play a prominent role in how projects are contested and justified. Measuring these costs, therefore, can be a highly contentious undertaking. Secondly, the question of *relative costs* are particularly important for the social acceptance of TWWR projects. It is rare for wastewater to be the only locally available water resource - although this may increasingly happen to be the case in the future, as water scarcity worsens in many places. The way in which TWWR costs compare with the costs of alternative water sources is therefore crucial.

However, the question of beneficiaries' social worth - do they deserve to be prioritized? - has no direct equivalent on the cost side. Compared with benefits, costs are both more diffuse and less transparent to the general public. “Payers” are less readily identifiable as a cohesive social group than beneficiaries. Negotiations regarding cost-sharing and the variety of cost-sharing arrangements therefore take place in specialized rather than public arena, as we analyze in section 3.2.

2.1. Measuring negative externalities

Broadly speaking, four categories of indirect costs can be distinguished. The first two are related to reallocation processes: first between human users; and second, from the environment to human uses. The third type of cost is linked to the effects of contamination on crops and soils. The fourth is linked to increased health risks.

The (hidden) costs of reallocation between human users

In national planning documents, TWWR is often lumped together with other water resources such as desalination, under the generic heading of "unconventional water". This conflation, however, is partly misleading. Seawater desalination, however questionable some of its ecological and economic effects may be, *does* generally constitute a new resource for human users. By contrast, the more an area is subject to water stress, the more likely it is that wastewater, whether treated or not, is *already* being used informally for human purposes (UNWWAP, 2017; Aït-Mouheb et al., 2020). In many cases, therefore, any new use implies a reallocation from one beneficial use to another, with the exception of particular cases where wastewater was directly discharged into the sea prior to the project.

In Morocco, most agricultural TWWR projects involve exactly these kinds of reallocations. In the city of Settat, an official project has been underway since 2009, based on a treatment plant that was completed in 2006. However, the project deliberately overlooked the fact that wastewater was already being reused before the plant was built. Discharged into a small stream, and accounting for most of its flow during the summer, Settat's wastewater had been collected since the 1980s by over a hundred farmers located along the stream, who used the wastewater to irrigate fodder crops on a total area of some 300 hectares, often in order to feed their own livestock. When the project was implemented, these irrigators lost access to the water, which is now conveyed directly to the plant via underground pipes. As a result, farmers not only lost access to a precious productive resource, but also saw the value of their land plummet. According to information obtained from interviews, the rental price of their former plots was halved following the loss of access to wastewater, while the price of newly irrigable plots doubled (Niard and Nidal, 2019). Some public measures were envisaged to compensate for the losses incurred, including the drilling of five collective boreholes that could have provided supplementary irrigation for 200 ha on some parts of the land previously irrigated with wastewater. However, these measures were never implemented, mainly for fear of setting a costly precedent that could have been applied to all future reuse projects (Mayaux and Bensmaïl, 2019).

Likewise, in the case of Tiznit, farmers located immediately downstream of the plant began to use the treated wastewater informally, without waiting for the construction of the planned distribution network. They are well aware that their access to the plant's abundant wastewater will be reduced once the official project is up and running, which will occur at some point in the future. Wastewater will then be partly reallocated and they will only remain with a minority share of it, since only 20 hydrants have been installed in the area, compared to 71 in other areas located upstream of the plant.

As a general rule, then, the positive bias of feasibility studies in favor of TWWR means that they do not account for the welfare losses associated with such reallocations. The costs borne by the losers of the reallocation thus remain hidden.

Reallocation away from the environment

In many places, wastewater plays a vital role in sustaining ecosystems, and TWWR projects can have negative impacts on them. In the case of Ablah (Lebanon), reusing treated water will reduce the ratio of clean water in the polluted Litani river (and further away in Qaraoun Lake). It will also reduce the volume of water available for farmers who pump directly from the river downstream, even though they have no alternative source of freshwater (Nassif, 2016).

Likewise, in Barcelona, a major project currently under consideration consists of pumping treated water from the wastewater plant near the mouth of the Llobregat River into the river some eight kilometres upstream, ahead of the purification plants (located next to the river) that serve the urban area. Thus, this project does correspond neatly to circular economy principles. However, in so doing, it will divert wastewater that is currently discharged into the lower Llobregat. These flows currently sustain the delta wetlands and act as a hydraulic barrier against saline intrusion. Even though the current demand for treated wastewater (13 Mm³ for urban services and industry, 44 Mm³ for agricultural irrigation) should theoretically leave enough to secure environmental flows in the river (up to 31.5 Mm³ per year) many uncertainties remain regarding how much will actually remain in the lower Llobregat in the event of an intense drought, like the one experienced by Catalonia between 2021 and 2024 (March et al., 2023).

Generally speaking, indirect costs to ecosystems are almost never taken into account in the economic evaluation of TWWR: a good illustration of how many of these evaluations are biased in favor of this “no-regret solution”.

Contamination costs

In some cases, generally in costal zones, reclaimed wastewater contains salts that may accumulate in the root zone of the plant, depending on soil properties and frequency of leaching, with possible harmful impacts on both soil health and crop yields. The problem of soil salinity and sodicity can be solved by applying natural or artificial soil conditioners (such as biochar and compost tea). However, soil reclamation measures are costly and may fail to restore the soil to its original level of productivity. As a consequence, irrigation with wastewater may have a long-term economic impact on soil, which, in turn, may affect market prices and the land value of saline agricultural plots.

Effluents emanating from urban wastewater treatment plants are suspected of being among the main anthropogenic sources of pathogens, antibiotics, antibiotic-resistance genes and antibiotic-resistant bacteria released into the environment (Ait-Mouheb et al., 2018). A number of technologies exist for the efficient removal of contaminants from treated wastewater, but several questions remain to be addressed. These include the dynamics of contaminants during water transport and distribution; and the effects on soil microbiota, which have also been neglected in most studies on irrigation with wastewater (Becerra Castro et al. 2015). Likewise, when the capacity of the soil to retain heavy metals declines, heavy metals may leach into the groundwater, hence becoming available for plant uptake (Frost and Ketchum 2000).

Internationally, health risks tend to be measured by the number of years of healthy life lost compared with theoretical healthy life expectancy (Disability Adjusted Life Years, DALYs). However, the costs associated with this possible reduction are rarely quantified in economic studies, since it is both economically difficult, and politically controversial, to attach a price tag on human life. Generally, then, indirect costs are less recognized and discussed than

indirect benefits throughout the decision-making process. This differential attention is emblematic of the economic bias in favor of TWWR.

2.2. How to assess relative costs?

Any potential user will be all the more willing to pay for wastewater if its cost-benefit ratio is better than the one of alternative water resources. Of course, the use of wastewater can simply be made legally compulsory. But if other, more advantageous sources of water are available locally, there will be resistance to TWWR, and its sustainability will be in doubt. Much like any “conventional” source of water, therefore, there is little point in analyzing TWWR costs in isolation: either because users do have actual, physical alternatives to TWWR; or because they compete with, or compare themselves with, users of other water sources.

The costs of water reuse vary greatly from one use to the next -as wastewater that is “fit for purpose” will have gone through varying levels of treatment- and from one location to the next. Comparing these costs to alternative water sources is also inherently complex: firstly, because the cost structures of different types of water provision tend to differ markedly, particularly the ratio between capital expenditures and operating expenditures, which makes their overall comparison intrinsically difficult; and secondly, because relative prices constantly fluctuate, in particular due to changes in public subsidies that partly determine them. Nonetheless, it is possible to indicate some typical orders of magnitude with which TWWR compares to other water sources, as this comparison is partly independent from the details of treatment processes and the exact level of water quality pursued, with many common features of TWWR related to particular treatment costs, infrastructure maintenance, energy costs and monitoring issues. In doing so, however, it is preferable to limit oneself to operating and maintenance costs. Investment costs vary enormously depending, for example, on the size of the project and the number of users involved, which makes general estimates very difficult. We are also faced with a dearth of information regarding total, actual investment costs, as well as the volume of water that is either demanded or actually produced. This makes it impossible, in this paper, to compare water projects using a ratio such as the investment cost per m³.

Table 1 thus breaks down the main types of operation and maintenance costs for TWWR. It compares them to three other typical sources of water used for agriculture and green spaces: surface water bodies (SWB); groundwater (GW); and drinking water (DW). In all three cases, we are referring to situations in which users already have access to some of these water sources: these are the most plausible situations since TWWR is generally intended to replace, complement or secure uses that *already* exist. Finally, it should be borne in mind that relative costs depend to a great extent on regulations, including water quality standards.

Our starting point is therefore a situation in which users already have access to some sources of water, and where a turn to wastewater is contemplated. How, then, do the operating and maintenance costs of TWWR compare with those of alternative water resources?

Table 1. Relative O&M costs of TWWR (source: the authors)

Cost categories	Differential vis-à-vis surface water bodies	Differential vis-à-vis groundwater	Differential vis-à-vis drinking water
Water treatment <i>Operation of tertiary treatment system (equipment, energy, chemicals, etc.)</i>	TWWR costs more	TWWR costs more	TWWR costs the same
Storage <i>Cleaning, dredging and maintaining the storage basin</i>	TWWR costs more (e.g. higher risks of eutrophication)	TWWR costs more (e.g. higher risks of eutrophication)	TWWR costs more (e.g. higher risks of eutrophication)
Infrastructure maintenance <i>Preventing clogging of filters, pumps, pipes and irrigation drippers</i>	TWWR costs more (e.g. higher risks of clogging)	TWWR costs more (e.g. higher risks of clogging)	TWWR costs more (e.g. higher risks of clogging)
Energy for water distribution <i>Energy used to pump water uphill, as WWTP tend to be more systematically located downhill</i>	TWWR costs more	TWWR cost more than some groundwaters (depending on groundwater depth and pumping mode)	TWWR costs more
Monitoring <i>Ensuring compliance with (usually) stricter water quality regulations (e.g. microbiological parameters and heavy metals), (more frequent) water tests and water "barriers"</i>	TWWR costs more (according to most -but not all- regulations).	TWWR costs more (according to most -but not all- regulations).	TWWR costs more (according to most -but not all- regulations).

First, additional treatment costs. As a rule, regulations that apply to standard treated wastewater only aim to ensure that it does not harm the receiving environment. However, the resulting quality may not be sufficient to minimize health risks for wastewater direct re-users, for nearby residents and, in the case of TWWR for agriculture, for the consumers of food products. Particular risks are related to bacteriological contamination (mostly through fecal coliforms, including *Escherichia coli* - *E. coli*) and contamination by parasitic worms (e.g. helminth eggs). Additional treatment processes are thus necessary for disinfection, for example using UV radiation or chlorination (Ait-Mouheb et al., 2020). In Morocco, the Ministry of Agriculture estimated that the cost of this “tertiary” treatment represented somewhere between 32% and 44% of the total operating costs of a water reuse project for agriculture (MAMF, 2016).

Second, storage costs. A treatment plant does not provide a regular volume of water over the course of the day. There is a significant contrast between peak hours (typically in the early morning and early evening) and off-peak hours. For this reason, relatively large storage basins must be built and properly maintained to allow for variabilities and adapt to consumption patterns (such as irrigation and watering green spaces). However, wastewater tends to contain more nitrates and phosphates than other waters. This is particularly true when plant managers want to keep part of its nutritional content for use for agriculture. In this case, algae proliferate in the storage basins, and their regular removal represents an additional cost.

Third, maintaining the irrigation infrastructure has its own challenges. The presence of nutrients and ions in wastewater increases the possibility of chemical precipitation, and of the development of biofilm in drippers (Lequette et al., 2020). The additional concentration of solids in wastewater compared to conventional water is not necessarily taken into account in the design and installation of equipments (e.g. the need for high-flow rate drippers, filtration, etc.) or in the maintenance protocols for drip irrigation systems (by chlorination and acidification). This can lead to the rapid loss of investment, and of the initial subsidies provided for drip irrigation, or the massive use of disposable dripper systems (3–6 months of shelf life) which is environmentally unsatisfactory (Ait-Mouheb et al., 2020).

Fourth: energy costs for water distribution. These tend to be higher for TWWR than for most water sources, as treatment plants are usually located on lower ground to enable discharge of wastewater by simple gravity. Some reuse sites may, of course, be located still further downhill; however on average, TWWR is located on lower ground than other water sources and so the probability of having to pump water uphill is higher. In Morocco, the cost of energy required for pressure distribution can represent up to a third of the total price of a cubic meter of water (MAMF, 2016). However, it should be noted that exploiting groundwater also involves significant energy costs, which according to Omole et al. (2019) can be even higher than for TWWR, although they are significantly lowered when solar energy is used.

Fifth, monitoring tends to generate additional costs. Due to the potential risks that TWWR pose to human health, TWWR usually has to comply with stricter regulations than conventional water sources, especially those regarding microbiological pathogens and heavy metals. Water needs to be tested more frequently, and, when TWWR is used for agriculture, specific monitoring may also be required for food products. However, the difference in the costs of monitoring TWWR vis-a-vis other water sources varies with the type of crops irrigated, irrigation technologies used, and the stringency of regulations. In many situations, there is a significant gap between regulatory frameworks applying to

TWWR and those applying to conventional waters, although it should be noted that in a minority of cases, regulations are identical. In this situation, which prevails in many Latin American countries, TWWR is considered a water source just "like any other". In Morocco, by contrast, the average annual cost of monitoring water quality was evaluated, a decade ago, at Dh 65,000 (US\$ 6,429), accounting for 5-10% of the total operating costs of large schemes (> 200 ha) and 10-25% for smaller schemes (MAMF, 2016).

Finally, O&M costs are not necessarily reduced when a "multi-barrier" approach is deployed. This approach has been promoted by the WHO (2006) and is the paradigm that informs various national legislations. It recognizes the difficulty in obtaining, on a permanent basis, a quality of wastewater that makes it "fit for use". As a solution, it seeks to complement water reuse with additional precautionary measures, known as "barriers". Originally, this approach was designed to reduce treatment costs, particularly in developing countries, while still ensuring safe water reuse. However, it generates a paradox since monitoring the adequate application of each and every barrier actually requires considerable organizational resources. In Tunisia, for example, regulations stipulate that "livestock feed must be stored long enough to minimize the risk of contamination"; and that "a fruit that falls to the ground shall not be eaten or sold"; whereas Algerian regulations stipulate that treated wastewater must "under no circumstances be the cause of water stagnation, unpleasant odors or larval breeding sites". Effectively implementing these rules is obviously very costly in administrative terms.

Thus, on the whole, TWWR tends to have higher operating costs than other types of water. These higher costs may be perfectly acceptable if wastewater has corresponding benefits, for example in terms of water reliability and reduced need for fertilizers. In the case of Ablah in Lebanon, for example, total operating and maintenance costs were estimated to be US\$ 53,278/year for 25 ha, or around US\$ 2,130/year/ha, since the cost of the operation in energy is very high, as wastewater is pumped uphill three separate times. This is considerably higher than individual pumping costs that, according to one survey, vary from US\$ 310 to US\$ 830/ha depending on the depth of the wells. At the same time, however, treated wastewater should be of better quality than the heavily polluted groundwater currently tapped by wells, and supply would be more reliable in the summer. Besides, nutrients contained in wastewater would reduce fertilization costs by around 25%, explaining why one farmer was reported to be willing to pay as much as US\$ 3,650/ha/year for TWWR (Nassif et al., 2022b). This highlights the relevance of adapting treatment levels to the desired use of nutrients. Removing nitrogen from wastewater requires additional energy throughout the treatment process, for example through aeration in an activated sludge treatment plant, which can represent up to 40% of total energy costs (Nassif et al., 2022b). This cost is reduced if some of the nitrogen remains in the water.

3. The politics of cost-sharing

Since most governments emphasize the importance of providing "new" water resources in water-stressed areas, they usually recognize the need to subsidize *investment* in TWWR. However, with respect to *operation and maintenance* costs, the prevailing international norm sets out the principle of full cost recovery through water rates (Eid-Sabbagh, 2014). In this section, we first emphasize the considerable difficulty of implementing this objective. Then, we highlight the diversity of actually existing cost-sharing arrangements. Finally, we explore how these various arrangements are normatively justified by stakeholders.

3.1. Full cost recovery: an impracticable standard?

The contrast between, on the one hand, investment that is readily subsidized, and on the other, operation and maintenance that should be fully funded by users, is a reflection, in the water sector, of the globally dominant paradigm of the investor state. According to Lepont (2024), this type of state does not hesitate to intervene massively to modernize and steer the economy. However, in so doing, it gives priority to supporting profitable activities. Because of this expected profitability, once the initial investments have been made, the state can offload the funding of operation and maintenance onto the target population. In line with this general principle, the US Environmental Protection Agency states that “revenues from rates and charges should be sufficient to provide annual operating maintenance and repair expenses, capital improvement costs, adequate working capital, and required reserves” (US EPA, 2012: 7-1).

However in practice, the objective of recovering the full O&M costs through water rates is rarely achieved. Many studies have pointed that, in most parts of the world, farmers who use treated wastewater are unlikely to agree to pay the corresponding rates (Hanjra et al. 2015; Gebrezgabher et al. 2022). This is particularly true when the first beneficiaries of TWWR are relatively poor smallholders (Mayaux and Ennabih, 2023). The WHO itself recognizes that there may be a trade-off between cost recovery and the overall attractiveness of water reuse, stating that “while it is, of course, desirable to ensure the maximum recovery of costs, an important consideration is to avoid discouraging the permitted use of wastewater. Some a priori investigation of the willingness and ability to pay is therefore essential in determining not only the level of charges, but also the frequency, timing and means of payment” (WHO, 2006:136).

As a result of this tension between international standards and the realities of economic viability on the ground, national legislations vary widely in their stipulations regarding the possibilities for subsidizing reuse.

Table 2 gives an overview of economic regulations in four Mediterranean countries (Morocco, Algeria, Tunisia and the Palestinian territories), both for investments (first two rows) and for operation and maintenance (two following rows).

As we can see, most, though not all, legislations provide for the (limited) possibility of subsidizing operation and maintenance. However, a number of legislations remain silent on important dimensions. This leaves the stakeholders of each particular project free to negotiate their own financial arrangements. As a consequence, TWWR may continue to be subsidized on an ongoing basis, with very different cost-sharing arrangements depending on the institutional context and local configurations.

Table 2. Financing mechanisms under 4 different legislations (source: Mayaux, 2022).

	Morocco	Algeria	Tunisia	Palestinian territories
Public financing of additional treatment and storage facilities	Possible: - Subsidies from river basin organizations (<i>Agence de Bassin Hydraulique</i>) - WWTP operator on a voluntary basis (in "mutual agreement" with users)	Not specified (NS)	NS	NS
Public financing of hydro-agricultural schemes using TWWR	Yes: Min. of Agriculture, both for infrastructure and localized irrigation	Yes: Min. of Agriculture	Yes: Min. of Agriculture (through its regional agricultural offices - CRDA)	Yes: Min. of Agriculture
Legal possibility of subsidies for O&M	No. Principle of full cost recovery for O&M from farmers	No ("The operator is responsible for the preventive maintenance and repair of the works and pipes of the treated wastewater distribution network")	Yes (CRDA for preventive maintenance and repairs)	Yes (pumping costs can be subsidized)
Legal possibility for public funding of water tests	Yes ("users, owners or managers of wastewater treatment plants" can bear the costs of water tests)	No (at the user's expense)	Yes (water provider, on a voluntary basis)	Yes (both through Min. of Agriculture and Min. of Health laboratories)
Eligibility criteria for public subsidies	The project must - achieve "water savings" - end "contamination of the receiving environment". Users' application submitted to a commission made up of representatives of: - Agences de Bassins Hydrauliques - Financing Authority - Infrastructure authority - the authority in charge of the Environment; - the authority responsible for the TWWR user sector.	NS	NS	NS

3.2. A variety of cost-sharing arrangements... and tensions

Contrary to what most economic TWWR studies seem to suggest, cost-sharing is fraught with uncertainty and tensions, resulting in highly heterogeneous arrangements and compromises.

In general, as mentioned above, the bulk of the investment is borne by the state, that is, by the taxpayer. This can take place at different scales, from the local to the national. In France, for example, river basin agencies have subsidized a large proportion of TWWR infrastructures as they took the lead in promoting this solution. In so doing, regional domestic users, coupled with local authorities, finance the bulk of these investments, since it is they who mostly fund the agencies (up to 80-85%).

This is a common pattern. In all the countries we studied, particularly those around the Mediterranean, users pay only a small share (typically less than 20%) of investment costs, and often no share at all. A partial exception of this is Morocco, where the National Integrated Sanitation Plan adopted in 2018, allocates 60% of investment expenditures to the State and 40% to users. However, while this breakdown has broadly been applied to green spaces and golf courses, it has not been applied to agriculture.

Concerning O&M costs, domestic users usually contribute significantly to cost recovery through their wastewater fees. This is true both for agricultural and for recreational projects (e.g. golf courses). Beyond this general pattern, however, arrangements vary greatly.

In Lebanon, for example, a study conducted for the rehabilitation and extension of a TWW system for agriculture in Ablah, in the Bekaa region, recognizes that the issue of cost-sharing is fraught with deep uncertainty. Its authors admit that their findings “challenge the objective of covering the reuse system’s operational costs from users and generating income for operators, as hoped by the municipality of Ablah and as promoted by donors” (Nassif et al., 2022b, p.29). They estimate that “more than half of the cost of operation will need to be subsidized”. To achieve this objective, they contemplate various options, including subsidies from the central government based on levied income tax, and taxes on big farms and commercial enterprises in the Bekaa region. However, they recognize that all these options would frontally challenge the neoliberal logic of full-cost recovery from users. The same orders of magnitude, and considerations, are at play in the Zahle project, located in the same Litani watershed.

In some cases, negotiations between public authorities and farmers are clearly antagonistic. In Tunisia, the case of the Zaouiet Sousse project is a good illustration of these ongoing tensions. Ever since the creation of the Water Users’ Association (WUA) in 2001 there have been endless negotiations between the WUA and the Regional Agricultural bureaucracy (French acronym CRDA) over the sharing of operational costs. To alleviate its financial burden, mainly related to energy costs, the CRDA has been pushing the WUA to increase the price of water, a move that has to be formally approved by the WUA, but to little avail. Until 2017, water prices remained at 20 millimes per m³ (less than one US cent). According to CRDA’s calculations, however, it should be about 85 millimes (2.7 US cents) if the WUA were to cover just a third of the electricity bills, which is the CRDA’s minimum stated objective. The WUA only agreed to increase the price to 35 millimes in 2018 and to 45 millimes (1.4 US cents) in 2019, after which it refused to comply with the CRDA’s demand. As one CRDA staff member recounts,

“In Zaouiet Sousse, the WUA does not want to increase the price of water because this is not financially convenient for board members (...). With the scheme’s long history and the number of repairs that have to be done, adjusting the price is absolutely necessary, but they don’t want to do it (...). Therefore, in order for the CRDA to recover at least a share, we suggested they cover a third of the energy bill, but they categorically refused. They said that there was already a lot of tension, that there wasn’t enough water. We said, okay, maybe not a third, but at least pay something, but again they refused. So, we said, listen, as an administration, we have decided to charge you for some of the water we distribute to you (...). So we sent

them subscription contracts with 20 additional millimes per cubic meter, but they refused to pay. And we left it at that”.

The same trend can be observed in projects currently underway in France, irrespective of the use (irrigation, golf courses). Most of the costs are borne by stakeholders other than the direct beneficiaries. Investment costs are often covered by basin agencies, or by the local authority that initiated the project, whereas treatment and storage are included in the wastewater treatment budget, with TWW users bearing only the cost of pressure distribution, in the same way as for conventional irrigation water.

These diverse arrangements are not simply the product of local power configurations. They are also justified by various arguments. Debates around pricing imply different conceptions of TWWR, especially along a continuum that ranges from considering TWWR as a purely market activity to viewing it as a public service.

3.3. Who should pay? *The cultural political economy of TWWR*

In many cases, it seems that TWWR would be economically viable only if a large proportion of its costs, including its running costs, were socialized. This has prompted some experts to redefine its rationale, placing greater emphasis on indirect benefits and its contribution to social welfare. As a Moroccan expert summed up, “if there is a political will to reuse, then you have to think macro, not purely in terms of internal economics. If I want to save on conventional sources, reduce the water deficit, etc., then that means I have to be willing to ‘put my hand in my pocket’ [*i.e.*: *be willing to pay*]. When it comes to water reuse, it is impossible to think in terms of conventional economics. You have to think in terms of positive externalities on a macro scale²”.

Subsidizing operation and maintenance can be justified by appealing to a variety of general interests, such as securing local family farming; using wastewater instead of other, over-exploited water resources; and ensuring better protection of receiving environments (in the latter case, based on the same rationale as sanitation itself, which, in most countries, is largely subsidized in the name of preserving a healthy environment for all). In Spain, for example, support for local agriculture justifies calling on the solidarity of urban users with the partial financing of TWWR by urban dwellers through their sanitation bills.

However, this general interest is rarely made explicit in legal frameworks or policy documents (Mayaux, 2022). Morocco is a partial exception, as public subsidies to TWWR projects are possible but explicitly subject to one of two conditions: they must enable “water savings”; and/or have the ability to stop contamination of the receiving environment. These conditions echo some of the arguments used to justify circular economy initiatives more broadly. However, the capacity of these very general criteria to effectively discriminate between alternative projects remains uncertain.

While taking note of the often vague justification for subsidies, we should pay attention to the way in which the notion of public service can be strategically, rather than distastefully deployed. For example, farmers have a vested interest in considering that

² Interview, Rabat, November 7, 2019.

water reuse carries so much collective benefit that it should be considered a public good. This would justify farmers' not (fully) paying for it, as they would in fact be providing a service to the whole community by using it.

Whatever the specific context, the local construction of a general interest can only be legitimate if it is properly debated and validated through an inclusive decision-making process, which presupposes involving civil society organizations early in the process.

Conclusion

Treated wastewater reuse is widely promoted as a key component of a circular economy. Its promoters argue that it would allow to intensify water uses all along the water cycle, while still better preserving the environment. However, so far, the question of how to assess its costs and benefits as comprehensively and holistically as possible has received little scrutiny. Few cases have had their actual – as opposed to ideal – political economies analyzed in detail. In this context, this article set out to challenge the assumption that the circular economy in general, and TWWR in particular, was bound to generate net benefits. By looking at TWWR through political-economic lenses, we explored how its economic dimensions (its different categories of costs and benefits, its cost-sharing arrangements) were profoundly shaped by its politics. We highlighted the way in which the assessment of its costs and benefits, far from being a settled affair, was persistently beset by debates, negotiations and contestations; and we explored the diversity of cost-sharing arrangements that resulted from these debates, negotiations and contestations. In this highly political economy, we showed that the interests of TWWR promoters tended to carry the most weight, resulting in a "positive economic bias" towards water reuse. The paper has highlighted three dimensions of this politicization in particular.

The first dimension is the recurrent discursive emphasis put on indirect benefits, while indirect costs receive comparatively less attention. Most consultancy reports, and most academic studies, emphasize positive externalities on health, the environment, the use of fertilizers and local job creation. On the other hand, they are much less forthcoming about hidden costs, such as the reallocation away from particular human users and ecosystems; declining yields in the medium term due to the accumulation of salts and pollutants; and the high running costs of monitoring and controlling water quality. Our point here is not that these indirect costs necessarily outweigh the benefits. For example, in the case of water reallocation, it might well be argued that former informal wastewater users should be aware that they are only making a windfall gain until the piped distribution system is built, and should therefore invest appropriately while it lasts. Rather, our point is that these estimates and comparisons should be developed further, and debated in public arenas, even if it implies a time-consuming process of deliberation and negotiation.

The second dimension concerns the consideration of alternative scenarios. More often than not, alternatives to water re-use are reduced to "business as usual" scenarios, which give an exaggeratedly degraded picture of alternative options and merely act as a convenient foil that help promote TWWR. On the contrary, policy-makers should design a variety of alternative, in-depth scenarios, and compare TWWR with the most favorable of these. It might be relevant, in particular, to design "no-TWWR" scenarios that include strategies for reducing water consumption, with their associated shifts in land use towards other forms of agriculture, and towards other types of land use altogether. More broadly, within these two general scenarios -with and without TWWR- it would be preferable to

design a range of specific, plausible scenarios, as this would more accurately reflect the inherent uncertainties underlying any projection (European Commission, 2015; Meunier and Marsden, 2009).

Thirdly, economic arguments are often put forward informally, without being properly quantified. This is particularly the case for indirect benefits. Vague references are often made to the benefits of safeguarding local value chains, or creating local jobs, without specifying their order of magnitude, let alone comparing them with alternative scenarios. The point here is not that all arguments used in public arenas should necessarily be quantified and economicized; but rather that whenever economic 'evidence' is invoked, with all the authority that it usually conveys, some quantification should be put on the table if it is to be properly debated. In so doing, economic analysis should foreground its own uncertainties and give an indication of their range, so that decision-makers are made fully aware of them (following the general principles that it is more difficult to assign a monetary value to indirect costs and benefits than to direct ones; and that the more temporally distant the costs and benefits, the more uncertain they are). Conducting a cost-benefit analysis also involves determining a discount rate that expresses, in numeric form, the intensity of stakeholders' preference for the present (the higher the rate, the more importance is attached to present actions relative to future ones). A sensitivity analysis should be carried out for the selected rate in order to identify, and place under public scrutiny, the value at which the overall decision would be reversed.

Again, the observation of these biases in favor of TWWR, which often go unexamined, is not an argument against TWWR as such. It is, rather, a case for setting up, as early as possible in the decision-making process, inclusive forums for debate and decision. Ideally, these should not be limited to the existing arenas of local representative democracy. Experience has shown that a few sessions of debate within a local council cannot suffice for a detailed examination of the political economy of TWWR, and of the circular economy more broadly, and make this eminently technical issue the subject of a truly democratic decision-making. Instead, project-specific participation arenas should be set up, where all relevant social interests can be adequately represented and have an effective influence on decisions.

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